AVIAN PREDATION SYNTHESIS REPORT

Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management

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A Synopsis of Ecology and Management

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PREFACE

Beginning in 1997, our group conducted numerous research, monitoring, and evaluation (RM&E) studies to assess the impact of predation by Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), and other native piscivorous colonial waterbirds on the survival of Endangered Species Act (ESA)-listed salmonids (*Oncorhynchus* spp.) from the Columbia River basin (CRB). As part of this work, well over 50 technical reports and 40 peer-reviewed scientific publications have been prepared that covered a wide range of topics regarding avian predation in the CRB (see www.birdresearchnw.org). In general, these studies investigated the nesting ecology of piscivorous colonial waterbirds, and their predation impacts on fish of conservation concern. The studies of the ecology of piscivorous colonial waterbirds have largely focused on nesting locations, colony sizes (number of breeding adults), nesting success (number of fledglings produced), foraging ecology, and inter-colony movement and dispersal patterns within the CRB and throughout the Pacific Flyway. Studies of the impacts of avian predation, on the other hand, have mostly focused on diet composition (proportion of different prey types consumed) and the number and percentage of available prey consumed (predation rates) by piscivorous waterbirds nesting at various colonies in the CRB. This long-term research program has identified predation by piscivorous colonial waterbirds at some breeding colonies as a factor that limits the survival of some ESA-listed salmonid populations in the CRB. Consequently, addressing concerns over avian predation rates has become a component of the Biological Opinions and other plans for restoration of ESA-listed salmonids issued by the Federal National Marine Fisheries Service (NMFS) for management of the Columbia River Power System.

Based on results from earlier studies, the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps (USACE), and the Bureau of Reclamation (BOR), with direction from the NMFS, developed and implemented three different management plans to reduce avian predation rates on juvenile salmonids from some of the largest breeding colonies of piscivorous waterbirds in the CRB: Caspian tern and double-crested cormorant colonies on East Sand Island in the Columbia River estuary and the Caspian tern colonies on Crescent Island in McNary Reservoir and Goose Island in Potholes Reservoir in the Columbia Plateau region. The primary goal of management was to reduce predation rates on ESA-listed salmonid populations by reducing the number of birds nesting at each of these four colony sites. All four colonies targeted for management had formed and expanded since 1980 and, while not implicated in declines of salmonid populations in the CRB, had been identified as factors limiting the recovery of some ESA-listed populations. Management initiatives implemented at Caspian tern colonies in the CRB were primarily non-lethal approaches (i.e. passive and active nest dissuasion), whereas management of the large double-crested cormorant colony on East Sand Island used a combination of lethal (i.e. culling and egg-oiling) and non-lethal (i.e. active nest dissuasion and reduction of nesting habitat) strategies. As part of the two management plans for Caspian terns, alternative nesting habitat was created or enhanced at various locations outside the CRB (i.e. interior Oregon, northeastern California, and southern San Francisco Bay) for nesting Caspian terns displaced from the managed colony sites at East Sand, Crescent, and Goose islands. No alternative nesting habitat was provided as part of the management plan for double-crested cormorants nesting on East Sand Island.

In addition to the managed breeding colony sites on East Sand Island, Crescent Island, and Goose Island, a number of other breeding colonies of Caspian terns and double-crested cormorants, as well as colonies of other species of piscivorous waterbirds, such as California gulls (*Larus californicus*), ring-billed gulls (*L. delawarensis*), glaucous-winged/western gulls (*L. glaucescens* x *occidentalis*), and American white pelicans (*Pelecanus erythrorhynchos*), are present in the CRB. Some of these gull and pelican colonies are quite large and previous research indicates that they may consume substantial numbers of available salmonids under certain conditions. While no gull or pelican colonies are currently managed in the region, data on colony locations, colony sizes, diet composition, and smolt consumption rates were determined for some colonies in some years and are presented herein.

The primary goal of this report, “Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management,” is to assemble and summarize results from previous published and unpublished studies of avian predation in the CRB and to present results from retrospective analyses of data from previous studies to identify long-term patterns and trends. Results of these analyses are vital for assessing the efficacy of management initiatives to reduce predation impacts on ESA-listed juvenile salmonids by Caspian terns and double-crested cormorants in the CRB and to inform an adaptive approach that ensures that management objectives are fully achieved and sustained into the future. These analyses are also essential for evaluating how management actions have and will continue to affect populations of piscivorous colonial waterbirds in the CRB and throughout the Pacific Flyway, populations of native species that are protected by the Migratory Bird Treaty Act. Finally, we hope that this report will serve as a comprehensive repository of results from RM&E of avian predation during 1997–2019 and provide a single-source document on the topic for regional managers, stakeholders, researchers, and the public.

This synoptic report is organized into 10 chapters and four appendices. Chapters 1–5 describe the results of management of colonies of piscivorous colonial waterbirds in the CRB to reduce predation rates on ESA-listed salmonids. ***Chapter 1*** focuses on management of Caspian terns nesting in the Columbia River estuary and how management actions have affected predation on juvenile salmonids as well as the breeding population of Caspian terns in the estuary. ***Chapter 2*** examines the management of Caspian terns in the Columbia Plateau region and how management actions have influenced predation rates on salmonid smolts and the breeding population of Caspian terns in the Plateau region. ***Chapter 3*** summarizes the results of efforts to provide alternative nesting habitat for Caspian terns outside the CRB to attract them away from the CRB concurrent with the managed reduction of Caspian tern nesting habitat and displacement of breeding Caspian terns from the CRB. ***Chapter 4*** describes the nesting and foraging ecology of double-crested cormorants in the Columbia River estuary and how management actions have affected that ecology and the impact of cormorant predation on survival of salmonid smolts in the estuary. ***Chapter 5*** details an investigation into how predation rates on salmonid smolts by Caspian terns and double-crested cormorants nesting in the Columbia River estuary have been influenced by the locations of nesting colonies within the estuary. ***Chapter 6*** summarizes what is known about colonies of piscivorous waterbirds in the CRB that have not been subjected to management to reduce predation rates on juvenile salmonids, including information on the distribution and size of colonies and what is known about their impacts on smolt survival.

Chapters 7–10 explore the effects of avian predation on the mortality and survival of salmonids of conservation concern from the CRB. ***Chapter 7*** describes the cumulative impacts of predation by piscivorous colonial waterbirds on survival of smolts from selected salmonid populations in the CRB. ***Chapter 8*** investigates whether avian predation on selected populations of salmonids is primarily an additive or a compensatory source of smolt mortality, which influences the extent to which managed reductions in avian predation would enhance smolt and smolt-to-adult survival. ***Chapter 9*** focuses on the question of whether avian predation on juvenile salmonids in the CRB is density-dependent by investigating functional responses in per capita predation rates. Finally, ***Chapter 10*** examines the factors that influence the susceptibility of salmonid smolts to avian predation in the CRB, including intrinsic (e.g., smolt condition) and extrinsic (e.g., river discharge/spill) factors. *Appendix A* provides details on the standardized methodologies used to collect project data; *Appendix B* lists the sizes and smolt predation rates associated with the various colonies of piscivorous waterbirds in the CRB; *Appendix C* describes an analytical procedure for estimating future predation rates on juvenile salmonids by Caspian terns nesting on East Sand Island in the Columbia River estuary; *Appendix D* is an annotated bibliography of key references cited in the report.

The management of avian predators in the CRB as an approach to enhance the survival of salmonids of conservation concern has caused considerable controversy within and among the various management agencies, stakeholder groups, publics, and scientists with interests in the CRB ecosystem. Anadromous salmon and steelhead are generally considered an invaluable natural resource in the Pacific Northwest and an icon of the quality of life for the residents of the region. Most managers consider the “4 Hs” (harvest, habitat, hydrosystem, and hatcheries) to be the main factors responsible for salmonid declines and the listing under the ESA of 13 of 20 populations of salmonids from the CRB. The magnitude of smolt losses to predation by some piscivorous waterbirds in the CRB, however, surprised many fisheries managers in the region and prompted swift action by management agencies to reduce those losses. Consensus among managers, scientists, and stakeholders on the science and policy of whether and to what extent avian predation should be managed, however, has been elusive. Some have viewed management of avian predation as a diversion of limited resources from addressing the root causes of salmonid declines (i.e. the “4 Hs”). These divergent viewpoints have put the science of avian predation on salmonids in a political hotseat, where one side views avian predation as a major threat to salmonid populations from throughout the CRB and the other views any attempts to reduce avian predation rates on CRB salmonids through management as a misguided effort destined to provide little in the way of benefits toward salmonid recovery, while raising conservation concerns for some protected populations of avian predators.

We have attempted to provide an objective synopsis of the topic of avian predation on juvenile salmonids in the Columbia River basin during 1997–2019 in this 10-chapter report, while acknowledging the controversy surrounding the issue. Previous management policy for avian predators in the basin has been shaped by political forces that reflect the diverse values of stakeholders and the differing resource management mandates of federal, state, and tribal agencies. Although we do not expect the scientific results described in this report to change the values and political views of stakeholders, we believe that the best policies are built on a solid foundation of science that informs management. We hope that this report will make a substantive contribution to that foundation.

CHAPTER 1: Caspian Tern Management in the Columbia River Estuary

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1.1 Summary

The primary objective of this synoptic study is to present and evaluate results of management actions designed to reduce predation rates by Caspian terns (*Hydroprogne caspia*) on juvenile salmonids (*Oncorhynchus* spp.) listed under the U.S. Endangered Species Act (ESA). Caspian terns nesting at the large breeding colony on East Sand Island near the mouth of the Columbia River were known to consume about 5 million juvenile salmonids/year, and predation rates on some ESA-listed salmonid populations were as high as 25%. The management objective was to reduce the size of the Caspian tern colony on East Sand Island by about 60%, and thereby reduce tern predation rates on juvenile salmonids in the Columbia River estuary by a similar percentage.

Our research on Caspian tern nesting and foraging ecology in the Columbia River estuary began during the 1997–1998 breeding seasons, when we investigated the large Caspian tern colony on Rice Island, an artificial dredged material disposal island 29 river km up-river from the mouth of the Columbia River. Then, during 1999–2001, we monitored management efforts to dissuade terns from nesting on Rice Island as a first step in reducing tern predation rates on ESA-listed salmonids. During the 1999–2007 breeding seasons, we studied the restored Caspian tern colony on East Sand Island, 21 river km closer to the mouth of the river, the managed alternative nesting site for terns displaced from the Rice Island colony. Finally, during the 2008–2019 nesting seasons, we implemented effectiveness monitoring of management efforts to reduce the size of the Caspian tern colony on East Sand Island, while precluding terns from nesting at Rice Island and other sites in the estuary other than the designated colony site on East Sand Island. We compared the nesting and foraging ecology of Caspian terns at the East Sand Island colony between the management period (2008–2019) and the pre-management period (2000-2007).

Our research, monitoring, and evaluation (RM&E) efforts on Caspian terns in the Columbia River estuary included recording tern nesting chronology; describing use of nesting habitat by terns; measuring tern colony size, nesting density, and nesting success; assessing factors limiting tern colony size and nesting success; describing tern foraging behavior, diet composition, and consumption of juvenile salmonids; and quantifying tern predation rates on ESA-listed salmonid populations. In addition, we banded over 7,200 Caspian terns at colonies in the Columbia River estuary, over 3,500 Caspian terns at colonies in the Columbia Plateau region of the Columbia Basin, and over 2,600 Caspian terns at colonies elsewhere in the Pacific Flyway to monitor vital rates of the population and assess inter-colony movements and dispersal patterns of terns to and from colonies in the Columbia River estuary. We also conducted aerial surveys throughout the study period to search for incipient Caspian tern colonies in the lower Columbia River and in estuaries on the outer coast of Washington.

The management plan "*Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary*" (hereafter the *Caspian Tern Management Plan*) was jointly developed by the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps of Engineers (USACE), and the National Marine Fisheries Service (NMFS). The *Caspian Tern Management Plan* was first implemented in 2008 and then updated in 2015, with ongoing implementation through the 2019 nesting season. While the goal of the *Caspian Tern Management Plan* was to reduce predation rates on ESA-listed juvenile salmonids, especially steelhead smolts, by Caspian terns in the Columbia River estuary, the objective of the *Plan* was to reduce the size of the Caspian tern breeding colony on East Sand Island to 3,125–4,375 breeding pairs by reducing the area of suitable nesting habitat on East Sand Island from 6.0 acres (2.4 ha) to 1.0 acres (0.4 ha), while preventing Caspian terns from colonizing other sites in the Columbia River estuary. This represented a ca. 83% reduction in bare-ground nesting habitat prepared for Caspian terns on East Sand Island compared to the pre-management period. Caspian terns were provided with just 1.0 acres (0.4 ha) of nesting habitat on East Sand Island during the 2015–2019 nesting seasons. Fence rows of landscape fabric, plus networks of stakes, ropes, and flagging, were installed to dissuade terns from nesting on otherwise suitable nesting habitat outside the designated colony area.

Peak tern colony size on the 1-acre (0.4-ha) colony area during 2019, the last year of this study, was about 3,860 breeding pairs (95% confidence interval = 3,667–4,055 breeding pairs) and within the range of target colony sizes stipulated in the *Caspian Tern Management Plan* (3,125–4,375 breeding pairs). Tern colony size in 2019 represented about a 57% decline in colony size compared with average colony size during the pre-management period (9,080 breeding pairs). Of the 1.0 acres (0.4 ha) of bare-ground habitat provided for tern nesting in 2019, 86% (0.86 acres or 0.35 ha) was occupied by nesting terns; the remaining area of tern habitat was occupied by nesting glaucous-winged/western gulls (*Larus glaucescens x L. occidentalis*), which are larger than Caspian terns and are major predators on tern eggs and young. During the last five years of this study (2015–2019), when the area of tern nesting habitat on East Sand Island was limited to 1.0 acres (0.4 ha), hundreds of pairs of Caspian terns prospected for nest sites and attempted to nest on East Sand Island outside of the designated colony area. These satellite tern colonies formed later in the breeding season, after the designated tern colony area was fully occupied. Although a few breeding pairs in satellite colonies on East Sand Island were successful in raising young in 2015 and 2016, during 2017–2019 all nesting attempts in satellite colonies failed. By deploying passive nest dissuasion materials (ropes, stakes, flagging) and intensive active dissuasion (human hazing) nearly all Caspian terns prospecting for nest sites on East Sand Island outside the 1-acre (0.4-ha) designated colony area were prevented from initiating nesting attempts. These same dissuasion techniques were used during the management period to prevent the re-establishment of a Caspian tern colony on Rice Island, where prospecting adults have congregated on the former colony site each breeding season during 2011–2019. Although some incipient nesting attempts were detected on islands in Grays Harbor, Washington, no new Caspian tern colonies formed in the lower Columbia River or along the outer coast of Washington or Oregon during the management period (2008–2019). Nevertheless, when the area of bare-ground nesting habitat for Caspian terns was gradually reduced at East Sand Island during the management period, average nesting density in the designated colony area roughly doubled, from an average of about 0.7 nests/m^2 during 2008–2010 to over 1.3 nests/m^2 during 2015–2016. By 2019, tern nesting density on the 1-acre (0.4-

ha) designated colony area had relaxed to an average of about 1.1 nests/m^2 but was still much greater than the pre-management average of about 0.55 nests/m^2.

Tern nesting success (average number of young raised/breeding pair) at the East Sand Island colony was highly variable during the study period, from a high of about 1.4 young raised/breeding pair in 2001 to a low of no young raised (i.e. complete colony failure) in 2011 and 2017. Average nesting success at the East Sand Island colony was much lower during the management period (0.27 young/pair) than during the pre-management period (0.89 young/pair). There was a significant correlation between colony size and nesting success at East Sand Island during the study period (2000–2019), and there was a significant negative correlation between nesting density and nesting success. But the decline in tern nesting success at East Sand Island was apparent before the implementation of reductions in nesting habitat, and those terns that nested in parts of the colony with relatively high nesting densities had higher nesting success than those that nested where nesting density was low. During the management period (2008–2019), two environmental factors emerged as causes of low nesting success at the East Sand Island Caspian tern colony: (1) reduced availability of marine forage fish in the estuary associated with high Columbia River discharge and poor ocean conditions (i.e. marine heat wave), and (2) frequent colony disturbances by bald eagles (*Haliaeetus leucocephalus*) and associated high predation rates on tern eggs and chicks by glaucous-winged/western gulls.

The average annual proportion of juvenile salmonids (percent of identified prey items) in the diet of Caspian terns nesting on East Sand Island averaged 33% during the study period (1999–2019) and varied only modestly among years. The proportion of salmonids in the diet (averaged across the nesting season) during the management period (2008–2019) averaged slightly more than during the pre-management period (1999–2007), but the difference was not significant. During both the pre-management and management periods more than half the identified prey items in the Caspian tern diet at East Sand Island consisted of marine or estuarine forage fishes (i.e. anchovy [Engraulidae], surf perch [Embiotocidae], smelt [Osmeridae], and clupeids [herring, sardines, and shad; Clupeidae]). This is in contrast to the diet composition of Caspian terns nesting at Rice Island during 1997–2000, where ca. 78% of prey items consisted of juvenile salmonids, and only a small percentage of the diet consisted of marine forage fishes.

Within a breeding season there was considerable variation in the proportion of salmonid smolts in the diet of East Sand Island terns; the consistent pattern each year was for salmonids to comprise more than half of all prey items during the early part of the nesting season (late April and the first three weeks of May), while during most of the remainder of the nesting season (June and July) juvenile salmonids comprised less than 20% of prey items. Thus, most of the tern predation pressure on juvenile salmonids was during the out-migration period for spring migrant stocks (steelhead trout [*Oncorhynchus mykiss*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*], and yearling Chinook salmon [*O. tshawytscha*]). During the chick-rearing portion of the tern nesting season, Caspian terns nesting on East Sand Island preyed primarily on northern anchovy (*Engraulis mordax*) and surf perch, while most of the juvenile salmonids consumed by terns were sub-yearling Chinook salmon.

Bioenergetics calculations to estimate total annual consumption of juvenile salmonids by Caspian terns nesting at East Sand Island during the pre-management period (2000–2007) indicated that an average of ca. 5.0 million juvenile salmonids was consumed annually. During the management period there was a significant decline in the estimated annual number of salmonid smolts consumed by Caspian terns nesting on East Sand Island; by 2019, the last year of this study, ca. 2.9 million juvenile salmonids were consumed by terns nesting on East Sand Island. This compares with an estimated average of 11.8 million salmonid smolts consumed per year when the tern colony was on Rice Island in 1998 and 1999. Bioenergetics modeling was also used to estimate the numbers of different salmonid species/age-classes that were consumed by Caspian terns nesting at East Sand Island during 2000–2019. The models indicated that in most years during the pre-management period coho salmon were the most frequently consumed salmonid species/age-class by terns nesting at East Sand Island (1.2–2.6 million/year), but there was considerable inter-annual variation in the relative consumption of coho salmon, steelhead trout (0.8–1.3 million/year), yearling Chinook salmon (0.8–1.6 million/year), and sub-yearling Chinook salmon (0.4–0.9 million/year). Estimated annual consumption of sockeye salmon by terns nesting on East Sand Island during the pre-management period (21–66 thousand/year) was much lower than for other salmonid species/age-classes. During the management period (2008–2019) the estimated annual consumption of each species/age-class declined with the notable exception of sub-yearling Chinook salmon; average estimated consumption of sub-yearling Chinook salmon pre-management was 0.7 million/year, whereas during the management period the average was 1.1 million/year.

Predation rates on specific ESA-listed evolutionarily significant units (ESUs) or distinct population segments (DPSs) of anadromous salmonids by Caspian terns nesting on East Sand Island were investigated by recovering smolt passive integrated transponder (PIT) tags from the surface of the tern colony after each breeding season. From 1999 to 2018, more than 530,000 smolt PIT tags were recovered (i.e. detections of unique tag codes from individual fish) on the Caspian tern colony at East Sand Island, representing the largest number of smolt PIT tags recovered on any breeding colony of piscivorous waterbirds in the Columbia River basin. These PIT tag recoveries were corrected for deposition rates (proportion of tags ingested by terns that were deposited on-colony) and detection rates (proportion of tags deposited on-colony that were detected post-nesting season) to provide unbiased estimates of predation rates (percentage of available tagged fish that were consumed) on each ESU/DPS of juvenile salmonid. Although not all ESUs/DPSs of ESA-listed juvenile salmonids were PIT-tagged in sufficient numbers to allow reliable estimates of predation rates by Caspian terns nesting on East Sand Island, some clear patterns and trends in predation rates were evident from the data.

First, steelhead trout smolts were much more susceptible to predation by East Sand Island terns than other salmonid species. Higher susceptibility to tern predation in the estuary was evident regardless of the DPS of steelhead, and steelhead susceptibility was not different for smolts raised in hatcheries vs. in the wild, or those that migrated in-river vs. those that were transported. During the pre-management period (2000–2007), average annual predation rates by East Sand Island terns on Upper Columbia River, Snake River, Middle Columbia River, and

Lower Columbia River steelhead DPSs were estimated at 17.2%, 25.3%, 17.1%, and 15.2%, respectively. This compares to pre-management predation rates on coho salmon smolts from the Lower Columbia River ESU of 2.6%, and predation rates on Chinook salmon smolts from the Upper Columbia River Spring, Snake River Spring/Summer, Snake River Fall, and Lower Columbia River ESUs of 4.3%, 5.2%, 2.9%, and 4.1%, respectively. Unlike steelhead smolts, yearling Chinook salmon smolts raised in a hatchery were significantly more susceptible than their wild counterparts to predation by East Sand Island terns.

Second, our results indicate that predation rates were, on average, significantly lower following implementation of management actions that reduced the size of the Caspian tern colony on East Sand Island. Reduction in the size of the East Sand Island Caspian tern colony from the pre-management period (2000–2007) to the last three years during the management period when predation rates were measured (2016–2018) resulted in much lower predation rates on ESA-listed steelhead smolts, with reductions of 50% to 76%, depending on the DPS. By 2016–2018 the average annual predation rates on steelhead DPSs by East Sand Island Caspian terns had declined from approximately 19% to 7%, or an average reduction in tern predation rates on steelhead smolts of about 63%. Significant reductions in tern predation rates were observed for all four ESA-listed steelhead DPSs and in most, but not all, salmon ESUs by the latter part of the management period (2016–2018) compared to the pre-management period (2000–2007). Thus, our results provide evidence that Caspian tern management actions on East Sand Island significantly reduced average annual predation rates on ESA-listed juvenile salmonids by terns nesting at this colony and have met the targeted reductions in predation rates on ESA-listed steelhead smolts that were specified in the *Caspian Tern Management Plan*.

Third, an investigation of the relationship between estimates of peak (maximum) annual size of the East Sand Island Caspian tern colony and estimates of annual predation rates on steelhead DPSs indicated that reductions in predation rates were generally commensurate with reductions in colony size. For all steelhead DPSs evaluated, comparisons indicate a linear relationship between annual predation rates and peak size of the East Sand Island colony. These results provide evidence that Caspian tern predation rates on juvenile salmonids, on average, can be reduced in the Columbia River estuary by reducing the number of Caspian terns breeding on East Sand Island, while simultaneously preventing terns from nesting at other sites in the estuary. Nevertheless, colony size alone was not the sole factor regulating predation rates on juvenile salmonids by Caspian terns nesting on East Sand Island; for example, estimates of peak colony size explained 52.0% of annual variation in predation rates on the Snake River steelhead DPS.

Finally, in a study of Caspian tern predation on PIT-tagged Upper Columbia River and Snake River steelhead smolts, our results indicated that decreases in tern predation rates on steelhead were associated with statistically significant increases in steelhead survival rates in all evaluated years and salmonid life-stages (i.e. smolt out-migration and smolt-to-adult returns). Results indicated that in the absence of tern predation significantly more steelhead smolts would have survived out-migration and, more importantly from a conservation perspective, more adult steelhead would have returned to the Columbia River to spawn. These results provide evidence that predation by Caspian terns is not primarily a compensatory source of mortality for steelhead, and instead can be a largely additive source of mortality during the smolt life-stage and a partially additive source of mortality for the smolt-to-adult life-stage.

Resightings of previously banded Caspian terns on East Sand Island during the 2005–2016 nesting seasons indicated that there is strong natal and breeding philopatry to the East Sand Island colony. Nevertheless, many Caspian terns that were banded at the East Sand Island colony were resighted at the alternative tern colony sites that were constructed by the USACE during 2008–2012, some located as far as 550 km from East Sand Island. Resightings indicated that most adult terns that recruited to the newly constructed colony sites were from the East Sand Island colony. Resightings of Caspian terns on East Sand Island that were previously banded at other colonies in the Pacific Flyway during the 2005–2015 nesting seasons indicated, however, that some terns were immigrating to the East Sand Island colony from other colonies. Resightings of banded terns also demonstrated that some adult terns continue to disperse from the East Sand Island colony to other colony sites in the Columbia Plateau region, where management has also been implemented to reduce Caspian tern predation rates on ESA-listed salmonid smolts from the upper Columbia River and Snake River basins.

A demographic model was developed for the Pacific Flyway population of Caspian terns, one of the most sophisticated population models available to date for colonial waterbirds, to project the Pacific Flyway population of Caspian terns under 16 different scenarios of management and environmental conditions for breeding. The model incorporated both life history complexity (e.g., delayed maturity, age structured survival and breeding propensity) and spatial specificity (e.g., natal and breeding dispersal, region-specific demographic rates across five distinct regions of the breeding range) and was based on empirical data collected throughout the Flyway over more than a decade. Population simulations from the model indicated resiliency of the Pacific Flyway population of Caspian terns under current vital rates and management regimes. Model population trajectories depended, however, on the quantity and quality of nesting habitat available for Caspian terns in the Pacific Flyway, the predominant limiting factor for this population. Also, certain combinations of management actions coupled with realistic changes in breeding conditions that follow recent trends (e.g., simultaneous reductions in nesting habitat in the Columbia River estuary coupled with recent unfavorable environmental conditions for reproduction on East Sand Island) would likely result in long-term declines in the Pacific Flyway population, even when assumptions are made for the continued availability of nesting habitat outside of the Columbia River estuary at the maximum levels observed to date. Circumstances that substantially reduce the unique capacity for fledgling production at the East Sand Island colony in the Columbia River estuary, if not mitigated by a similar high capacity for fledgling production elsewhere in the Pacific Flyway, are likely to result in a population of substantially reduced size, and potentially a long-term downward trend that could put the Pacific Flyway population of Caspian terns at risk of greatly reduced resiliency. Avoiding such a long-term downward trajectory will depend on maintaining suitable Caspian tern nesting habitat on East Sand Island, as stipulated in the *Caspian Tern Management Plan*, or creating an alternative coastal nesting site in the Pacific Flyway capable of supporting a large and productive Caspian tern breeding colony.

1.2 Introduction

1.2.1 Caspian Terns in the Pacific Flyway

Caspian terns (*Hydroprogne caspia*) are the largest species of tern in the world, with a nearly cosmopolitan distribution; they nest at scattered sites in North America, Eurasia, Africa, Australia, and New Zealand (Cramp 1985, Cuthbert and Wires 1999). The species has a widespread but disjunct breeding distribution in North America, where five breeding populations of Caspian terns are recognized: (1) Pacific coast/Western states, (2) Central Canada, (3) Great Lakes, (4) Atlantic coast, (5) Gulf coast (Wires and Cuthbert 2000). Caspian terns breeding in the Columbia River estuary are part of the Pacific coast/Western states population (Wires and Cuthbert 2000), also known as the Pacific Flyway population, which includes breeding colonies along the Pacific coast from Alaska to Baja California. Resightings and recoveries of banded Caspian terns indicate that there is widespread movement of individuals among colonies in the Pacific Flyway (Gill and Mewaldt 1983, Suryan et al. 2004, Suzuki et al. 2018).

Annual survival rates of adult Caspian terns in the Pacific Flyway population are high, generally over 95% and sometimes as high as 98% (Suzuki et al. 2018). Maximum known longevity is 26 years based on a banded adult recovered on Rice Island (Suryan et al. 2004), but it is likely that a few individuals survive to at least 30 years. Age at first reproduction averages 5–6 years (Suzuki et al. 2018), but some individuals nest as early as 3 years (Cuthbert and Wires 1999). Clutch size is generally 1–3 eggs; supernormal clutches of 4–6 eggs are uncommon and may reflect female-female pairs (Thompson and Tabor 1981, Cuthbert and Wires 1999). The incubation period averages 26–27 days (Penland 1981); at hatching young are semi-precocial, down-covered, and semi-nidifugous, able to leave their nest scrape within two days of hatching (Cuthbert and Wires 1999). Eggs and young chicks are especially vulnerable to predation by gulls; older chicks may succumb to starvation, abandonment, aggression from adult terns, and gull predation. Young tend to form crèches when predators (e.g., bald eagles) flush adults from the colony, leaving chicks vulnerable to predators. Re-nesting is common following loss of a clutch of eggs to predators or to disturbance (Cuthbert 1988). As with most other tern species, Caspian terns have an extended period of post-fledging parental care; parents may continue to feed young for several months after they fledge (Cuthbert and Wires 1999), and adults feeding young can be seen hundreds of miles from the nearest active breeding colony.

Caspian terns are almost entirely piscivorous and forage opportunistically on fish at or near the surface. The species normally forages alone or, if prey are aggregated, in loose foraging flocks, often with other piscivorous birds (e.g., double-crested cormorants [*Phalacrocorax auritus*]). Caspian terns forage by plunge-diving, and dive from heights up to 30 m (100 ft) into water to capture prey (Cuthbert and Wires 1999). The species is found in a variety of marine, brackish, and freshwater habitats, usually on or near large water bodies, where forage fishes 10–25 cm (4-10 in) in length are readily available at the surface. Caspian terns are most numerous along coasts, especially in bays and estuaries, and rarely beyond sight of land. The species usually breeds in colonies, often in association with other colonial waterbirds, especially gulls (*Larus* spp.), and for nesting prefers unvegetated sites on islands that are isolated, free of mammalian predators, and in large bodies of water.

Early in the 20th century the population of Caspian terns in the Pacific Flyway nested primarily in colonies at inland lakes and marshes (e.g., Malheur Lake in the Harney Basin, Upper Klamath Basin; Finley 1907, Willet 1919), frequently amidst emergent vegetation or on mats of floating vegetation. By the mid-20th century breeding colonies of Caspian terns in the Pacific Flyway were mostly at inland lakes and reservoirs, with a few small colonies on coastal bays and estuaries, and many of the inland colonies were in decline and of conservation concern (Bent 1947, Gabrielson and Jewett 1940). During the latter half of the 20th century, however, breeding habitat/distribution shifted from inland lakes and reservoirs to coastal sites (Penland 1981), including the Columbia River estuary. The Pacific Flyway population increased from about 3,500 pairs in 1960 to about 14,000 pairs in 2000 (Gill and Mewaldt 1983, Suryan et al. 2004). By the end of the 20th century, the Pacific Flyway population of Caspian terns nested mostly on sandy islands along the Pacific coast, including dredged material disposal sites, and in more urban environments Caspian terns began nesting on warehouse rooftops and on vacant barges. Caspian tern nests now typically consist of a shallow scrape in sand, gravel, or other fine debris with little or no nest material, but nest scrapes are often adjacent to driftwood or other larger debris, when available (Roby et al. 2003a).

The factors responsible for the dramatic increase in the Caspian tern breeding population in the Pacific Flyway during the second half of the 20th century included (1) the shift from former inland breeding colonies subject to wetland reclamation, invasive predators, drought, and anthropogenic development, to coastal colony sites; (2) the creation of secure nesting habitat on dredged material disposal islands and other anthropogenic sites along the Pacific coast; and (3) enhanced availability of forage fishes for Caspian terns, in particular hatchery-raised anadromous salmonid (*Oncorhynchus* spp.) smolts (Collis et al. 2002a), at coastal colony sites (Suryan et al. 2004). During the 2000s, the Pacific Flyway population of Caspian terns increased to nearly 19,000 breeding pairs (Peterson et al. 2017a), followed by a decline during the 2010s; by 2018 the Pacific Flyway population consisted of about 11,000 breeding pairs (Peck-Richardson 2019).

1.2.2 Caspian Terns Nesting in the Columbia River Estuary

For the last three decades, the largest breeding colony of Caspian terns in the Pacific Flyway has been in the Columbia River estuary (Collis et al. 2002, Roby et al. 2002, Roby et al. 2020). The first record of a breeding colony of Caspian terns in the estuary was in 1984 on East Sand Island (river km 8; *Figure 1.1*); this colony consisted of approximately 1,000 breeding pairs and was located on dredged material that had been deposited on the east end of the island in 1983 (G. Dorsey, USACE, pers. comm.). These terns likely relocated from a colony on Gunpowder Sands at the mouth of Willapa Bay, Washington, about 48 km to the north. Caspian terns nested on the east end of East Sand Island again during the 1985 and 1986 nesting seasons, but colony size was smaller than in 1984. Nesting by Caspian terns on Rice Island, an artificial dredged material disposal island at river km (Rkm) 29 in the Columbia River estuary (*Figure 1.1*), occurred during the 1986–2000 nesting seasons (A. Clark, USFWS, unpubl. data). In 1998, a small colony of Caspian terns also formed on Miller Sands Spit (Rkm 30; *Figure 1.1*), another dredged material disposal island located across the shipping channel from Rice Island. The incipient colony at Miller Sands Spit formed following the deployment conspecific attraction (i.e. decoys and acoustic vocalization playback) on suitable nesting substrate (Collis et al. 2000). At least 17 pairs of Caspian terns laid eggs in nest scrapes amongst the decoys, but the tern colony was abandoned by 11 June due to egg predation by glaucous-winged/western gulls (*Larus glaucescens* X *L. occidentalis*). No young terns were raised in 1998 at the Miller Sands Spit colony, which was not occupied in subsequent years when no conspecific attraction was deployed. This attempt to use conspecific attraction techniques to establish a Caspian tern colony on Miller Sands Spit was a pilot study to determine whether conspecific attraction could be used to relocate the large tern colony on Rice Island to an alternative colony site (Collis et al. 2000). Dredged material disposal sites on East Sand Island, Rice Island, and Miller Sands Spit are the only three locations where Caspian terns are known to have nested in the Columbia River estuary.

1.2.3 Caspian Terns Nesting on Rice Island

Nesting by Caspian terns at Rice Island was first recorded in 1986, when the colony consisted of approximately 1,000 breeding pairs (A. Clark, U.S. Fish and Wildlife Service, pers. comm.); terns that had formerly nested at East Sand Island in 1984 and 1985 were presumably the first to nest on Rice Island. The colony was located at the downstream (western) tip of Rice Island, and quickly increased in size (i.e. number of breeding pairs) over the next decade. Much of the rapid increase in the size of the Rice Island colony was thought to be due to emigrants from former colonies in Grays Harbor, Washington (Penland 1981). By 1998 the Rice Island colony consisted of about 8,000 breeding pairs and was the largest known Caspian tern breeding colony anywhere in the world (Suryan et al. 2004). During 1997–2000, average annual nesting success at the Rice island colony varied from 0.05 to 0.55 young raised per breeding pair (Roby et al. 1998; Collis et al. 2000, 2002b).

The diet of Caspian terns nesting on Rice Island consisted mostly of juvenile salmonids (74–90% of prey items), including coho salmon (*O. kisutch*), Chinook salmon (*O. tshawytscha*), sockeye salmon (*O. nerka*), and steelhead trout (*O. mykiss*; Collis et al. 2001, Collis et al. 2002a, Roby et al. 2002). During 1997 and 1998, Caspian terns nesting on Rice Island annually consumed an estimated 5.4–14.2 million juvenile salmonids (Roby et al. 2003b), which represented about 5–15% of all salmonid smolts reaching the estuary during each of those two migration years. Due to growing concern regarding the impact of avian predation on recovery of Columbia Basin salmonid populations listed under the U.S. Endangered Species Act (ESA), regional fish and wildlife managers called for immediate management action in 1999 to reduce losses of juvenile salmonids to Caspian tern predation in the Columbia River estuary.

Figure 1.1. Map of the Columbia River estuary on the border of Oregon and Washington State, showing the location of Rice Island (river km 29) and Miller Sands Spit (river km 30) in the upper estuary and East Sand Island (river km 8) in the lower estuary, the three dredged material disposal sites where Caspian terns have been known to nest in the Columbia River estuary.

Beginning in 1999, federal, state, and tribal resource management agencies attempted to relocate the Rice Island colony to the former colony site on East Sand Island, 21 river km closer to the river mouth (*Figure 1.1* above), to reduce Caspian tern predation on juvenile salmonids in the Columbia River estuary (USACE 2001).

1.2.4 Managed Colony Relocation from Rice Island to East Sand Island

Management actions first implemented in 1999 sought to relocate the Caspian tern colony on Rice Island to a restored colony site on East Sand Island (*Figure 1.1* above), which is owned by the U.S. Army Corps of Engineers (USACE). By shifting the Caspian tern colony closer to the ocean, biologists believed terns would consume more marine forage fish and fewer juvenile salmonids. The approach used to relocate the large tern colony involved (1) restoring suitable tern nesting habitat on East Sand Island at the former dredged material disposal site (Quinn and

Sirdevan 1998), (2) deploying social attraction (i.e. tern decoys and audio playback of tern colony sounds; Kress 1983) at the restored nesting habitat, (3) close monitoring of the restored colony site to identify factors limiting colony restoration, (4) limited lethal control of gulls if they were observed depredating tern eggs and young (Kress 1998), and (5) installing passive tern nest dissuasion materials (i.e. rows of landscape fabric fencing; Pochop et al. 1999) at the colony site on Rice Island. Over 94% of the Caspian terns nesting on Rice Island relocated to nest on East Sand Island in 2000, which resulted in a reduction by more than half in the proportion of the tern diet that consisted of salmonids (Roby et al. 2002). By 2001, all Caspian terns nesting in the Columbia River estuary had shifted from Rice Island to East Sand Island. During 2001–2010, all nesting by Caspian terns in the Columbia River estuary occurred at East Sand Island; in 2011 three nesting pairs of Caspian terns laid a total of four eggs and hatched at least 3 chicks on Rice Island (Roby et al. 2012). During 2012–2019, terns attempted to re-establish a breeding colony on Rice Island, but dissuasion efforts at that island conducted by the USACE and its contractors effectively deterred terns from nesting successfully.

During the first three breeding seasons after restoration of the Caspian tern colony on East Sand Island, nesting success was high, much higher than at the former colony on Rice Island. Average annual nesting success at East Sand Island was from 0.57 to 1.40 young raised per breeding pair during 1999–2001 (Roby et al. 1998, Roby et al. 2002). At 8,900 pairs, the newly restored colony of Caspian terns on East Sand Island was home to about two-thirds of the Pacific Flyway population, one-quarter of the North American metapopulation, and roughly one-tenth of the worldwide population (Suryan et al. 2004).

1.2.5 Caspian Tern Management Plan for the Columbia River Estuary

Beginning in 2008, the USACE – Portland District implemented management described in the January 2005 Final Environmental Impact Statement (FEIS) and November 2006 Records of Decision (RODs) for *Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (hereafter the *Caspian Tern Management Plan*; USFWS 2005, USFWS 2006, USACE 2006). The *Caspian Tern Management Plan* was developed jointly by the U.S. Fish and Wildlife Service (USFWS; lead), the USACE – Portland District, and the National Marine Fisheries Service (NMFS), and sought to redistribute most Caspian terns nesting at the colony on East Sand Island in the Columbia River estuary to alternative colony sites (artificial islands) in interior Oregon, northeastern California, and the San Francisco Bay area (see *Chapter 3*). The goal of the plan was to reduce Caspian tern predation on out-migrating juvenile salmonids in the Columbia River estuary, and thereby enhance recovery of salmonid stocks from throughout the Columbia River basin, while ensuring the long-term conservation of the Pacific Flyway population of Caspian terns.

The specific objectives of the *Caspian Tern Management Plan* were to reduce the area of the East Sand Island Caspian tern colony from ca. 6 acres (2.4 ha) to 1.0 acres (0.4 ha) of suitable nesting habitat, with an expected reduction in tern colony size to 3,125–4,375 breeding pairs, while providing new nesting habitat for Caspian terns at alternative colony sites outside the Columbia River basin. These objectives were identified as the preferred alternative in the Final

Environmental Impact Statement released in early 2005 and updated in the Final Environmental Assessment and Finding of No Significant Impact, released in April 2015 (USFWS 2005, USACE 2015a, USACE 2015b). Caspian terns displaced by reductions in nesting habitat on East Sand Island were expected to relocate to 14 alternative colony sites: nine USACE-constructed tern islands in interior Oregon and northeastern California (i.e. Fern Ridge Reservoir, Crump Lake, Summer Lake Wildlife Area [3 separate islands], Tule Lake National Wildlife Refuge (NWR), Lower Klamath NWR [2 separate islands], and Malheur NWR), plus five USACE-constructed tern islands in Don Edwards San Francisco Bay NWR in central California (see *Chapter 3*). For every two acres (0.8 ha) of alternative Caspian tern nesting habitat that was created or restored outside the Columbia River basin, the available nesting habitat for Caspian terns on East Sand Island was reduced by one acre (0.4 ha) from its initial size (approximately 6 acres [2.4 ha] in 2007) down to a lower limit of 1.0 acre (0.4 ha).

The *Caspian Tern Management Plan* assumed that reductions in tern nesting habitat on East Sand Island would be proportional to reductions in the size of the Caspian tern colony, and that reductions in tern colony size would be proportional to reductions in tern predation rates on juvenile salmonids (USFWS 2005). More specifically, the *Plan* was implemented to increase the survival of four ESA-listed steelhead trout populations in the Columbia River basin (Upper Columbia River steelhead; Snake River steelhead; Middle Columbia River steelhead, and Lower Columbia River steelhead), populations that previous research indicated were especially susceptible to predation by Caspian terns in the estuary (Collis et al. 2001, Roby et al. 2003b, Ryan et al. 2003, USFWS 2005). Benefits to other anadromous salmonid species and populations were also expected because of management actions under the *Caspian Tern Management Plan* (USFWS 2005). The *Plan*, however, did not target a specific reduction in steelhead predation rates, but rather a reduction in the amount of suitable tern nesting habitat and tern colony size to ultimately achieve reductions in predation rates that would enhance annual population growth rates (λ) for steelhead by 1 percent (USACE 2015a). Consequently, as part of this study we monitored the effectiveness of the *Caspian Tern Management Plan* for reducing predation rates on ESA-listed populations of juvenile salmonids in the Columbia River estuary. Predation rates on evolutionarily significant units (ESUs) or distinct population segments (DPSs) of juvenile salmonids were compared prior to management actions at the East Sand Island colony (pre-management period) and following the initiation of management actions on East Sand Island (management period) to determine whether benefits to salmonid smolt survival were achieved and to inform decisions regarding the potential need for adaptive management.

Thirteen of the 20 ESUs/DPSs of Columbia Basin salmonids are currently listed as either threatened or endangered under the U.S. Endangered Species Act (ESA), whereas Caspian terns are protected under the Migratory Bird Treaty Act. Studies of Caspian tern predation rates on juvenile salmonids have been conducted at breeding colonies located throughout the Columbia River basin, both in the Columbia River estuary and in the Columbia Plateau region (see *Chapter 2*). Avian predation in the Columbia River estuary is of particular concern to fisheries managers because it affects fish belonging to all 13 ESA-listed anadromous salmonid populations in the Columbia River basin (NMFS 2019b); fish that have survived freshwater out-migration to the estuary have a higher probability of survival to adulthood compared to fish that have yet to complete out-migration (Roby et al. 2003b; USFWS 2005; Evans et al. 2019b). As part of this chapter, we summarize and synthesize predation rate studies based on smolt PIT tag recoveries on Caspian tern colonies in the Columbia River estuary, with a focus on the colony at East Sand Island. More specifically, we summarize predation rates on specific ESUs or DPSs of ESA-listed, PIT-tagged juvenile salmonids by Caspian terns breeding in the estuary. In addition, we summarize Caspian tern nesting ecology, foraging ecology, diet composition, colony connectivity, and factors limiting colony size and nesting success at the colony on East Sand Island in the Columbia River estuary during 1999–2019.

1.3 Methods

1.3.1 Management of Tern Nesting Habitat

In March 1999, approximately 7.4 acres (3 ha) of bare-sand nesting habitat was restored on a former dredged material disposal site at the east end of East Sand Island by the U.S. Marine Corps using heavy equipment that had been barged to the island. The dredged material, which had been deposited in 1983, was completely vegetated by 1999. During late March and early April of each subsequent year, habitat restoration at the Caspian tern colony site on East Sand Island was accomplished by the U.S. Army Corps of Engineers (USACE). During 2000–2006, approximately 6.5 acres (2.6 ha) of bare-sand nesting habitat was prepared before the nesting season by disking and harrowing the colony site to mechanically remove encroaching European beach grass (*Ammophila arenaria*) and other invasive plants. Without annual restoration of the bare sand nesting habitat that Caspian terns prefer, the East Sand Island colony would likely have been eliminated within a few years by rapidly encroaching pioneer vegetation. To aid in vegetation suppression on the colony site, the USACE treated the tern colony site with herbicides (Rodeo) during the pre-nesting season in 2001 and 2012 and post-nesting in 2005 and 2010. Because the area of provided nesting habitat that was used by nesting Caspian terns during the 2000–2006 nesting seasons never exceeded 5 acres (2 ha), in 2007 only 6.0 acres (2.4 ha) of tern nesting habitat was provided (*Figure 1.2*). Starting in 2008, with implementation of the *Caspian Tern Management Plan*, the area of bare-ground nesting habitat provided for Caspian terns at East Sand Island was gradually reduced *(Figure 1.2*) as alternative colony sites for terns (artificial islands) were prepared outside the Columbia River estuary (see *Chapter 3*).

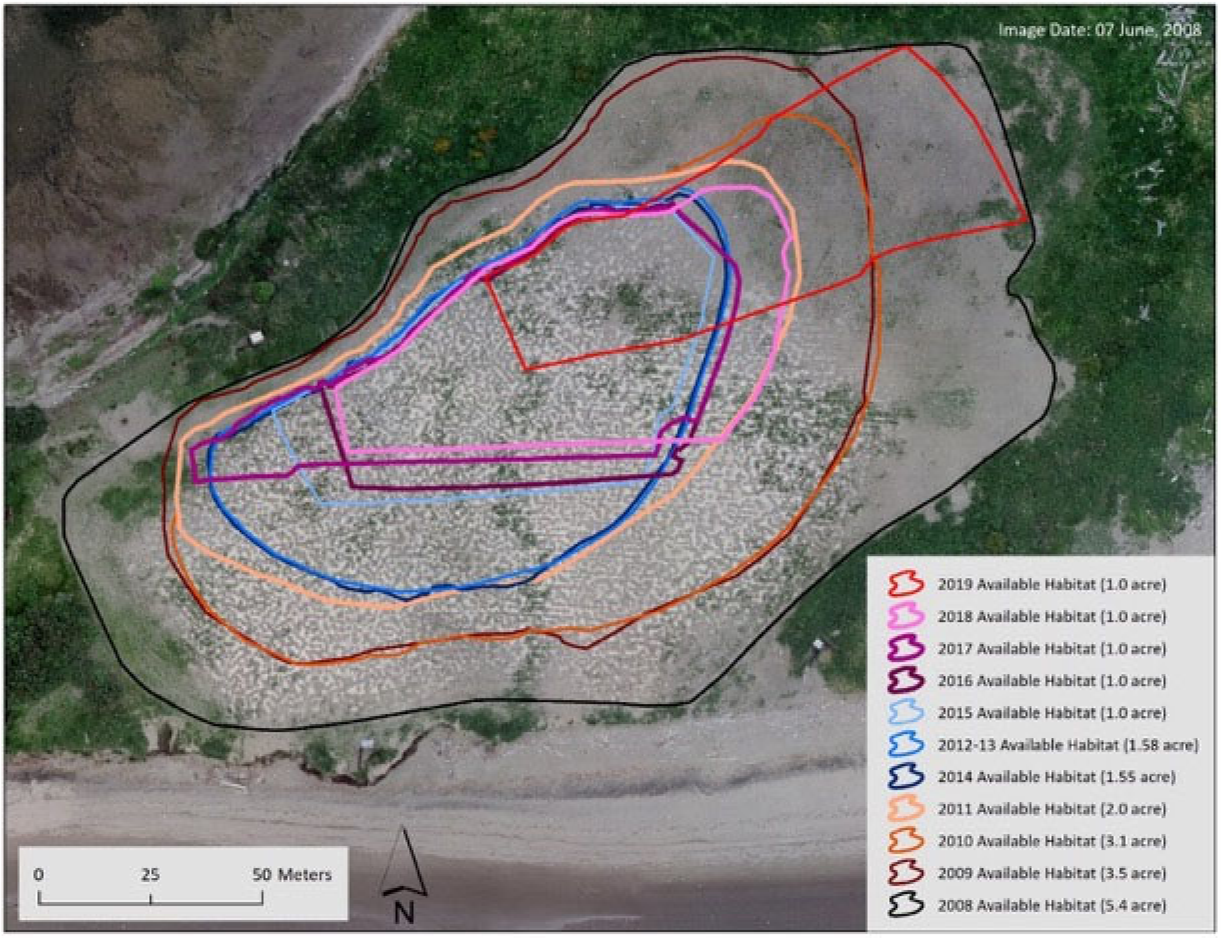

Figure 1.2. Aerial photograph taken in 2008 of the east end of East Sand Island in the Columbia River estuary showing the incremental reduction of bare-ground nesting habitat provided for Caspian terns and the gradual shift of the designated colony area to the northwest.

Caspian tern conspecific attraction (i.e. Caspian tern decoys and acoustic playback systems continuously broadcasting Caspian tern vocalizations) were deployed on the East Sand Island colony site starting in 1999 to restore a breeding colony on the newly restored nesting habitat (Kress 1983, Jones and Kress 2012). Prior to the breeding season in 1999, 385 tern decoys and 4 acoustic systems were deployed on the restored nesting habitat; during subsequent years, the following social attraction was deployed prior to the nesting season: 415 decoys and 4 acoustic systems in 2000, 253 decoys and 2 acoustic systems in 2001, 252 decoys and 2 acoustic systems in 2002, 100 decoys and 2 acoustic systems in 2003, 58 decoys in 2004, 40 decoys in 2005, 50 decoys in 2006, and 50 decoys in 2007. After 2007, a small number of tern decoys (< 50) were occasionally deployed on the colony in some years to help attract adult terns back to the colony following disturbances, but no audio playback systems were deployed on the tern colony site after 2003.

Prior to each tern nesting season during 1999–2018, a camp was set up on East Sand Island that was continuously occupied by two colony monitors throughout the nesting season. These colony monitors visited the colony daily to collect observational data from the blinds at the edge of the colony. Prior to each nesting season, USACE personnel delineated the designated area of prospective nesting habitat for Caspian terns on East Sand Island and prepared the colony surface for tern nesting by removing existing vegetation. We installed 2–4 observation blinds at the edge of the tern colony prior to each tern breeding season, as well as above-ground tunnels used to access observation blinds without disturbing nesting terns. Starting in 2006, the entire East Sand Island colony site of bare-ground nesting habitat for Caspian terns was marked off with wooden stakes and ropes prior to the arrival of terns on the colony to assist in efforts to census and monitor the colony each season. In 2019, colony monitors did not reside on East Sand Island during the nesting season, but instead visited the island every other day, weather permitting, to collect data from the observation blinds.

Limited lethal control of glaucous-winged/western gulls was performed on the East Sand Island tern colony during the first two seasons of restoration efforts; 183 gulls were removed in 1999 and 44 gulls were removed in 2000. Gull control was implemented to enhance prospects for tern colony restoration by limiting the numbers of tern eggs and chicks that were depredated on-colony by gulls. Gulls were shot on the tern colony under permit using .22 caliber high-powered rifles. No gull control was conducted on the East Sand Island tern colony during the 2001–2011 nesting seasons, once the tern colony had become established. In 2012, however, 50 glaucous-winged/western gulls were shot under permit on the East Sand Island tern colony. This limited removal of gulls was conducted to prevent the tern colony from completely failing to produce young, as occurred in 2011, and avoid the potential abandonment of the East Sand Island tern colony site and the return of the colony to Rice Island. No lethal gull control was conducted on the East Sand Island Caspian tern colony during 2013–2019.

The managed reduction of Caspian tern nesting habitat on East Sand Island began in 2008 and was accomplished by erecting parallel rows of landscape fabric fencing (ca. 4 m between rows), stringing rope between rows of fence posts, and securing caution tape to the rope. Caspian terns avoided the bare-ground nesting habitat between fence rows, and this habitat gradually became unsuitable for tern nesting due to vegetation encroachment. Beginning in 2010 and concurrent with the reduction in area of prepared bare-ground nesting habitat for terns on East Sand Island, Caspian terns prospected for new nest sites on East Sand Island outside the prepared colony area. This necessitated the deployment of passive nest dissuasion (stakes, ropes, and flagging) to prevent tern nesting in these areas, mostly on bare sand above the rack line on beaches near the main colony area. The use of landscape fabric fencing as passive nest dissuasion on upper beach habitat was problematic because the fabric fencing would frequently wash away during spring tide series.

Caspian terns formed incipient satellite colonies on the upper beach near the main Caspian tern colony on East Sand Island starting in 2010, and this led to in-season deployment of passive nest dissuasion materials. Then, starting in 2013, pre-season deployment of extensive areas of passive nesting dissuasion materials was used to prevent the establishment of new satellite colonies on East Sand Island. The continued formation of incipient satellite tern colonies also led to the initiation of active nest dissuasion efforts (human hazing) in 2013, efforts that intensified in 2015 to prevent satellite tern colonies from forming at sites where passive dissuasion had not been deployed. Active dissuasion of prospecting terns to prevent the formation of satellite colonies was allowed under permit on East Sand Island, but only until a tern egg had been laid; once a tern egg was detected, hazers were required to avoid the area to prevent failure of the incipient nesting attempt. Nevertheless, most tern eggs laid in incipient satellite colonies were taken by gulls or washed away during spring high tide series soon after they were laid.

1.3.2 Nesting Chronology & Colony Attendance

Resident colony monitors observed the colony on East Sand Island from observation blinds located at the edge of the prepared colony area daily starting soon after the first Caspian terns were seen on-colony. The colony monitors recorded the date when adult Caspian terns were first seen on-colony, when the first tern egg was seen in a nest scrape, when the first hatchling chick was seen, and when the first fledgling tern capable of sustained flight was observed on the prepared colony site. In addition, colony monitors estimated twice each day, once at high tide and once at low tide, the numbers of adult Caspian terns attending the East Sand Island colony site during the nesting season. The large numbers (thousands) of adult Caspian terns frequently present on the East Sand Island colony made it impractical to count terns individually, so counts were conducted in groups of 10s, 50s, or 100s, and thus when large numbers of terns were present, estimates were less precise and likely less accurate. Late in the incubation period, when the numbers of adult terns attending nests with eggs was at its peak, the colony monitors signaled that it was time to conduct the aerial photo census of the tern colony, which was used to obtain a precise and accurate estimate of the peak colony size in each year.

1.3.3 Colony Size, Nesting Density, & Nesting Success

The number of Caspian tern breeding pairs using the colony on East Sand Island was estimated each year using low-altitude, high-resolution vertical aerial photography of the colony taken near the end of the incubation period (late May or early June), when colony attendance is at its peak. During 1997–2017, vertical aerial photography was taken from fixed-wing aircraft, and during 2018–2019 from unmanned aerial vehicles (UAV). The average of 2–3 direct counts of all adult terns in aerial photography of the prepared tern nesting habitat at the east end of East Sand Island was corrected to estimate the number of breeding pairs using ground counts of incubating and non-incubating terns on at least 12 different sample plots (6 m x 6 m) within the managed colony area. Confidence intervals for the number of breeding pairs were calculated using a Monte Carlo simulation procedure to incorporate the variance in the counts from the aerial photography and the variance in the ratios of incubating to non-incubating adult terns among the 12 plots.

Average nesting density of Caspian terns on the prepared colony area at East Sand Island was estimated based on the estimate of peak colony size divided by the area (m^2 of habitat) occupied by nesting Caspian terns on the prepared colony site. Area of occupied habitat was measured on ortho-rectified aerial photography by integrating the minimum concave polygon that included all nesting terns during late incubation. Variability in nesting density across the colony was quantified by measuring nest density on the 12 sample plots, each measuring 6 m x 6 m, scattered over the prepared colony site.

Nesting success (average number of young raised per breeding pair) at the East Sand Island tern colony was estimated using low-altitude, high-resolution vertical aerial photography taken of the colony just prior to the peak of fledging. The average of 2–3 direct counts of all terns (adults and juveniles) in aerial photography was corrected to estimate the number of fledglings on the colony using ground counts of the ratio of adults to fledglings on at least 12 different sample plots (6 m x 6 m) within the colony area. The estimated total number of fledglings was then divided by the estimated number of breeding pairs from aerial photography taken at the peak to estimate nesting success (the average number of young raised per breeding pair) in that year. The confidence intervals for nesting success were calculated using a Monte Carlo simulation procedure to incorporate the variance of the multiple counts from the aerial photography and from the sample plot counts used to generate the estimate. See *Appendix A.1.1* for a detailed description of methods used to estimate colony size.

1.3.4 Banding to Allow Measures of Dispersal & Colony Connectivity

During 1997–2016, a total of 7,247 Caspian terns, including both adults and flightless young, were banded at the Rice Island and East Sand Island breeding colonies (*Table 1.1*). In addition, during this period a total of 3,520 Caspian terns were banded at three different colonies in the Columbia Plateau region of the Columbia Basin, and 2,362 Caspian terns were banded at other colonies outside the Basin from Alaska to the San Francisco Bay area. These banding efforts were part of our objective to measure survival and other vital rates, post-breeding dispersal rates, and movement rates among colonies for Caspian terns in the Pacific Flyway population. Each banded tern was first banded under federal permit with a metal leg band provided by the U.S. Geological Survey – Bird Banding Lab. During 1997–1999, terns were banded only with federal metal leg bands (n = 1,074), each with a unique numerical code.

Starting in 2000, most banded terns (n = 2,815) were also marked with a unique color combination of five acrylic plastic leg bands that could be used to identify banded individuals in the field (i.e. at roosts or on colonies). Young terns that were too small to be banded with five plastic leg bands were only banded with a federal metal leg band. Some terns were able to subsequently shed one or more of their five colored leg bands, making them no longer identifiable to the level of the individual. Consequently, starting in 2005 for adult terns and 2006 for young flightless terns, we used a Darvic plastic leg band engraved with a unique alphanumeric code on one leg and a federal numbered metal band and two plastic colored leg bands on the other leg (n = 2,873). This new banding protocol for individually marking Caspian terns was longer lasting and more reliable for field identification of individuals at a distance than the five-band color combination method.

As part of these banding studies, flightless tern chicks that were near fledging age were captured on-colony at the Rice Island and East Sand Island colonies. Tern chicks were captured by herding flightless young into holding pens. Adult terns were captured on-colony for banding using monofilament noose mats placed around active nests late in the incubation period. Once captured, terns were immediately transferred to holding crates until they were banded and released. Terns that were color-banded or banded with field-readable alphanumeric bands were resighted on East Sand Island and at various other tern breeding colonies by colony monitors using binoculars and spotting scopes throughout each breeding season. Resightings of banded terns at other locations were reported to us through our project web page (www.birdresearchnw.org), by phone, or by e-mail, either via the US Geological Survey – Bird Banding Lab or directly from the public.

Table. 1.1. Numbers of Caspian terns banded under federal permit in the Columbia River estuary during 1997–2016. Starting in 2000, most banded terns were also banded with a field-readable unique color combination of five acrylic plastic leg bands. Starting in 2005, we switched to field-readable Darvic leg bands engraved with a unique alphanumeric code.

	Metal USGS Band Only				**Plastic Color Bands**			**Alphanumeric Band**		
Year	Rice Is. (Adult)	Rice Is. (Chick)	East Sand Is. (Adult)	East Sand Is. (Chick)	Rice Is. (Adult)	East Sand Is. (Adult)	East Sand Is. (Chick)	East Sand Is. (Adult)	East Sand Is. (Chick)	**TOTAL**
1997	--	358	--	--	--	--	--	--	--	**358**
1998	34	508	--	--	--	--	--	--	--	**542**
1999	30	114	22	8	--	--	--	--	--	**174**
2000	1	--	1	19	586	29	341	--	--	**977**
2001	--	--	--	59	47	22	288	--	--	**416**
2002	--	--	--	6	--	--	366	--	--	**372**
2003	--	--	--	12	--	--	438	--	--	**450**
2004	--	--	1	26	--	29	425	--	--	**481**
2005	--	--	--	42	--	--	209	38	--	**289**
2006	--	--	2	165	--	--	--	43	262	**472**
2007	--	--	1	35	--	--	35	56	378	**505**
2008	--	--	1	42	--	--	--	51	406	**500**
2009	--	--	--	37	--	--	--	55	406	**498**
2010	--	--	1	13	--	--	--	124	223	**361**
2011	--	--	--	--	--	--	--	--	--	**0**
2012	--	--	--	--	--	--	--	--	--	**0**
2013	--	--	--	2	--	--	--	--	264	**266**
2014	--	--	--	--	--	--	--	--	--	**0**
2015	--	--	--	--	--	--	--	--	276	**276**
2016	--	--	--	19	--	--	--	--	291	**310**
Total	**65**	**980**	**29**	**485**	**633**	**80**	**2102**	**367**	**2506**	**7247**
			Subtotal	**1559**		**Subtotal**	**2815**	**Subtotal**	**2873**	

1.3.5 Limiting Factors for Colony Size & Nesting Success

Tern colony size and nesting success can be limited by a variety of factors, including availability of suitable nesting habitat, food availability, disturbance, nest predation, and severe weather. Resident colony monitors on East Sand Island collected systematic data and anecdotal observations pertaining to some of these factors during each nesting season. Systematic data collection included measures of (1) gull predation rates on tern eggs and chicks, (2) rates of disturbance to the tern colony when at least 10% of attending adult terns were flushed, and (3) rates of gull kleptoparasitism of fish from adult terns transporting bill-load fish to their nests. Anecdotal observations included (1) rain- and/or wind-storms that caused loss of tern eggs and/or chicks through exposure or inundation of parts of the colony, (2) major disturbance or predation events on-colony due to terrestrial predators (e.g., mink [*Neovison vison*], river otters [*Lontra canadensis*], raccoons [*Procyon lotor*]) or raptors (e.g., great horned owls [*Bubo virginianus*], bald eagles [*Haliaeetus leucocephalus*], peregrine falcons [*Falco peregrinus*]), (3) human disturbance (e.g., recreational boaters, fishers, beach combers), and (4) episodes of acute and widespread nest failure on-colony. Other factors that could potentially limit average nesting success (e.g., colony size, nesting density, diet composition) were measured on an annual basis as part of effectiveness monitoring. Finally, some climate-related factors (e.g., El Niño/Southern Oscillation Index, Pacific Decadal Oscillation, timing of spring transition, up-welling index, and river discharge) have been shown to affect the abundance and availability of marine forage fish in the Columbia River estuary, and thus can have a strong influence on prey availability for Caspian terns nesting in the estuary.

1.3.6 Diet Composition

Caspian terns transport whole fish in their bills to their mates (i.e. courtship meals) and young (i.e. chick meals) at the breeding colony. Thus, taxonomic composition of the diet can be determined by direct observation of adults as they return to the colony with fish (i.e. bill-load observations). Prey items in tern bill-loads were identified to the taxonomic level of family. We were confident in our ability to distinguish salmonids from non-salmonids and to distinguish among most non-salmonid taxa based on direct observations from blinds, but we did not attempt to distinguish the various salmonid species in observed bill-loads. The percent of the identifiable prey items in tern bill-loads that belonged to each prey category was calculated for each 2-week period throughout the nesting season. The diet composition of terns over the entire breeding season was based on the average of the percentages for the 2-week periods from 22 April to 30 July. Most juvenile salmonids consumed by Caspian terns in the Columbia River estuary are depredated early in the nesting season, during the out-migration of spring-migrant smolts through the estuary. To investigate inter-annual variability in the proportion of salmonid smolts in tern diets, we focused on diet composition from May 1 until June 15, when the proportion of salmonids in the diet is the greatest.

To assess the relative proportion of the various salmonid species in tern diets, we collected bill-load fish near the East Sand Island or Rice Island tern colonies by either shooting adult terns with shotguns (1997–2011) or startling terns with pyrotechnics (2011–2015) that were flying over land while returning to the colony with whole fish carried in their bills (referred to hereafter as "collected bill-loads"). During 2016–2019, no bill-load fish were collected from Caspian terns nesting on East Sand Island. Salmonid bill-loads were identified as either Chinook salmon, sockeye salmon, coho salmon, steelhead trout, or unknown based on analyses of morphometrics, diagnostic bones, and molecular genetics.

Adult stomach contents were also collected from adult terns that were shot to collect bill-load fish. Samples were separated into major fish prey categories, and further identified to genus and species, whenever possible. Items from each taxonomic category were weighed and enumerated. The taxonomic composition of stomach contents was expressed as % of identifiable biomass and % of identifiable prey items. The stomach contents samples from individual Caspian terns were compared to the fish they were transporting in their bills (collected bill-load fish) to test the hypothesis that tern bill-loads are a representative sample of adult tern diets. To obtain an adequate sample size for this comparison, the samples collected during 1997 and 1998 on Rice Island were pooled. Of the 48 paired stomach contents/bill-load samples examined, 38 (74.5%) contained similar prey items. Of the other 10 paired samples, half had salmonids in stomach contents but not in the bill, while the other half had salmonids in the bill but not in stomach contents. A statistical comparison revealed no significant difference in the proportion of salmonids vs. non-salmonids in the two samples ($X^2 = 0.05$, $P = 0.82$), suggesting that in the Columbia River estuary the proportion of salmonids consumed by adult terns is similar to the proportion of salmonids in bill-loads provisioned to mates and young.

1.3.7 Foraging Behavior

During the 1998 and 1999 nesting seasons, data were collected on the foraging distribution of Caspian terns in the Columbia River estuary using point count surveys of foraging birds at 40 sites along the river's banks, and using aerial strip transect counts throughout the estuary for terns (Lyons et al. 2007). All nesting by Caspian terns in the Columbia River estuary occurred at Rice Island in 1998, and nearly all nesting occurred on Rice Island in 1999, the first year of efforts to restore the Caspian tern colony on East Sand Island (Roby et al. 2002).

We used VHF radio-telemetry to determine the foraging patterns and habitat use of adult Caspian terns breeding at both the East Sand Island and Rice Island colonies in 1999, 2000, and 2001. Tail-mounted radio-transmitters were attached to adult terns trapped at the two breeding colonies. Adult terns were caught for radio-tagging late in the incubation period by placing monofilament noose mats around active nests at the Rice Island colony in 1999 and at the East Sand Island colony in 1999, 2000, and 2001. Adult terns were also captured on the Rice Island colony in 2000 and 2001 using a rocket-propelled net prior to the onset of egg-laying. Radio-tagged terns were relocated during foraging trips using fixed-wing aircraft, and stationary tracking stations were also used to relocate foraging terns within the estuary and when attending the colony where they were nesting. Terns that were radio-tagged were also color-banded with a unique color combination of plastic leg bands for visual identification while attending the breeding colony. Radio-tagged terns were determined to be nesting on Rice

Island or East Sand Island based on repeated detections of a tagged individual at the same colony location on successive nights (using fixed receiving stations located on Rice and East Sand islands).

Off-colony locations of radio-tagged terns were assumed to be foraging locations and were categorized as one of the following: (1) the Columbia River between the Astoria-Megler Bridge and Kelso/Longview (i.e. the freshwater portion of the estuary), (2) the Columbia River between the Astoria-Megler Bridge and the mouth (i.e. the mixed/marine portion of the estuary), (3) Willapa Bay and the Washington Coast between the mouth of the Columbia River and Willapa Bay, and (4) Grays Harbor and the Washington Coast between Willapa Bay and Grays Harbor.

1.3.8 Smolt Consumption Estimates Based on Bioenergetics Calculations

We constructed bioenergetics models to obtain supportable estimates of the number of juvenile salmonids consumed by Caspian terns in the Columbia River estuary (Madenjian and Gabrey 1995, Glahn and Brugger 1995, Derby and Lovvorn 1997). Estimates of annual smolt consumption by Caspian terns nesting at the East Sand Island and Rice Island colonies were calculated using a bioenergetics modeling approach (Roby et al. 2003b). Briefly, the model begins with estimates of the energy expenditure rates of individual birds, expanded by the estimated number of birds present at the colony during each 2-week period of the nesting season. Multiplying by metabolic efficiency produced the total energy requirements of the colony. Estimates of the numbers of juvenile salmonids and other prey types consumed to meet energy demands of the tern colony were then derived from measurements of the proportion (% of prey biomass) of salmonids and other prey categories in the diet, the mean mass of individual prey items in each prey category, and the mean energy density of items in each prey category. The length of the breeding season (tern presence in the estuary) was broken into 11 two-week time periods, in each of which salmonid consumption was calculated. We used a Monte Carlo simulation procedure to calculate reliable 95% confidence intervals for estimates of smolt consumption by terns. By estimating the proportion of each salmonid species/age-class (i.e. coho salmon, sockeye salmon, yearling Chinook salmon, sub-yearling Chinook salmon, steelhead trout) in the salmonid smolt portion of the tern diet based on collected bill-loads, it was possible to estimate using the bioenergetics model the numbers of each salmonid species/age-class consumed by Caspian terns in the Columbia River estuary on an annual basis.

1.3.9 Predation Rates on Juvenile Salmonids

Methods to estimate avian predation rates (percentage of available fish consumed) by Caspian tern colonies in the Columbia River estuary on PIT-tagged juvenile salmonids are provided in *Appendix A.2*. It should be noted that accurate predation rate estimates based on PIT tag recoveries on Caspian tern colonies in the Columbia River basin have not always been available and previously reported (Hostetter et al. 2015). As a consequence, many of the predation rate estimates reported in technical reports, management plans, and peer-reviewed manuscripts, particularly those published prior to 2014, were reported as minimum estimates of predation (e.g., Collis et al. 2001; Ryan et al. 2003; USFWS 2005; Antolos et al. 2005; Evans et al. 2012; Sebring et al 2013). Here we report accurate estimates of predation by using recently developed models that account for the proportion of consumed tags that were deposited by nesting birds on their breeding colony (referred to as deposition probabilities) and the proportion of deposited tags subsequently recovered by researchers following the breeding season (referred to as detection probabilities).

Because all anadromous juvenile salmonids in the Columbia River basin migrate through the Columbia River estuary, all 13 ESA-listed ESUs/DPSs in the basin are susceptible to predation by Caspian terns nesting in the estuary (Roby et al. 2002, Roby et al. 2003b, USFWS 2005, Lyons 2010). The timing of the Caspian tern breeding season (April to September) also coincides with the peak smolt out-migration period (April to August; Lyons et al. 2007, Evans et al. 2012), resulting in near complete spatial and temporal over-lap between migrating juvenile salmonids and breeding Caspian terns (see *Chapter 7*). Not all salmonid ESUs/DPSs, however, are PIT-tagged in sufficient numbers for predation rate analyses. See *Appendix A.2* for a detailed description of methods used to estimate predation rates.

1.4 Results

1.4.1 Tern Nesting Ecology

1.4.1.1 Nesting habitat use – Excess bare-sand nesting habitat for Caspian terns was provided at the east end of East Sand Island prior to implementation of the *Caspian Tern Management Plan* in 2008. During 2000–2007, between 3.4 acres (1.4 ha) and 4.8 acres (2.0 ha) of prepared tern nesting habitat was occupied by nesting terns, and from 1.75 acres (0.7 ha) to 3 acres (1.2 ha) of suitable nesting habitat was unoccupied by nesting terns (*Figure 1.3*). From 2008 to 2015 the area of bare-ground nesting habitat provided for Caspian terns was intentionally and gradually reduced to 1.0 acres (0.4 ha) to induce Caspian terns to disperse to alternative colony sites outside the Columbia River estuary and achieve a target colony size of 3,125 to 4,375 breeding pairs (USFWS 2005, 2006). During 2015–2019 the area of bare-ground nesting habitat provided for Caspian terns remained at the target area of 0.4 ha; however, the location of the 1-acre (0.4-ha) colony area was shifted to the north and east in most years due to erosion along the southern edge of the colony area during winter storms (*Figure 1.2* above).

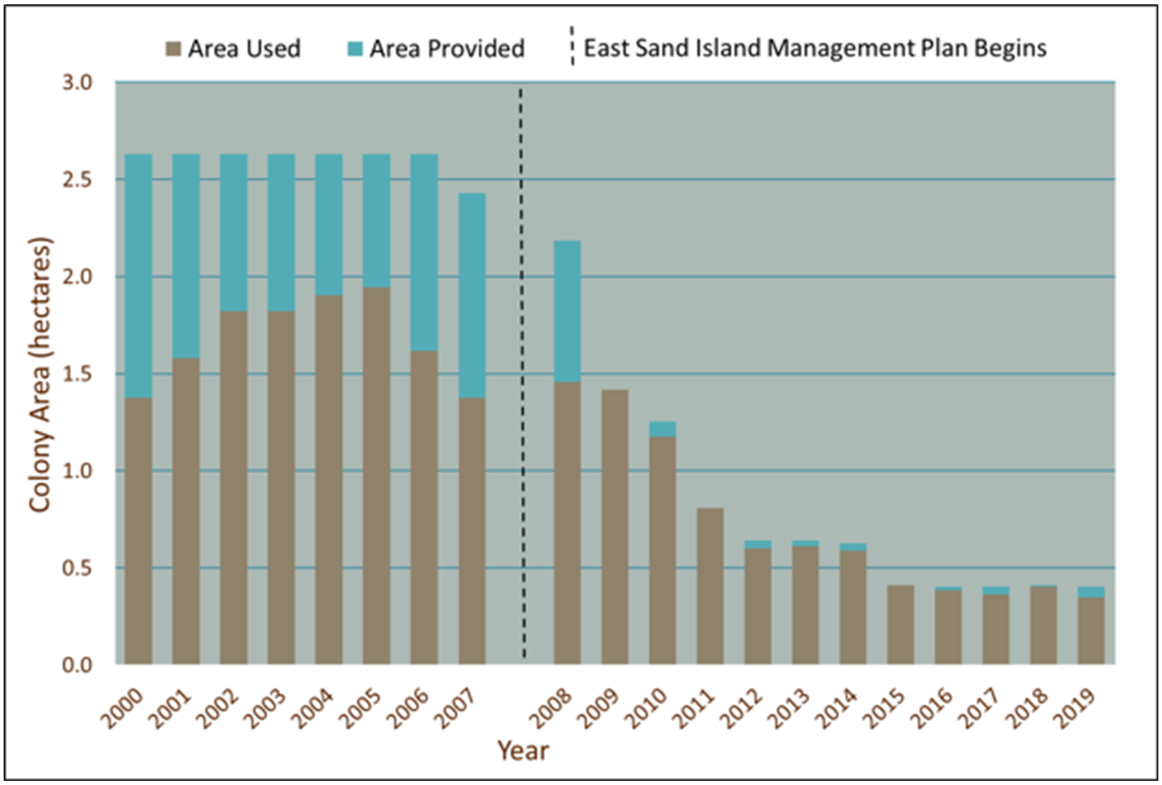

Figure 1.3. Area of suitable nesting habitat provided for Caspian terns on East Sand Island during the pre-management (2000–2007) and management (2008–2019) periods, and the area of provided nesting habitat occupied by nesting Caspian terns in each year. The vertical dashed line marks the implementation of the Caspian Tern Management Plan.

Despite the reduction in available tern nesting habitat from 6 acres (2.4 ha) in 2007 to 5.5 acres (2.2 ha) in 2008, nearly 2.0 acres (0.8 ha) of suitable habitat remained unused by nesting terns. Not until 2009, when only 3.5 acres (1.4 ha) of bare-sand habitat was provided for nesting terns, did Caspian terns nest in virtually all the available habitat (*Figure 1.3* above). As the area of available tern nesting habitat was reduced until 2015, the nesting density of terns on the designated colony area increased and the numbers of nesting terns declined (see *below*). Starting in 2011, some Caspian terns nesting on East Sand Island prospected for nest sites outside the prepared area of nesting habitat, mostly on the upper beach just to the east of the designated colony site.

The prepared tern nesting habitat on East Sand Island was originally restored by removing all vegetation in 1999 from sandy dredged material that had been deposited on the east end of East Sand Island in 1983. The sand component of this dredged material was gradually reduced by wind action that selectively removed sand particles from the surface of the prepared colony area, leaving a hardpan of fine-grained material. Precipitation was less readily absorbed by this fine-grained substrate, causing pooling, rill-wash, and partial flooding on the colony surface during heavy rain events. The progressive de-sanding of the prepared colony substrate necessitated disking the colony each spring before the arrival of nesting terns. Nevertheless, the colony substrate has gradually become less suitable for successful tern nesting and will remain so until the 1-acre (0.4-ha) prepared colony area has sand added to enhance the porosity and friability of the existing substrate.

1.4.1.2 Nesting chronology – The first egg date for Caspian terns nesting at the East Sand Island colony each year was during the last two weeks of April (*Figure 1.4*). Median first egg date during the pre-management period (2000–2007) was on 19 April, eight days earlier than the median first egg date during the management period (2008–2019). The date when the first tern chick hatched was more variable than the first egg date, ranging from 13 May to 20 June, or a period of more than five weeks (*Figure 1.4*). The difference in median first hatch date between the pre-management and management periods was greater than for first egg date; median first hatch date was 18 May during 2000–2007 and 6 June during 2008–2019, a 19-day difference. Because the length of the incubation period for Caspian tern eggs varies by only a few days (Cuthbert and Wires 1999), the later median first hatch date during the management period was mostly a consequence of lower survival of tern eggs laid early in the nesting season. The date when the first young tern fledged (achieved sustained flight) was also highly variable among years (*Figure 1.4*), ranging from 19 June to 31 July (a 6-week period). Median first fledge date was over three weeks later during the management period (17 July) than during the pre-management period (24 June).

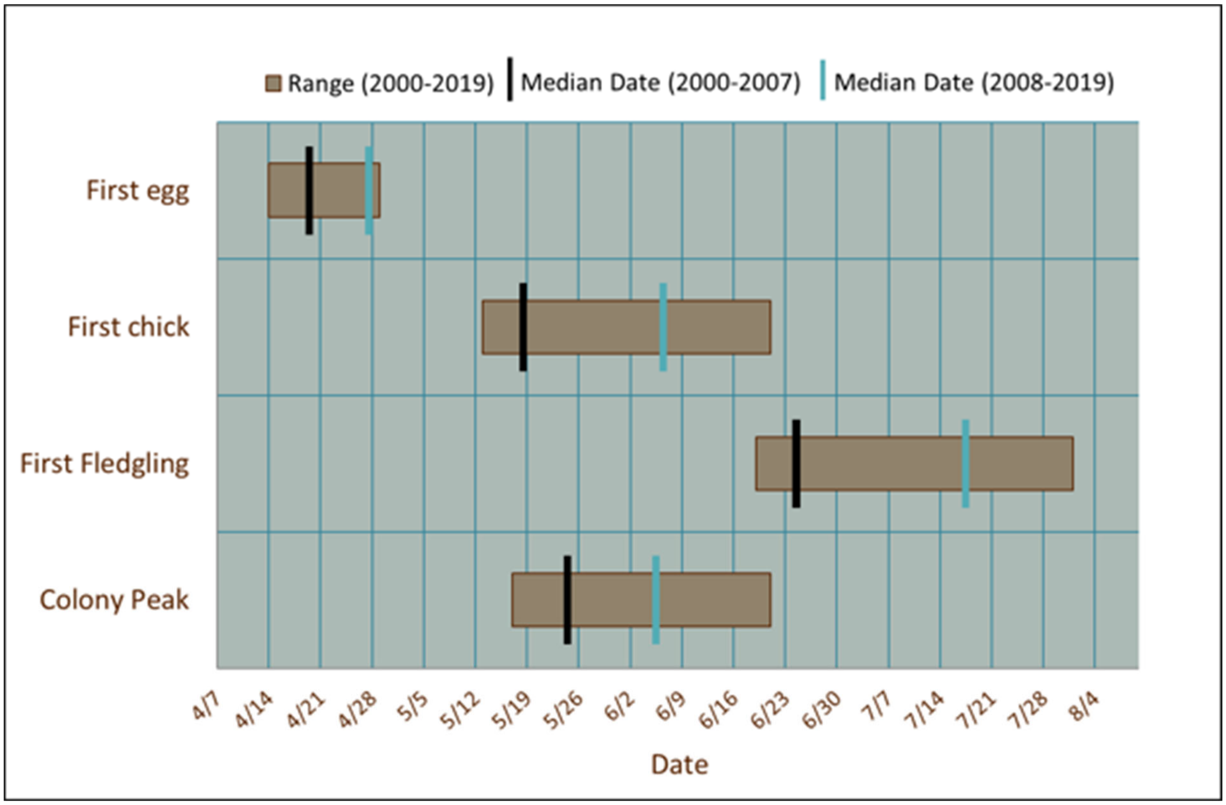

Figure 1.4. Nesting chronology for Caspian terns nesting on East Sand Island in the Columbia River estuary during 2000–2019. Median dates for each nesting chronology measure are presented for both the pre-management period (2000–2007) and the management period (2008–2019).

Each year, colony monitors attempted to identify the date when the peak number of active tern nests were present on the East Sand Island colony, and that date was selected for aerial photography of the tern colony to estimate colony size (number of breeding pairs). The date of peak colony attendance ranged from 17 May to 20 June during the study period, but as with other measures of nesting chronology, the median date of peak colony attendance was later during the management period than during the pre-management period. Median date of peak colony attendance during the management period (5 June) was 12 days later than during the pre-management period (24 May). The later nesting chronology during the management period compared to the pre-management period was largely a reflection of a gradual delay in the timing of nesting events across the 20-year study period (*Figure 1.5*). Most of the variation in the first hatch date was explained by year ($P < 0.0001$), indicating a clear temporal trend toward later nesting by Caspian terns at the East Sand Island colony. Onset of nesting (first egg date) was delayed much less, however, than first hatch date or first fledge date, indicating that much of the delay in nesting chronology during the management period was due to factors that reduced nesting success of early nesters and first nesting attempts.

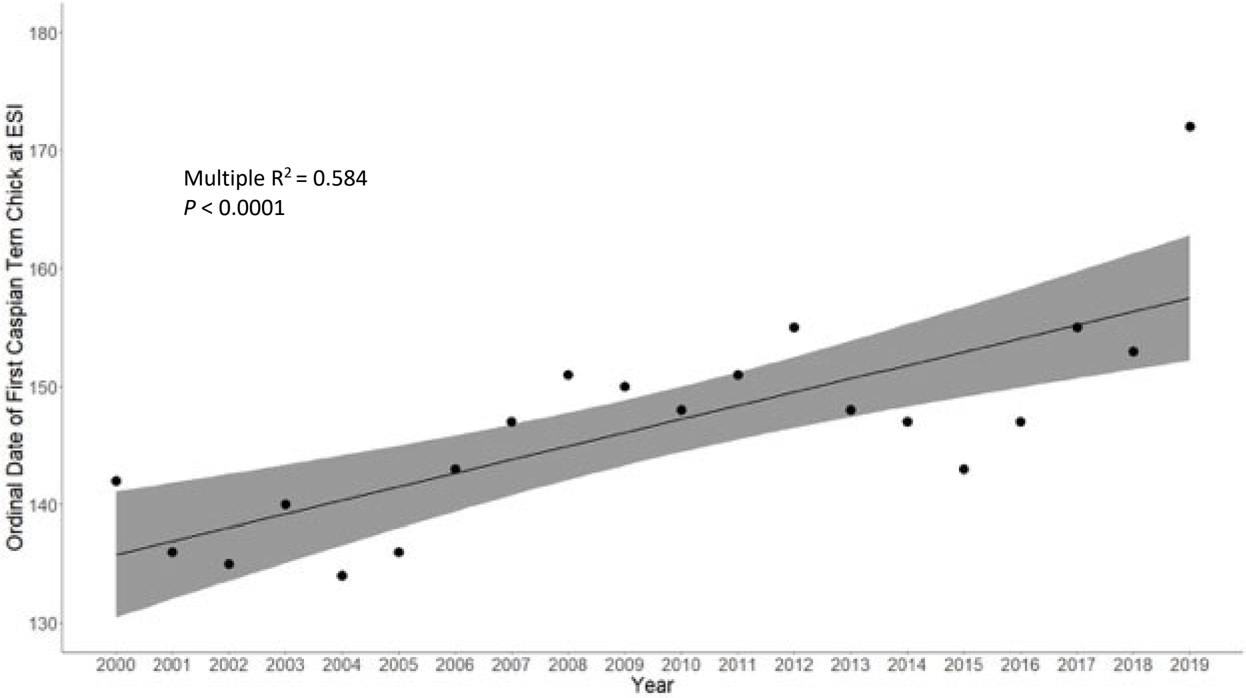

Figure 1.5. Trend in the date when the first Caspian tern chick appeared on the breeding colony (first hatch date) at East Sand Island (ESI) in the Columbia River estuary during 2000–2019. Solid line represents the least squares linear regression, and the shaded area represents the 95% confidence interval about the slope of that line.

The trend of delay in nesting chronology for Caspian terns at the East Sand Island colony over the 20-year study period was also reflected in the numbers of adult terns attending the colony over the nesting season. During the pre-management period, the numbers of terns attending the colony increased rapidly from late March until mid-May and then, after peaking in late May, declined precipitously during June and July (*Figure 1.6*). By August, few adult terns remained on-colony during the day and most young terns had fledged and left the colony. In 2019, 12 years after the initiation of management to reduce the size of the East Sand Island colony, the seasonal pattern of colony attendance had changed dramatically. Colony attendance started later, increased more slowly, peaked in late June and early July, and gradually declined through late July and August (*Figure 1.6*). Some adult terns continued to attend the colony into September. The extended nesting season during the management period was primarily due to breeding pairs re-nesting repeatedly, sometime up to three times, if earlier nesting attempts failed (Bailey 2018), rather than new breeding pairs arriving to nest later in the season as earlier breeding pairs fledged their young and left the colony.

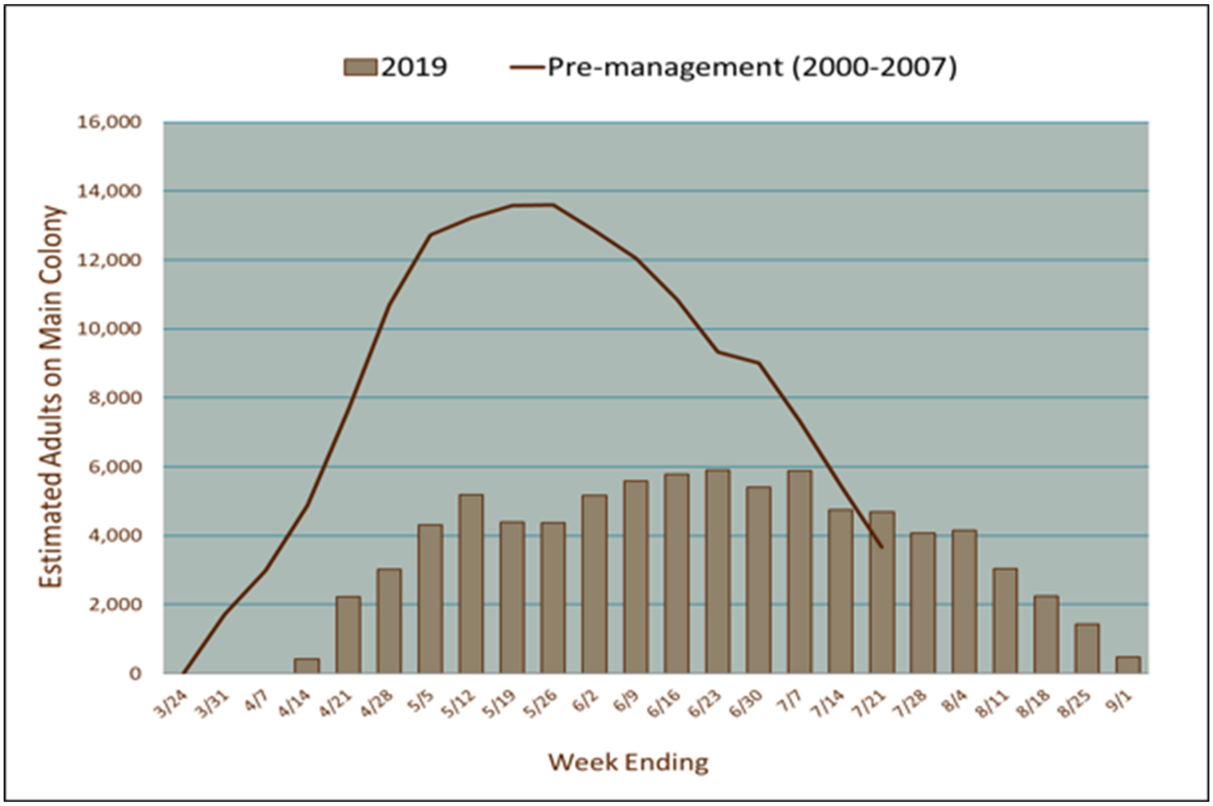

Figure 1.6. Weekly estimates from the ground of the number of adult Caspian terns on the designated 1-acre (0.4-ha) colony area at East Sand Island during the 2019 breeding season compared to the weekly averages during the pre-management period (2000–2007).

1.4.1.3 Colony size & nesting density – The numbers of Caspian tern breeding pairs nesting in the Columbia River estuary were estimated from 1997 to 2019. During 1997–1999 nearly all nesting by Caspian terns in the estuary was at the colony on Rice Island, and during 2000–2019 nearly all nesting by Caspian terns was at the colony on East Sand Island (*Figure 1.7*). The managed relocation of the tern colony from Rice Island to East Sand Island took place during 1999 and 2000; by 2001 no nesting by Caspian terns occurred on Rice Island and all known nesting occurred on East Sand Island. The tern colony on Rice Island consisted of about 8,000 breeding pairs (Collis et al. 2000) before relocation to East Sand Island, whereas the colony on East Sand Island gradually increased to over 10,000 breeding pairs by 2008 (*Table 1.2*), the first year of implementation of the *Caspian Tern Management Plan*.

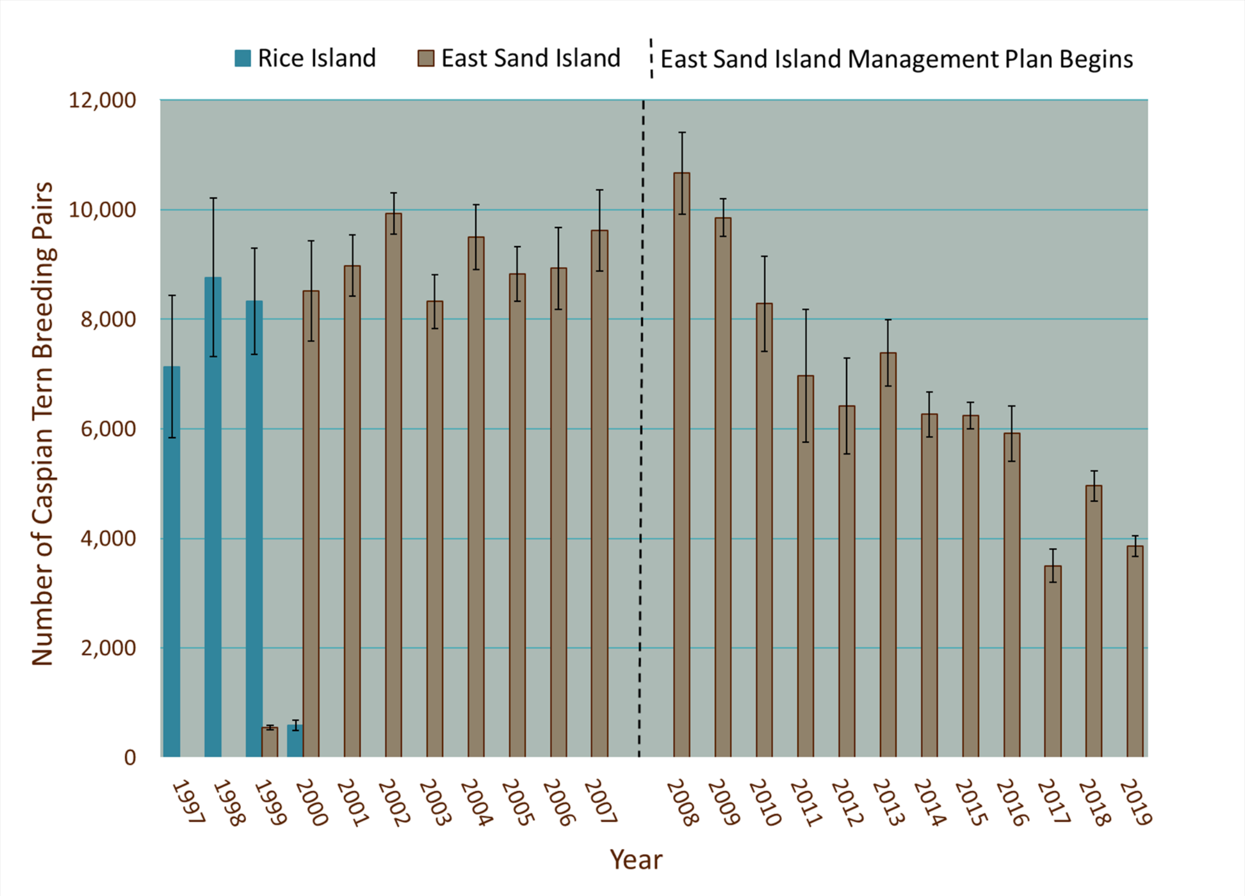

Figure 1.7. Size of the Caspian tern breeding population (number of breeding pairs) in the Columbia River estuary before, during, and after the shift of the colony from Rice Island in the upper estuary to East Sand Island in the lower estuary, 1997–2019. Error bars represent the 95% confidence interval about the annual estimate derived using a Monte Carlo simulation procedure that incorporated the variance in the counts from the aerial photography and the variance in the ratios of incubating to non-incubating adult terns among 12 on-colony plots. Note: Estimates of colony size on East Sand Island during the management period were restricted to the prepared colony area and do not include any Caspian terns that may have nested in satellite colonies on East Sand Island.

Table 1.2. Estimated colony size (number of breeding pairs) and nesting density (nests/m^2) for Caspian terns nesting on East Sand Island in the Columbia River estuary during 2000–2019. Potential error of the estimates is expressed as the 95% confidence interval (c.i.). Average colony size and nesting density are presented for the pre-management period (2000–2007) and the management period (2008–2019) periods. Note: Estimates of colony size on East Sand Island during the management period were restricted to the prepared colony area and do not include any Caspian terns that may have nested in satellite colonies on East Sand Island. The dashed horizontal line indicates implementation of the Caspian Tern Management Plan.

Year	Colony Size	Lower 95% c.i.	Upper 95% c.i.	Nesting Density	Lower 95% c.i.	Upper 95% c.i.
2000	8,513	7,597	9,429	0.62	0.55	0.69
2001	8,982	8,427	9,537	0.57	0.53	0.61
2002	9,933	9,552	10,314	0.55	0.53	0.57
2003	8,325	7,838	8,812	0.45	0.42	0.48
2004	9,502	8,905	10,099	0.50	0.47	0.53
2005	8,822	8,325	9,319	0.45	0.42	0.48
2006	8,929	8,188	9,670	0.55	0.50	0.60
2007	9,623	8,880	10,366	0.70	0.65	0.75
2008	10,668	9,923	11,413	0.72	0.67	0.77
2009	9,854	9,509	10,199	0.70	0.68	0.72
2010	8,283	7,412	9,154	0.70	0.63	0.77
2011	6,969	5,759	8,179	0.85	0.75	0.95
2012	6,416	5,545	7,287	1.06	0.92	1.20
2013	7,387	6,776	7,998	1.17	1.06	1.28
2014	6,269	5,858	6,680	1.06	0.99	1.13
2015	6,240	6,000	6,480	1.32	1.26	1.37
2016	5,915	5,410	6,425	1.36	1.31	1.41
2017	3,500	3,200	3,900	0.97	0.87	1.06
2018	4,959	4,682	5,236	1.23	1.16	1.29
2019	3,861	3,667	4,055	1.11	1.06	1.17
Average (2000-2007)	**9,079**	**8,464**	**9,693**	**0.55**	**0.51**	**0.59**
Average (2008-2019)	**6,693**	**6,145**	**7,251**	**1.02**	**0.95**	**1.09**

Following implementation of the *Caspian Tern Management Plan* and the reduction in area of bare-ground nesting habitat for terns on East Sand Island (*Figure 1.2* above), the size of the colony gradually declined to slightly less than 4,000 breeding pairs by 2019. Colony size in 2019 was well within the target range of colony sizes (3,125–4,375 breeding pairs) stipulated in the Final EIS and Records of Decision for the *Caspian Tern Management Plan* (USFWS 2005, 2006; USACE 2006). *Figure 1.8* compares the trends in colony size at East Sand Island during the pre-management period and the management period. The trend in colony size during the pre-management period (2000–2007) was only slightly positive and not significant (*Figure 1.8A*), whereas the trend during the management period (2008–2019) was strongly negative and highly significant ($P < 0.0001$; *Figure 1.8B*).

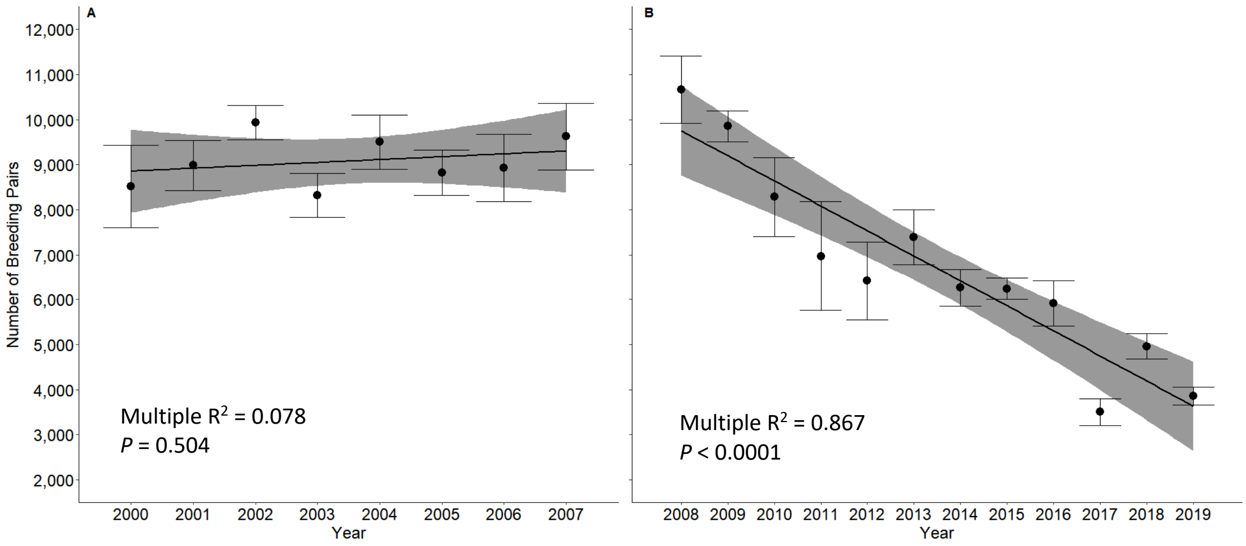

Figure 1.8. Trends in size (numbers of breeding pairs) of the Caspian tern breeding colony on East Sand Island in the Columbia River estuary before (2000–2007; A) and after (2008–2019; B) implementation of management to reduce colony size. Error bars represent the 95% confidence intervals. Solid lines represent the least squares linear regression and shaded areas represent the 95% confidence interval about the slope of that line.

During the management period, as the area of available tern nesting habitat (*Figure 1.2* above) and the size of the tern colony (*Figure 1.7* above) declined, the nesting density of Caspian terns on the East Sand Island colony increased markedly (*Figure 1.9*, *Table 1.2* above). Average nesting density during the pre-management period was 0.55 nests/m^2, whereas during the management period nesting density averaged 1.02 nests/m^2, nearly twice as much. There was no significant trend in nesting density ($P = 0.865$) during the pre-management period (*Figure 1.10*), but during the management period the trend was significant ($P = 0.0043$) and positive (*Figure 1.10*). The trend of increasing nesting density suggests that nesting Caspian terns partially compensated for the decline in area of available nesting habitat by nesting at higher densities. Nesting density appeared to peak in 2016 at an average of 1.36 nests/m^2, and then relaxed in the subsequent three years to between 0.97 and 1.23 nests/m^2 (*Figure 1.9*).

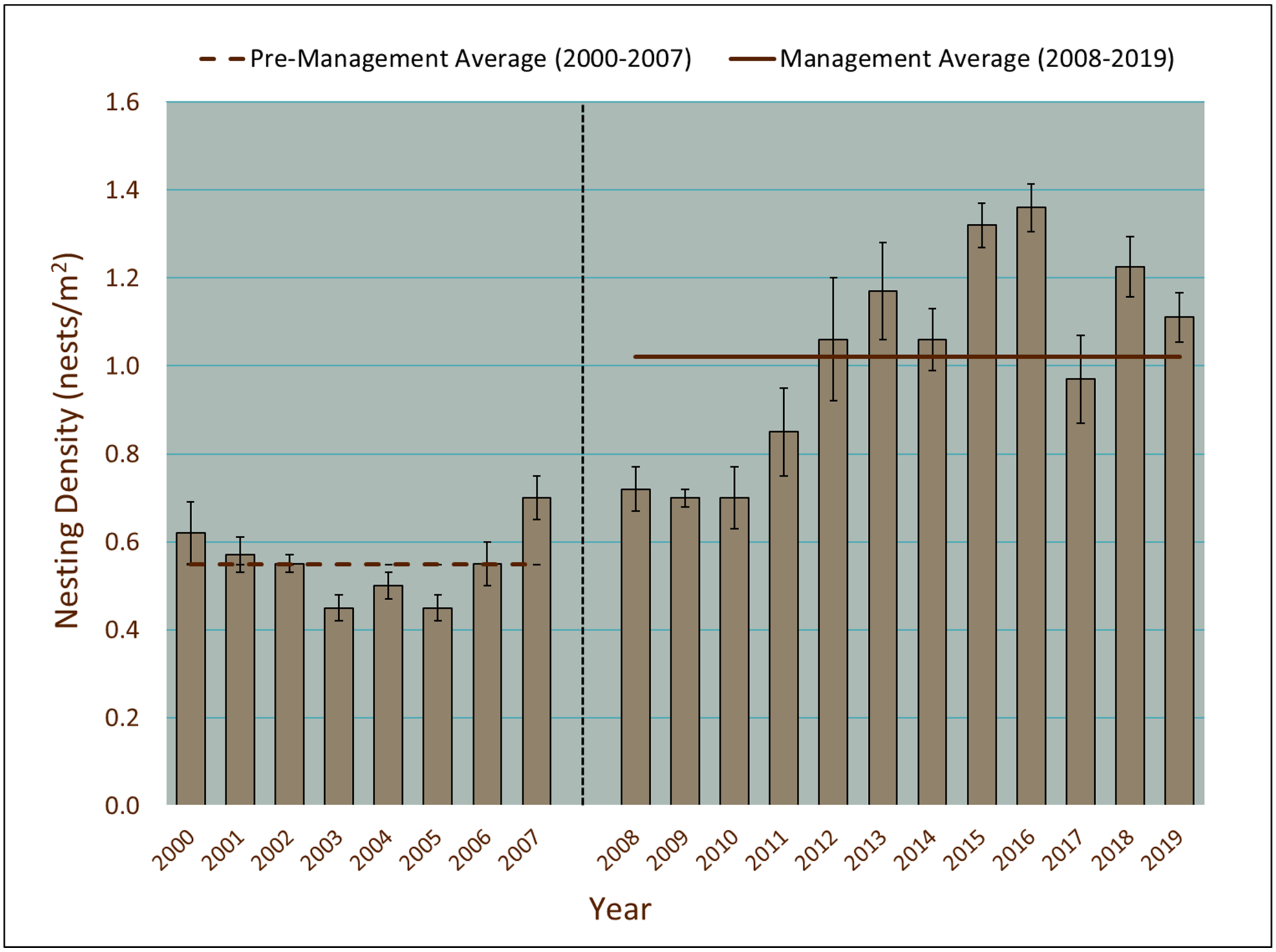

Figure 1.9. Caspian tern nesting density (average number of nests/m^2) at the colony on East Sand Island in the Columbia River estuary during the pre-management period (2000–2007) and during the management period (2008–2019). The Caspian Tern Management Plan presumed that the average nesting density during the management period would remain constant at the average nesting density observed during the pre-management period (i.e. 0.55 nests/m^2; USACE 2015). Error bars represent the 95% confidence intervals; the vertical dashed line indicates the implementation of the Caspian Tern Management Plan.

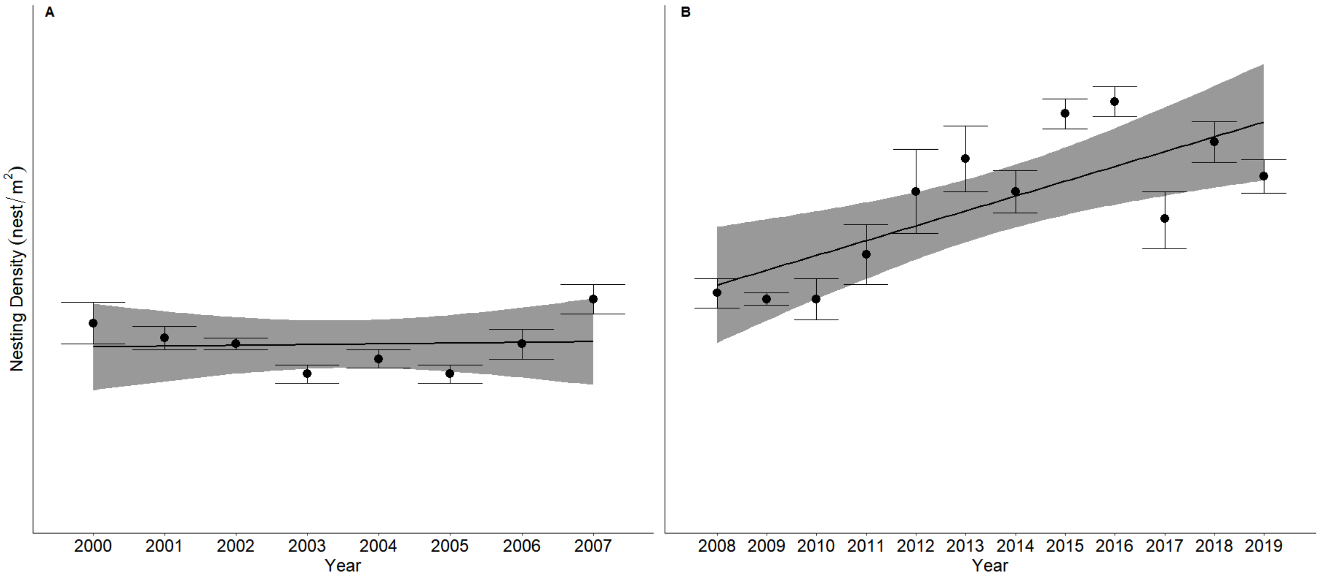

Figure 1.10. Trends in nesting density (average number of nests/m^2) of Caspian terns at the breeding colony on East Sand Island in the Columbia River estuary during the pre-management period (2000–2007) and during the management period (2008–2019). Error bars represent 95% confidence intervals; solid lines represent the least squares regression line, and the shaded areas represent the 95% confidence interval for the slope of the line.

1.4.1.4 Nesting success – Inter-annual variability in average nesting success at the Caspian tern colony on East Sand Island remained high throughout the 20-year study period (*Figure 1.11*). Nesting success at the East Sand Island colony varied from a high of about 1.4 young raised/breeding pair in 2001 to a low of no young raised (complete colony failure) in 2011 and 2017. Nesting success at the East Sand Island colony during the pre-management period (1999–2007) averaged 0.89 young raised per breeding pair. This level of nesting success is considered well above average for Caspian terns (Cuthbert and Wires 1999), and well above average nesting success at the Rice Island colony during 1997–2000 (0.30 young raised/breeding pair). Average nesting success at the East Sand Island colony was much lower during the management period (2008–2019), when nesting success averaged only 0.27 young raised/breeding pair (*Figure 1.11*). During the management period at East Sand Island there were two years when no young terns were raised at the colony (*Figure 1.11*).

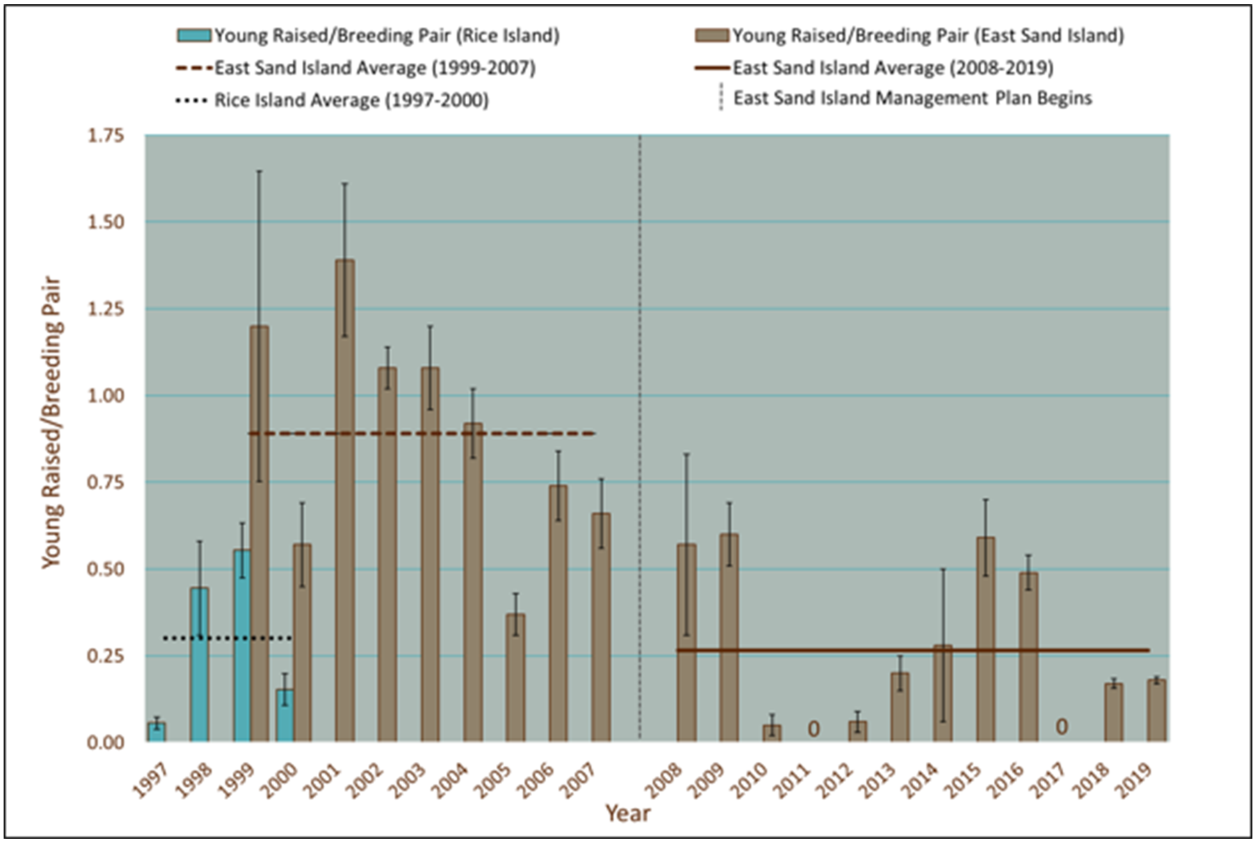

Figure 1.11. Nesting success (average number of young raised per breeding pair) for Caspian terns nesting at Rice Island (1997–2000), at East Sand Island during the pre-management period (1999–2007), and at East Sand Island during the management period (2008–2019) in the Columbia River estuary. Error bars represent 95% confidence intervals.

An investigation of trends in nesting success for Caspian terns at the East Sand Island colony during 2000–2019 revealed that average nesting success was in decline soon after the colony was relocated from Rice Island to East Sand Island (*Figure 1.12*). While the trend in average nesting success during the pre-management period is clearly downward, there was little or no apparent downward trend in nesting success during the management period (*Figure 1.12*).

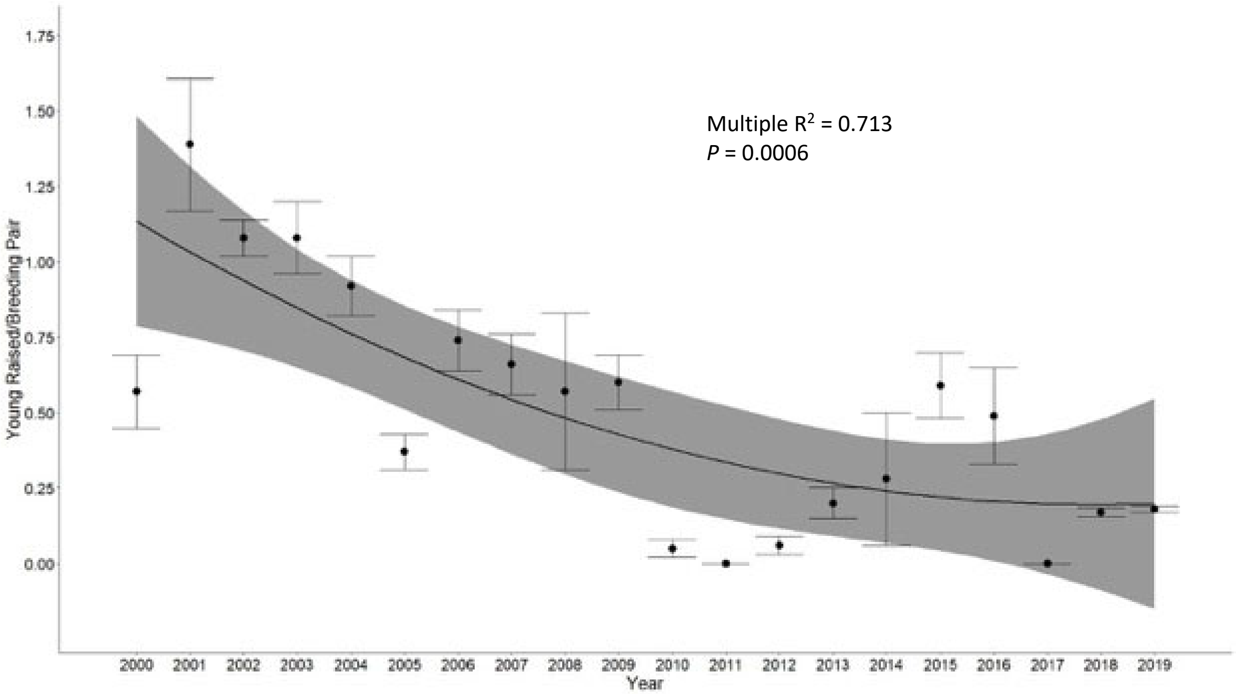

Figure 1.12. Trend in nesting success (average number of young raised per breeding pair) for Caspian terns nesting at East Sand Island in the Columbia River estuary during the pre-management period (2000–2007) and during the management period (2008–2019). Error bars represent 95% confidence intervals; the curved solid line represents the best fit quadratic equation, and the shaded area represents the 95% confidence interval for the slope of the curve.

1.4.1.5 Factors limiting colony size & nesting success – Tern colony size (number of breeding pairs) and nesting success (average number of young raised/breeding pair) at the East Sand Island colony were highly variable across the study period (2000–2019). Colony size varied from a high of about 10,700 breeding pairs in 2008, the first year of implementation of the *Caspian Tern Management Plan*, to a low of 3,500 breeding pairs in 2017 (*Figure 1.7* above). Most of the variation in tern colony size at East Sand Island can be explained by the gradual reduction in available tern nesting habitat during the management period; the managed reduction in tern nesting habitat limited the numbers of Caspian tern nesting pairs that could secure nesting space on East Sand Island. *Figure 1.13* depicts colony size as a function of the area of bare-ground habitat prepared for Caspian terns prior to the nesting season. A quadratic equation fits the data well (multiple $R^2 = 0.875$, $P < 0.0001$), indicating that variation in colony area had little or no effect on colony size when area exceeded 3.5 acres (1.4 ha), while the negative effect on colony size increased rapidly as colony area was reduced to 1.0 acres (0.4 ha). Nevertheless, some of the variation in colony size, both the pre-management period and the management period, could not be explained by changes in available nesting habitat (*Figure 1.13*). This residual variation in colony size, unexplained by variation in available nesting habitat, was likely caused by a combination of bottom-up factors (e.g., forage fish availability) and top-down factors (e.g., pressure from nest predators; Collar et al. 2017, Bailey 2018).

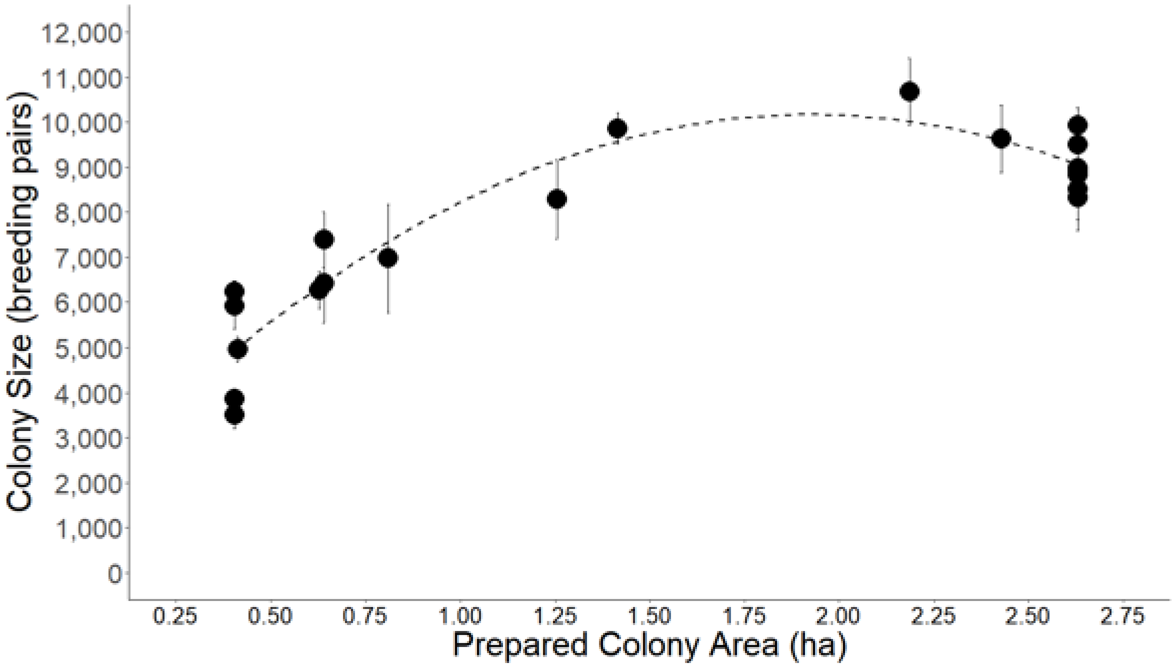

Figure 1.13. Size of the Caspian tern breeding colony (number of breeding pairs) as a function of the area of bare ground nesting habitat prepared for terns prior to the nesting season on East Sand Island in the Columbia River estuary. Vertical bars represent the 95% confidence interval for colony size and the dashed line represents the best fit quadratic equation.

An examination of trends in nesting success of Caspian terns at the East Sand Island colony during 2000–2019 revealed that average nesting success was in decline soon after the colony was relocated from Rice Island to East Sand Island (*Figure 1.12* above). While the trend in average nesting success during the pre-management period is clearly downward, during the management period there was little or no apparent trend and nesting success remained at a low level (*Figure 1.12* above). The factors responsible for the decline in nesting success of Caspian terns at the East Sand Island colony were at work well before the implementation of the *Caspian Tern Management Plan*. There was a significant positive correlation between colony size and average nesting success at East Sand Island during the study period (2000–2019), and there was a significant negative correlation between nesting density and nesting success (Bailey 2018). But the decline in tern nesting success at East Sand Island was apparent starting in 2002, well before the implementation of reductions in nesting habitat (*Figure 1.12* above), suggesting that the factors causing reduced nesting success were unrelated to the managed reduction in area of suitable nesting habitat. Also, those terns that nested in parts of the colony with relatively high densities of nesting terns had higher nesting success than those that nested where nesting density was low (Bailey 2018). During the management period (2008–2019), two environmental factors emerged as causes of low nesting success at the East Sand Island Caspian tern colony: (1) reduced availability of marine forage fish in the estuary associated with high Columbia River discharge and (2) frequent colony disturbances by bald eagles and associated high predation rates on tern eggs and chicks by glaucous-winged/western gulls (Collar et al. 2017, Bailey 2018). These two factors both increased in intensity during the study period and are negatively correlated with nesting success at the tern colony.

River discharge into the Columbia River estuary has a negative influence on tern nesting success through both bottom-up and top-down factors (Collar et al. 2017). Freshwater discharge into the estuary has a negative bottom-up influence on tern nesting success by reducing the availability of marine forage fish in the estuary (Weitkamp et al. 2012), prey types that collectively comprise more than half of the diet of Caspian terns nesting on East Sand Island (see *below*). River discharge also has an indirect negative effect on tern nesting success by reducing food availability to tern nest predators, a top-down effect (Collar et al. 2017); when marine prey are less available to predators, they switch more to seabirds and their eggs and young as a source of food. Gulls nesting near the tern colony at East Sand Island devote more time and effort to kleptoparasitizing terns (stealing bill-load fish) as the availability of marine forage fish declines (Bailey 2018), further reducing the food energy that terns can allocate toward successfully raising young. Bald eagles switch to preying on terns and their nest contents, plus kleptoparasitizing terns for bill-load fish, as the availability of marine forage fish declines (Bailey 2018). Bald eagle activity at the tern colony caused major disturbances, when virtually all breeding terns flushed from the colony, leaving their eggs and young unattended and vulnerable to predation by gulls. The combination of high colony disturbance rates by eagles and high nest predation rates by gulls was the cause of total reproductive failure at the East Sand Island tern colony in 2011 and 2017, two years of exceptionally high river discharge early in the nesting season (Collar et al. 2017, Bailey 2018).

These results, although correlative, suggest that interactions between bottom-up and top-down regulating factors, driven largely by variable freshwater input to the Columbia River estuary, limited Caspian tern nesting success at the breeding colony on East Sand Island during 2000–2019. We found that river discharge was highly negatively correlated to nesting success at this colony, with average nesting success at the East Sand Island tern colony declining significantly as freshwater input during the tern breeding season increased. Years of higher-than-average river discharge (2011 and 2017) were also years when the Caspian tern colony on East Sand Island failed to produce any young, apparently driven by declines in the abundance and availability of marine forage fishes in the estuary, which in turn limited food availability to many estuary consumers. The decline in prey availability likely contributed to the increase in bald eagle disturbance and gull kleptoparasitism at the East Sand Island tern colony, and to the corresponding increase in depredation rates on tern nests by gulls due to the synergistic impact of eagle disturbance and gull predation on tern nesting success (Collar et al. 2017). In particular, Columbia River discharge was highly and positively correlated with gull kleptoparasitism rates on terns during the chick-rearing period (*Figure 1.14*), providing a strong link between bottom-up and top-down controls on nesting success at the Caspian tern colony on East Sand Island (Bailey 2018).

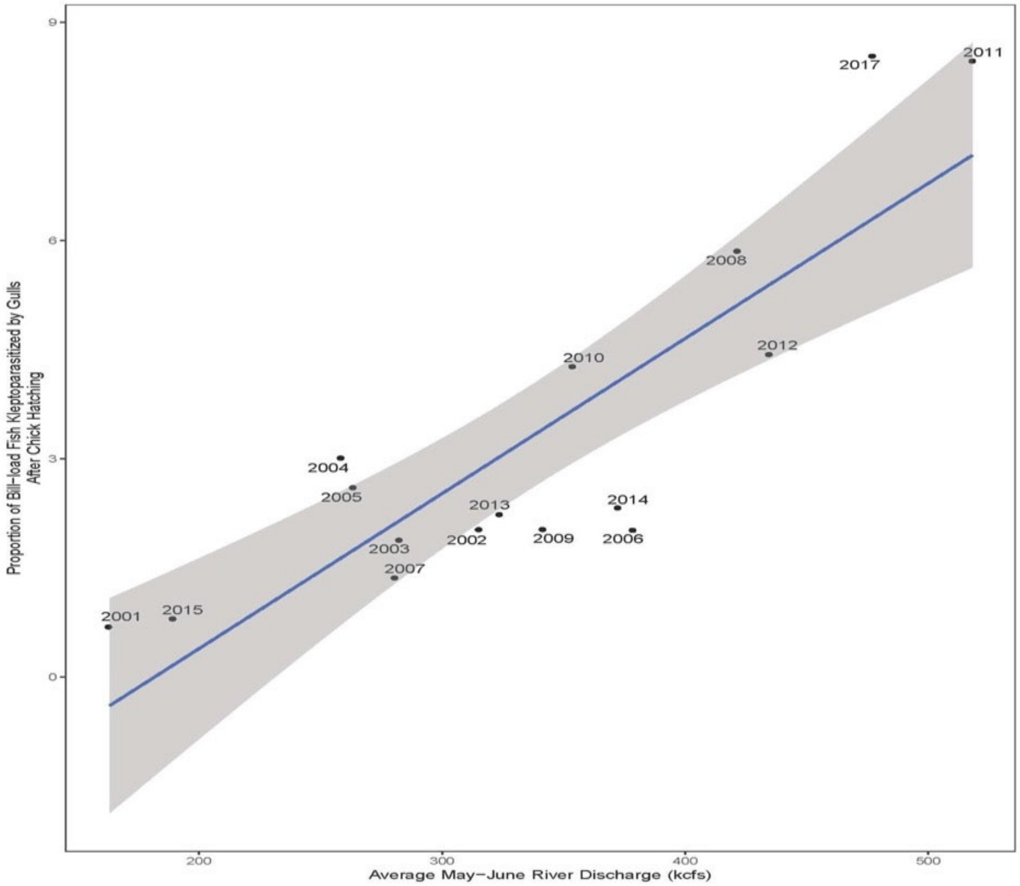

Figure 1.14. Gull kleptoparasitism rate after hatch (proportion of tern bill-load fish kleptoparasitized by gulls/all recorded bill-load fish observed on-colony after observation of the first tern chick) at the Caspian tern colony on East Sand Island in the Columbia River estuary as a function of average Columbia River discharge during May/June (kcfs) during 2001–2017 (R = 0.85, P < 0.001). Data points are labeled by year. The solid line represents the least squares linear regression, and the shaded area represents the 95% confidence interval for the slope of the line. After Bailey (2018).

1.4.1.6 Dispersal and colony connectivity – Of the 50 adult Caspian terns radio-tagged on Rice Island in 2000, 7 (14%) and 37 (74%) subsequently nested on Rice Island and East Sand Island, respectively (Collis et al. 2002). Efforts to capture and mark terns using rocket-nets on Rice Island prior to the nesting season may have helped encourage terns to relocate to East Sand Island to nest in 2000. Also, the large number of terns that settled at the East Sand Island colony site prior to rocket-netting on Rice Island probably attracted terns that were tagged on Rice Island to relocate to East Sand Island and nest there. Fifty-two percent (n = 12) of the 23 terns radio-tagged on East Sand Island in 2000 whose eggs were removed failed to renest (Collis et al. 2002). Of those 11 terns that renested, 8 (73%) and 3 (27%) renested at the East Sand Island and Rice Island colonies, respectively. These results suggest that, under the conditions of the experiment, failed breeders at the East Sand Island colony are more likely to renest at that colony than elsewhere. The results may have been different, however, if there had been alternative colony sites in the Columbia River estuary available for failed breeders to immigrate to other than Rice Island.

Caspian tern chicks that were banded pre-fledging with field readable leg-bands during the 2000–2016 nesting seasons (*Table 1.1* above) were used to assess dispersal of terns raised in the Columbia River estuary to colony sites in other regions of the Pacific Flyway. Despite limited resighting effort, Caspian terns that were banded pre-fledging at the Rice Island or East Sand Island breeding colonies were resighted during 2003–2007 at nesting and roosting sites in coastal Washington State, the Columbia Plateau region in interior Washington State, the San Francisco Bay area in California, and at Salton Sea in southeastern California (*Table 1.3*). Resighting effort during 2008–2015 expanded in both intensity and geographic extent, including much of the breeding range of the Pacific Flyway population of Caspian terns. Monitoring the alternative Caspian tern colony sites that were built by the USACE in the SONEC (southern Oregon & northeastern California) region during 2008–2012 (see *Chapter 3*) led to large numbers of resightings in the SONEC region of terns banded as pre-fledged young at the East Sand Island colony (*Table 1.3*). The discovery of a new Caspian tern colony in the Copper River Delta of Alaska resulted in the resighting of a number of terns that were banded as pre-fledged young on East Sand Island; this represents a natal dispersal distance of more than 2,000 km. Together with the large numbers of resightings throughout the Pacific Flyway of terns that were banded pre-fledging at East Sand Island (*Table 1.3*), these data demonstrate the high natal dispersal ability of Caspian terns raised at the East Sand Island colony, and the high connectivity of that colony to all breeding colonies in the Flyway.

Table 1.3. Numbers of Caspian terns that were individually marked as pre-fledged young at the colony on East Sand Island in the Columbia River estuary and subsequently resighted elsewhere in the Pacific Flyway during 2003–2019. Terns were marked with either a unique color combination of plastic leg-bands or a field-readable alphanumeric leg-band. Numbers include resightings of individuals that were resighted in more than one region within a year, and that were resighted during more than one year. "—" indicates no resighting effort within that region during that nesting season.

	Region Where Resighted						
Year	Coastal Alaska	Coastal Washington	Columbia Plateau, WA	SONEC, OR/CA	San Francisco Bay, CA	Salton Sea, CA	**Yearly Total**
2003*	--	19	--	--	--	1	**20**
2004*	--	5	--	--	3	--	**8**
2005*	--	13	--	--	2	--	**15**
2006*	--	--	4	--	--	--	**4**
2007*	--	--	2	--	--	--	**2**
2008	--	18	4	5	8	--	**35**
2009	--	20	3	24	11	--	**58**
2010	5	62	17	34	--	31	**149**
2011	2	58	21	79	--	36	**196**
2012	10	26	53	130	--	20	**239**
2013	5	--	21	124	--	--	**150**
2014	2	--	43	102	--	18	**165**
2015	4	74	41	55	6	--	**180**
2016*	--	83	34	9	--	--	**126**
2017*	--	29	11	--	--	--	**40**
2018*	--	--	8	--	--	--	**8**
2019*	--	--	6	--	--	--	**6**

* Indicates reduced resighting effort throughout the Pacific Flyway during that year.

Resighting a banded tern at a particular breeding colony does not necessarily indicate that it has recruited to that colony as a breeding adult. Individual marked terns at a breeding colony are not considered to be nesting until they are observed performing at least one of several behaviors associated with nesting (e.g., incubation, brooding young, feeding young). Resighting of terns banded as pre-fledglings at colonies throughout the Pacific Flyway and confirmed as breeders on East Sand Island demonstrated that terns raised at colonies throughout the Flyway were recruiting to the East Sand Island colony as breeders during 2009–2019 (*Table 1.4*). This included terns raised at colonies in the Columbia Plateau region, coastal Washington, the San Francisco Bay area, the SONEC region, and coastal Alaska. But the vast majority of banded young terns that recruited to nest on East Sand Island had been banded on East Sand Island, and therefore were demonstrating natal philopatry.

Table 1.4. Numbers of Caspian terns that were individually marked as pre-fledged young at various colonies in the Pacific Flyway and subsequently resighted at the colony on East Sand Island in the Columbia River estuary and observed displaying nesting behavior on-colony during 2001–2019. Terns were marked with either a unique color combination of plastic leg-bands or a field-readable alphanumeric leg-band. Nesting behaviors included attending a nest scrape, incubating egg(s), brooding chick(s), attending a chick, or feeding a chick. Blank cells indicate no marked individuals from the region were observed nesting at the East Sand Island colony in that year.

	Banding Location						
Year	Coastal Alaska	Coastal Washington	Columbia Plateau, WA	Columbia River Estuary, OR	SONEC, OR/CA	San Francisco Bay, CA	**Yearly Total**
2003				1			**1**
2004							
2005							
2006				7			**7**
2007			1	8			**9**
2008				8			**8**
2009			3	67			**70**
2010			1	5		1	**7**
2011*	--	--	--	--	--	--	**--**
2012		3	2	63		1	**69**
2013		2	1	69		0	**72**
2014		1	2	89		1	**93**
2015		9	5	183		5	**202**
2016		6	7	192		4	**209**
2017*	1	5	5	111		2	**124**
2018		6	5	226	1	1	**239**
2019			2	111		2	**120**

*There was complete nesting failure at East Sand Island during the 2011 and 2017 breeding seasons.

In addition to the large numbers of Caspian terns that were banded as pre-fledged young at colonies across the Pacific Flyway, we also banded some adult terns at a few colonies in the Flyway, but mostly at the colonies on Rice Island and East Sand Island in the Columbia River estuary (*Table 1.1* above). All adult terns that were caught and banded at colonies in the Columbia River estuary were assumed to be nesting there. Terns that were banded as nesting adults in the Columbia River estuary were the most frequently resighted adults at the East Sand Island (*Table 1.5*), indicating a high degree of breeding philopatry. Some individuals that were banded as breeding adults at other colonies were, however, resighted at the East Sand Island colony. Immigrants from colonies in coastal Washington were especially prevalent in the early 2000s, when colonies in that region were abandoned or in decline. Immigrants from colonies in the Columbia Plateau region were most prevalent in the mid-2010s, when the *Inland Avian Predation Management Plan* (*IAPMP*) was implemented to displace nesting Caspian terns from the two largest breeding colonies in the Columbia Plateau region (USACE 2014; see *Chapter 2*).

Only a very few individuals that were banded as breeding adults in other regions of the Pacific Flyway were resighted on East Sand Island (*Table 1.5*), providing more support for the hypothesis of high breeding philopatry in the Pacific Flyway population of Caspian terns.

Table 1.5. Numbers of Caspian terns that were individually marked as adults at various colonies in the Pacific Flyway and subsequently resighted at the colony on East Sand Island in the Columbia River estuary during 2001–2019. Terns were marked with either a unique color combination of plastic leg-bands or a field readable alphanumeric leg-band. Blank cells indicate no marked individuals from the region were resighted at the East Sand Island colony in that year.

	Banding Location					
Year	Coastal Washington	Columbia Plateau, WA	Columbia River Estuary, OR	SONEC, OR/CA	San Francisco Bay, CA	**Yearly Total**
2001	8		474			**482**
2002	18		276			**294**
2003	22	1	56			**79**
2004	16	1	14			**31**
2005	2	2	29			**33**
2006	3	6	48			**57**
2007	1	6	85			**92**
2008	2	5	123		1	**131**
2009	1	8	165			**174**
2010		6	209	1		**216**
2011		11	319			**330**
2012		17	292			**309**
2013		15	243			**258**
2014		27	241	1		**269**
2015		40	226	1		**267**
2016		34	188			**222**
2017		29	176			**205**
2018		26	152			**178**
2019		17	98			**115**

When only those adult terns banded as breeders that were subsequently observed nesting at the colony on East Sand Island were included (*Table 1.6*), the pattern of resightings indicating the prevalence of breeding philopatry was evident. Resightings on East Sand Island of terns banded as adults were almost always of terns that had been captured at the colony on East Sand Island. The few exceptions were terns that had been banded as adults at colonies in coastal Washington in the early 2000s and terns that had been banded as adults at colonies in the Columbia Plateau region prior to implementation of the *IAPMP* in the mid-2010s.

Table 1.6. Numbers of Caspian terns marked as adults at various colonies in the Pacific Flyway and subsequently resighted at the East Sand Island colony in the Columbia River estuary and observed displaying nesting behavior on-colony during 2001–2019. Nesting behaviors included attending a nest scrape, incubating egg(s), brooding chick(s), attending a chick, or feeding a chick. Terns were marked with either a unique color combination of plastic leg-bands or a field-readable alphanumeric leg-band. Blank cells indicate no marked individuals from the region were observed nesting at the East Sand Island colony in that year.

	Banding Location				
Year	Coastal Washington	Columbia Plateau, WA	Columbia River Estuary, OR	SONEC, OR/CA	**Yearly Total**
2001	3		173		**176**
2002	5		60		**65**
2003	3		14		**17**
2004	3		1		**4**
2005			13		**13**
2006	1	1	23		**25**
2007		1	25		**26**
2008			9		**9**
2009		1	70		**71**
2010		9	22		**31**
2011*	--	--	--		**--**
2012		2	85		**87**
2013		1	90		**91**
2014			79		**79**
2015		15	120	1	**136**
2016		10	103		**113**
2017*		8	72		**80**
2018		15	97		**112**
2019		11	61		**72**

*There was complete nesting failure at the East Sand Island colony during the 2011 and 2017 breeding seasons.

1.4.2 Tern Foraging Ecology

1.4.2.1 Foraging behavior – In 1998 and 1999, the distribution of foraging Caspian terns and their selection of foraging habitats within the Columbia River estuary was investigated using point count surveys and aerial surveys (Lyons et al. 2007). Based on point counts, terns selected tidal flats and sites with roosting beaches nearby for foraging, making greater use of the marine/mixing zone of the estuary later in the season, particularly areas near the ocean jetties. During aerial surveys we observed 50% of radio-tagged terns either foraging or commuting within 8 river km of the Rice Island colony, and ≤ 5% of activity occurred ≥ 27 km from this colony in both years (Lyons et al. 2007). These results indicated that relocation of the Rice Island colony to a site or sites at sufficient distance from areas of relatively high salmonid availability would be a more straightforward means of reducing impacts of tern predation on salmonids than habitat alterations within the Columbia River estuary.

Comparisons were made of the foraging ecology of Caspian terns nesting on Rice Island and East Sand Island in the Columbia River estuary during 1999 using radio telemetry and observations of prey fed to chicks and mates at each colony (Lyons et al. 2005). Early in the chick-rearing period, radio-tagged terns nesting at Rice Island (river km 34) foraged mostly in the freshwater zone of the estuary close to the colony, while terns nesting on East Sand Island (river km 8) foraged in the marine or estuarine mixing zones close to that colony. Late in the chick-rearing period, Rice Island terns moved more of their foraging to the marine and estuarine mixing zones, lower in the estuary, while East Sand Island terns continued to forage in these areas. Tern diets at each colony corresponded to the primary foraging zone (freshwater vs. marine/mixing) of radio-tagged individuals: Rice Island terns relied heavily on juvenile salmonids (71% of identified prey) early in chick-rearing, but this declined late in chick-rearing (46%). East Sand Island terns relied less upon salmonids (42% and 16%, early and late in chick-rearing, respectively), and instead utilized marine fishes such as northern anchovy (*Engraulis mordax*) and juvenile herring (*Clupea pallasi*). Throughout chick-rearing, Rice Island terns foraged farther from their colony (median distance: 12.3 km during early chick-rearing and 16.9 km during late chick-rearing) than did East Sand Island terns (9.6 and 7.7 km, respectively).

During the 2000 nesting season, the locations of all off-colony detections for radio-tagged terns nesting on Rice Island (n = 21) were: 14 (67%) on the Columbia River above the Astoria-Megler Bridge, 6 (28%) on the Columbia River below the Astoria-Megler Bridge, 0 (0%) in Willapa Bay or vicinity, and 1 (5%) in Grays Harbor or vicinity (Collis et al. 2002). By comparison, the location of all off-colony detections for radio-tagged terns nesting on East Sand Island (n = 225) were: 19 (8%) on the Columbia River above the Astoria-Megler Bridge, 151 (67%) on the Columbia River below the Astoria-Megler Bridge, 43 (19%) in Willapa Bay and vicinity, and 12 (5%) in Grays Harbor and vicinity. These data indicate that off-colony distribution, and presumably foraging distribution, of terns nesting at the two colonies in the estuary were markedly different. Terns nesting on East Sand Island were found outside the estuary along the Washington Coast more frequently (24% of all detections) than terns nesting on Rice Island (5% of all detections). During the brood-rearing period, Rice Island terns foraged significantly more in freshwater habitats higher in the estuary, where juvenile salmonids were apparently the primary prey type available. Terns nesting on East Sand Island foraged primarily in brackish estuarine and marine habitats, where alternative prey were apparently more abundant, as indicated by diet composition (Collis et al. 2002). These results led to the conclusion that Caspian terns are generalist foragers and make use of the most available forage fish resources near their nesting colony when raising young (Lyons et al. 2005).

Comparisons were also made of the foraging ecology of Caspian terns nesting at East Sand Island between the 2000 and 2001 nesting seasons (Anderson et al. 2007). Average foraging distance from the East Sand Island colony was 38% (6.6 km) greater in 2000, associated with lower availability of marine forage fish near East Sand Island and lower prevalence of marine prey in tern diets during the 2000 nesting season. Colony attendance was much lower (37.0% vs. 62.5% of daylight hours), average trip duration was 40% longer (38.9 min), and nesting success was much lower (0.57 young fledged/pair vs. 1.40 young fledged/pair) in 2000 compared to 2001. Higher proportions of juvenile salmonids in the diet were associated with relatively high use of the freshwater zone of the estuary by radio-tagged terns, which occurred prior to chick-rearing and when out-migrating salmonid smolts were relatively abundant. Lower availability of marine prey in 2000 compared to 2001 apparently limited Caspian tern nesting success by markedly reducing colony attendance and lengthening foraging trips by nesting terns, thereby increasing chick mortality rates from predation, exposure, and starvation (Anderson et al. 2007).

1.4.2.2 Diet composition – The average annual proportion of juvenile salmonids (percent of identified prey items) in the diet of Caspian terns nesting on East Sand Island was about 33% during the study period (1999–2019) but varied from 18.7% to 43.7% of prey items, depending on the year (*Table 1.7*). The proportion of salmonids in the diet averaged across the nesting season (last week of April to end of July) during the management period (2008–2019) averaged slightly more than during the pre-management period (1999–2007), but the difference was not significant (*Table 1.7*). During both the pre-management period and the management period, juvenile salmonids were the single most prevalent prey type, followed by anchovies (Engraulidae) the second most prevalent prey type. In some years during both the pre-management period (2003–2007) and the management period (2013), anchovies were more prevalent in the tern diet than salmonids. During the last three years of our study (2017–2019), however, the prevalence of anchovies in the tern diet was less than half that of juvenile salmonids, suggesting a trend of declining availability of anchovies in the Columbia River estuary. In those latter years, anchovies were less prevalent in tern diets than clupeids (herring, sardines, and shad) and surfperch (Embiotocidae; *Table 1.*7).

Table 1.7. Average annual diet composition (percent of prey items) of Caspian terns nesting at East Sand Island as identified by bill-load fish delivered to the colony from the last week of April to the end of July. "Salmonids" = Oncorhynchus spp.; "Anchovy" = northern anchovy (Engraulis mordax); "Clupeids" = Pacific herring (Clupea pallasi), Pacific sardine (Sardinops sagax), American shad (Alosa sapidissima); "Surfperch" = Embiotocidae; "Smelt" = Osmeridae. Horizontal dashed line indicates implementation of the Caspian Tern Management Plan.

Year	Salmonids	Anchovy	Clupeids	Surfperch	Smelt	Other
1999	43.6%	19.2%	9.5%	11.3%	4.1%	12.3%
2000	43.7%	13.5%	11.8%	11.8%	5.3%	14.0%
2001	32.6%	26.7%	18.7%	6.3%	4.5%	11.2%
2002	31.4%	12.1%	19.2%	13.6%	7.7%	16.0%
2003	26.4%	28.3%	19.6%	7.5%	11.7%	6.7%
2004	18.7%	27.6%	26.2%	13.4%	5.8%	8.3%
2005	22.0%	38.9%	9.4%	18.9%	5.5%	5.3%
2006	33.1%	34.0%	5.0%	17.4%	3.1%	7.4%
2007	28.8%	30.8%	12.1%	15.6%	3.8%	8.7%
2008	31.7%	19.3%	10.2%	22.5%	7.4%	8.8%
2009	35.6%	20.6%	15.6%	18.2%	3.2%	6.8%
2010	33.7%	26.1%	9.2%	22.2%	3.6%	5.3%
2011	36.5%	26.3%	5.3%	16.1%	10.6%	5.1%
2012	34.2%	30.9%	1.8%	12.8%	13.9%	6.4%
2013	30.9%	39.0%	4.3%	11.3%	8.4%	6.1%
2014	34.6%	30.4%	8.0%	14.4%	7.2%	5.4%
2015	38.5%	21.5%	9.7%	12.7%	13.1%	4.5%
2017	35.1%	16.2%	10.8%	26.7%	0.6%	10.7%
2018	40.1%	9.7%	22.1%	16.2%	2.0%	9.9%
2019	33.5%	12.4%	17.2%	22.3%	5.4%	9.2%
Average Pre-Management (1999–2007)	**31.1%**	**25.7%**	**14.6%**	**12.9%**	**5.7%**	**10.0%**
Average Management (2008–2019)	**35.0%**	**22.9%**	**10.4%**	**17.8%**	**6.9%**	**7.1%**

Marine or estuarine forage fishes (i.e. anchovy, surfperch, clupeids [Clupeidae], and smelt [Osmeridae]) collectively comprised more than half the identified prey items in the Caspian tern diet at East Sand Island during both the pre-management period and the management period (*Figure 1.15*). This is in stark contrast to the diet composition of Caspian terns nesting at Rice Island during 1997–2000, where ca. 78% of prey items consisted of juvenile salmonids, and only a small percentage of the diet consisted of marine forage fishes. *Figure 1.15* also indicates the annual variability in diet composition of Caspian terns nesting on East Sand Island. Juvenile salmonids, anchovies, surfperch, clupeids, and smelts were the dominate prey types in all years, but a variety of other prey types, including sculpins (Cottidae), flatfish (Pleuronectidae), sand lance (Ammodytidae), lamprey (Petromyzontidae), northern pikeminnow (*Ptychocheilus oregonensis*), peamouth (*Mylocheilus caurinus*), sticklebacks (Gasterosteidae), and suckers (Catostomidae), each comprised less than 5% of prey items in any given year. These relatively minor prey types were grouped in the category "Other" for the purpose of investigating trends in diet composition (*Table 1.7* above, *Figure 1.15*).

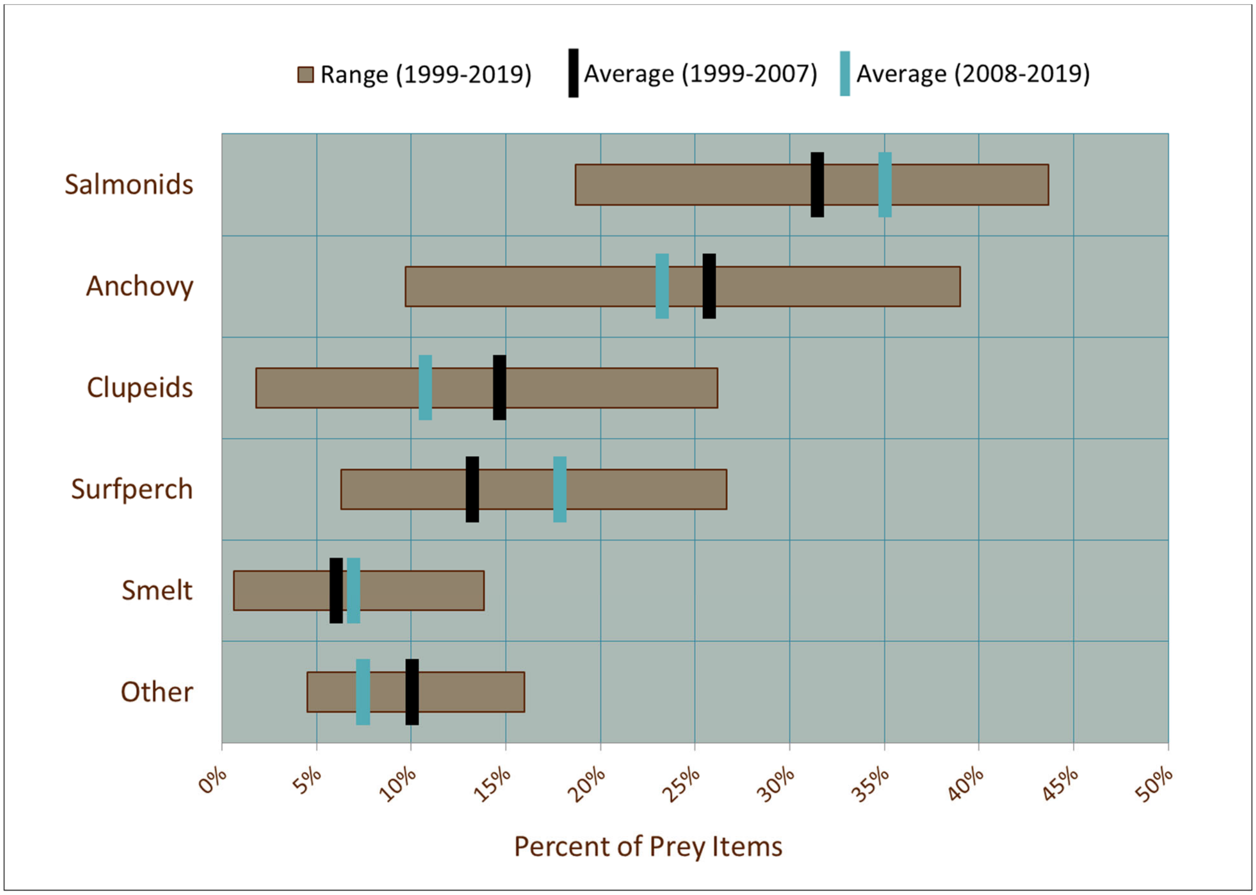

Figure 1.15. Percentage of different major prey types in the diet of Caspian terns nesting on East Sand Island in the Columbia River estuary. Figure shows the range of percentages for 1999–2019 and the median percent for the pre-management period (1999–2007) and the management period (2008–2019). Values are based on identified bill-load fish from the last week of April to the end of July. "Salmonids" = Oncorhynchus spp.; "Anchovy" = northern anchovy (Engraulis mordax); "Clupeids" = Pacific herring (Clupea pallasi), Pacific sardine (Sardinops sagax), and American shad (Alosa sapidissima); "Surfperch" = Embiotocidae; "Smelt" = Osmeridae. No diet composition data are available for 2016.

Within a breeding season there was considerable variation in the proportion of salmonid smolts in the diet of East Sand Island terns; the consistent pattern each year was for salmonids to comprise more than half of all prey items during the early part of the nesting season (late April and the first three weeks of May), while during most of the remainder of the nesting season (June and July) juvenile salmonids comprised less than 20% of prey items (*Figure 1.16*). Thus, most of the tern predation pressure on juvenile salmonids was during the out-migration period for spring migrant stocks (steelhead, coho salmon, and yearling Chinook salmon). During the chick-rearing portion of the tern nesting season, Caspian terns nesting on East Sand Island preyed primarily on northern anchovy and surfperch, while most of the juvenile salmonids consumed by terns were sub-yearling Chinook salmon.

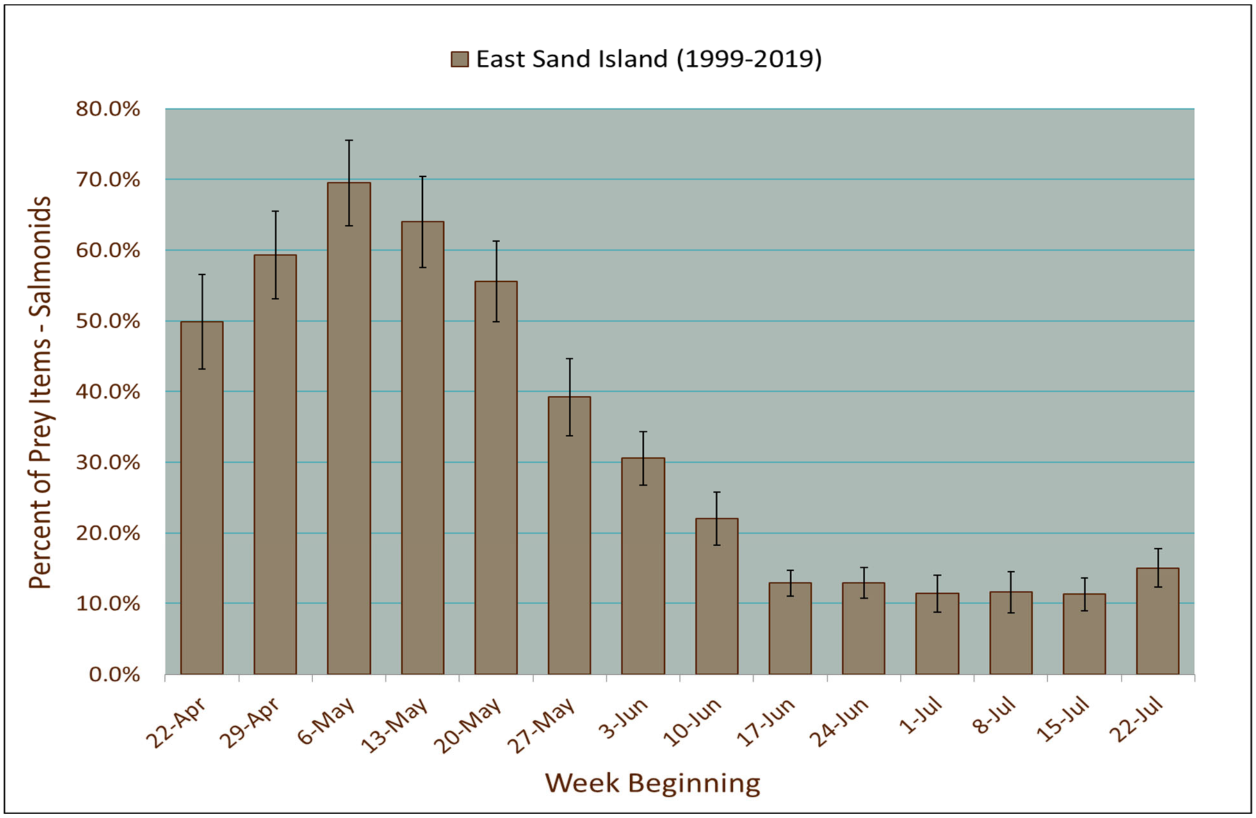

Figure 1.16. Average percentage of salmonids in the diet (proportion of prey items) of Caspian terns nesting at East Sand Island as a function of week, averaged across the 1999–2019 nesting seasons. Error bars represent ± 2SE. Diet data are not available for 2016.

Because juvenile salmonids are far more prevalent in the diet of Caspian terns in the Columbia River estuary during the early part of the nesting season, we investigated the variability in the proportion of salmonids in the tern diet averaged across May and the first half of June (*Figure 1.17*). Average percent salmonids in the diet of terns early in the nesting season was 86.2% at Rice Island (1998–2000), 42.1% at East Sand Island during the pre-management period (1999–2007), and 45.8% at East Sand Island during the management period (2008–2019). These data indicate that, even during the early part of the tern nesting season when juvenile salmonids were most prevalent in the diet, the proportion of salmonids in the diet of terns nesting on East Sand Island was approximately half what it was for terns nesting on Rice Island. Also, despite a slightly higher average proportion of salmonids in the diet of terns nesting on East Sand Island during the management period compared to the pre-management period, this difference was not significant.

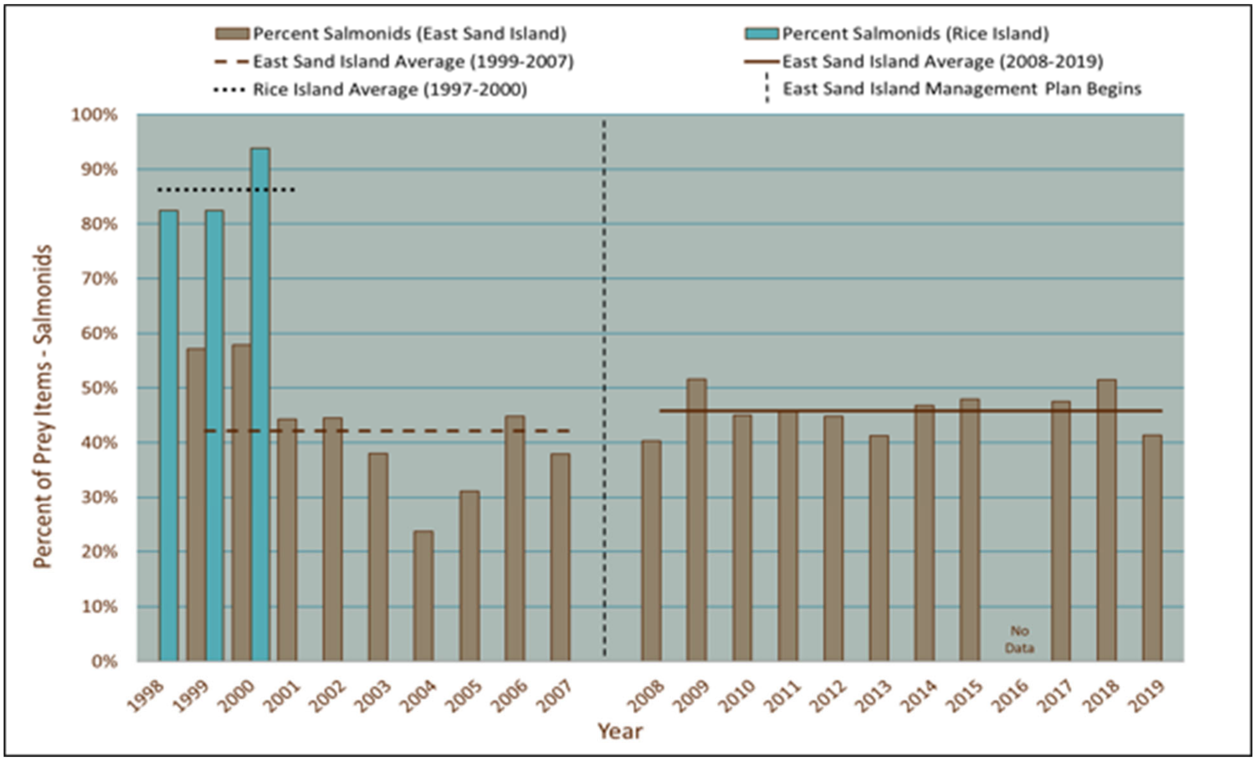

*Figure 1.17. Percent salmonids (*Oncorhynchus *spp.) of all identified prey items in bill-loads of Caspian terns nesting in the Columbia River estuary at either Rice Island (1997–2000), East Sand Island during the pre-management period (1999–2007), or East Sand Island during the management period (2008–2019) in the peak smolt out-migration period (May and the first half of June); no data on tern diet composition were available in 2016.*

1.4.2.3 Numbers of juvenile salmonids consumed – Bioenergetics calculations to estimate total annual consumption of juvenile salmonids by Caspian terns nesting at East Sand Island during the pre-management period (2000–2007) indicated that an average of ca. 5.0 million juvenile salmonids were consumed annually (*Figure 1.18*), and the trend was not significantly different from zero (multiple r^2 = 0.11, P = 0.425; *Figure 1.19*). This compares with an average of about 11.8 million juvenile salmonids consumed per year by Caspian terns nesting on Rice Island during 1998 and 1999 (*Figure 1.18*). During the management period at East Sand Island, there was a significant decline in the estimated annual number of salmonid smolts consumed by Caspian terns (multiple r^2 = 0.81, P = 0.002; *Figure 1.19*). By 2019, the last year of this study, ca. 2.9 million juvenile salmonids were consumed by terns nesting on East Sand Island (*Table 1.8*).

This compares with an estimated 6.7 million smolts consumed in 2008, the first year of implementation of the *Caspian Tern Management Plan* (*Figure 1.18*, *Table 1.8*).

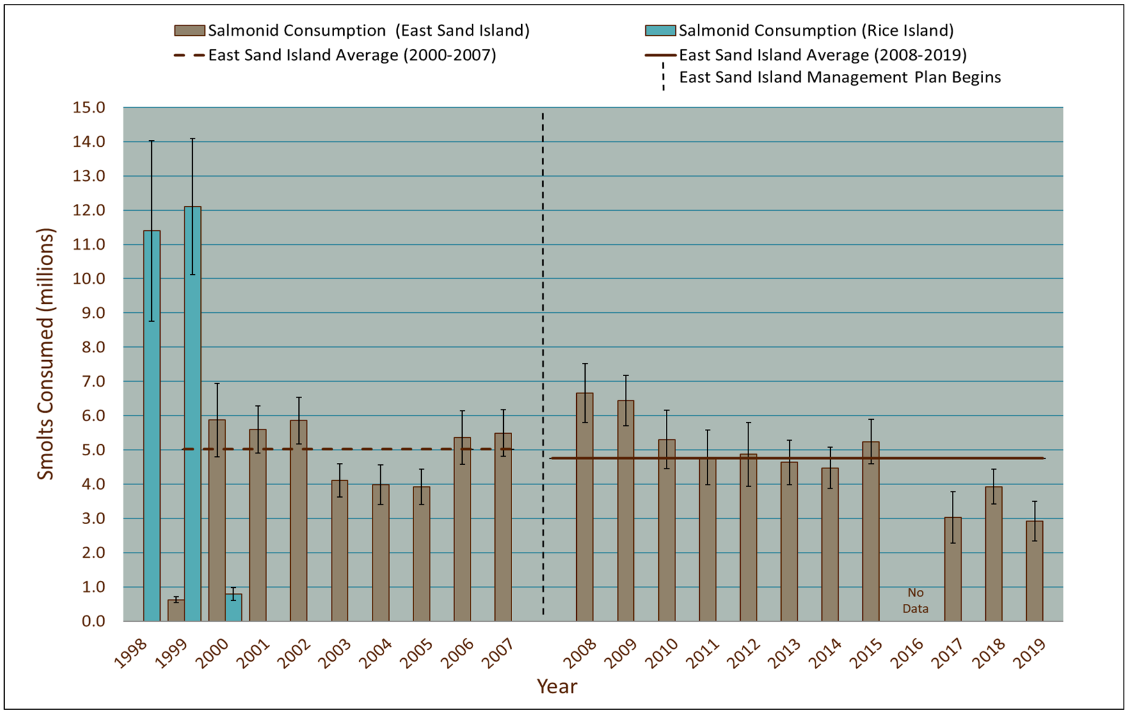

Figure 1.18. Estimated number of juvenile salmonids consumed by Caspian terns nesting at colonies on Rice Island (1998–2000) and on East Sand Island (1999–2019) in the Columbia River estuary based on bioenergetics modeling. Error bars represent 95% confidence intervals.

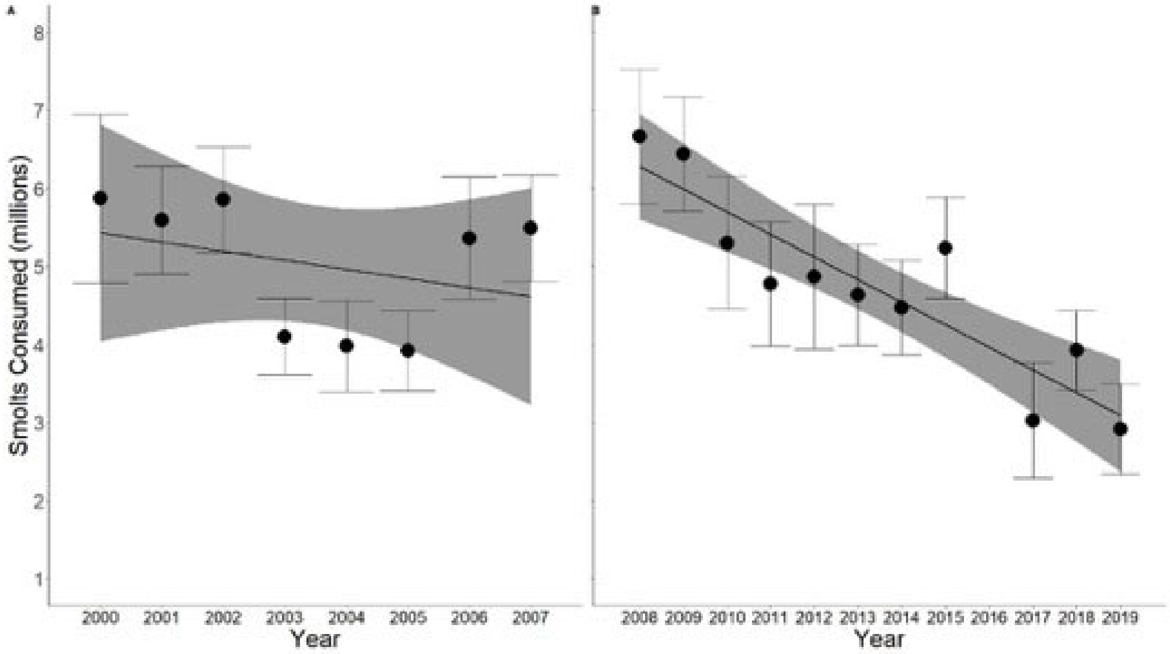

Figure 1.19. Estimated total number of juvenile salmonids consumed by Caspian terns nesting on East Sand Island in the Columbia River estuary as a function of year. Trend analysis is separated into the pre-management period (2000–2007) and the management period (2008–2019). Error bars depict the 95% confidence interval about each annual point estimate, and the 95% c.i. for the trendline based on least squares linear regression is depicted in gray shading. No diet composition data were available for 2016.

Table 1.8. Estimated number (95% c.i.) of juvenile salmonids consumed by Caspian terns nesting on East Sand Island in the Columbia River estuary during 1999–2019 based on bioenergetics modeling. Model estimates are in millions of smolts consumed.

	Salmonid Species/Age-class					
Year	Sub-yearling Chinook Salmon	Yearling Chinook Salmon	Coho Salmon	Sockeye Salmon	Steelhead Trout	**Total Smolts Consumed**
1999	0.21 (0.18-0.24)	0.11 (0.09-0.13)	0.22 (0.19-0.25)	0.004 (0.003-0.005)	0.08 (0.06-0.10)	**0.62 (0.54-0.71)**
2000	0.77 (0.61-0.92)	1.49 (1.18-1.80)	2.30 (1.90-2.70)	0.044 (0.034-0.053)	1.27 (1.01-1.53)	**5.87 (4.82-6.93)**
2001	0.87 (0.74-1.00)	1.37 (1.18-1.55)	2.22 (1.93-2.51)	0.048 (0.040-0.056)	1.08 (0.93-1.23)	**5.59 (4.90-6.28)**
2002	0.70 (0.58-0.81)	1.33 (1.16-1.51)	2.62 (2.31-2.94)	0.034 (0.028-0.039)	1.17 (1.02-1.32)	**5.86 (5.17-6.54)**
2003	0.53 (0.45-0.60)	1.46 (1.26-1.66)	1.26 (1.09-1.43)	0.049 (0.039-0.058)	0.81 (0.69-0.92)	**4.10 (3.62-4.59)**
2004	0.51 (0.40-0.61)	0.77 (0.65-0.89)	1.33 (1.10-1.55)	0.066 (0.047-0.085)	1.31 (1.11-1.51)	**3.98 (3.40-4.56)**
2005	0.36 (0.30-0.42)	1.15 (0.97-1.33)	1.18 (1.02-1.35)	0.021 (0.017-0.024)	1.20 (1.02-1.39)	**3.92 (3.41-4.43)**
2006	0.76 (0.64-0.89)	1.64 (1.37-1.91)	1.94 (1.65-2.23)	0.034 (0.028-0.040)	0.99 (0.83-1.15)	**5.36 (4.57-6.15)**
2007	0.79 (0.67-0.91)	1.14 (0.97-1.30)	2.61 (2.27-2.96)	0.033 (0.028-0.038)	0.91 (0.78-1.04)	**5.49 (4.81-6.17)**
2008	1.23 (1.04-1.43)	1.41 (1.21-1.60)	2.80 (2.43-3.17)	0.037 (0.031-0.044)	1.19 (1.02-1.35)	**6.66 (5.81-7.52)**
2009	1.05 (0.90-1.19)	1.47 (1.28-1.65)	2.66 (2.35-2.97)	0.032 (0.027-0.036)	1.23 (1.07-1.39)	**6.44 (5.71-7.17)**
2010	0.78 (0.63-0.92)	1.26 (1.04-1.47)	2.18 (1.82-2.54)	0.041 (0.034-0.049)	1.04 (0.87-1.22)	**5.30 (4.45-6.15)**
2011	1.27 (1.05-1.50)	0.77 (0.62-0.92)	1.39 (1.15-1.62)	0.028 (0.023-0.033)	1.32 (1.09-1.55)	**4.78 (3.99-5.57)**
2012	1.35 (1.08-1.62)	1.05 (0.83-1.27)	1.56 (1.25-1.88)	0.020 (0.016-0.024)	0.89 (0.71-1.07)	**4.87 (3.94-5.80)**
2013	1.11 (0.95-1.27)	0.88 (0.75-1.02)	1.73 (1.48-1.97)	0.024 (0.021-0.028)	0.88 (0.75-1.02)	**4.63 (3.99-5.28)**
2014	0.96 (0.78-1.14)	0.87 (0.71-1.03)	1.73 (1.46-1.99)	0.025 (0.005-0.045)	0.89 (0.75-1.04)	**4.48 (3.87-5.08)**
2015	1.69 (1.41-1.98)	0.84 (0.69-0.98)	1.84 (1.58-2.11)	0.023 (0.003-0.042)	0.84 (0.71-0.97)	**5.24 (4.59-5.89)**
*2016	--	--	--	--	--	**--**
2017	0.53 (0.37-0.68)	0.66 (0.48-0.84)	1.11 (0.82-1.39)	0.015 (0.002-0.028)	0.72 (0.53-0.91)	**3.03 (2.29-3.77)**
2018	1.02 (0.84-1.20)	0.73 (0.60-0.85)	1.42 (1.21-1.63)	0.019 (0.003-0.034)	0.74 (0.62-0.86)	**3.93 (3.42-4.44)**
2019	0.68 (0.52-0.84)	0.58 (0.44-0.72)	1.03 (0.81-1.26)	0.013 (0.002-0.025)	0.61 (0.47-0.76)	**2.92 (2.34-3.50)**

** No diet composition data available in 2016.*

Bioenergetics modeling was also used to estimate the numbers of smolts, separated by salmonid species/age-class, that were consumed by Caspian terns nesting at East Sand Island during 2000–2019 (*Table 1.8* above). Models indicated that in most years during the pre-management period coho salmon were the most frequently consumed species of salmonid by terns nesting at East Sand Island (1.2–2.6 million/year), but there was considerable inter-annual variation in the relative consumption of coho salmon, steelhead trout (0.8–1.3 million/year), yearling Chinook salmon (0.8–1.6 million/year), and sub-yearling Chinook salmon (0.4–0.9 million/year; *Figure 1.20*). Estimated annual consumption of sockeye salmon by terns nesting on East Sand Island was comparatively much lower in all pre-management years (21–66 thousand). During the management period (2008–2019) the estimated annual consumption of each species/age-class declined, with the notable exception of sub-yearling Chinook salmon; average estimated consumption of sub-yearling Chinook salmon during the pre-management period was 0.7 million/year, whereas during the management period the average was 1.1 million/year.

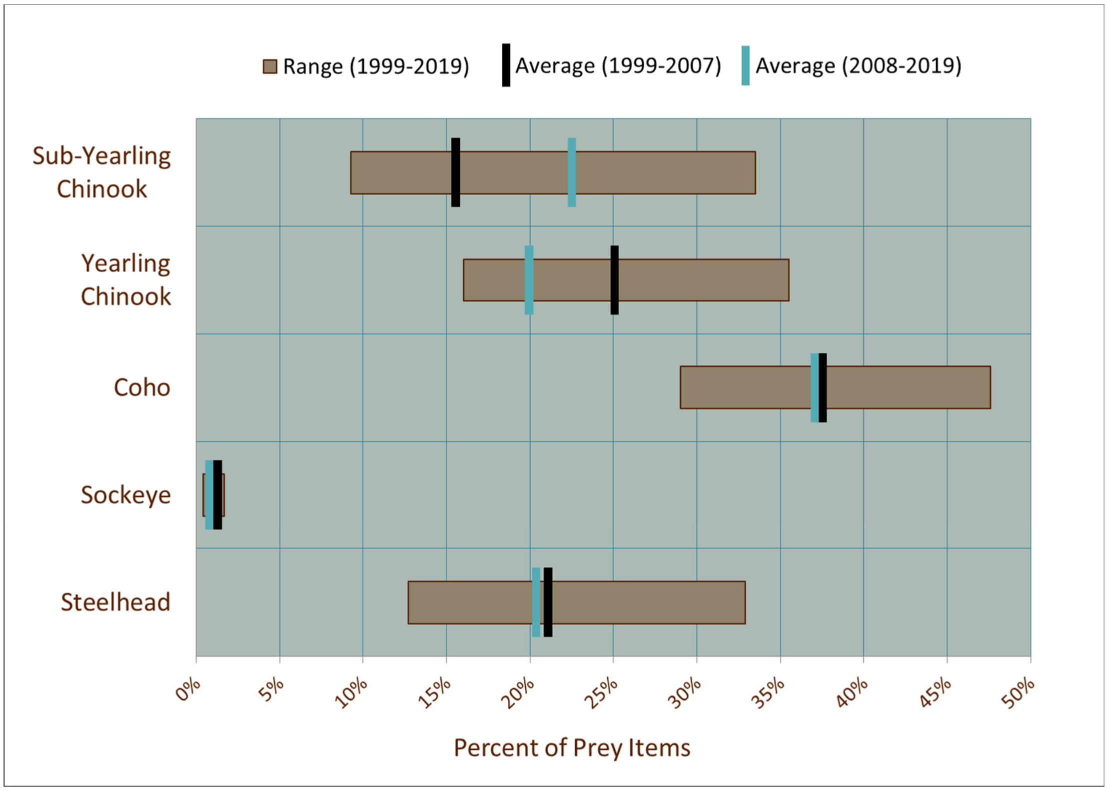

Figure 1.20. Annual proportions of five salmonid species/age-classes among juvenile salmonids consumed by Caspian terns nesting at East Sand Island in the Columbia River estuary. Figure shows the range of proportions for each salmonid species/age-class during 1999–2019 (exclusive of 2016), and the average proportions during the pre-management period (1999–2007) and during the management period (2008–2019).

1.4.2.4 Predation rates on juvenile salmonids by terns nesting on East Sand Island – Predation rate studies based on juvenile salmonid (smolt) PIT tag recoveries were first conducted on the Caspian tern colony on East Sand Island in 1999 (Ryan et al. 2001), with studies continuing each year through 2018 (Evans et al. 2019c). During 1999 to 2018, 531,058 PIT tags (detections of unique fish) were recovered on the East Sand Island Caspian tern colony, representing the largest number of fish PIT tags recovered on an individual colony in the CRB. Most of these tags were from anadromous juvenile salmonids (> 99%), but small numbers of tags from other fishes (Pacific lamprey [*Lampetra tridentate*] and northern pikeminnow [*Ptychocheilus oregonensis*]) have also be recovered on the East Sand Island tern colony (Evans et al. 2019c).

Herein we summarize predation rates by Caspian terns nesting at the East Sand Island colony on each ESA-listed salmonid population (i.e. ESU/DPS) where adequate data were available during 2000–2018. To evaluate the efficacy of the *Caspian Tern Management Plan* to reduce predation rates on juvenile salmonids, predation rates were compared between the pre-management period (2000–2007) and the management period (2008–2018), following implementation of management actions aimed at reducing the size of the tern colony on East Sand Island.

Upper Columbia River steelhead trout: Estimates of annual predation rates on Upper Columbia River (UCR) steelhead by East Sand Island Caspian terns were amongst the highest of any ESA-listed ESU/DPS evaluated, ranging from 6.5% (95% credible interval = 4.7–9.6%) to 23.4% (18.1–34.1%) of available PIT-tagged fish, depending on the year (*Figure 1.21*, see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) steelhead are part of the ESA-listed UCR population (NMFS 2019b) and comparisons of weekly and annual predation rates indicated that both rear-types were equally susceptible to predation by East Sand Island Caspian terns (*Figure 1.22*). Predation rates by East Sand Island Caspian terns on UCR steelhead declined coincident with management actions to reduce the size of the East Sand Island colony, with average annual predation rates decreasing from 17.2% (15.2–19.5%) during 2000–2007 to 11.0% (10.0–12.5%) during 2008–2018. Decreases were greatest during 2011–2018, which reflected the period of the greatest decline in the size of the Caspian tern colony on East Sand Island (*Figure 1.7* above).

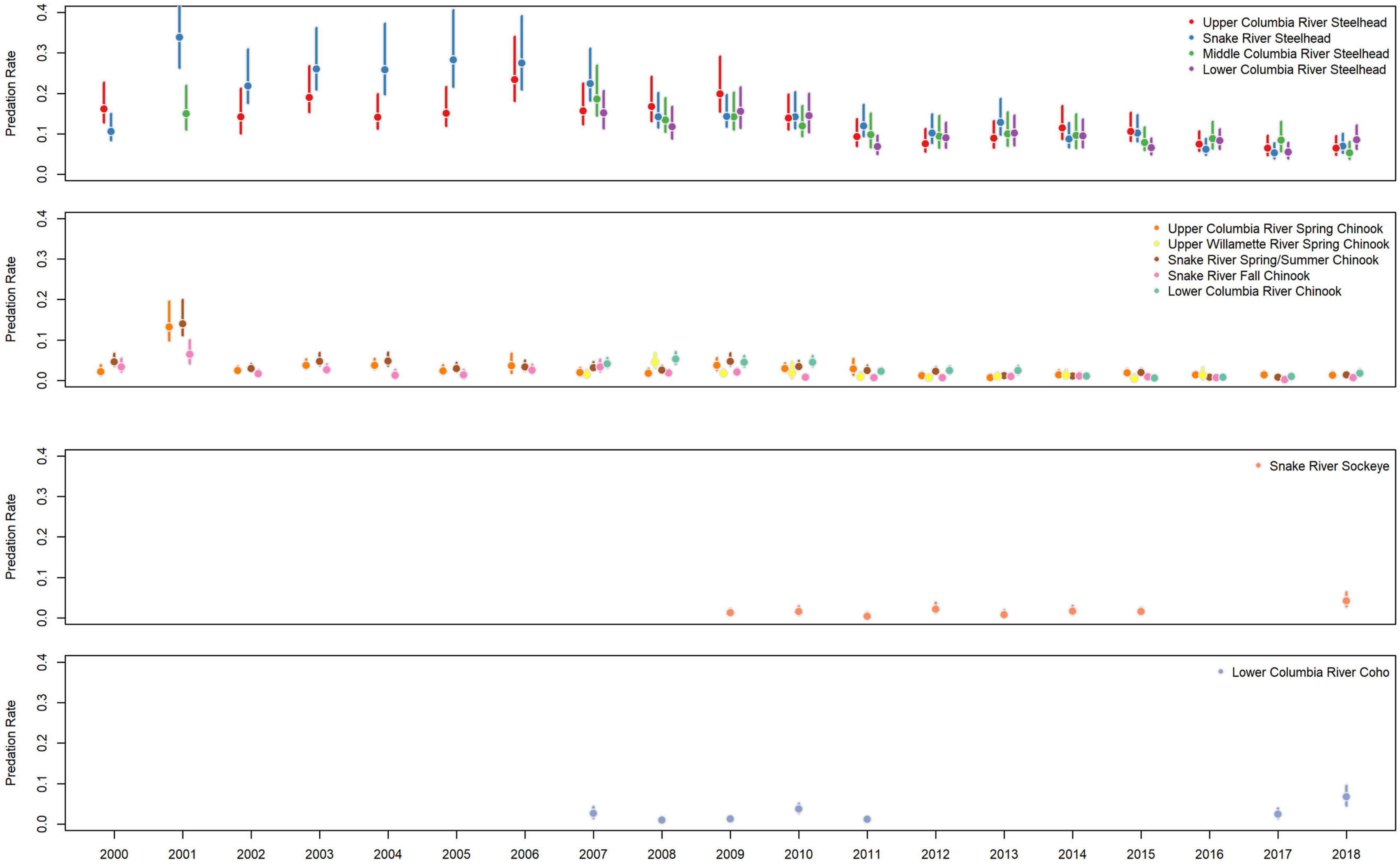

Figure 1.21. Annual predation rates on ESA-listed juvenile salmonids by Caspian terns nesting on East Sand Island in the Columbia River estuary during 2000–2018. Error bars denote 95% credible intervals. Results are those of Evans et al. (2019a) or are newly reported for Lower Columbia River ESUs/DPSs.

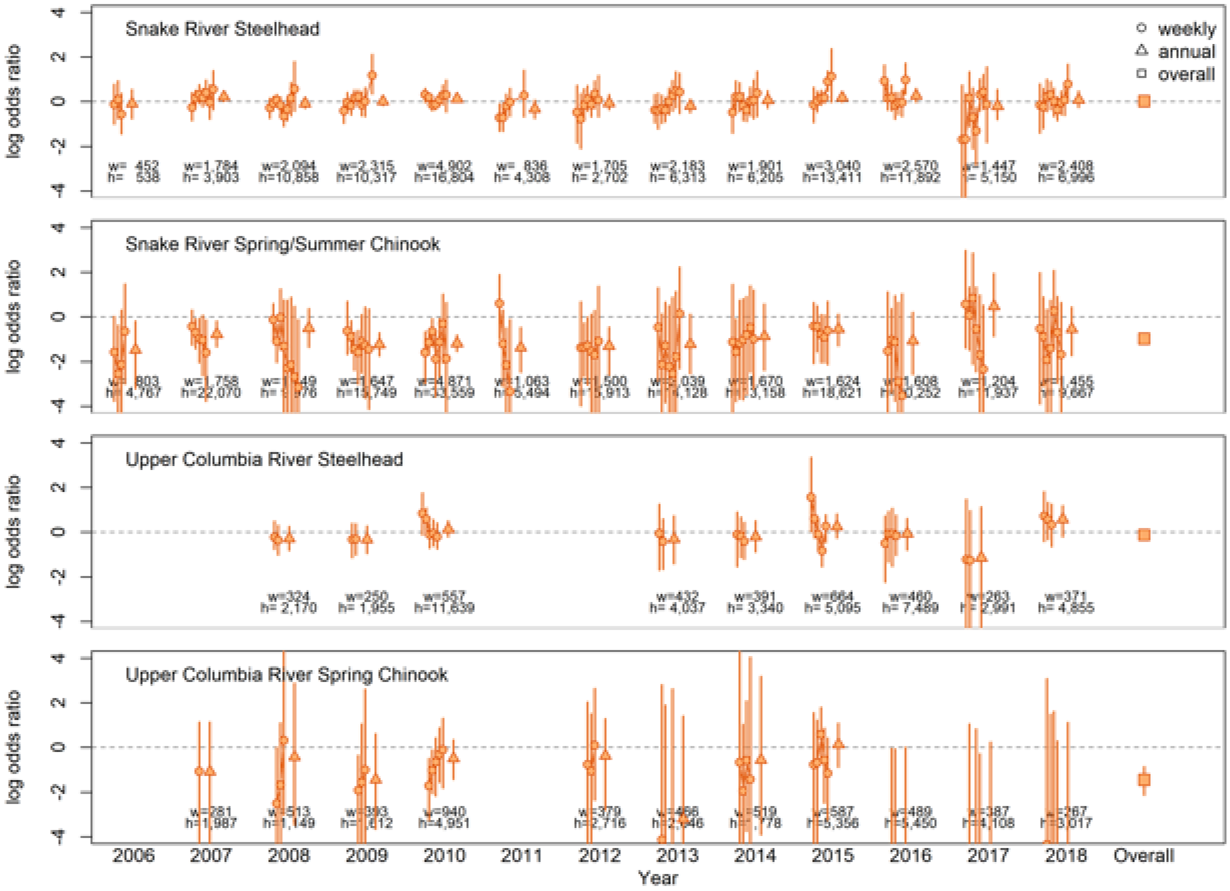

Figure 1.22. Relative susceptibility of hatchery versus wild salmonid smolts to predation by Caspian terns nesting on East Sand Island in the Columbia River estuary during 2006–2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for hatchery fish and values > 0 indicating greater predation odds for wild fish. Error bars represent 95% credible intervals; uncertainty ranges over-lapping 0 were associated with relative differences that were not considered statistically significant. Only years when > 500 PIT-tagged fish of each rearing type were available, and only weeks when > 50 PIT-tagged fish of each rearing type were available are included in the plot. Within each year, weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right. Figure from Evans et al. (2019a).

Upper Columbia River spring Chinook salmon: Estimates of annual predation rates on UCR spring Chinook salmon by East Sand Island Caspian terns were significantly lower than those on UCR steelhead, ranging from just 1.7% (1.0–2.9%) to 13.2% (9.9–19.5%), depending on the year (*Figure 1.21* above; see also *Appendix B, Table B2*). The estimate of 13.2% was from 2001, a year where predation rates on all ESUs/DPSs were significantly higher than those from all other years (*Figure 1.21* above). For example, predation rates on UCR spring Chinook salmon were less than 3.8% in all other years. Both wild and hatchery (from select hatcheries) UCR spring Chinook salmon are part of the ESA-listed population (NMFS 2019b), and Evans et al. (2016a, 2019c) found that hatchery UCR spring Chinook salmon were significantly more susceptible to predation by East Sand Island Caspian terns compared with their wild counterparts (*Figure 1.22* above). An analysis of predation rates based on the size (fork-length) of spring Chinook salmon smolts by Roby et al. (2017) indicated that East Sand Island Caspian terns disproportionately consumed larger-size Chinook salmon, and hatchery Chinook salmon smolts were, on average, larger than wild Chinook salmon smolts (see also *Chapter 10*). Predation rates on UCR spring Chinook by Caspian terns nesting on East Sand Island declined following implementation of the *Caspian Tern Management Plan*, with average annual predation rates decreasing from 4.3% (3.7–5.1%) during 2000–2007 to 1.9% (1.6–2.2%) during 2008–2018 (*Table 1.9*).

Table 1.9. Average annual predation rates (95% credible intervals) by Caspian terns nesting on East Sand Island prior to (Pre-Management) and during implementation of management actions (Management) to reduce colony size. Salmonid populations (ESUs/DPSs) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. Asterisks denote statistically credible differences between management periods. NA denotes that estimates were not available for that ESU/DPS during that time period.

Salmonid ESU/DPS	Pre-Management	Management
	2000-2007	2008-2018
Upper Columbia River Steelhead [1]	17.2% (15.2-19.5)	11.0% (10.0-12.5) *
Upper Columbia River Sp Chinook	4.3% (3.7-5.1)	1.9% (1.6-2.2) *
Snake River Steelhead	25.3% (22.7-28.3)	10.7% (9.8-12.0) *
Snake River Sp/Su Chinook	5.2% (4.6-6.0)	2.1% (1.9-2.4) *
Snake River Fall Chinook	2.9% (2.4-3.4)	1.0% (0.8-1.2) *
Snake River Sockeye [2]	NA	1.8% (1.4-2.2)
Middle Columbia River Steelhead [3]	17.1% (14.0-22.0)	10.1% (9.1-11.4) *
Upper Willamette River Steelhead	NA	NA
Upper Willamette River Sp Chinook [4]	1.4% (0.7-2.4)	1.7% (1.3-2.1)
Lower Columbia River Steelhead [5]	15.2% (11.7-20.7)	10.4% (9.4-11.4) *
Lower Columbia River Chinook [5]	4.1% (3.2-5.6)	2.5% (2.2-2.8) *
Lower Columbia River Coho [5]	2.6% (1.4-4.3)	3.1% (2.6-3.8)
Lower Columbia River Chum	NA	NA

[1] *Predation rate estimates were not available during 2001.*

[2] *Predation rate estimates were not available during 2016-2017.*

[3] *Predation rate estimates were not available during 2000-2006.*

[4] *Predation rate estimates were not available during 2000-2006 and in 2017.*

[5] *Predation rate estimates were not available during 2000-2006.*

Snake River steelhead trout: Estimates of annual predation rates on Snake River (SR) steelhead by East Sand Island Caspian terns were similar to those on UCR steelhead and were among the highest of all ESUs/DPSs evaluated, with predation rates ranging annually from 5.3% (3.9–7.7%) to 33.9% (26.3–49.1%) of available fish during 2000–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) steelhead are part of the ESA-listed SR steelhead population (NMFS 2019b) and analogous to UCR steelhead, studies comparing predation rates between rear-types found no evidence of a difference in relative susceptibility to predation by East Sand Island Caspian terns, with wild SR steelhead just as likely to be consumed as their hatchery counterparts (*Figure 1.22* above; see also *Chapter 10*). As a strategy to reduce mortality of juvenile salmonids during out-migration through the Federal Columbia River Power System (FCRPS), the USACE collects smolts at dams on the Snake River and transports them to release sites in the tailrace of Bonneville Dam, the furthest downstream dam that SR smolts encounter during out-migration (see *Appendix B, Map B1).* A comparison of predation rates on transported vs. in-river (naturally migrating) SR steelhead smolts by East Sand Island Caspian terns found no evidence of a difference in relative susceptibility, with both groups, on average, equally susceptible (*Figure 1.23*). Prior to implementation of the *Caspian Tern Management Plan*, the average annual predation rate by East Sand Island Caspian terns on SR steelhead was 25.3% (22.7–28.3%), indicating that over a quarter of all Snake River steelhead that survived out-migration through the FCRPS were consumed by Caspian terns nesting on East Sand Island (*Table 1.9* above). Predation rates significantly declined following implementation of the *Caspian Tern Management Plan*; average annual predation rate was 10.7% during 2008–2018 (*Table 1.9* above). Similar to UCR steelhead, the greatest declines in predation rates on SR steelhead occurred during 2011–2018, a period that coincided with the greatest decrease in the size of the Caspian tern colony on East Sand Island (*Figure 1.7* above).

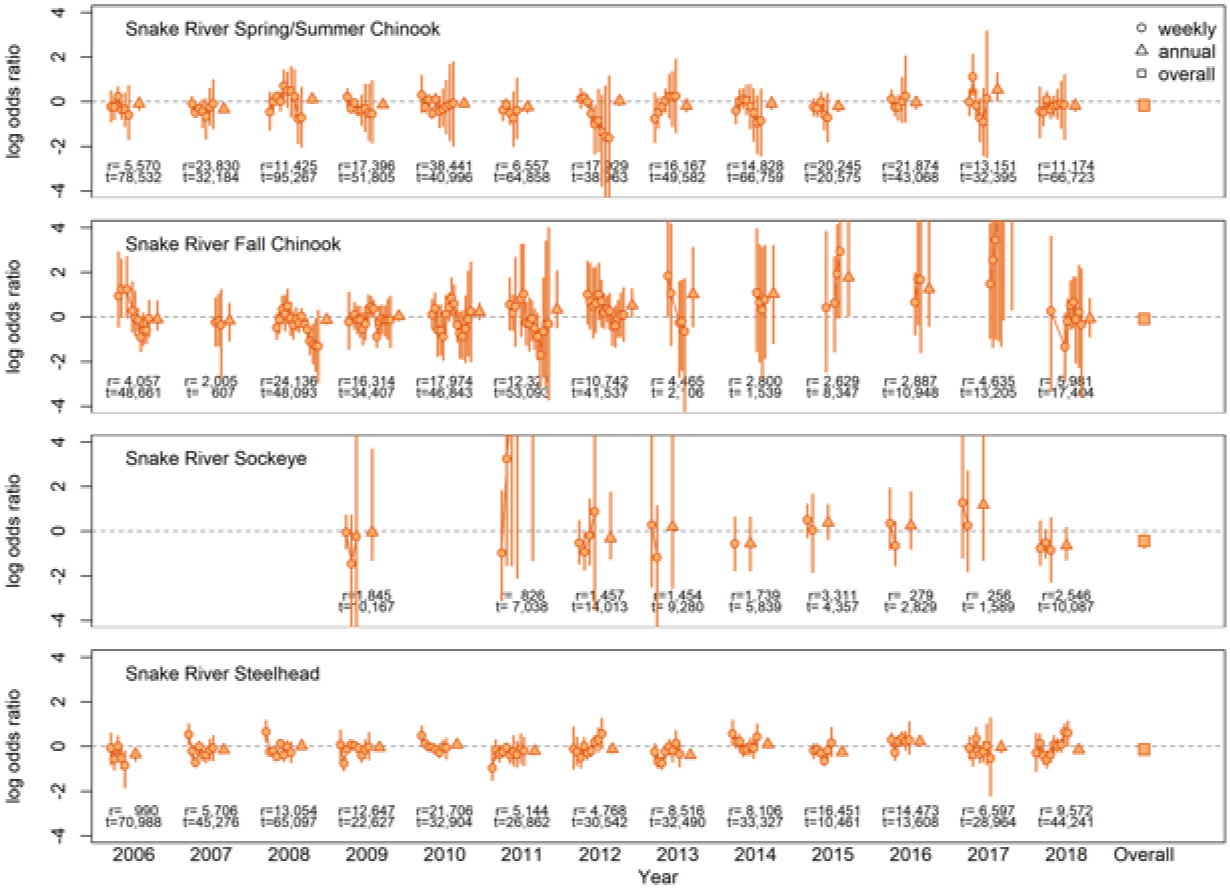

Figure 1.23. Relative susceptibility of in-river and transported smolts to predation by Caspian terns nesting on East Sand Island in the Columbia River estuary during 2006–2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for in-river fish and values > 0 indicating greater predation odds for transported fish. Error bars represent 95% credible intervals; uncertainty ranges over-lapping 0 were associated with relative differences that were not considered statistically significant. Only years when > 500 PIT-tagged fish of each migration history were available, and only weeks when > 50 PIT-tagged fish of each migration history were available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right. Figure from Evans et al. (2019a).

Lower Columbia River steelhead trout: Estimates of annual predation rates on Lower Columbia River (LCR) steelhead by East Sand Island Caspian terns were similar to those of the other steelhead DPSs, with annual predation rates ranging from 5.8% (4.0–8.3%) to 15.5% (11.3–22.0%) of available PIT-tagged fish during 2007–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) steelhead are part of the ESA-listed LCR population (NMFS 2019b) and, similar to other steelhead DPSs, hatchery and wild LCR steelhead were equally susceptible to predation by East Sand Island Caspian terns. Prior to implementation of the *Caspian Tern Management Plan*, annual predation rates on LCR steelhead were only available in 2007, when an estimated 15.2% (11.4–20.7%) of available tagged smolts were consumed (*Figure 1.21* above, see also *Appendix B, Table B3*). Estimates of annual predation rates on LCR steelhead were available in most years following implementation of the *Caspian Tern Management Plan*; the average annual predation rate during the management period (2008–2018) was 10.4% (9.4–11.4%; *Table 1.9* above). Given that estimates of annual predation rates for LCR steelhead prior to implementation of the *Caspian Tern Management Plan* were available for only one year, comparison of predation rates pre-management vs. during management is tenuous.

Snake River spring/summer Chinook salmon: Estimates of annual predation rates on SR spring/summer Chinook by East Sand Island Caspian terns ranged from 2.9% (2.3–4.1%) to 14.0% (11.1–20.0%) during 2000–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Similar to predation on UCR spring Chinook salmon, the estimate in 2001 (14.0%) was significantly higher than in all other years, with rates in other years less than 4.9% of available smolts (*Figure 1.21* above). Both wild and hatchery (from select stocks) SR spring/summer Chinook salmon are part of the ESA-listed population (NMFS 2019b). Analogous to results from UCR spring Chinook salmon, a comparison of tern predation rates on SR spring/summer Chinook salmon between wild and hatchery smolts found consistent differences in relative susceptibility depending on rear-type, with hatchery fish significantly more likely to be consumed compared to their wild counterparts (*Figure 1.22* above). These differences are likely attributable to the difference in relative size of hatchery vs. wild spring Chinook salmon smolts (Roby et al. 2017). Unlike other ESUs/DPSs of Snake River salmonids, there was some evidence that in-river migrating SR spring/summer Chinook salmon smolts were more likely, on average, to be consumed by East Sand Island Caspian terns than their transported counterparts (*Figure 1.23* above). These differences in relative susceptibility between in-river and transported smolts were small, however, and no consistent trend across weeks and years was identified (*Figure 1.23* above). Similar to predation on UCR spring Chinook salmon, predation rates on SR spring/summer Chinook salmon by Caspian terns nesting on East Sand Island declined following implementation of the *Caspian Tern Management Plan*, with average annual predation rates decreasing from 5.2% (4.6–6.0%) during 2000–2007 to 2.1% (1.9–2.4%) during 2008–2018 (*Table 1.9* above).

Snake River fall Chinook salmon: Estimates of annual predation rates on SR fall Chinook salmon by East Sand Island Caspian terns were similar to, but generally lower than, those on the other salmon ESUs evaluated, ranging from just 0.2% (0.1–0.5%) to 6.4% (4.2–10.0%) during 2000–2018 (*Figure 1.21* above; see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) SR fall Chinook salmon are part of the ESA-listed population (NMFS 2019b), but there were not enough tagged wild SR fall Chinook salmon available for comparison of tern predation rates on hatchery vs. wild smolts (Evans et al. 2016a, 2019c). There was no consistent evidence of a difference in relative susceptibility of in-river vs. transported SR fall Chinook salmon, with both groups, on average, equally susceptible to predation by East Sand Island Caspian terns (*Figure 1.23* above). Comparisons of predation rates prior to and following implementation of the *Caspian Tern Management Plan* indicated an overall reduction in average annual predation rates, with estimates of 2.9% (2.4–3.4%) during the pre-management period (2000–2007) and 1.0% (0.8–0.2%) during the management period (2008–2018).

Snake River sockeye salmon: Sample sizes of PIT-tagged SR sockeye salmon were inadequate for generating reliable estimates of predation rates in most years, with less than 500 tagged fish last detected passing Bonneville Dam during 2000–2007 and again during 2016–2017. For those years with adequate sample sizes, all of which were after implementation of the *Caspian Tern Management Plan*, average annual predation rates on SR sockeye salmon by East Sand Island Caspian terns ranged from 1.3% (0.7–2.2%) to 4.2% (2.9–6.4%; *Figure 1.21* above, see also *Appendix B, Table B2*); average annual predation rate during the management period was 1.8% (1.4–2.2%; *Table 1.9* above). Both wild and hatchery sockeye smolts are part of the SR population, but inadequate numbers of PIT-tagged wild smolts prevented comparisons of relative susceptibility based on rear-type (Evans et al. 2019c). Small sample sizes of PIT-tagged SR sockeye salmon, particularly in-river fish, also prevented comparisons of susceptibility based on out-migration history (in-river vs. transported) in most, but not all, study years. Limited data from those years with adequate samples sizes found no consistent trend in relative susceptibility of in-river and transported SR sockeye salmon across weekly and yearly comparisons (*Figure 1.23* above). Small sample sizes, however, resulted in imprecise (based on the width of 95% CRI) comparisons of predation rates on in-river vs. transported SR sockeye salmon.

Middle Columbia River steelhead trout: Estimates of annual predation rates on Middle Columbia River (MCR) steelhead smolts by East Sand Island Caspian terns were similar to those on UCR and SR steelhead, ranging from 5.3% (3.8–8.0%) to 18.7% (14.6–26.8%) during 2000–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Both wild and hatchery (from select stocks) steelhead are part of the ESA-listed MCR population (NMFS 2019b), but inadequate numbers of PIT-tagged wild fish prevented comparisons by rear-type (Evans et al. 2019a). There were also inadequate sample sizes of PIT-tagged Middle Columbia River (MCR) steelhead in most, but not all, years prior to implementation of the *Caspian Tern Management Plan* (*Figure 1.21* above). For those years with adequate samples size, comparisons indicated that average annual predation rates on MCR steelhead by East Sand Island Caspian terns declined following implementation of the *Caspian Tern Management Plan*, from 17.1% (14.0–22.0%) during the pre-management period to 10.1% (9.1–11.1%) during the management period (*Table 1.9* above).

Upper Willamette River Chinook salmon: Estimated annual predation rates on Upper Willamette River (UWR) Chinook salmon by East Sand Island Caspian terns were consistently the lowest of all salmon ESUs evaluated, ranging from just 0.4% (0.1–1.5%) to 4.4% (3.2–6.7%) during 2007–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) UWR Chinook salmon are part of the ESA-listed population (NMFS 2019b), but small samples sizes of wild UWR Chinook salmon prevented comparison of predation rates on wild vs. hatchery smolts (Evans et al. 2019a). Due to inadequate sample sizes of tagged UWR Chinook salmon, estimated annual predation rates prior to implementation of the *Caspian Tern Management Plan* were only available for 2007, when an estimated 1.4% (0.8–2.5%) of available smolts were consumed by East Sand Island Caspian terns (*Figure 1.21* above, see also *Appendix B, Table B2*). Adequate sample sizes of tagged UWR Chinook salmon smolts were available in most years following implementation of the *Caspian Tern*

Management Plan (2008–2018), when the average annual predation rate was 1.7% (1.3–2.1%; *Table 1.9* above). Because estimates of annual predation rates prior to implementation of the *Caspian Tern Management Plan* were available in just one year, comparison of tern predation rates on UWR Chinook salmon between the pre-management period and the management period is tenuous.

Lower Columbia River Chinook salmon: Estimated annual predation rates on LCR Chinook salmon by East Sand Island Caspian terns were similar to, but generally higher than, those on other Chinook salmon ESUs, ranging from 0.6% (0.4–1.0%) to 5.3% (4.1–6.9%) during 2007–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) smolts are part of the ESA-listed LCR Chinook salmon population (NMFS 2019b), but insufficient numbers of wild LCR Chinook salmon were PIT-tagged, preventing comparisons of tern predation rates on wild vs. hatchery smolts. For example, less than 500 PIT-tagged wild LCR Chinook salmon were available for estimates of annual predation rates, but several thousand (range = 11,734–33,031) PIT-tagged hatchery LCR Chinook salmon were available for estimation of annual predation rates (see also section *1.5.3 Adaptive Management Considerations*). The vast majority of LCR Chinook salmon smolts available to terns in the estuary are sub-yearling hatchery smolts (Sebring et al. 2013, Lyons et al. 2014, NMFS 2019a), so estimates of predation rates herein are based on the most abundant rear-type. Potential differences between hatchery and wild LCR Chinook salmon in susceptibility to Caspian tern predation in the Columbia River estuary are currently unknown.

Estimates of annual predation rates on LCR Chinook salmon were not available for years prior to implementation of the *Caspian Tern Management Plan*, with the exception of 2007, when Caspian terns consumed an estimated 4.1% (3.2–5.6%) of available LCR Chinook salmon smolts (*Figure 1.21* above, see also *Appendix B, Table B3*). Estimated annual predation rates were available for all years following implementation of the *Caspian Tern Management Plan* (2008–2018) and were highly variable depending on the year, ranging from just 0.6% (0.4–1.0%) to 5.3% (4.1–6.9%; *Table 1.9* above). Decreases in predation rates were coincident with decreases in the size of the East Sand Island Caspian tern colony, with the lowest estimates all occurring during 2014–2018, the period when colony sizes were lowest during the study period (*Figure 1.7* above).

Lower Columbia River coho salmon: Estimated annual predation rates on LCR coho salmon by East Sand Island Caspian terns were similar to those of other salmon ESUs, ranging from 0.9% (0.6–1.5%) to 6.7% (4.5–9.7%) during 2007–2018 (*Figure 1.21* above, see also *Appendix B, Table B2*). Both wild and hatchery (from select hatcheries) smolts are part of the ESA-listed LCR coho population (NMFS 2019b), but analogous to LCR Chinook salmon, very few wild LCR coho salmon smolts were PIT-tagged, preventing comparisons of predation rates on hatchery vs. wild smolts. Numerically, hatchery LCR coho salmon smolts are more abundant than their wild counterparts (Lyons et al. 2014; NMFS 2019a). Unlike LCR Chinook salmon and LCR steelhead, however, the vast majority of PIT-tagged LCR coho salmon were released into tributaries that enter the Columbia River downstream of Bonneville Dam, preventing comparisons of predation rates by release location (upstream vs. downstream of Bonneville Dam). Estimates of tern predation rates on LCR coho salmon were not available in most years prior to implementation of the *Caspian Tern Management Plan*, with the exception of 2007, when an estimated 2.6% (1.4–4.3%) of available smolts were consumed by Caspian terns in the estuary (*Figure 1.21* above, see also *Appendix B, Table B3*). Estimates of tern predation rates were also lacking in several years following implementation of the *Caspian Tern Management Plan*, with data available in only five of the 11 years during the management period (2008–2018). Average annual predation rate on LCR coho salmon by East Sand Island Caspian terns was 3.1% (2.6–3.8%) during the management period (*Table 1.9* above). Since estimates of annual predation rates were available in just one year during the pre-management period and that estimates were only available for less than half the years during the management period, comparison of predation rates between the pre-management period vs. the management period is tenuous.

Other ESA-listed salmonid populations: Two other ESA-listed salmonid populations, UWR steelhead and Columbia River chum salmon (*O. keta*), also migrate through the Columbia River estuary, but estimates of predation rates were not available due to a lack of PIT-tagging studies in these two populations. PIT tags implanted in wild UWR steelhead have been recovered on the East Sand Island Caspian tern colony, confirming that smolts from this population are susceptible to tern predation to some degree. UWR steelhead out-migrate as yearlings in the spring and given the similarities in predation rates observed in all other steelhead DPSs (SR, UCR, MCR), predation rates on UWR steelhead by East Sand Island Caspian terns may be similar to those on other steelhead populations. Chum salmon fry out-migrate as sub-yearlings shortly after emergence in late winter to early spring (NMFS 2019a) and although they may reside and rear in the estuary before entering the open ocean (Groot and Margolis 1991), most fish are too small to PIT tag (< 80 mm fork length: PTAGIS 2019). Based on genetic analyses of Caspian tern prey (see *above*), there has been no confirmation of chum salmon in the diet of Caspian terns nesting on East Sand Island, suggesting that tern predation rates on ESA-listed chum salmon in the estuary were extremely low (Lyons 2010).

Predation rates as a function of East Sand Island colony size: The over-riding assumption behind management actions described in the *Caspian Tern Management Plan* is that the number of Caspian tern breeding pairs nesting at the colony on East Sand Island is directly related to tern predation rates on juvenile salmonids in the Columbia River estuary, whereby reductions in colony size are proportional to reductions in predation rates (USFWS 2005; NMFS 2008). An investigation of the relationship between estimates of peak (maximum) annual tern colony size and estimates of annual predation rates on steelhead DPSs supports this assumption, with reductions in colony size generally commensurate with reductions in predation rates (*Figure 1.24*). Trends were particularly evident when large changes in colony size occurred (i.e. several thousand breeding pairs: *Figure 1.24*). Using data from all study years (i.e. the full range of colony sizes and associated predation rates), plots of predation rates as a function of colony size indicate a linear relationship between peak colony size and annual predation rates for all steelhead DPSs evaluated ($p < 0.01$ in all comparisons; Evans et al. 2019a; *Figure 1.24*). Collectively, these results support the conclusion that, if given sufficient time and a large enough reduction in the size of Caspian tern colony on East Sand Island, significant reductions in tern predation rates on steelhead smolts in the Columbia River estuary would occur.

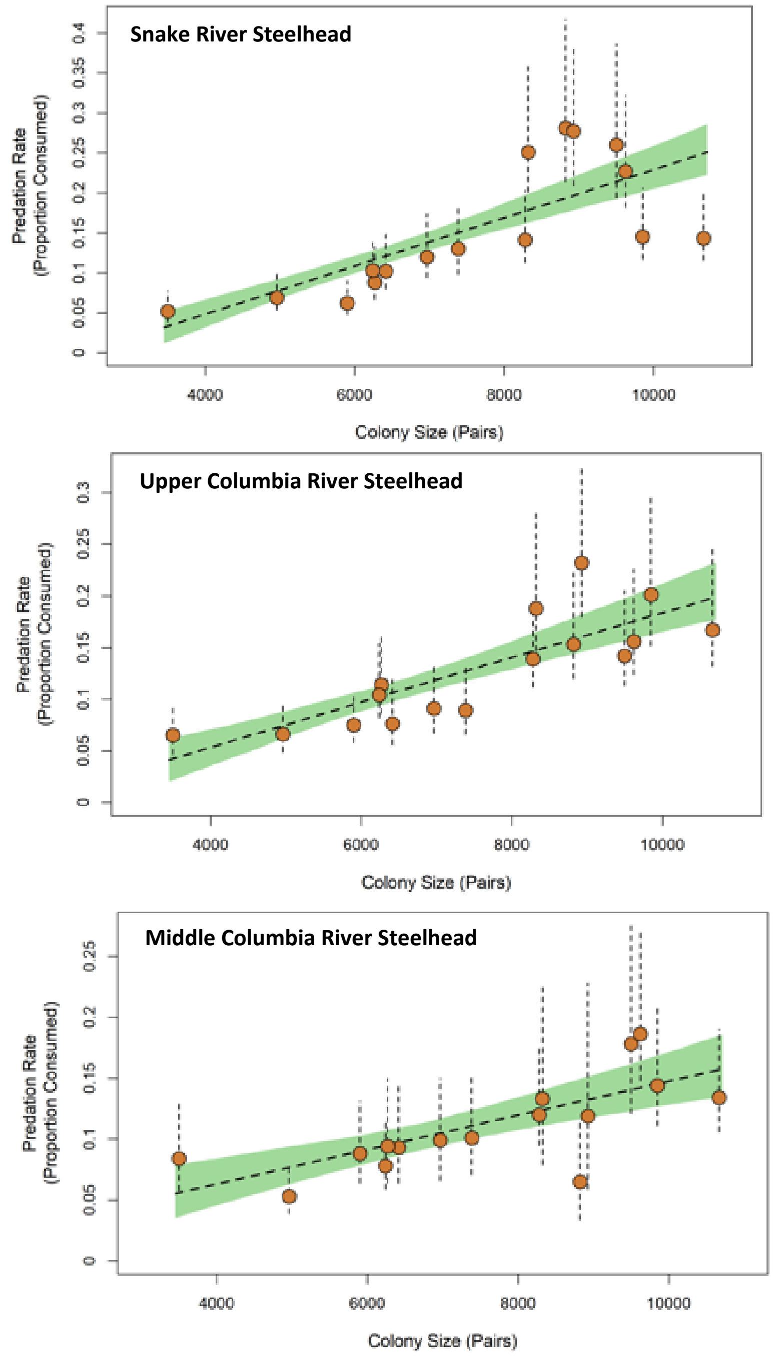

Figure 1.24. Estimated annual predation rates on PIT-tagged Snake River, Upper Columbia River, and Middle Columbia River steelhead smolts by East Sand Island Caspian terns as a function of the peak size of the tern colony on East Sand Island in the Columbia River estuary during 2000–2018 (for years with adequate sample sizes). Error bars about predation rates represented the 95% credible interval. The dashed lines represent the best fit least squares linear regression between predation rate and colony size. Results are from Evans et al. (2019a).

Although results provide evidence that predation rates, on average, can be reduced by reducing the number of Caspian terns nesting on East Sand Island, there was still considerable residual variation in predation rates. Significantly different predation rates on steelhead DPSs were observed at similar tern colony sizes (*Figure 1.24* above), suggesting that colony size alone was not the only factor causing variation in predation rates during the study period (Lyons 2010, Evans et al. 2016a). For example, estimates of peak colony size alone explained 52.0% (25.1–72.0%) and 70.6% (38.4–94.4%) of variation in annual predation rates on SR and UCR steelhead DPSs, respectively. *Appendix C* provides results of a new analysis aimed at better predicting predation rates on steelhead smolts by Caspian terns nesting on East Sand Island using both estimates of colony size and other factors or covariates that are known to influence tern predation rates.

Per capita (per breeding pair) estimates of ESU/DPS-specific predation rates by East Sand Island Caspian terns are reported in *Table 1.10*. Per capita predation rates provide information on the foraging ecology of colonies and have also been used to help prioritize which colonies to potentially manage (Roby et al. 2002, Evans et al. 2012, USACE 2015, Evans et al. 2019a). Estimates of per capita predation rates provide further evidence that steelhead trout were more susceptible to predation from East Sand Island Caspian terns compared with other salmonid species. There was no evidence that per capita predation rates varied significantly by management period, with similar, albeit variable, rates observed during both the pre-management and management periods. Relative comparisons of per capita predation rates among different predator taxa (Caspian terns, double-crested cormorants, California and ring-billed gulls) and different nesting colonies in the Columbia River basin, including the East Sand Island Caspian tern colony, are presented in *Chapter 10.*

Table 1.10. Average annual per capita (per nesting pair) predation rates (95% credible intervals) by Caspian terns nesting on East Sand Island in the Columbia River estuary during 2000–2018. Salmonid populations (ESUs/DPSs) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. NA denotes that estimates for that ESU/DPS were not available.

Salmonid ESU/DPS	Per Capita Predation Rates
Upper Columbia River Steelhead [1]	0.0018% (0.0016-0.0020)
Upper Columbia River Sp Chinook	0.0003% (0.0003-0.0004)
Snake River Steelhead	0.0021% (0.0020-0.0023)
Snake River Sp/Su Chinook	0.0003% (0.0003-0.0004)
Snake River Fall Chinook	0.0002% (0.0001-0.0002)
Snake River Sockeye [2]	0.0004% (0.0003-0.0006)
Middle Columbia River Steelhead [3]	0.0015% (0.0014-0.0017)
Upper Willamette River Steelhead	NA
Upper Willamette River Sp Chinook [4]	0.0002% (0.0001-0.0003)
Lower Columbia River Steelhead [5]	0.0018% (0.0016-0.0020)
Lower Columbia River Chinook [5]	0.0004% (0.0003-0.0004)
Lower Columbia River Coho [5]	0.0008% (0.0006-0.0009)
Lower Columbia River Chum	NA

[1] *Predation rate estimates were not available during 2001.*

[2] *Predation rate estimates were not available during 2016-2017.*

[3] *Predation rate estimates were not available during 2000-2006.*

[4] *Predation rate estimates were not available during 2000-2006 and in 2017.*

[5] *Predation rate estimates were not available during 2000-2006.*

1.4.2.5 Predation rates by terns nesting on Rice Island – Studies of avian predation rates based on recoveries of smolt PIT tags on-colony were first conducted at the Rice Island Caspian tern colony in 1996, the first study of its kind in the Columbia River basin (Collis et al. 2001). Standardized, corrected estimates of predation rates (i.e. those corrected for PIT tag deposition and detection probabilities) by Rice Island Caspian terns were available during 1998–2000 and again in 2017. Predation rates measured during 1998–2000 were associated with the management plan to move the Rice Island tern colony to East Sand Island. Predation rates on juvenile salmonids by Rice Island Caspian terns during 1998–2000 varied significantly by salmonid ESU/DPS and year (se *Appendix B, Table B2*). Similar to predation rates by Caspian terns nesting on East Sand Island, predation rates by terns nesting on Rice Island were significantly higher on steelhead DPSs compared with salmon ESUs. In 1998, prior to the shift of Caspian terns from the Rice Island colony to the restored colony site on East Sand Island and when about 8,000 tern pairs nested on Rice Island (Collis et al. 2000), predation rates on SR and UCR steelhead were estimated at 21.2% (16.0–42.0%) and 23.2% (15.8–42.2%), respectively (see also *Appendix B, Table B2*). In comparison, predation rates on all ESUs of salmon evaluated were less than 3.4% in 1998. By 2000, when only 588 pairs of Caspian terns remained at the Rice Island colony, tern predation rates on all ESUs/DPSs evaluated were significantly lower, with predation rates on steelhead DPSs and salmon ESUs reduced to less than 3.1% and less than 0.5%, respectively (see also *Appendix B, Table B2*).

Since implementation of the *Caspian Tern Management Plan*, active and passive nest dissuasion techniques have been successful at preventing formation of a sustained Caspian tern breeding colony on Rice Island. Likely linked to management actions to reduce the size of the East Sand Island tern colony, however, several hundred to several thousand Caspian terns have attempted to nest on Rice Island since 2011. In 2017, upwards of 3,000 Caspian terns were observed digging nest scrapes, laying eggs, and incubating eggs on Rice Island (Harper and Collis 2018). Smolt PIT tags were recovered from the area on Rice Island used by nesting Caspian terns in 2017, providing a single year of data during the management period. Results indicated that ESU/DPS-specific predation rates on juvenile salmonids ranged from 0.2% (0.2–0.7%) for UCR spring Chinook salmon to 3.0% (2.3–4.4%) for SR steelhead (see *Appendix B, Table B2*). Analogous to predation rates by Rice Island terns during 1998–2000, predation rates were significantly higher for steelhead DPSs (ranging from 2.4% to 3.0% annually, depending on the DPS) compared with salmon ESUs (ranging from just 0.2% to 0.4% annually, depending on the ESU). Estimates of predation rates by Rice Island Caspian terns in 2017, however, are not directly comparable to those by terns nesting on East Sand Island in 2017 because terns were not nesting at the incipient Rice Island colony site throughout the entire smolt out-migration period of April through August and because terns on Rice Island were hazed throughout much of the 2017 breeding season. Regardless, PIT tag recoveries from Rice Island in 2017 provide evidence that Caspian terns that prospected for nest sites on islands in the upper estuary consumed substantial numbers of steelhead smolts. Results also indicate that the overall impact of Caspian tern predation on survival of juvenile steelhead in the Columbia River estuary was greater than that of Caspian terns nesting on East Sand Island alone.

An investigation of the relative susceptibility of PIT-tagged juvenile salmonids to predation by Caspian terns nesting on Rice Island vs. East Sand Island during 1999–2000, years when terns nested successfully on both islands, is presented in *Chapter 5.*

1.5 Discussion

1.5.1 Management Action Effectiveness

Incremental reduction in the area of Caspian tern nesting habitat provided on East Sand Island was initiated in 2008 with the goal of reducing predation rates on ESA-listed salmonid smolts in the Columbia River estuary. By 2015, seven years later, the area of tern nesting habitat on East Sand Island had been reduced to 1.0 acres (0.4 ha), the minimum area of nesting habitat allowable under the January 2005 Final Environmental Impact Statement (FEIS) and November 2006 Records of Decision (RODs) for *Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (USFWS 2005, USFWS 2006, USACE 2006), and the

Final Environmental Assessment and Finding of No Significant Impact, released in April 2015 (USACE 2015a, USACE 2015b). The managed reduction in tern nesting habitat under the *Caspian Tern Management Plan* was intended to reduce the size of the Caspian tern colony on East Sand Island to within the range of 3,125 to 4,375 breeding pairs, or 34–48% of average colony size during the pre-management period (9,080 breeding pairs). The rate at which nesting habitat for Caspian terns could be reduced on East Sand Island was dictated under the terms of the *Caspian Tern Management Plan* by the rate at which alternative nesting habitat for Caspian terns was provided outside the Columbia River estuary. For every acre of tern nesting habitat created elsewhere the area of nesting habitat on East Sand Island could be reduced by half an acre. This approach was adopted based on the assumptions that (1) the Pacific Flyway population of Caspian terns was primarily limited by availability of suitable nesting habitat and (2) not all alternative colony sites created for Caspian terns would be occupied in all years (USFWS 2005; see *Chapter 3* for the results of efforts to provide alternative tern nesting habitat outside the Columbia River estuary).

1.5.1.1 Reduction of nesting habitat & colony size – Despite the reduction in tern nesting habitat on East Sand Island to 1.0 acres (0.4 ha) by 2015, the size of the tern colony did not reach the target colony size (3,125–4,375 breeding pairs) until 2017, when colony size was estimated at 3,500 breeding pairs. During the 2015 nesting season, the size of the Caspian tern colony on East Sand Island was approximately 6,240 breeding pairs, or about 43% larger than the upper limit of the range of target colony sizes. During the 2016 nesting season tern colony size was about 5,920 breeding pairs (*Table 1.2* above), or about 35% larger than the upper limit. The failure to achieve the target colony size despite reducing the area of tern nesting habitat to the target (1 acre or 0.4 ha) was due to two factors. First, the area of available tern nesting habitat on East Sand Island far exceeded the area used by nesting terns when the *Caspian Tern Management Plan* was implemented starting in 2008. The area of designated tern nesting habitat was reduced by about 2.5 acres (42%) before terns occupied all of the available nesting habitat that was prepared for them on East Sand Island; it was not until the area of nesting habitat had been reduced from the 6 acres (2.4 ha) available in 2007 to 3.5 acres (1.4 ha) in 2009 that there was no unoccupied nesting habitat on the tern colony (*Figure 1.3* above). Second, the density of tern nests on the East Sand Island colony increased starting in 2011 and by 2016 nesting density had nearly doubled to 1.36 nests/m^2 (*Table 1.2* above). This level of increased nesting density was unexpected, exceeded that measured at any other colony in the Pacific Flyway (Antolos et al. 2006, Bailey 2018), and provided a strong testimonial to the motivation of Caspian terns to nest on East Sand Island.

In 2017, the first year that the size of the East Sand Island colony of Caspian terns was within the range of targeted colony sizes stipulated in the *Caspian Tern Management Plan*, tern nesting density relaxed to 0.97 nests/m^2, the lowest nest density recorded at the colony since 2011. The lower colony size and nesting density in 2017 coincided, however, with the complete failure of the colony to raise any young in that year. The only other year when the Caspian tern colony on East Sand Island failed to raise any young was in 2011. In 2018, following the poor nesting season of 2017, the size of the tern colony rebounded to about 4,960 breeding pairs, 13% larger than the upper limit of the range of target colony sizes, and nesting density increased to 1.23 nests/m^2. Nevertheless, in 2019 tern colony size declined to 3,860 breeding pairs and nesting density declined to 1.11 nests/m^2, suggesting that colony size at East Sand Island might settle into the range stipulated in the *Caspian Tern Management Plan*.

Factors other than the area of suitable nesting habitat provided by managers clearly influence the size of the Caspian tern colony on East Sand Island. The residuals about the curve shown in *Figure 1.13* above, especially in years when colony area was 1.0 acres (0.4 ha), illustrate the magnitude of variation in colony size that is unexplained by the area of available nesting habitat. The factors other than area of nesting habitat that influence annual colony size are likely similar to those that influence annual nesting success (see *above*), and include river discharge, availability of marine forage fish (especially early in the nesting season), bald eagle disturbance rates, and nest predation rates and kleptoparasitism rates by glaucous-winged/western gulls (Collar et al. 2017, Bailey 2018).

1.5.1.2 Consumption of juvenile salmonids – Estimated annual smolt consumption by Caspian terns nesting on East Sand Island, based on bioenergetics modeling, averaged only slightly less during the management period (2008–2019) than during the pre-management period (2000–2007; *Figure 1.18* above). Nevertheless, the trend in estimated annual smolt consumption was significantly and strongly negative during the management period (*Figure 1.19* above). By the end of the study period (2019) estimated smolt consumption had declined to about 2.9 million smolts/year, compared with average annual smolt consumption during the pre-management period of 5.0 million smolts/year. Although this represents a ca. 42% decline in annual smolt consumption by Caspian terns nesting at the East Sand Island colony, it is considerably less than the ca. 57% reduction in annual smolt consumption that was anticipated based on the managed reduction in tern colony size by about 57% (from the pre-management average of 9,080 breeding pairs to the 2019 colony size of 3,860 pairs). Based on the range of target colony sizes specified in the *Caspian Tern Management Plan* (3,125–4,375 breeding pairs), the anticipated benefits from *Plan* implementation were to reduce annual smolt consumption by 52–66%.

The less than anticipated reduction in annual smolt consumption by Caspian terns nesting at the East Sand Island colony was caused by several factors. First, the composition of the salmonid portion of the diet of Caspian terns nesting on East Sand Island changed from the pre-management to the management period (*Table 1.9* above, *Figure 1.20* above). During the management period the proportion of sub-yearling Chinook salmon in the salmonid portion of the diet increased substantially, while the proportion of yearling Chinook salmon declined. This shift was largely driven by the delay in nesting chronology of terns during the management period (*Figure 1.4* above), which increased the overlap of the tern nesting season with the run timing of sub-yearling Chinook salmon and decreased the overlap with the run timing of yearling Chinook salmon. Sub-yearling Chinook salmon smolts are considerably smaller than yearling Chinook salmon smolts, necessitating consumption of greater numbers of sub-yearling smolts to meet tern energy demands.

Second, poor ocean conditions associated with the marine heat wave (“the Blob”) in the northern California Current (Peterson et al. 2017b, 2019) apparently reduced the availability of schooling marine forage fishes, especially northern anchovies, for terns foraging in the Columbia River estuary during 2015–2019. The impact of the marine heat wave was manifest in unexpectedly high smolt consumption by terns during the 2015 nesting season (*Figure 1.18* above), the complete failure of the tern colony to produce any young in 2017, and the sparsity of northern anchovies in the tern diet during the 2017–2019 nesting seasons (*Table 1.7* above). Although data on the size and lipid content of marine forage fish consumed by Caspian terns nesting at East Sand Island were not available during the marine heat wave, it is likely that the quality (energy content) of marine forage fish in the Columbia River estuary also was below average. This would shift the tern diet generally from marine forage fish to juvenile salmonids, especially during the chick-rearing period for terns, when marine forage fish typically dominate the tern diet (*Figure 1.16* above). This factor was completely unrelated to implementation of the *Caspian Tern Management Plan* and could not have been anticipated.

Finally, although there was not a significant increase in the proportion of juvenile salmonids in the diet of Caspian terns nesting at East Sand Island from the pre-management period to the management period, there was a trend toward increasing reliance on salmonid smolts (*Figures 1.15* & *1.17* above). In particular, if the percent salmonids in the tern diet during 1999 and 2000, the first two seasons after terns started shifting from nesting on Rice Island to nesting on East Sand Island, are eliminated from the pre-management period, the average proportion of juvenile salmonids in the diet was 27.6% during 2001–2007, significantly less than the average of 35.0% during the management period (2008–2019; Wilcoxon rank sum test, W = 72, *P* = 0.001; *Table 1.7* above). At the time that the *Caspian Tern Management Plan* was prepared, salmonids were estimated to comprise, on average, about 33% of the diet of Caspian terns nesting at East Sand Island based on diet data collect during the 1999–2004 nesting seasons (USFWS 2005). This dietary shift toward a somewhat greater reliance on juvenile salmonids by East Sand Island Caspian terns was unrelated to implementation of the *Plan* and, similar to the impact of the marine heat wave, could not have been anticipated.

1.5.1.3 Predation rates on juvenile salmonids – To help evaluate the efficacy of the *Caspian Tern Management Plan* for reducing predation rates on juvenile salmonids by Caspian terns in the Columbia River estuary, we compared ESU/DPS-specific predation rates during the pre-management period (2000–2007) and during the management period (2008–2018). *Results indicated that predation rates* were, on average, significantly lower for most but not all ESUs/DPSs following implementation of management actions that reduced the amount of suitable nesting habitat and, ultimately, the size of the *East Sand Island* Caspian tern colony. For example, the average annual predation rate by East Sand Island Caspian terns on Snake River steelhead during the pre-management period was 25.3% (22.7–28.3%) compared with 10.7% (9.8–12.0%) during the management period, an approximate 58% reduction in average annual predation rate (*Table 1.9* above). Similar but less dramatic reductions in predation rates were observed for all other ESA-listed steelhead DPSs, with reductions in average annual predation rate of approximately 32% to 41% (*Table 1.9* above). Reductions were also significant in some but not all UCR, SR, and UWR Chinook salmon ESUs. For SR sockeye salmon there were no estimates of tern predation rates available during the pre-management period due to inadequate sample sizes of PIT-tagged sockeye salmon smolts (see *above*). For all Lower

Columbia River (LCR) ESUs/DPSs, estimates of tern predation rates during the pre-management period were limited to just a single year (2007), so comparisons of predation rates on LCR stocks during the pre-management period vs. the management period should be interpreted cautiously.

For salmonid ESUs/DPSs with adequate samples sizes of tagged smolts and where data were available for multiple years during both the pre-management period and the management period, these results indicate that implementation of the *Caspian Tern Management Plan* significantly reduced average annual predation rates on juvenile salmonids by terns nesting on East Sand Island. This was particularly evident in 2017 and 2018, the last two years when estimates of predation rates from PIT tag recoveries on the East Sand Island tern colony were available. During 2017 and 2018 the number of Caspian terns recorded nesting on East Sand Island (3,500 and 4,959 nesting pairs, respectively) were the lowest since 1999, and coincided with the lowest estimated tern predation rates on steelhead DPSs (*Figure 1.21* above). During the pre-management period (2000–2007), the East Sand Island Caspian tern colony averaged 9,080 breeding pairs (range = 8,330–9,930 pairs; *Table 1.2* above). During the management period (2008–2019), the East Sand Island Caspian tern colony averaged 6,690 breeding pairs (range = 3,500–10,670 pairs; *Table 1.2* above). It should be noted that reductions in the area of suitable tern nesting habitat (i.e. management actions) on East Sand Island had little or no effect on tern colony size for the first three years (2008–2010) of the management period because during the pre-management period terns only nested on a portion of the nesting habitat that was prepared for them (see *above*). As such, the management period used herein corresponds to when management actions were first implemented (policy-based criterion) and not when those actions first caused a reduction in the size of the East Sand Island Caspian tern colony (biological-based criterion).

Implementation of the *Caspian Tern Management Plan* eventually reduced the size of the East Sand Island tern colony to within the range of sizes specified (3,125–4,375 breeding pairs) and smaller colony size was associated with much lower predation rates on juvenile salmonids, especially steelhead smolts, by Caspian terns nesting on East Sand Island (*Table 1.9* above and *Figure 1.24* above). In particular, the reduction in size of the East Sand Island Caspian tern colony from the pre-management period (2000–2007) to the last three years of the management period when predation rates were measured (2016–2018) resulted in major declines in predation rates on ESA-listed steelhead smolts, with reductions of 50% to 76%, depending on the DPS. By 2016–2018 the average annual predation rates on steelhead DPSs by East Sand Island Caspian terns had declined from approximately 19% to 7%, or an average reduction in tern predation rates on steelhead smolts in the estuary of about 63%. This decline in predation rates on steelhead smolts exceeds the targeted reduction of 60% that was stipulated in the *Caspian Tern Management Plan* (USFWS 2005).

LCR Chinook salmon exhibit both a sub-yearling and yearling life history, with yearlings out-migrating in the spring and sub-yearlings in the late-spring and summer (NMFS 2019a). Variations in these life histories correspond to different population groups or strata, groups that originate from distinct areas within the geographic range of the population (referred to as "Coastal, Cascade, and Gorge"; NMFS 2019a). A large number and percentage (> 90%) of PIT-tagged hatchery LCR Chinook salmon included in predation rate analyses consisted of sub-yearling smolts that were released *en masse* in the Coastal and Cascade regions downstream of Bonneville Dam (see *Appendix A*). An investigation of how smolt release location influenced tern predation rates indicated that predation rates by East Sand Island Caspian terns were consistently higher on sub-yearling Chinook salmon released downstream of Bonneville Dam relative to sub-yearling and yearling Chinook salmon released upstream of Bonneville Dam, with statistically significant differences observed in most years. Sebring et al. (2013) also observed significantly higher rates of predation by East Sand Island Caspian terns on LCR Chinook salmon relative to other Chinook salmon ESUs. Sebring et al. (2013) theorized that slower migration rates and prolonged estuarine residence times increased the susceptibility of LCR Chinook salmon smolts, particularly those originating from Coastal and Cascade regions to predation from Caspian terns nesting on East Sand Island (see also *Chapter 10*).

Comparisons of predation rates on hatchery vs. wild steelhead by Caspian terns nesting at the East Sand Island colony revealed no significant differences in susceptibility to tern predation. For UCR and SR spring/summer Chinook salmon, however, hatchery smolts were significantly more susceptible to tern predation in the estuary than their wild counterparts. This difference in susceptibility was attributed to the larger size of Chinook salmon smolts that are raised in hatcheries, along with the selectivity for larger sized smolts exhibited by Caspian terns (Hostetter et al. 2012, Roby et al. 2017). Unlike UCR and SR spring/summer Chinook salmon, which predominately migrate in April and May, the out-migration timing of UWR Chinook salmon varies considerably, with smolts out-migrating nearly year-round (NMFS 2019a). Differences in run-timing and relative availability of UWR Chinook salmon in the estuary may explain why tern predation rates in the estuary on this ESU of Chinook salmon were so low compared with other salmon ESUs (Evans et al. 2016).

1.5.1.4 Compensatory vs. additive smolt mortality – A critical assumption of the *Caspian Tern Management Plan* is that reductions in Caspian tern predation rates on juvenile salmonids will result in increased smolt survival rates in the estuary. At the time when the *Caspian Tern Management Plan* was prepared, however, data regarding the degree to which Caspian tern predation was an additive source of mortality (i.e. tern predation adds to smolt mortality) versus a compensatory source of mortality (i.e. tern predation is compensated for by other, non-tern sources of mortality) were lacking (USFWS 2005). Recent advances in mark-recapture-recovery models now make it possible to jointly estimate predation rates and survival rates of PIT-tagged fish to measure the strength, magnitude, and direction of this relationship. Results of these studies provide the means to estimate what proportion of depredated smolts would have otherwise survived in the absence of Caspian tern predation (i.e. the additive effects of tern predation). A detailed description of these studies is provided in *Chapter 8* (see also Payton et al. 2020). Briefly, in a study of Caspian tern predation on PIT-tagged UCR and SR steelhead smolts, results indicated that increases in tern predation rates on steelhead were associated with statistically significant decreases in steelhead survival rates in all evaluated years and in two salmonid life-stages (smolt out-migration and smolt-to-adult returns). Results indicated that, in the absence of tern predation, significantly more steelhead smolts would have survived out-migration and, more importantly from a conservation perspective, more adult steelhead would have returned to the Columbia River to spawn. These results provide novel and robust evidence that predation by Caspian terns was largely an additive source of mortality during the smolt life-stage and a partially additive source of mortality during the smolt-to-adult life-stage (see *Chapter 8* for a detailed discussion of these results).

1.5.2 Adaptive Management

1.5.2.1 East Sand Island – The target size for the East Sand Island colony of Caspian terns (3,125–4,375 breeding pairs) that was stipulated in the *Caspian Tern Management Plan* was not immediately achieved by reducing the area of suitable nesting habitat from 6 acres (2.4 ha) to 1.0 acres (0.4 ha). During the first two nesting seasons after the area of tern nesting habitat had been reduced to 1.0 acres, tern colony size was about 6,240 breeding pairs and about 5,920 breeding pairs in 2015 and 2016, respectively, or 43% and 35% greater than the maximum target colony size (4,375 breeding pairs). The larger tern colony size than planned was due primarily to tern nesting densities on the East Sand Island colony that were about twice those seen at the East Sand Island colony pre-management, and 50% greater than the maximum nesting density anticipated when the *Caspian Tern Management Plan* was prepared (USFWS 2005). Nesting densities at the East Sand Island colony during 2015 and 2016 were much higher than any nesting density previously reported for Caspian terns nesting in the Columbia River estuary. Because the *Caspian Tern Management Plan* only covered reduction in the area of tern nesting habitat on East Sand Island down to 1.0 acres (0.4 ha; USFWS 2006, USACE 2015), achieving the target colony size of 3,125–4,375 breeding pairs by further reducing the area of managed tern nesting habitat on East Sand Island was not permitted. Instead, the management agencies attempted to reduce nesting density and colony size during 2017–2019 by reducing the pre-season preparation (i.e. disking and scraping) of bare-ground nesting substrate for terns on the 1-acre (0.4-ha) colony site and by adjusting the shape of the 1-acre (0.4-ha) colony site to increase the area of edge nesting habitat relative to core nesting habitat (*Figure 1.2* above).

In addition to high tern nesting densities on the 1-acre (0.4-ha) prepared colony site on East Sand Island during 2015 and 2016, in both years terns formed satellite colonies on the eastern end of East Sand Island outside the prepared colony site. In 2015, a total of 810 pairs of terns nested in two satellite colonies and in 2016 a total of 700 pairs of terns nested in two satellite colonies. Terns nesting in satellite colonies represented just 13% and 12% of the total number of terns nesting on East Sand Island in 2015 and 2016, respectively. Plus, these satellite colonies formed late in the nesting season when tern predation rates on juvenile salmonid were relatively low, and all or most tern nests in satellite colonies failed to fledge young or even hatch eggs. Nevertheless, the formation of tern satellite colonies on East Sand Island was partly responsible for the failure of the *Caspian Tern Management Plan* to achieve the objective of reducing tern colony size to no more than 4,375 nesting pairs in 2015 and 2016.

Adaptive management efforts to prevent the formation of satellite tern colonies on East Sand Island near the managed colony site were initiated in 2010 using passive tern nest dissuasion techniques (i.e. stakes, ropes, and flagging). Despite this effort, in 2010 a small satellite tern colony (56 breeding pairs) formed on the upper beach near the main tern colony, outside the designated area of tern nesting habitat. Then, beginning in 2013, passive nest dissuasion materials were deployed over extensive areas of potential tern nesting habitat at East Sand Island prior to the nesting season. Also beginning in 2013, active nest dissuasion techniques (i.e. hazing) were implemented to enhance efforts at preventing the formation of tern satellite colonies outside the designated colony area. Both passive and active nest dissuasion techniques were enhanced and expanded in 2015, the first year when the area of designated tern nesting habitat was just 1.0 acres (0.4 ha), but two satellite tern colonies nonetheless formed outside the main colony in 2015 and again in 2016. No collection of tern eggs was permitted on East Sand Island, however, so tern eggs laid outside the managed colony area on East Sand Island could not be collected to help deter formation of tern satellite colonies, unlike on Goose Island in Potholes Reservoir (see *Chapter 2*). Once a Caspian tern egg was laid outside the managed tern colony area on East Sand Island, any human hazing of terns attending an egg was prohibited. Despite this restriction, and continued prospecting by hundreds of terns outside the designated colony area, no satellite tern colonies formed on East Sand Island during the 2017–2018 nesting seasons, when a combination of both intensive passive and active nest dissuasion techniques was deployed.

After the failure of the Caspian tern colony on East Sand Island to produce any young in 2011, limited lethal removal of glaucous-winged/western gulls from the tern colony was implemented in 2012 in an effort to prevent the colony from failing for the second year in a row. Limited gull control had not been implemented at the East Sand Island tern colony since 2000, as part of efforts to relocate the Caspian tern colony on Rice Island to East Sand Island. The lethal removal of 50 gulls that were foraging on tern eggs and chicks in the East Sand Island tern colony early in the 2012 nesting season was credited with preventing the colony from failing a second year in a row, as only about 400 young terns (average of 0.06 young raised per breeding pair) were raised that season (Roby et al. 2013). Part of the motivation for managers to seek limited gull control on the East Sand Island tern colony in 2012 was to avoid a major increase in numbers of Caspian terns attempting to nest at the former colony site on Rice Island, where a few pairs of Caspian terns had successfully nested in 2011 for the first time since 2000.

1.5.2.2 Columbia River estuary – Efforts to dissuade and otherwise discourage terns from nesting at sites in the Columbia River estuary other than East Sand Island, particularly sites in the freshwater portion of the estuary where salmonids are more prevalent in tern diets (Roby et al. 2002; see *Chapter 5*), were an important component of adaptive management actions in the estuary. In 2011, a small number of Caspian terns successfully nested at the former colony site on Rice Island for the first time since 2000. Because the diet of Caspian terns nesting on Rice Island consisted primarily of juvenile salmonids, managers sought to prevent the re-establishment of a Caspian tern colony on Rice Island, a large dredged-material disposal island in the upper estuary. Despite Caspian terns attempting to nest on Rice Island in large numbers during most years since 2011, efforts to prevent a Caspian tern colony from forming on Rice Island were successful during 2012–2018 (USACE, unpubl. data; Harper and Collis 2018).

An exceptionally large number of Caspian terns prospected for nesting opportunities at Rice Island in 2017. During this nesting season, the Caspian tern colony on East Sand Island failed to raise any young, and the designated colony site on East Sand Island was completely abandoned by terns for 10 days in the middle of the nesting season. This unprecedented event at the East Sand Island tern colony was no doubt a contributing factor to the large numbers of terns prospecting at the former Rice Island colony site. Estimates of predation rates on steelhead DPSs by terns that attempted to nest on Rice Island in 2017, but failed, ranged from 2.4% to 3.0%, depending on the DPSs. Given the large numbers of Caspian terns that have attempted to nest on Rice Island since 2011 (several hundred to several thousand breeding adults, depending on the year; USACE, unpubl. data; Harper and Collis 2018) and the much higher per capita smolt predation rates by Caspian terns nesting on Rice Island compared to Caspian terns nesting on East Sand Island (Roby et al. 2003b; see also *Chapter 5*), the overall net benefit of reducing the size of the East Sand Island Caspian tern colony has been offset to some unknown degree by the consumption of smolts by terns that have attempted to nest elsewhere in the Columbia River estuary. As such, continued adaptive management efforts to prevent terns from nesting on Rice Island and other dredged material disposal islands in the upper Columbia River estuary may be paramount to achieving the overall goal of reducing predation rates on juvenile salmonids by reducing the number of Caspian terns nesting in the estuary. One component of this adaptive management effort is to ensure that intense gull predation on tern eggs and chicks does not result in such chronically low nesting success at the East Sand Island colony that the colony is abandoned, thereby intensifying tern nesting attempts on Rice Island and the risk of a new Caspian tern colony forming on Rice Island.

1.5.3 Adaptive Management Considerations

In our professional opinion, there are several key uncertainties that may potentially impact future action effectiveness and the necessity for adaptive management in order to fully achieve the goals and objectives of the *Caspian Tern Management Plan*. These uncertainties include (1) ensuring that Caspian tern predation rates on ESA-listed juvenile salmonids, particularly steelhead smolts, do not exceed acceptable levels in the Columbia River estuary; (2) preventing the re-establishment of a Caspian tern nesting colony on Rice Island, or on other dredged material disposal sites in the upper Columbia River estuary; (3) maintaining the size of the Caspian tern colony on East Sand Island within the stipulated range of 3,125–4,375 breeding pairs; (4) ensuring the long-term viability of the East Sand Island Caspian tern colony; and (5) avoiding the long-term decline of the Pacific Flyway population of Caspian terns as a consequence of management in the Columbia River estuary.

1.5.3.1 Monitor predation rates by terns on juvenile salmonids – Predation rate estimates provide crucial information on the relative impact of Caspian tern predation on specific groups of tagged fish. Without monitoring predation rates based on smolt PIT tag recoveries/detections on Caspian tern colonies in the Columbia River estuary, it is not possible to detect changes in predation rates that could threaten the conservation status of ESA-listed populations of salmonids from the Columbia River basin, especially populations of steelhead. Predation rates, however, are not equivalent to estimates of consumption (number of fish consumed) and they require sufficient sample sizes of tagged fish to generate accurate estimates of predation rates (see *below*). *Appendix A, Predation Rate Methods* provides a list of key assumptions and caveats of models to estimate predation rates, including information on the validity of these assumptions and caveats.

Studies of Caspian tern predation rates on juvenile salmonids that are based on on-colony PIT tag recoveries require that sufficient numbers of fish in a population are tagged, and that tagged fish are representative of the population at-large (tagged and untagged fish; Evans et al. 2012, Evans et al. 2014, Evans et al. 2019c). Adequate numbers of some, but not all, ESA-listed salmonid populations were PIT-tagged and available for analyses of predation rates in the present study. Notable exceptions include Upper Willamette River (UWR) steelhead trout and Columbia River chum salmon, where no estimates of predation rates were available because no or very few tagged fish were available for analyses of predation rates. Annual sample sizes of PIT-tagged fish were also limited for several other Evolutionarily Significant Units or Distinct Population Segments (ESUs/DPSs) of Columbia Basin salmonids, including Snake River (SR) sockeye salmon, UWR Chinook salmon, and Lower Columbia River (LCR) coho salmon.

For some ESUs/DPSs of Columbia Basin salmonids, the run-timing and abundance of PIT-tagged fish was in close agreement with the run-timing and abundance of untagged fish passing Bonneville Dam and Sullivan Dam (Evans et al. 2012). For a few populations, such as Upper Columbia River (UCR) steelhead, fish were also randomly selected for tagging and were tagged in concert and in proportion to the number of fish (tagged and untagged) available in-river (Evans et al. 2019a). For other groups of fish, however, individuals were selected for tagging based on their condition or size, or only a small number or proportion of available fish were PIT-tagged from a given ESU/DPS (Evans et al. 2012, Sebring et al. 2013). The latter is of particular concern for Lower Columbia River ESUs/DPSs, where PIT-tagging was often limited to fish from selected hatcheries and tagging efforts varied substantially by salmonid population and year.

Estimates of predation on LCR steelhead and LCR Chinook salmon were based on fish that originated from tributaries located both upstream and downstream of Bonneville Dam (NMFS 2019b; see also *Appendix A, Map 1*). Analogous to estimates of predation in UCR, SR, and Middle Columbia River (MCR) ESUs/DPSs, estimates of LCR ESU/DPS-specific predation rates account for fish survival to Bonneville Dam and they account for within-season differences in run-timing and the relative availability of tagged fish each week (see *Appendix A, Predation Rate Methods*). For those smolt releases into tributaries downstream of Bonneville Dam, however, estimates do not account for mortality from the release site to the Columbia River estuary. If large numbers of PIT-tagged Lower Columbia River smolts that were released into tributaries downstream of Bonneville Dam died prior to reaching the foraging range of Caspian terns nesting on East Sand Island, estimated predation rates would be biased low to an unknown degree.

1.5.3.2 Prevent formation of tern colonies in the upper Columbia River estuary – As the area of designated Caspian tern nesting habitat on East Sand Island was reduced to 1.0 acres (0.4 ha) to reduce the size of the East Sand Island colony to 3,125–4,375 breeding pairs, tern prospecting and nest initiation behavior on Rice Island intensified. Although no nesting attempts by Caspian terns were detected on Rice Island during 2001–2010, by 2011 terns that were unsuccessful in nesting attempts at East Sand Island resumed efforts to initiate nests at the former colony site on Rice Island. Without intensive nest dissuasion efforts on Rice Island throughout much of the nesting season during 2012–2019, a breeding colony would certainly have become re-established at the downstream end of Rice Island. Given the current limited area of nesting habitat for terns on East Sand Island and the poor average nesting success experienced by terns nesting at the East Sand Island colony, continued attempts by hundreds or even thousands of Caspian tern breeding pairs to nest at the former colony site on Rice Island should be expected. If a colony of Caspian terns becomes established on Rice Island, most of the diet of terns nesting there is expected to be juvenile salmonids (70–90% of prey items), as was the case during 1997–1999. The much higher per capita predation rates on juvenile salmonids by terns nesting on Rice Island compared to those nesting on East Sand Island pose a significant threat to Columbia Basin salmonids, especially steelhead, if a large Caspian tern colony is allowed to form on Rice Island, as was the case during the 1990s.

Fortunately, non-lethal techniques have been developed and refined that are highly effective in deterring Caspian terns from nesting at colony sites where they pose a risk to fish stocks of conservation concern (see *Section 1.3.1* above). By using a combination of passive nest dissuasion techniques (i.e. landscape fabric fencing, stakes, ropes, and flagging), active nest dissuasion techniques (i.e. hazing using humans, kites, and lasers), and collection of any Caspian tern eggs that are laid, prospecting Caspian terns can be precluded from nesting at undesirable colony sites. Effective and extensive tern nest dissuasion on Rice Island is particularly challenging because of the presence of ESA-listed streaked horned larks (*Eremophila alpestris strigata*), which nest on the island and whose designated critical habitat includes portions of Rice Island. Human activity on the island has been severely restricted to minimize the impact on nesting success of streaked horned larks (USACE, unpubl. data: Harper and Collis 2018), but so far these restrictions have not led to the formation of a Caspian tern colony on Rice Island.

1.5.3.3 Limit colony size for terns on East Sand Island – Two potential factors could result in the colony of Caspian terns nesting on East Sand Island to exceed the size range designated in the *Caspian Tern Management Plan* (3,125–4,375 breeding pairs): (1) the return of high tern nesting densities at the 1-acre (0.4-ha) designated colony site and (2) the formation of satellite tern colonies on East Sand Island outside the 1-acre (0.4-ha) designated colony site. As confirmed during the 2015, 2016, and 2018 nesting seasons, if nesting Caspian terns occupy the entire 1-acre (0.4-ha) colony site, and if terns nest at average densities that are in excess of 1.08 nests/m^2, then the size of the tern colony will exceed 4,375 breeding pairs. In 2017 and 2019, the two years when tern colony size on East Sand Island was less than 4,375 breeding pairs, nesting density was 0.97 and 1.11 nests/m^2, respectively, but in 2019 terns used only about 0.86 acres (0.35 ha) for nesting, instead of the full 1 acre (0.4 ha). The reason that Caspian terns did not nest throughout the 1-acre (0.4-ha) designated colony area in 2019 is because about 15% of the prepared nesting habitat was occupied by nesting glaucous-winged/western gulls, which are larger than Caspian terns and depredate the contents of tern nests. In order to ensure that tern colony size does not exceed 4,375 breeding pairs in the future it will likely be necessary to reduce the area of suitable nesting habitat on East Sand Island from 1.0 acres (0.4 ha) to between 0.50 and 0.75 acres (0.2 to 0.3 ha). Such a reduction will require additional environmental analysis and NEPA compliance by federal management agencies. The results presented in this chapter strongly suggest that attempts to reduce tern nesting density and colony size on East Sand Island by altering the shape of the designated 1-acre (0.4-ha) colony site or by reducing the suitability of nesting substrate will likely reduce tern nesting success more than nesting density and colony size. Sustained periods of poor nesting success are a threat to the long-term viability of the Caspian tern colony on East Sand Island (see *below*), which puts the long-term success of the *Caspian Tern Management Plan* at risk.

The other key for limiting colony size to less than 4,375 breeding pairs on East Sand Island is to prevent Caspian terns from nesting outside the designated colony area and thereby establishing satellite colonies. By using a combination of passive and active nest dissuasion techniques, we have demonstrated that Caspian terns can be nondestructively deterred from nesting outside the designated colony site on East Sand Island. Nest dissuasion techniques would be much more effective and less labor-intensive, however, if tern eggs laid outside the designated colony area could be collected under permit. The presence of tern eggs or incubating terns at an incipient satellite colony site is a major attractant for other terns prospecting for nest sites, especially if the designated colony site is fully occupied by nesting terns. Removing this attractant by collecting any tern eggs laid outside the designated colony area would greatly reduce the risk of satellite colony formation, as demonstrated by management of the Caspian tern colony on Goose Island in Potholes Reservoir, Columbia Plateau region (see *Chapter 2*).

1.5.3.4 Maintain viability of the East Sand Island tern colony – Although Caspian terns have shown higher than expected site fidelity to the colony on East Sand Island (Collar 2013, Collar et al. 2017, Bailey 2018, Suzuki et al. 2018), Caspian terns are generally considered to have relatively low colony site fidelity (Cuthbert 1988). Repeated breeding seasons of poor nesting success (i.e. 2010, 2012, 2018, 2019) or complete nesting failure (i.e. 2011, 2017, 2020) at the Caspian tern colony on East Sand Island enhances the prospect of the designated colony site being completely abandoned. Should abandonment of the East Sand Island colony site occur, it would likely result in two undesirable outcomes: (1) increase the numbers of Caspian terns prospecting for alternative colony sites outside the designated colony area on East Sand Island, both on the upper beaches of East Sand Island and especially at dredged material disposal sites in the upper estuary, such as Rice Island, and (2) eliminate a breeding colony that is key to avoiding a long-term decline in the Pacific Flyway population of Caspian terns.

The large Caspian tern colony on Rice Island in the upper Columbia River estuary was moved to East Sand Island during 1999–2001 because the diet of Caspian terns nesting on Rice Island consisted primarily of juvenile salmonids (74–90% of prey items). During 1997 and 1998, Caspian terns nesting on Rice Island annually consumed an estimated 5.4–14.2 million juvenile salmonids (Roby et al. 2003b), which represented about 5–15% of all salmonid smolts reaching the estuary during each of those two migration years. Predation rates on Snake River and Upper Columbia River steelhead were estimated at 21.2% (16.0–42.0%) and 23.2% (15.8–

42.2%), respectively, of all the smolts from those two populations that survived to the estuary (see section *1.4.2.5 Predation rates by terns nesting on Rice Island* above; see also *Chapter 5*). The Caspian tern colony on East Sand Island has the highest colony connectivity with the former colony site on the downstream end of Rice Island, and this has been demonstrated each year since 2011 by the large numbers of Caspian terns that have prospected for nest sites on Rice Island, especially in years when nesting success on East Sand Island has been poor. If the Caspian tern colony site on East Sand Island was abandoned, as it was before in 1986, most terns dispersing from East Sand Island will likely seek new nest sites on Rice Island.

The objective of the *Caspian Tern Management Plan* was to reduce colony size on East Sand Island to 3,125–4,375 breeding pairs to achieve the goal of enhancing recovery of ESA-listed populations of salmonids in the Columbia River basin, especially steelhead. The objective was not to eliminate the Caspian tern colony on East Sand Island. Before implementation of the *Caspian Tern Management Plan*, the East Sand Island colony represented nearly 70% of the entire Pacific Flyway population of Caspian terns, and young terns raised on the East Sand Island colony were the primary source of new recruits for the Flyway-wide population. Using population projection models, Suzuki et al. (2018) showed that the abandonment of the East Sand Island tern colony, or even just the reduction in colony size below 1,000 breeding pairs, would result in a long-term decline in the Pacific Flyway population of Caspian terns. While we recognize and support the primary goal of the *Caspian Tern Management Plan*, which is to enhance the recovery prospects for ESA-listed salmonids in the Columbia River basin, especially steelhead, by reducing Caspian tern predation on juvenile salmonids in the Columbia River estuary, a secondary objective is to conserve the Pacific Flyway population of Caspian terns. Based on 23 years of conducting research, monitoring, and evaluation of management of Caspian terns nesting in the Columbia River estuary, we believe that achieving the primary goal of the *Caspian Tern Management Plan* and conserving the Caspian tern population are not incompatible.

In order to conserve the East Sand Island Caspian tern colony and maintain it within the designated size range, we recommend (1) reducing the area of tern nesting habitat further from its current area of 1.0 acres (0.4 ha) to between 0.50 and 0.75 acres (0.2 and 0.3 ha), (2) moving the colony site further to the northeast on East Sand Island so it is not adjacent to the beach (to avoid attracting terns to nest on the upper beach immediately adjacent to the designated colony area), and (3) ensuring that the nesting substrate on the designated colony area is sandy and well-drained so that tern nests do not experience widespread failure due to flooding and erosion. These steps for enhancing the long-term viability of the Caspian tern colony on East Sand Island may not by themselves be sufficient to ensure that the colony persists, however. Average nesting success of Caspian terns at the East Sand Island colony has declined drastically in the last two decades, and the colony has failed to raise any young in three of the last 10 years. The proximate factor responsible for the vast majority of tern nest failures at the East Sand Island colony is predation on tern eggs and chicks by glaucous-winged/western gulls. Gull predation on tern nests is due almost entirely to a small subset of the thousands of glaucous-winged/western gulls that nest on East Sand Island, those that nest on or adjacent to the tern colony. By lethally removing gulls that habitually depredate tern nests on the colony, predation from those individual gulls can be eliminated and other gulls that witness the lethal removal will learn to avoid the tern colony. Limited gull control was not only used to successfully restore the Caspian tern colony on East Sand Island in 1999 and 2000, but also to prevent the East Sand Island tern colony from failing to produce any young in 2012.

One additional approach to reducing the numbers of Caspian terns prospecting for nest sites on Rice Island and reducing the impact to the Flyway-wide Caspian tern population of poor nesting success at East Sand Island is to create alternative nesting habitat at or near former colony sites along the coast of Washington. Several sites in Grays Harbor, the Strait of Juan de Fuca, and Puget Sound formerly supported large (> 1,000 breeding pairs) and productive colonies of Caspian terns that did not prey primarily of juvenile salmonids. Establishing coastal colonies of Caspian terns outside the Columbia River basin with high connectivity to the colony on East Sand Island would likely facilitate greater breeding dispersal away from the Columbia River estuary and support efforts to reduce Caspian tern predation on juvenile salmonids from the Columbia and Snake rivers.

1.5.3.5 Avoiding a long-term decline in the Pacific Flyway population of Caspian terns – The Caspian Tern Management Plan was designed to achieve the goal of dramatically reducing tern predation rates on ESA-listed juvenile salmonids in the Columbia River estuary without causing a long-term decline in the size of the Pacific Flyway population of Caspian terns. To this end, the Plan included the provision of alternative nesting habitat for Caspian terns that were displaced from the East Sand Island colony by the managed reduction in nesting habitat. The new alternative nesting habitat provided, however, has only accommodated a fraction of the ca. 5,400 breeding pairs of Caspian terns that have been displaced from the East Sand Island colony due to management. Also, the nesting success of Caspian terns that continue to nest on East Sand Island, which formerly supported nearly 70% of the Pacific Flyway population of the species, has steadily declined.

From 2000 to 2011 the U.S. Fish and Wildlife Service conducted annual surveys of breeding colonies of Caspian terns throughout the Pacific Flyway to census the Flyway-wide population. Population size peaked in 2009 at a census count of about 18,872 breeding pairs, with 52% of the Flyway-wide population nesting on East Sand Island (USFWS 2015). In 2015 the U.S. Fish and Wildlife Service initiated a monitoring strategy for the Caspian tern population in the Pacific Flyway with the objective of estimating population size every three years using a dual frame sampling approach intended to detect any significant trends in population size (USFWS 2015). Minimum census counts of the Pacific Flyway population of Caspian terns in 2015 (Peterson et al. 2017a) and 2018 (Peck-Richardson et al. 2019) indicated that the breeding population had declined by about 34% from 2009 to 2015 (12,407 breeding pairs), and by about another 15% from 2015 to 2018 (10,580 breeding pairs), although the later decline was not statistically significant. Nevertheless, based on the minimum census estimate of population size in 2018, the Pacific Flyway population of Caspian terns is apparently at its lowest level in the last 25 years (Peck-Richardson et al. 2019) and the declining trend is likely to persist.

A population projection model developed for the Pacific Flyway population of Caspian terns (Suzuki et al. 2018) found that, while the population was likely resilient to the management actions described in the *Caspian Tern Management Plan* (i.e. reduction in colony size at East Sand Island to 3,125–4,375 breeding pairs), further reductions to a colony of less than 1,000 breeding pairs and/or consistently poor nesting success of terns on East Sand Island would result in long-term declines in the breeding population throughout the Flyway. With the size and nesting success of the colony of Caspian terns on East Sand Island so unpredictable, the future of the colony and the population as a whole is uncertain. No new colonies of Caspian terns that come close to the size or nesting success of the East Sand Island colony during the 2000s have formed in the aftermath of implementing the *Caspian Tern Management Plan*, and in the absence of such a colony the Pacific Flyway population is at risk of a substantial, prolonged decline.

Given the key role that the East Sand Island colony of Caspian terns plays in the status of the Pacific Flyway population of the species and given recent trends in colony size and nesting success at East Sand Island, plus recent trends in overall population size in the Pacific Flyway, the continued close monitoring of the size of the Pacific Flyway population is clearly warranted. The dual frame sampling approach currently used by the U.S. Fish and Wildlife Service as its monitoring strategy for the Caspian tern population in the Pacific Flyway (USFWS 2015) has struggled to provide accurate and precise estimates of the census population size (Peck-Richardson et al. 2019) due to the high rate of change in the distribution and size of breeding colonies. But sustaining and updating the Caspian tern monitoring strategy is crucial for detecting major population declines before the species becomes of conservation concern in the Pacific Flyway.

1.6 Conclusions

1. The goal of the *Caspian Tern Management Plan*, which was implemented starting in 2008, was to reduce predation rates on juvenile salmonids, especially steelhead smolts, by reducing the size of the large Caspian tern breeding colony on East Sand Island. Reduction in colony size was achieved by reducing the area of suitable nesting habitat on East Sand Island from a baseline of approximately 5.0 acres (2.02 ha) to 1.0 acres (0.4 ha).

2. The area of tern nesting habitat on East Sand Island that was stipulated by the *Caspian Tern Management Plan* (1.0 acres or 0.4 ha) was achieved by 2015 and sustained for the next four breeding seasons. Two separate tern satellite colonies of moderate size formed, however, late in the 2015 and 2016 nesting seasons.

3. The target size for the Caspian tern colony on East Sand Island (3,125–4,375 breeding pairs) that was stipulated in the *Caspian Tern Management Plan* was achieved in two of the five years when the area of nesting habitat was 1.0 acres (0.4 ha); colony sizes in excess of 4,375 breeding pairs were the result of higher-than-expected tern nesting densities in the designated colony area.

4. Reduction in the size of the East Sand Island Caspian tern colony resulted in a reduction in consumption of juvenile salmonids by approximately 42%, or from about 5.0 million smolts/year down to about 2.9 million smolts/year.

5. During the pre-management period (2000–2007), average annual predation rates on ESA-listed steelhead populations (DPSs) by Caspian terns nesting at the East Sand Island colony were approximately 19%, whereas average annual predation rates on other ESA-listed salmon populations (ESUs) were much lower, approximately 4.0%.

6. The reduction in the size of the East Sand Island Caspian tern colony from the pre-management period to that during 2016–2018 (the last three years when predation rates were measured) resulted in reductions in predation rates on ESA-listed steelhead smolts by 50% to 76%, depending on the DPS. On average, the annual predation rates on steelhead DPSs by East Sand Island Caspian terns declined from approximately 19% during the pre-management period to 7% during 2016–2018, or an average reduction in predation rates on steelhead smolts by East Sand Island terns of about 63%.

7. While the *Caspian Tern Management Plan* achieved the targeted reductions of 60% in predation rates on steelhead populations by Caspian terns nesting on East Sand Island, reductions in Caspian tern predation rates on steelhead smolts throughout the Columbia River estuary were less substantial due to (1) persistent prospecting by Caspian terns displaced from the East Sand Island colony at the former colony site on Rice Island, where juvenile salmonids are the predominant prey type; (2) the formation of incipient satellite tern colonies on East Sand Island, outside the designated colony area; and (3) increased reliance on juvenile salmonids as a food source by Caspian terns in the Columbia River estuary due to increased river discharge, declining ocean conditions, and consequent lower availability of marine forage fishes as alternative prey.

8. Caspian tern nesting success at the East Sand Island colony has declined dramatically since the colony was re-established in 1999; lower nesting success is due to increased gull predation on tern eggs and young, decreased availability of marine forage fish in the estuary, increased bald eagle disturbance to the tern colony, and delays in tern nesting chronology.

9. Caspian tern nesting attempts at sites outside the designated colony area on East Sand Island, elsewhere on East Sand Island and at the former colony site on Rice Island, intensified as the area of designated nesting habitat on East Sand Island was reduced to 1.0 acres (0.4 ha). Non-lethal tern nest dissuasion techniques succeeded in preventing the formation of satellite colonies on East Sand Island during 2017–2019 and the re-formation of a tern colony on Rice Island during 2012–2019.

10. In order to fully achieve the objectives of the *Caspian Tern Management Plan,* we recommend (1) continued monitoring of Caspian tern predation rates on juvenile salmonids, especially steelhead smolts, using smolt PIT tag recoveries on-colony to ensure that per capita predation rates do not exceed acceptable levels; (2) continued use of tern nest dissuasion techniques to prevent tern nesting outside the designated colony area on East Sand Island, especially on Rice Island; (3) further reductions in the area of tern nesting habitat on East Sand Island from 1.0 acres (0.4 ha) to between 0.50 and 0.75 acres (0.2 to 0.3 ha); (4) management of the designated Caspian tern colony site on East Sand Island to reduce the risk of permanent colony abandonment by enhancing nesting substrate and controlling gull predation on tern nests; (5) restoration of a large Caspian tern nesting colony (ca. 1,000 breeding pairs) at or near a former colony site in coastal Washington; and (6) continued close monitoring of the Pacific Flyway population of Caspian terns for early detection of major declines.

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CHAPTER 2: Caspian Tern Management in the Columbia Plateau Region

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2.1 Summary

The purpose of this chapter is to summarize results of the *Inland Avian Predation Management Plan* (*IAPMP*) which focuses on reducing the number of Caspian terns (*Hydroprogne caspia*) nesting at the two largest breeding colonies in the Columbia Plateau region (CPR) during 2014-2019, namely the colonies on Goose Island in Potholes Reservoir and on Crescent Island in McNary Reservoir on the Columbia River. The *IAPMP* was implemented by the U.S. Army Corps of Engineers – Walla Walla District (USACE) and the U.S Bureau of Reclamation (BOR) to reduce tern predation rates on juvenile salmonids (*Oncorhynchus* spp.) listed under the U.S. Endangered Species Act (ESA). The research, monitoring, and evaluation (RM&E) studies described herein include data collected at the Caspian tern colonies on Goose Island and Crescent Island during both the pre-management period (2000-2013) and the management period (2014-2019), as well as at other Caspian tern colonies in the CPR throughout the entire study period (2000-2019).

The *IAPMP* has four main objectives, namely to (1) reduce impacts of Caspian tern predation on ESA-listed salmonids in the CPR, including Upper Columbia River and Snake River steelhead trout (*O. mykiss*), Chinook salmon (*O. tshawytscha*), and sockeye salmon (*O. nerka*); (2) dissuade Caspian terns from nesting on Goose and Crescent islands, and other islands where tern colonies might increase or become established (“at-risk” islands), if necessary; (3) preclude the re-establishment of Caspian tern nesting colonies at Goose and Crescent islands in the long-term; and (4) provide suitable alternative nesting habitat outside the Columbia River basin for Caspian terns displaced from colonies in the CPR. As described in the Environmental Assessment for the *IAPMP*, the plan was implemented using a two-phase approach starting in 2014. Phase 1 of the *IAPMP* included: (a) passive nest dissuasion and active hazing on Goose Island to discourage Caspian terns and gulls from nesting on the island, including limited take of any Caspian tern eggs laid at the site; (b) passive nest dissuasion and active hazing on Crescent Island to discourage Caspian terns and gulls from nesting on the island, including limited take of any Caspian tern eggs laid at the site; (c) a feasibility study to assess the use of willow plantings as a means to permanently dissuade Caspian terns and gulls from nesting on Crescent Island; (d) passive nest dissuasion and active hazing on any at-risk islands should Caspian tern colonies over 40 breeding pairs become established; and (e) completion of supplemental/tiered National Environmental Policy Act (NEPA) analysis and assessment to create new nesting habitat for Caspian terns outside the Columbia River basin and in the Pacific Flyway as compensation for the managed reduction of tern nesting habitat in the CPR. Phase 2 of the *IAPMP* included: (f) preparation of suitable alternative nesting habitat for Caspian terns outside the Columbia River basin; (g) a feasibility study to investigate large cobble and/or drought-tolerant vegetation plantings as a means to permanently discourage Caspian terns and gulls from nesting on Goose Island; (h) widespread planting of native vegetation and construction of a berm at the shoreline to permanently dissuade Caspian terns and gulls from nesting on Crescent Island; and (i) continued passive nest dissuasion and active hazing on at-risk islands should Caspian tern colonies over 40 breeding pairs become established.

RM&E studies at the Caspian tern colonies on Goose and Crescent islands included the following specific metrics: (1) nesting chronology; (2) type of nesting habitat used; (3) colony attendance, colony size, and nesting success; (4) factors limiting colony size and nesting success; (5) connectivity with other tern colonies in the Pacific Flyway; (6) foraging behavior and diet composition; and (7) predation rates on ESA-listed salmonid populations (hereafter referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]). Aerial surveys were conducted throughout the study period to search for incipient Caspian tern colonies in the CPR and once identified, data was collected on the nesting colony metrics listed above. To monitor vital rates of the tern population (see *Chapter 1*) and assess inter-colony movements and dispersal patterns of terns to and from colonies in the CPR, over 3,500 Caspian terns were banded at the Crescent Island nesting colony and at colonies in Potholes Reservoir, plus over 7,000 terns were banded at colonies in the Columbia River estuary and over 2,300 terns at colonies elsewhere in the Pacific Flyway. To better understand the foraging ecology of Caspian terns nesting on Goose Island, a nesting colony located at least 33 km from the Columbia River, a sample of terns were tagged with GPS data-logger tags that provided detailed tracks of complete foraging trips by breeding adult terns from this colony. Finally, as part of a comprehensive effort to assess the efficacy of the *IAPMP* in reducing the numbers of Caspian terns nesting and foraging in the CPR, the movements and habitat use of adult terns that formerly nested at the Goose Island and Crescent Island colonies was evaluated using satellite telemetry. The movements of satellite-tagged terns from each nesting colony were tracked both within and outside the CPR to assess whether management caused terns to shift nesting and foraging outside the Columbia River basin.

The *IAPMP* was implemented during 2014–2018 by the USACE and BOR, with implementation in 2019 by Grant County Public Utility District and the Priest Rapids Coordinating Committee. A metric of this management plan was to reduce the number of Caspian terns in the CPR to less than 40 breeding pairs per colony and to less than 200 pairs for all colonies combined, with the goal of reducing the impact of predation by Caspian terns on ESA-listed juvenile salmonids to less than 2.0% per colony, per salmonid ESU/DPS and to less than 5.0% for all tern colonies combined (i.e. cumulative predation), per salmonid ESU/DPS. The management approach focused on reducing predation through actions that sought to influence Caspian tern abundance, distribution, and behavior in the CPR. Specifically, a primary objective of the *IAPMP* was to redistribute most, if not all, nesting Caspian terns from the Goose Island and Crescent Island colonies to colonies outside the Columbia River basin, thereby reducing predation rates of Caspian terns on ESA-listed juvenile salmonids in the CPR. To reduce the size of the Caspian tern colonies on Goose and Crescent islands, the strategy was to use passive nest dissuasion materials (i.e. stakes, rope, flagging, fences, vegetation, and woody debris) that were installed, planted, or placed on each island in order to eliminate suitable nesting habitat for terns. Active nest dissuasion (i.e. human hazing) was then used to further discourage terns from nesting at Goose and Crescent islands. Monitoring was conducted at these two nesting colony sites, as well as at other active and prospective nesting sites in the CPR, to better understand the response of Caspian terns to management at Goose and Crescent islands and to assess any changes in predation rates on salmonid ESUs/DPSs. When directed by regional resource managers, adaptive management actions were implemented to prevent formation of Caspian tern colonies at other sites within the CPR where the impacts of predation by Caspian terns on the survival of ESA-listed juvenile salmonids was demonstrated and/or expected to be high.

Passive and active nest dissuasion actions on Goose Island and elsewhere in Potholes Reservoir were successful in preventing tern nesting colony formation in each of the last three years of the study (2017–2019). In 2014, the first year of management at Goose Island, 159 breeding pairs nested on a small rocky islet immediately adjacent to the main island, and in 2015 two breeding pairs of Caspian terns nested under the passive dissuasion near the former nesting colony site on Goose Island. Satellite telemetry indicated that most terns displaced from the Goose Island nesting colony remained in the CPR during the 2014–2017 nesting seasons, immediately after implementation of the *IAPMP*. In 2016, an incipient Caspian tern nesting colony (144 breeding pairs) became established on a small island in northern Potholes Reservoir (approximately 5.5 km to the north-northwest of Goose Island) before it ultimately failed due to nest predation and predator disturbance that led to colony abandonment. Throughout the management period (2014–2019), some Caspian terns continued to display high fidelity to Potholes Reservoir as a nesting area. Caspian tern use of Potholes Reservoir was largely limited to roosting in shoreline areas that gradually became exposed during the nesting season as reservoir levels receded. Despite high fidelity of terns to Goose Island and sites in northern Potholes Reservoir, placement of passive nest dissuasion materials, active nest dissuasion (hazing), nest depredation by California gulls (*Larus californicus*), and collection of any tern eggs laid (under permit) were successful in preventing the formation of a Caspian tern nesting colony at Potholes Reservoir during 2017–2019. Prior to management, an average of 386 breeding pairs of Caspian terns nested on Goose Island during 2007–2013, years when nesting colony size data for Caspian terns were available.

The fidelity of Caspian terns to the Potholes Reservoir area is likely due to the species' longevity (up to 26 years), a long history of nesting in and around Potholes Reservoir (over 50 years), and the presence of a large California and ring-billed gull (*L. delawarensis*) nesting colony on Goose Island, both before and during implementation of the *IAPMP*. Caspian terns usually nest in association with large gull colonies and the presence of nesting gulls likely attracts prospecting Caspian terns to Goose Island. Another factor that might explain the strong fidelity of Caspian terns to Potholes Reservoir is its large area, the largest waterbody in southeastern Washington State, and the high density of forage fish that the reservoir supports in most years. Also, the tracking of Goose Island Caspian terns using GPS data-loggers indicated that foraging trips to the mid-Columbia River to prey primarily on juvenile salmonids, especially steelhead smolts, were prevalent. The preference for foraging on the mid-Columbia River by terns nesting at the Goose Island nesting colony suggests that salmonid smolts were a primary prey resource despite the more than 33-km overland commute required to forage there.

Passive nest dissuasion techniques alone were successful in preventing all nesting and roosting by Caspian terns on Crescent Island during the management period (2015–2019). Prior to implementation of the *IAPMP* at Crescent Island (2000–2014), an average of 461 breeding pairs of Caspian terns nested at the Crescent Island nesting colony. The complete abandonment of the Crescent Island nesting colony by Caspian terns beginning in the first year of management

(2015) was somewhat unexpected because terns and gulls had nested consistently on Crescent Island for nearly three decades. A major contributor to the early success of efforts to dissuade Caspian terns from nesting on Crescent Island was the abandonment of the large California gull nesting colony on Crescent Island in 2015. The gull nesting colony on Crescent Island was apparently abandoned due to a combination of major habitat alterations on the island to reduce bare-ground nesting substrate for terns and persistent human hazing on the island. At Goose Island, where the large gull nesting colony persisted despite the use of both passive and active nest dissuasion techniques, Caspian terns were far more persistent in attempting to nest. At Crescent Island, gulls never attempted to nest during the management period (2015–2019), likely due to the use of vertical fence rows and extensive revegetation efforts as tern nest dissuasion techniques and the availability of suitable nest sites for gulls nearby. When California gulls abandoned Crescent Island as a nesting colony site, some, if not most, of these birds established a new nesting colony on Badger Island in the Columbia River, one km upriver from Crescent Island. Similarly, resighting of banded terns indicated that most Caspian terns displaced from the Crescent Island nesting colony relocated to an unmanaged nesting colony site in the Blalock Islands in John Day Reservoir (70 river km downriver from Crescent Island on the Columbia River), where Caspian terns had nested in small numbers intermittently over the previous decade. As with the Goose Island nesting colony, most Caspian terns displaced from the Crescent Island nesting colony remained in the CPR during the 2015–2017 nesting seasons following the elimination of the Crescent Island nesting colony.

Surveys were conducted throughout the CPR to assess the region-wide breeding population of Caspian terns both before and during implementation of the *IAPMP*. Together with satellite-tracking of terns tagged at Goose and Crescent islands, these surveys were effective at identifying all active Caspian tern nesting colonies in each year and for assessing where terns displaced from the managed colonies on Goose and Crescent islands might attempt to nest. Nesting colony size at each active Caspian tern nesting colony in the region was estimated, as was the size of the regional breeding population during each year of the study (2005–2019) with the aim of evaluating the efficacy of the *IAPMP*. The expectation was that that successful implementation of the *IAPMP* would results in reductions in the colony size for terns at Goose and Crescent islands, but also a decline in the regional breeding population for terns in the CPR. During the study period, Caspian terns attempted to nest at a total of 14 different colony sites in the CPR; as many as seven nesting sites were active in 2010 and as few as three colony sites were active in 2019, the last year of study. The total estimated breeding population of Caspian terns in the CPR prior to management (2005–2013) averaged 875 breeding pairs. In 2019, the region-wide breeding population of Caspian terns was 445 breeding pairs, the lowest recorded during the study period, and a 49% decline in the size of the regional breeding population of Caspian terns compared to the pre-management period.

It was expected that some terns displaced from nesting at Goose and Crescent islands might relocate to nest at other colonies within the CPR. Increases in colony size at unmanaged colony sites following the implementation of management was observed at the Blalock Islands on the Columbia River, North Rock in Lenore Lake, and Harper Island in Sprague Lake. The largest increase in nesting colony size for Caspian terns was seen at the Blalock Islands, where colony size increased more than 10-fold in 2015, from a pre-management average of 57 breeding pairs to 677 breeding pairs, the single largest Caspian tern nesting colony ever recorded in the CPR. Resighting of terns at the Blalock Islands nesting colony during the 2015 and 2016 nesting seasons that had been banded on Crescent Island prior to implementation of the *IAPMP* indicated that most of the increase in nesting colony size at the Blalock Islands was due to immigration from Crescent Island. Nevertheless, by 2018–2019 the Blalock Islands nesting colony of Caspian terns had declined to an average of 346 breeding pairs.

To help evaluate the efficacy of the *IAPMP* to reduce predation impacts by Caspian terns on ESA-listed juvenile salmonids, predation rates were compared between the pre-management period (2007–2013) and the management period (2014–2019) at both managed and unmanaged tern colonies in the CPR. Results indicated that the goal of achieving predation rates of less than 2.0% per colony, per salmonid ESU/DPS was met for most, but not all, Caspian tern colonies in the region and for most, but not all, ESA-listed salmonid ESUs/DPSs. In the first breeding season following implementation of the *IAPMP* at Goose Island (2014), the tern nesting colony remained active on a small rocky islet just off Goose Island and terns consumed an estimated 2.9% (95% confidence interval = 1.9–5.1%) of available Upper Columbia River (UCR) steelhead smolts, but predation rates on all other ESUs/DPS were less than 1.0%. In all subsequent years during the management period (2015–2019), the Goose Island tern nesting colony was either nearly or entirely eliminated, effectively reducing predation rates on ESA-listed juvenile salmonids by terns from this nesting colony to zero, or nearly so. In 2016, an incipient Caspian tern nesting colony formed on an unnamed island in northern Potholes Reservoir, a colony that consumed an estimated 4.1% (2.9–6.3%) of UCR steelhead smolts, but again predation rates on all other ESUs/DPSs were less than 1.0%. Adaptive management eliminated the tern nesting colony site in northern Potholes Reservoir in 2017, and since then (2018–2019) no Caspian terns have nested on Goose Island or elsewhere in Potholes Reservoir, effectively eliminating predation by Caspian terns associated with an active nesting colony in this waterbody. The former Caspian tern nesting colony on Crescent Island was successfully dissuaded in all years of the management period (2015–2019), effectively eliminating predation impacts on salmonid smolts from this tern nesting colony and immediately achieving the objective of reducing tern predation rates on all ESA-listed ESUs/DPSs of salmonids to less than 2.0% at this managed nesting colony site.

Concurrent with reductions in Caspian tern nesting colony size and predation rates on smolts at Goose and Crescent islands, survival rates for UCR steelhead were significantly higher on average in the river reaches where terns from these two managed colonies had previously foraged. An unintended consequence of management actions at the Goose Island and Crescent Island Caspian tern colonies, however, was an increase in the size of unmanaged Caspian tern colonies in the region. Many of the Caspian terns dissuaded from nesting at managed colony sites dispersed to unmanaged colony sites in the CPR, where terns continued to consume ESA-listed salmonids. Most notable was the dramatic increase in the size of the Caspian tern nesting colony in the Blalock Islands in John Day Reservoir. Consequently, tern predation rates on ESA-listed salmonids, particularly predation on steelhead, increased significantly downstream of McNary Dam (where terns from the Blalock Islands colony predominately forage) following implementation of management actions at the Crescent Island nesting colony. For instance, since implementation of management actions at the Crescent Island nesting colony, Caspian tern predation rates as high as 2.3% (1.2–4.1%), 8.0% (6.0–11.4%), and 8.2% (5.9–12.4%) on Snake River (SR) sockeye, SR steelhead, and UCR steelhead, respectively, were documented at the Blalock Islands nesting colony. During the last three years of the study period (2017–2019), average predation rates on UCR steelhead and SR steelhead by terns nesting at the Blalock Islands nesting colony were 4.5% (3.4–6.1%) and 3.1% (2.4–4.1%), respectively, above the 2.0% predation threshold specified in the *IAPMP*. Predation rates by Caspian terns nesting at all other unmanaged colonies in the CPR, however, have consistently been less than 2.0% per colony per salmonid ESU/DPS during this same three-year period (2017-2019). For most, but not all, salmonid ESUs/DPSs there was no significant decrease in the cumulative effects of predation by all Caspian tern colonies combined in the CPR following implementation of the *IAPMP*. The two notable exceptions were for UCR steelhead and UCR spring Chinook, where the average annual cumulative effects of predation were reduced from 20.0% (17.3–23.0%) and 3.7% (2.7–4.8%) prior to management to 7.6% (6.5–9.2%) and 1.1% (0.7–1.7%) following management, respectively. Reductions in predation on UCR steelhead and UCR spring Chinook were directly related to the management of Caspian tern colonies in Potholes Reservoir, colonies that disproportionately consumed UCR salmonid populations compared with SR populations and colonies that were eliminated or nearly so.

The results indicate that the *IAPMP* accomplished the objective of reducing predation rates by Caspian terns nesting at the managed colonies on Goose Island and Crescent Island. The overall benefits to survival of ESA-listed salmonid smolts from tern management, however, were offset to some degree by increases in tern predation and decreases in smolt survival associated with terns from managed nesting colonies relocating to unmanaged colony sites within the CPR, most notably at the Blalock Islands where the largest Caspian tern nesting colony in the CPR resided at the end of the study (2019). In the case of SR salmonid populations, populations that were especially susceptibility to predation by Caspian terns nesting at Crescent Island, predation rates by terns nesting on the Blalock Islands were similar to or greater than those of terns nesting at the Crescent Island colony prior to management. Results from predation rate studies indicate that significant reductions in Caspian tern predation and increases in smolt survival have been achieved for UCR steelhead and UCR spring Chinook, but that a basin-wide (as opposed to colony-specific) adaptive management approach will be necessary to fully achieve the stated goals of the *IAPMP*.

Despite the successful elimination of the Caspian tern colonies on both Goose and Crescent islands and a nearly 50% decrease in the total number of breeding pairs of terns nesting in the CPR, there remained 445 breeding pairs of terns in the CPR in 2019, which exceeds the 200-breeding pair threshold specified in the *IAPMP*. The slow dispersal of Caspian terns from the CPR following the elimination of the Goose Island and Crescent Island colonies was unexpected for a species that is known to engage in long-distance dispersal from breeding sites. The philopatry of Caspian terns to the CPR was not, however, due to the failure of adult terns to prospect for breeding opportunities outside the region; satellite-tracking of terns that formerly nested at these two managed colonies indicated that most prospected widely for alternative nesting colony sites. Greater than expected philopatry to the CPR by terns that formerly nested at Goose or Crescent islands was apparently a reflection of a lack of suitable nesting opportunities elsewhere in the Pacific Northwest. Nevertheless, the numbers of Caspian terns nesting in the CPR are slowly declining, as they appear to be throughout the Pacific Flyway.

2.2 Introduction

2.2.1 Caspian Terns Nesting in the Columbia Plateau Region

Caspian terns (*Hydroprogne caspia*) have a well-documented history of nesting in the Columbia Plateau region (CPR) of southeastern Washington and northeastern Oregon (Kitchin 1930, Decker and Bowles 1932, Thompson and Tabor 1981, Penland 1982). Kitchin (1930) noted the first breeding record for Caspian terns in the region at Moses Lake in 1929; however, Gill and Mewaldt (1983) suggested that Caspian terns were established as a breeding species in inland Washington prior to 1929. In 1932, a Caspian tern nesting colony of ~50 breeding pairs was found on an island in the mid-Columbia River in Benton County, Washington (Decker and Bowles 1932). These two colonies disappeared in the mid-1950s and were subsequently replaced by a nesting colony in Potholes Reservoir, Washington, after the dam that created the reservoir was built in the late 1950s (Penland 1982). The numbers of Caspian terns nesting on islands in Potholes Reservoir fluctuated and colony sites shifted several times during the ensuing decades (Penland 1982). Five pairs of Caspian terns nested on Cabin Island above Priest Rapids Dam on the mid-Columbia River in 1975 (Penland 1982). Thompson and Tabor (1981) thoroughly surveyed the Columbia River between Priest Rapids, Washington and Portland, Oregon in 1977 and 1978, and while no Caspian terns were found nesting on Cabin Island, a colony of ~200 breeding pairs was discovered on Three Mile Canyon Island, near Boardman, Oregon. Crescent Island, located in the Columbia River near Wallula, Washington, was created in 1985 from dredge-spoil as a nesting site for waterfowl, and was soon after colonized by Caspian terns (Ackerman 1994). Two small Caspian tern colonies (~20 pairs) were first reported in 1997, one in Banks Lake, Grant County, Washington, and the other in Sprague Lake, Adams County, Washington (Shuford and Craig 2002). As was the case in the Columbia River estuary (see *Chapter 1*), all Caspian tern colonies in the CPR (*Figure 2.1*) were associated with much larger gull colonies (Collis et al. 2002); in the case of the Caspian tern colonies in the CPR, the gull nesting associates were California gulls (*Larus californicus*) and ring-billed gulls (*L. delawarensis*). For more details on the basic biology of Caspian terns and their nesting history throughout the Columbia Basin see *Chapter 1* (see also Cuthbert and Wires 1999).

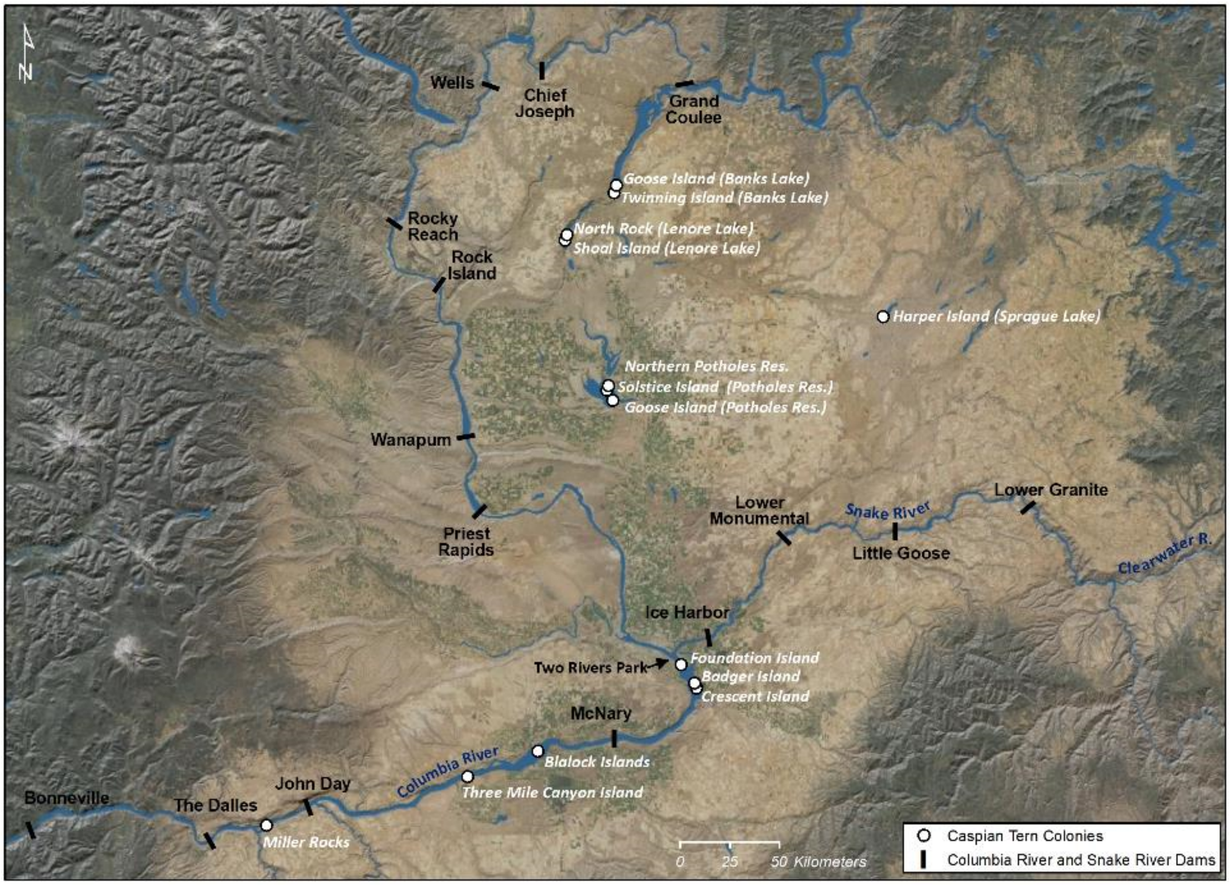

Figure 2.1. Map of the Columbia Plateau region showing the locations of Caspian tern breeding colonies during the study period (1997–2019).

2.2.2 Caspian Tern Nesting at Potholes Reservoir

Nesting by Caspian terns at Potholes Reservoir, located near Moses Lake, Washington, was first documented in the mid-1950s, with subsequent nesting occurring on several different small, low-lying sandy islands in the northern end of the reservoir (Penland 1982). Caspian terns were first recorded nesting on Solstice Island, a 4-acre (1.6-ha) sand dune island located at the northern end of Potholes Reservoir in 2000 (Antolos et al. 2004). Solstice Island was an active tern nesting colony until 2004 when the colony was abandoned due to a flooding event (Adkins et al. 2014). In 2002, Caspian terns began nesting on Goose Island, a 1-ha, steep-sided rocky island located near the southern end of Potholes Reservoir. After the abandonment of the Solstice Island nesting colony in 2004, Goose Island was the only active Caspian tern nesting colony within Potholes Reservoir during 2005–2013, and at an average size of 309 breeding pairs was the second largest Caspian tern nesting colony in the CPR. Only the Caspian tern nesting colony on Crescent Island in the mid-Columbia River was larger, at an average size of 417 breeding pairs (Adkins et al. 2014).

In 2000, passive integrated transponder (PIT) tags from juvenile salmonids (*Oncorhynchus* spp.) out-migrating in the Columbia River were discovered on the Solstice Island tern nesting colony, indicating that some Caspian terns flew at least 50 km from the nesting colony to forage on juvenile salmonids in the Columbia River (BRNW 2001). Subsequent tracking studies verified that Caspian terns were commuting from their breeding colonies on Solstice and Goose islands to the mid-Columbia River where they consumed ESA-listed juvenile salmonids, with the Upper Columbia River steelhead (*O. mykiss*) population (hereafter referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]) being particularly vulnerable to predation by terns nesting at Potholes Reservoir (Maranto 2010, Evans et al. 2012). Estimated predation rates on ESA-listed steelhead from the Upper Columbia River DPS (percentage of available fish consumed) by Caspian terns nesting on Goose Island averaged 15.7% annually and were as high as 22.6% in some years (Evans et al. 2012, BRNW 2014, Evans et al. 2019; see also *Appendix B, Table B21*). Results from this and related studies (see *below*) prompted resource managers to investigate management actions that might reduce the impacts of Caspian terns nesting at the Goose Island colony on the survival of ESA-listed juvenile salmonids.

2.2.3 Caspian Tern Nesting on Crescent Island

Crescent Island, located in the Columbia River just downstream of the confluence with the Snake River near Wallula, Washington, was created in 1985 with dredged material from the nearby Georgia-Pacific mill site as mitigation for loss of shoreline habitat and to enhance waterfowl nesting habitat (Ackerman 1994). The island, which is approximately 7.9 acres (3.2 ha) in area, was colonized almost immediately by nesting California and ring-billed gulls, followed soon thereafter by nesting Caspian terns. While these gull species will nest in habitats with shrubby and herbaceous vegetation, Caspian terns prefer to nest on bare ground or sparsely vegetated substrate (Quinn and Sirdevan 1998) that is soft enough for them to dig small depressions (nest scrapes) where they lay their eggs (Cuthbert and Wires 1999). Since at least the late 1990s, Crescent Island was the site of the largest Caspian tern nesting colony in the CPR (average of 417 breeding pairs during 2004—2010; Adkins et al. 2014).

Diet studies conducted at the Caspian tern nesting colony on Crescent Island indicated that juvenile salmonids were the predominate prey type, representing about 74% of prey items (Antolos et al. 2005). During 2000–2001, it was estimated that Caspian terns nesting on Crescent Island consumed between 465,000 and 679,000 juvenile salmonids annually (Antolos et al. 2005). As with the tern nesting colony on Goose Island, juvenile steelhead were particularly susceptible to predation by Caspian terns nesting on Crescent Island, with juvenile steelhead from the Snake River DPS being more susceptible than those from the Upper Columbia River DPS. Estimated predation rate on juvenile steelhead from the Snake River DPS by Caspian terns nesting on Crescent Island was 4.5% on average, with predation rates as high as 6.1% in some years (see *Appendix B, Table B21*). Although the size of the Caspian tern colonies on Crescent Island and Goose Island (ca. 400 breeding pairs/colony) were the largest in the CPR, combined they were an order of magnitude smaller than the Caspian tern colonies in the Columbia River estuary (over 8,000 breeding pairs; BRNW 2014). Despite this, the impacts of predation by Caspian terns nesting on Goose and Crescent islands on survival of some ESA-listed salmonid ESUs/DPSs were comparable to the impacts of terns nesting in the estuary (Evans et al. 2012), prompting calls from some resource managers to consider management of these two inland Caspian tern breeding colonies (see *below*).

2.2.4 Overview of the Inland Avian Predation Management Plan

An analysis entitled "Benefits to Columbia River anadromous salmonids from potential reductions in avian predation on the Columbia Plateau" (hereafter referred to as the "Benefits Analysis") was conducted to estimate the benefits to salmonid populations (as measured by increases in the population growth rate) associated with reductions in avian predation by piscivorous waterbirds nesting at colonies in the CPR (Lyons et al. 2011b). This investigation was focused primarily on five species of native piscivorous colonial waterbirds, all having historically nested in the region: Caspian terns, double-crested cormorants (*Phalacrocorax auritus*), American white pelicans (*Pelecanus erythrorhynchos*), California gulls, and ring-billed gulls. Preliminary data were collected from 18 breeding colonies used by these five species during 2004–2010 (Adkins et al. 2011) to identify colonies with the greatest potential to reduce survival of juvenile salmonids (smolts) during out-migration through the CPR. Using a variety of measures, including nesting colony size (Adkins et al. 2011), diet composition (Lyons et al. 2011a), and recovery rates of smolt PIT tags (Evans et al. 2011), colonies were ranked based on their potential impact on smolt survival. Using predation rate data based on recoveries of smolt PIT tags at bird colonies and the framework of a simple deterministic, age-structured, matrix population growth model, potential changes in smolt survival due to reductions in predation by piscivorous waterbirds from different colonies were translated into increases in the average annual population growth rate (λ) at the level of salmonid ESUs/DPSs. Estimates were produced for a range of reductions in avian predation and for a range of levels of compensatory mortality (i.e. 0%, 25%, 50%, and 75%).

The Benefits Analysis indicated that management to reduce predation on juvenile salmonids by Caspian terns nesting at the colony on Goose Island in Potholes Reservoir would offer the greatest benefits to salmonid populations, particularly Upper Columbia River steelhead, per managed bird (Lyons et al. 2011b). Management to reduce predation by Caspian terns nesting at two other colonies in the CPR, Crescent Island and the Blalock Islands (both located on the mid-Columbia River), would provide the next largest incremental benefit, with the greatest benefits accruing to the Snake River steelhead DPS. These analytical results along with all of the available data at the time were used by the Action Agencies (i.e. USACE and BOR) to develop a management plan to reduce the impacts of avian predators on smolt survival in the CPR (USACE 2014). This plan had as its general goal/target to reduce the size of or eliminate the Caspian tern colonies on Goose and Crescent islands so that fewer than 40 breeding pairs remained at each colony site. The management strategy was to use passive and active nest dissuasion techniques at each nesting colony site to reduce and limit colony size, with the expectation that displaced terns would relocate to breeding colonies at sites created as alternative tern nesting habitat outside the Columbia River basin (see *Chapter 3*).

It should be noted that at the time the Benefits Analysis was conducted, the estimates of predation rates on juvenile salmonids by double-crested cormorants, American white pelicans, and California and ring-billed gulls nesting at colonies in the CPR were minimum estimates that had not been corrected for species-specific PIT tag deposition probabilities; these estimates have recently been corrected and updated (see *Chapter 6* and *Appendix A2*). The updated estimates indicate that smolt consumption rates by double-crested cormorants and California and ring-billed gulls nesting at certain colonies in the CPR are similar to those of Caspian terns that formerly nested on Crescent and Goose islands; however, Caspian tern predation rates on juvenile salmonids in the CPR, particularly those on steelhead DPSs, remain the highest on a per capita (per bird) basis (see *Chapter 6*, *Figure 6.19*).

Beginning in 2014, the U.S. Army Corps of Engineers – Walla Walla District (USACE) and the U.S Bureau of Reclamation (BOR) implemented management described in the Final Environmental Assessment for the *Inland Avian Predation Management Plan* (*IAPMP*; USACE 2014). The overriding goal of the *IAPMP* was to "reduce predation on salmonids in the inland Columbia River Basin" by management of the two largest Caspian tern colonies in the region, namely Crescent Island on the Columbia River and Goose Island in Potholes Reservoir. This goal was to be achieved by (1) altering the habitat on Crescent Island and Goose Island to create unfavorable nesting conditions for Caspian terns (hereafter referred to "passive nest dissuasion"), (2) conducting daily active hazing at Crescent Island and Goose Island to discourage Caspian terns and gulls from nesting on either island (hereafter referred to as "active nest dissuasion"), (3) collecting a limited number of Caspian tern eggs (under permit) to prevent nesting colony formation at Crescent and Goose islands, (4) developing new or enhanced nesting habitat for Caspian terns (referred to as "habitat enhancement sites") to provide alternative nesting habitat for Caspian terns outside the Columbia River basin (Collis et al. 2012a; see Chapter 3 of this Synthesis Report), (5) conducting adaptive management (i.e. using passive and active nest dissuasion and egg collection) to limit Caspian terns from forming new colonies and/or expanding existing colonies within the CPR (referred to as "at-risk islands"), if allowed, and (6) monitoring the outcomes of management to determine progress. Finally, the *IAPMP* established targets for both Caspian tern colony size (i.e. < 40 breeding pairs/colony) and breeding population size for terns in the CPR (i.e. < 200 breeding pairs for all colonies combined). Similarly, the IAPMP established targets for Caspian tern predation rates on juvenile salmonids, both per colony (i.e. < 2% per ESA-listed salmonid ESU/DPS, per colony, per year, averaged over 3 years) and among all tern colonies in the CPR combined (i.e. < 5% per ESA-listed salmonid ESU/DPS, for all tern colonies combined, per year, averaged over 3 years; see section 2.2.4 of the *IAPMP* for further details on the management plan (USACE 2014).

The *IAPMP* was implemented in two phases. Phase 1 of the *IAPMP* included: (a) passive nest dissuasion and active hazing on Goose Island to discourage Caspian terns and gulls from nesting on the island, and, if needed, limited take of any Caspian tern eggs laid; (b) passive nest dissuasion and active hazing to prevent the formation of an incipient tern colony on Crescent Island, and, if needed, limited take of any Caspian tern eggs laid at the new site; (c) feasibility study to assess the use of willow plantings on Crescent Island as a means to passively dissuade terns and gulls from nesting; (d) passive nest dissuasion and active hazing on at-risk islands, if feasible; and (e) conduct an assessment and complete the supplemental/tiered National Environmental Policy Act (NEPA) analysis needed to create nesting habitat for terns within the Pacific Flyway, but outside the Columbia River basin. Phase 2 of the *IAPMP* included: (f) preparation of suitable nesting habitat for Caspian terns outside the Columbia River basin; (g) feasibility study to investigate large cobble and/or drought-tolerant vegetation plantings as an additional means to discourage tern and gull nesting on Goose Island; (h) passive nest dissuasion, including widespread plantings of native vegetation and/or construction of berm at the island's periphery, and active hazing on Crescent Island to discourage terns and gulls from nesting on their primary colony sites, and, if needed, limited take of any Caspian tern eggs laid; and (i) passive nest dissuasion and active hazing on at-risk islands, if feasible.

The primary focus of this chapter is to evaluate the efficacy of the *IAPMP* in meeting its stated goals, objectives, and targets. To do this we present data collected at Caspian tern colonies in the CPR during the pre-management period (2000-2013) and management period (2014-2019), at both the managed and unmanaged tern colonies. Specifically, we summarize predation rates on specific ESUs or DPSs of ESA-listed, PIT-tagged juvenile salmonids by Caspian terns breeding in the CPR. In addition, we summarize Caspian tern nesting ecology, foraging ecology, diet composition, colony connectivity, and factors limiting colony size and nesting success at colonies in the CPR during 2000–2019.

2.3 Methods

2.3.1 Nesting Habitat Reduction & Active Dissuasion

The general goal of the *IAPMP* was to limit the numbers of Caspian terns breeding at Goose Island and other islands in Potholes Reservoir, and on Crescent Island in McNary Reservoir, to less than 40 breeding pairs (USACE 2014). To accomplish this task, the availability of suitable Caspian tern nesting habitat was nearly eliminated at these sites by using a variety of permanent passive nest dissuasion techniques on Goose Island and elsewhere in Potholes Reservoir, and on Crescent Island on the mid-Columbia River. Passive nest dissuasion materials and/or vegetation covered all areas on the two islands where Caspian terns had previously nested, as well as all areas of open, sparsely vegetated habitat that might be used by ground-nesting Caspian terns. Once Caspian terns arrived at the Goose Island and Crescent Island colony sites to begin breeding each spring, temporary nest dissuasion materials (primarily bamboo stakes, rope, and flagging), active nest dissuasion (i.e. human hazing), and collection (under permit) of any tern eggs laid were also used to dissuade terns from nesting at these sites. See *below* for further details.

2.3.1.1 Passive nest dissuasion at Potholes Reservoir – During 2014-2019, a matrix of concrete pier blocks, rebar, PVC pipe, rope, and flagging was used as the primary (semi-permanent) passive nest dissuasion method to prevent Caspian terns from nesting on Goose Island (*Figure*

2.2; Roby et al. 2015; Collis et al. 2016, 2017, 2018, 2019, 2020). Concrete pier blocks (Mutual Materials; 12’’ x 12’’, 63 lbs. each) were placed in a 3-m x 3-m square grid in nearly all open areas on the island. The center of each concrete pier block was drilled out vertically to accommodate a 48” length of 0.5” (outside diameter) rebar and a 42” length of 0.5” (inside diameter) PVC pipe that was slipped over the rebar. Twisted polypropylene rope (0.25”) was then attached to the PVC pipe at approximately 42” above ground level (AGL) using clove hitch knots, and the rope was further secured to the pipe using UV-resistant cable ties. Ropes were fastened to the vertical PVC pipes to form a 3-m x 3-m grid, with each grid square also bisected diagonally with a section of rope. Four-foot-long pieces of industrial barricade tape (Mutual Industries; 3 mil “polyethylene flagging”) were inserted between the strands of the rope at approximately 3’ intervals and allowed to flutter in the wind as a visual and auditory deterrent to prospecting Caspian terns and gulls. A second layer of rope and flagging was added below the initial layer forming a “double layer” in areas where Caspian terns were considered most likely to attempt nesting, and in all new areas of passive dissuasion installed on the main island during the nesting season. A 3 m to 5 m buffer of double layer passive nest dissuasion was installed around the perimeter of all contiguous areas of passive dissuasion. Each year, repairs to the existing passive nest dissuasion array were conducted, as well as the construction of new passive nest dissuasion arrays in areas used by prospecting terns the previous year, were completed prior to the arrival of Caspian terns to the island (mid-March).

Figure 2.2. Passive nest dissuasion arrays installed on Goose Island in 2019. Semi-permanent nest dissuasion consisted of concrete blocks, stakes, rope, and flagging and was assembled prior to the breeding season. Temporary nest dissuasion (i.e. bamboo stakes installed directly in the ground, rope, and flagging) was installed prior to and during the breeding season as reservoir levels receded, exposing shoreline habitat for prospecting terns. Photo was taken in late July, after reservoir levels had dropped from levels at the onset of the breeding season (April). Adapted from Collis et al. (2020).

In 2018–2019, the feasibility of using planted vegetation on Goose Island as a sustainable passive nest dissuasion technique was evaluated at that site. Experimental plantings using native seed (2018; Hoag 2017) and plugs (2019; Collis et al. 2020) in test plots with and without irrigation were unsuccessful in establishing new vegetation on Goose Island. It was concluded that the substrate of Goose Island, primarily rock with little topsoil, is unsuitable for large-scale revegetation of island.

After the arrival of Caspian terns to Potholes Reservoir each spring, temporary passive nest dissuasion was erected during the breeding season in areas where terns were prospecting for nest sites. On Goose Island, this included shoreline habitat that became available for nesting terns as reservoir levels dropped during the breeding season, as well as the rocky islets located just off the main island that became exposed. In northern Potholes Reservoir, this included several low-lying islands where nesting behaviors by Caspian terns were observed during the 2016–2019 nesting seasons. Immediately following the 2014–2019 breeding seasons (end of July), all temporary nest dissuasion materials and all flagging was removed from the nest dissuasion arrays to prevent those materials from becoming detached over-winter and entering the reservoir. All other components of the nest dissuasion array (i.e. pier blocks, stakes, and ropes) were left on the island over-winter.

2.3.1.2 Passive nest dissuasion on Crescent Island – In 2015, the first year of implementation of the *IAPMP* at Crescent Island, permanent fence rows were installed to create a visual barrier for any prospecting Caspian terns that might land on the ground. A series of parallel fence rows spaced 5 m apart were constructed across the former Caspian tern nesting colony site and nearby sparsely vegetated areas, as well as in a second large, sparsely vegetated area in the southern part of the island (*Figure 2.3*). Additional fence rows were constructed along the perimeter of the island where continuous vegetation was not present at the island's edge, and to bisect other large open areas.

Figure 2.3. Distribution of passive nest dissuasion materials on Crescent Island, Columbia River, during 2015–2019. In 2016, willows and other native vegetation were planted in all open unvegetated habitat. This photo, taken in 2015, does not show the extensive vegetation growth that occurred on the island during 2016–2019. Adapted from Collis et al. (2019).

Fence rows were constructed by driving commercial-grade, painted steel, 6’ fence posts into the ground to depths of at least two feet. Along each fence row, fence posts were spaced no more than 6’ apart, and each fence row was securely anchored at both ends using specially designed angle brackets (Wedge-Loc®). Runs of taut, barbless wire were then secured to the fence posts at ground level, at 18” AGL, and at 36” AGL. Commercial grade knitted material (PAK Unlimited Inc.; 90% privacy screen) was then zip tied to the top and bottom wire strands to create a visual barrier for terns that landed on the ground. Fence rows were constructed across the entirety of the “Primary Dissuasion Area” and much of the “Secondary Dissuasion Areas” identified in the *IAPMP* (USACE 2014). Additionally, twisted polypropylene rope (0.25”) was then attached to the fence posts at approximately 42” AGL using clove hitch knots. Ropes were fastened to alternating fence posts diagonally between two adjacent fence rows, and then 4’ lengths of industrial barricade tape (see *above*) were inserted between strands of the rope at approximately 1-m intervals.

Finally, in open areas where Caspian terns were less likely to prospect for nest sites due to the proximity of mature woody vegetation, passive dissuasion consisted of stakes, rope, and flagging or placement of woody debris. Ropes and flagging were deployed in a 3-m x 3-m square array using 6’ steel fence posts driven into the ground, and with diagonal strands of rope and flagging bisecting each square. A double layer of rope and flagging was deployed at or near the high waterline around the island’s perimeter, where fence rows could not be constructed. Woody debris was collected from downed dead trees and was placed primarily on the west side of Crescent Island, where nest prospecting was considered possible but unlikely, and in open areas below the high waterline. Like was done at Potholes Reservoir, all temporary nest dissuasion materials and all flagging were removed from the nest dissuasion arrays following each breeding season. All other components of the nest dissuasion array (i.e. fencing, stakes, and rope) were left on the island over-winter.

In 2016, there was widespread planting of native vegetation, felling of non-native Russian olive (*Elaeagnus angustifolia*) trees, and subsequent spreading of woody debris as additional permanent passive nest dissuasion on Crescent Island (*Figure 2.4*). An exclusion fence made from chicken wire was erected and maintained around the planted areas to prevent American beaver (*Castor canadensis*) herbivory. The planting and subsequent growth of willows and other native vegetation on Crescent Island was successful in eliminating all open habitat on Crescent Island, rendering unnecessary the placement of temporary passive nest dissuasion after the arrival of terns to the site each spring.

In 2019 and 2020, all the permanent passive dissuasion materials that were placed on Crescent Island (i.e., fencing, stakes, and rope) were removed, since they were no longer needed to dissuade terns from nesting on that island. The permanent passive nest dissuasion materials installed on Goose Island (i.e., cinder clocks, stakes, and rope) remain on the island and will be reused to prevent tern nesting in 2021, and perhaps beyond.

Figure 2.4. Aerial imagery from June 2015 (top) and June 2016 (bottom) showing the results of vegetation manipulation that included removal of Russian olive and willow planting on and around the of the historical Caspian tern colony area at Crescent Island.

2.3.1.3 Active nest dissuasion – In accordance with the *IAPMP*, active nest dissuasion methods (also referred to as "active hazing") were used to supplement passive dissuasion measures to further deter nesting attempts by Caspian terns on islands in Potholes Reservoir and on Crescent Island (USACE 2014). Active nest dissuasion was conducted to disrupt nesting attempts by Caspian terns, including (1) island walk-throughs, (2) approaching the shoreline of the island by boat, (3) use of a green laser during low light conditions, (4) waving a 10' PVC pole with caution tape tied to each end, (5) flying a peregrine falcon kite on the island, (6) destruction of all Caspian tern and gull nests not containing eggs, and, as a last resort, (7) collection under permit of any Caspian tern eggs that were discovered.

Observation blinds were installed on both Goose Island and Crescent Island, adjacent to the former Caspian tern nesting colony sites. The blinds were used to monitor Caspian tern use of the former tern nesting colony sites and surrounding areas used by gulls, areas that cannot be readily seen from a boat. Beginning with the arrival of both Caspian terns and gulls intent on nesting at these sites, hazing activities were conducted daily or weekly (schedule depending on tern activity at each site) through July, weather permitting. These hazing activities were focused primarily during the dawn and dusk periods (starting 30 min before dawn and ending at least 30 min after dusk), or whenever it was determined to be most effective in keeping Caspian terns off the islands. Efforts were made during this time to prevent Caspian terns from using the managed sites as an overnight roost. The methods and duration of active hazing sessions were adjusted based on tern numbers and breeding activities observed on islands in Potholes Reservoir and on Crescent Island. A tent camp and a houseboat were used for overnight stays by field crews at Goose Island and Crescent Island, respectively.

The widespread establishment of active gull nests (those containing eggs) precluded island walk-throughs on Goose Island because it would cause nesting failure by gulls, which was not allowed under the permit issued by the U.S. Fish and Wildlife Service [USFWS]). After the cessation of island walk-throughs, the primary techniques used to actively dissuade prospecting Caspian terns were the use of a green laser (Agrilaser®; LEM 50) during low-light conditions and approaches by boat to the island shoreline to flush Caspian terns that were prospecting there. During low light conditions, green lasers were used to haze Caspian terns that were prospecting on Goose Island from a distance, without disturbing gulls attending nests nearby. Once reservoir water levels began to recede in early June, foot travel around Goose Island was restored, but limited to the shoreline to avoid disturbing nesting gulls on the upland portion of the island. Because no gulls or waterbirds other than Caspian terns were identified nesting on islands in northern Potholes Reservoir, island walk-throughs and motorboat approaches were the only hazing techniques used to dissuade prospecting Caspian terns in that area.

When Caspian tern eggs were laid on islands in Potholes Reservoir despite all nest dissuasion efforts, a take permit issued to the USACE and BOR by the USFWS allowed for the collection of tern eggs. The collection of Caspian tern eggs was intended to enhance the prospects for successfully dissuading Caspian terns from forming breeding colonies at managed sites.

2.3.2 Nesting Ecology

To evaluate the *IAPMP*, monitoring of Caspian tern colonies was conducted at managed nesting colony sites (i.e. Crescent Island, Goose Island, and islands in northern Potholes Reservoir) and unmanaged nesting colony sites (i.e. Miller Rocks, Three Mile Canyon Island, Blalock Islands, Badger Island, and Foundation Island on the Columbia River; Harper Island in Sprague Lake; Shoal Island and North Rock in Lenore Lake; and Twinning Island and Goose Island in Banks Lake; *Figure 2.1* above). These Caspian tern nesting colony sites were all the sites known to support an active Caspian tern nesting colony in the CPR during the study period (2005–2019). Monitoring at the managed nesting colony sites was conducted both before and after implementation of management; management was first implemented on Goose Island and islands in northern Potholes Reservoir in 2014, and on Crescent Island in 2015. Monitoring of the nesting ecology of Caspian terns in the CPR during the study period included: (1) fixed-wing aerial surveys to locate all active tern nesting colonies in the region; (2) ground-based surveys to assess status, including nesting chronology (i.e. dates of first egg, first chick, first fledging, and peak in colony attendance) and colony attendance (i.e. number of adults on-colony); (3) ground-based and aerial surveys to estimate peak nesting colony size (i.e. number of breeding pairs), nesting success (i.e. average number of young raised to fledging per breeding pair), and factors that limit colony size and nesting success (e.g., availability of suitable nesting substrate, nest site encroachment by other colonially nesting birds, and avian and mammalian predation); and (4) banding, tagging, and resighting of Caspian terns to assess inter-colony movements and colony connectivity. See *below* for a brief description of these methods and *Appendix A* for more details on methodologies to estimate nesting colony size and predation rates on juvenile salmonids that were used in this study.

2.3.2.1 Aerial surveys to locate tern colonies – Reconnaissance aerial surveys were conducted early in the incubation period (late April), to determine the distribution of Caspian terns (both nesting and roosting) throughout the Columbia Plateau region. These surveys were conducted over two consecutive days from a manned fixed-wing aircraft that covered the Columbia River from Bonneville Dam to Chief Joseph Dam, the Snake River from the confluence with the Columbia River to the mouth of the Clearwater River, and waterbodies off the Columbia and Snake rivers that are within tern foraging range (~90 km) of the Federal Columbia River Power System (FCRPS; *Figure 2.5*).

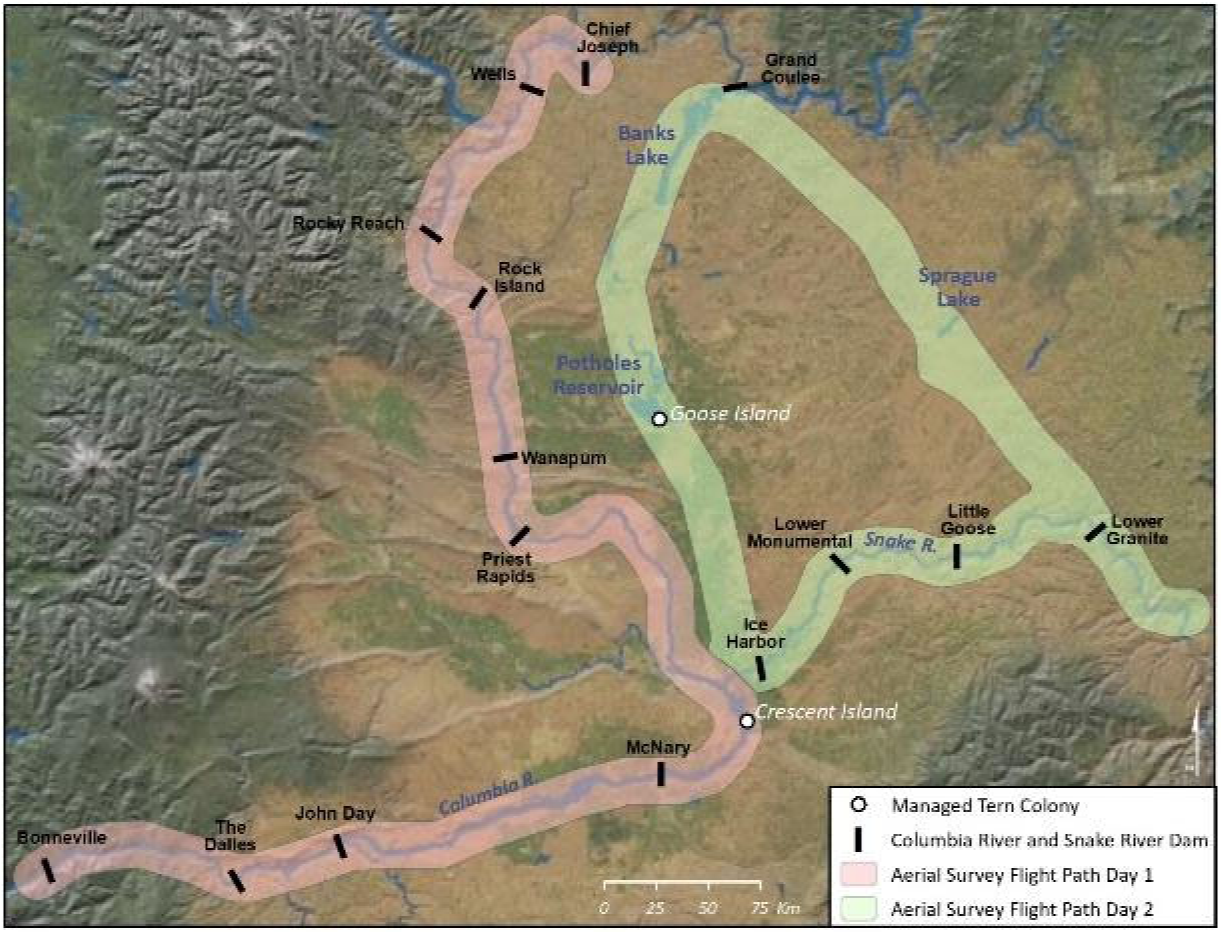

Figure 2.5. Flight path followed during aerial reconnaissance surveys conducted to locate active Caspian tern colonies in the Columbia Plateau region. Adapted from Collis et al. (2019).

At each locale where Caspian terns were observed, data was collected on the site (i.e., GPS coordinates, island name/description, habitat/substrate type), birds (i.e., count of adult terns, presence of nesting associates [e.g., gulls]), and bird behavior (i.e. nesting or loafing). When Caspian terns were observed in upland habitat on islands and on substrate that was potentially suitable for nesting, oblique aerial photographs were taken using a digital SLR camera with an image-stabilizing, zoom lens. When in-flight observations or post-flight inspection of digital images suggested the presence of a potential Caspian tern nesting colony, ground- or boat-based surveys were conducted at the site to assess breeding status and other colony metrics (see *below*).

2.3.2.2 Nesting chronology & colony attendance – Field crews visited each site that was identified as a potential Caspian tern nesting colony immediately after the completion of reconnaissance aerial surveys (see *above*). Most island sites were visited by boat, while some sites inaccessible by boat (i.e. Harper Island in Sprague Lake and North Rock and Shoal Island in Lenore Lake) were observed from the mainland (see *below*). At the larger Caspian tern nesting colonies in the region (i.e. Crescent Island, Goose Island, Blalock Islands, and Three Mile Canyon Island) observation blinds were constructed at the edge of each so that nesting birds could be observed by researchers without causing disturbance. All other tern colonies in the region were observed from a sufficient distance (from boats or on land) that nesting birds were not disturbed. Visits to active tern colonies occurred daily or weekly (depending on the site) throughout the breeding season (April–July) to assess nesting chronology, colony attendance, and other colony metrics (see *below*). Nesting chronology at Caspian tern colonies was monitored by recording the date when the first tern egg was seen in a nest scrape on-colony (i.e. first egg), the date when the first hatchling chick was seen (i.e. first chick), and the date when the first fledgling tern capable of sustained flight was observed (i.e. first fledging). In addition, field crews counted the numbers of adult Caspian terns (i.e. colony attendance) and attended tern nests throughout the breeding season. Late in the incubation period, when the numbers of adult terns attending nests with eggs was at its peak (i.e. peak colony attendance), field crews conducted a census of the tern nesting colony to obtain a precise and accurate estimate of colony size (i.e. number of breeding pairs; see *below*).

2.3.2.3 Nesting colony size, nesting success, & limiting factors – The size of Caspian tern nesting colonies (i.e. number of breeding pairs) in the CPR was estimated from counts of nesting birds in digital aerial photographs or ground counts of attended nests late in incubation (mid-May), when nesting colony size was at its peak (Bullock and Gomersal 1981, Gaston and Smith 1984). Nesting colony size estimates from digital photography were either direct counts of all incubating birds or direct counts of all adults on-colony corrected using simultaneous ground counts of incubating and non-incubating birds in seven 5 m x 5 m plots. All ground counts were made from an observation blind or a distant vantage so that the birds on-colony were not disturbed. Except for nesting colony size estimates for Goose Island in Potholes Reservoir during 2005, which were from Maranto et al. (2010), all nesting colony size estimates presented here were generated as part of this studies conducted by the OSU/RTR research team. Nesting success (i.e. average number of young raised per breeding pair) was estimated at some Caspian tern colonies in the CPR by dividing a count of all juveniles present on-colony just prior to fledging by the number of breeding pairs estimated in late incubation. See *Appendix A.1.2* for a detailed description of these methods.

Tern nesting colony size and nesting success can be limited by a variety of factors, including availability of suitable nesting habitat, food availability, disturbance, nest predation, and severe weather. Field crews collected systematic data and anecdotal observations pertaining to some of these factors during each nesting season. Systematic data collection included measures of (1) gull predation rates on tern eggs and chicks, (2) rates of disturbance to the tern nesting colony, and (3) rates of gull kleptoparasitism of fish from adult terns transporting bill-loads to their nests. Anecdotal observations included (1) rain and/or windstorm events that caused loss of tern eggs and/or chicks through exposure or inundation of parts of the nesting colony, (2) major disturbance or predation events on-colony due to terrestrial predators (e.g., American mink [*Neovison vison*]) or raptors (e.g., great horned owls [*Bubo virginianus*]), and (3) human disturbance (e.g., recreational boaters, fishers).

2.3.2.4 Colony dispersal & connectivity – The movements and connectivity among nesting colonies for Caspian terns both within and outside the CPR were evaluated by marking and tagging terns at colonies in the CPR. Data on nesting colony connectivity, both within and outside the CPR, was particularly relevant in the context of management to reduce or eliminate the colonies at Goose Island and Crescent Island. Two data sources were used in this in evaluation, namely (1) resightings of previously banded Caspian terns (BRNW annual reports) and (2) results from a satellite telemetry study (Roby et al. 2019a, 2019b). Brief descriptions of the methods used to collect these data are presented separately *below*.

Band resightings: During 1997–2016, Caspian terns were banded at numerous breeding colonies for the species in the Pacific Flyway to measure survival and other vital rates, post-breeding dispersal rates, and movement rates among colonies (BRNW annual reports). During 2000–2015 (years when banding occurred in the CPR), a total of 3,539 Caspian terns, including both adults and flightless young, were banded at the Crescent Island and two islands (Goose Island and Solstice Island) in Potholes Reservoir (*Table 2.1*). In addition, 7,247 Caspian terns were banded at two different colonies in the Columbia River estuary and 2,362 Caspian at other colonies outside the Basin from Alaska to the San Francisco Bay area.

Table 2.1. Numbers of Caspian terns banded under federal permit in the Columbia Plateau region during 2000–2015. In 2000–2004, most banded terns were also banded with a field-readable unique color combination of five acrylic plastic leg bands. Starting in 2005, field readable Darvic® leg bands engraved with a unique alphanumeric code were also used. Caspian terns were banded as chicks or adults at Crescent Island (CSI) and at two colonies in Potholes Reservoir (PTI), Solstice Island (2000–2004; 172 individuals) and Goose Island (2006–2015; 795 individuals). -- denotes years when banding did not occur.

	Metal Band only				**Plastic Color Bands**			**Alphanumeric Band**				
	CSI	CSI	PTI	PTI	CSI	CSI	PTI	CSI	CSI	PTI	PTI	
Year	(Adult)	(Chick)	(Adult)	(Chick)	(Adult)	(Adult)	(Chick)	(Adult)	(Chick)	(Adult)	(Chick)	**Total**
2000	--	1	--	--	--	68	44	--	--	--	--	**113**
2001	--	10	--	28	--	87	88	--	--	--	--	**213**
2002	--	12	--	--	--	60	--	--	--	--	--	**72**
2003	--	--	--	--	16	100	21	--	--	--	--	**137**
2004	--	9	10	--	28	214	--	--	--	--	--	**261**
2005	--	14	--	--	--	150	--	57	--	--	--	**221**
2006	2	11	--	20	--	--	--	59	60	--	40	**192**
2007	--	10	--	14	--	--	--	--	203	--	124	**351**
2008	--	13	--	--	--	--	--	--	86	--	--	**99**
2009	--	20	--	17	--	--	--	--	132	--	124	**293**
2010	--	29	--	--	--	--	--	--	186	49	--	**264**
2011	--	12	--	11	--	--	--	32	147	61	124	**387**
2012	--	4	--	2	--	--	--	61	85	54	36	**242**
2013	--	16	--	36	--	--	--	67	153	53	116	**441**
2014	--	5	3	--	--	--	--	--	147	44	--	**199**
2015	--	--	1	--	--	--	--	34		19	--	**54**
Total	**2**	**166**	**14**	**128**	**44**	**679**	**153**	**310**	**1,199**	**280**	**564**	**3,539**

Each banded tern was first banded under federal permit with a metal leg band provided by the U.S. Geological Survey - Bird Banding Lab. During 2000–2005, most terns banded at colonies in the CPR (91% or 876 individuals) were also marked with a unique color combination of five acrylic plastic leg bands that could be used to identify banded individuals in the field (i.e. on colonies or at roosts). Young terns that were too small to be banded with five plastic leg bands were banded with only one federal metal leg band. Some terns were able to subsequently shed one or more of their five acrylic leg bands, making them no longer identifiable to the level of the individual. Consequently, starting in 2005 for adult terns and 2006 for young flightless terns, Darvic® plastic leg band engraved with a unique alphanumeric code used on one leg and a federal numbered metal band and two acrylic colored leg bands on the other leg to band terns in the CPR (n = 2,353 terns banded with Darvic®). This new banding protocol for individually marking Caspian terns was longer lasting and more reliable for field identification of individuals at a distance than the method using a unique color combination of five acrylic leg bands.

As part of these banding studies, tern chicks that were captured on-colony prior to fledging by herding flightless young into holding pens. Adult terns were captured on-colony for banding using monofilament noose mats placed around active nests late in the incubation period, or by pneumatic net guns prior to the egg-laying period. Once captured, terns were immediately transferred to holding crates until they were banded and released.

Banded Caspian terns were resighted at various locations throughout the Pacific Flyway. At tern colonies monitored as part of this and other related studies, banded terns were resighted by colony monitors using binoculars and spotting scopes throughout each breeding season. Resightings of banded terns at other locations were reported to us through the project web page (www.birdresearchnw.org), by phone, or by e-mail, either via the US Geological Survey – Bird Banding Lab or directly from the public. Using a comprehensive database containing banding and resighting records for Caspian terns dating back to 2005, assessments were made of (a) colony connectivity among sites in the Columbia Plateau region; (b) emigration rates of banded individuals from nesting colony sites in the CPR to nesting colony sites outside the region; and (c) immigration rates of banded individuals from nesting colony sites outside the CPR to sites in the region. Systematic resightings of banded Caspian terns were used to evaluate the efficacy of the *IAPMP*.

The high degree of connectivity among Caspian tern colonies in the Pacific Flyway is reflected in the movements of banded individuals among colonies and regional sub-populations (Suzuki 2012). The banding dataset was evaluated in this chapter to determine if managed reductions in nesting habitat on Goose and Crescent islands as part of the *IAPMP* would result in Caspian terns with previous breeding experience in the CPR to leave the region and attempt to nest elsewhere, as well as banded birds in the region that might shift from being a breeder to being a non-breeder, depending on an individual's physiological condition or opportunities for breeding (Dobson and Jouventin 2010, Giudici et al. 2010). To accomplish this, band resighting data during the 2011–2016 breeding seasons were used and set criteria were applied to identify "breeders" (i.e. ≥ 5 resightings at a nesting colony over a period ≥ 3 weeks and/or confirmed breeding status) and "non-breeders" both within and outside the CPR. Transitions between consecutive breeding seasons in the breeding status of banded individuals were investigated, namely shifts between four different breeding statuses identified in this study, which were: "breeder in the CPR", "non-breeder in the CPR", "breeder outside the CPR", and "non-breeder outside the CPR." See Schniedermeyer et al. 2020 for details on study methods.

Satellite telemetry: To better understand the response of Caspian terns to management at Goose and Crescent islands (i.e. how management affected dispersal and foraging behavior [see *below*]), a satellite telemetry study during 2014–2018 was conducted (Roby et al. 2019a, 2019b). A Net Blaster™ compressed air-powered net launcher (Wildlife Control Supplies, East Granby, CT) was used to capture adult Caspian terns soon after their arrival in spring at the nesting colony sites on Goose Island in Potholes Reservoir in 2014 and 2015 and on Crescent Island in McNary Reservoir in 2015. Social attraction (i.e. decoys and audio playback) was used to attract pre-breeding adult Caspian terns to land within the capture footprint of the net launcher as they returned to the nesting colony site from their wintering range in March and April. Once captured, adult terns were temporarily held in crates prior to applying leg-bands and attaching satellite tags. Captured adult terns were fitted with an U.S. Geological Survey metal leg-band and two small colored acrylic leg bands on the left leg, and a field readable Darvic® leg band with a unique alphanumeric code on the right leg. To track captured individuals, Platform Transmitting Terminal (PTT) tags were used that transmit to the ARGOS satellite network and employ the Doppler Shift effect to estimate tag location, with precision varying from several 10s of meters to a km or more, depending on satellite position during location fixes. PTT tags were attached to healthy adult terns weighing ≥ 550 g using either a leg-loop or backpack harness of 0.25-inch tubular Teflon tape (Bally Ribbon Mills, Bally, PA). Tags (model: PTT-100-12g solar; Microwave Telemetry Inc., Columbia, MD) were factory-programmed to operate on a 32-hour duty cycle, with 6 hours "on" and 26 hours "off," transmitting at a 60-second repetition rate during the "on" period of each cycle. Each tag included a small solar panel to recharge the battery, which allowed for extended tag life and transmission of location data during daylight or nighttime hours. Tags weighed 12.4–12.9 g, not including harness materials, and were ≤ 2.3% of body mass for all individual tagged terns (body mass of tagged terns ranged from 560 g to 720 g). In 2014, PTT tags were attached to 30 adult Caspian terns that were captured at the nesting colony site on Goose Island in Potholes Reservoir prior to the onset of nesting. In 2015, prior to the breeding season, an additional 18 PTT tags were deployed on terns captured at Goose Island and deployed another 28 PTT tags on terns captured at Crescent Island in McNary Reservoir on the Columbia River.

Raw position fixes of tagged terns were reported daily by the Argos System (CLS America, Inc., Largo, MD). The Douglas Argos-Filter Algorithm (Douglas et al. 2012) was used to remove spurious locations from the raw data, using criteria like other seabird telemetry studies (e.g., Courtot et al. 2012). Location precision of tags using the ARGOS satellite network can range from ≤ 150 m (Class 3) to > 1 km (Class 0); however, we removed any location with > 1 km estimated accuracy from the data set used for analyses following the filtering process. Consistent with the management objective of minimizing Caspian tern predation on out-migrating steelhead smolts, the data analyses were focused on the period when most steelhead smolts out-migrate through the CPR (i.e. the 95% runtime, as measured at Bonneville Dam; FPC 2014, 2015, 2016, 2017, 2018) during each year (Roby et al. 2019a). In addition, the data set was limited to those individuals that had actively transmitting tags for ≥ 75% of the duration of each yearly steelhead runtime. The 2018 data from individuals tagged at Goose Island was censored because only three individuals tagged at Goose Island were still providing location fixes in 2018 (i.e. the fifth breeding season following initiation of management actions at that colony site), which was considered an inadequately small sample size.

The effects of *IAPMP* management actions on Caspian tern use of nesting colony sites both within (*Figure 2.1* above) and outside the CPR was examined by creating a circular buffer with a 1-km radius around each Caspian tern colony of interest using ArcMap GIS software. The proportion of high-quality locations were calculated for tagged bird that fell within the buffered area around colonies each year during the breeding season. Using this measure of nesting colony use as a response, a linear mixed-effects models in the R statistical software package (R Core Team 2018; LME4 package [Bates et al. 2015]) was applied, with the individual as a random effect. Seven *a priori* explanatory variables were selected based on their presumed influence on philopatry to nesting sites, the same seven explanatory variables that were selected to investigate the effects of management on foraging behavior (see *below*) and evaluated candidate models that included a combination of those variables as fixed effects. The *a priori* explanatory variables used in model selection were: (1) nesting colony where tern captured, (2) year when tern captured, (3) sex, (4) number of years since management initiated, (5) starting date of the steelhead smolt out-migration period, (6) duration of the steelhead smolt out-migration period, and (7) the interaction between starting date and duration of the steelhead smolt out-migration period. Model selection was based on the Akaike Information Criterion corrected for small sample sizes (AIC_c) to choose among candidate models and considered any model with a $\Delta AIC_c \leq 2$ to be a supported candidate model; model averaging was used of the supported candidate models to calculate parameter estimates (Burnham and Anderson 2002). See Roby et al. (2019b) for further details on this study.

The satellite telemetry dataset was used to explore the dispersal capacity and nesting colony network structure of Caspian terns in the Pacific Flyway of North America. See Roby et al. (2019b) for details of the network analyses conducted using the satellite telemetry data set. In brief, 65 active or historical Caspian tern colonies were identified within the Pacific Flyway, ranging from Baja California, Mexico, to south-central Alaska, drawing from prior flyway-wide census efforts (Suryan et al. 2004, Peterson et al. 2017, Peck-Richardson et al. 2019). A 10-km buffer was placed around each nesting colony using ArcMap GIS software. The intersections of interpolated tracks with colony buffers were used to determine which nesting colonies each tern visited. Terns were considered resident at a nesting colony if there were consecutive intersected locations at a colony. Any time a tern had locations at a different nesting colony it was treated as a movement between colonies.

A network was constructed using the tern tracking and colony datasets. Colonies that were visited by terns during this study became the vertices in the network. Tern movements between colonies were the edges in the network. Edges were given weights based on the total number of terns that moved between each pair of colonies at least one time during the study. The network was created using the 'igraph' package (Csárdi and Nepusz 2006) for R (R Core Team 2018). Colony connectivity within the network was quantified using a suite of network centrality measures: number of terns, neighborhood size, betweenness, and distance centrality. The number of terns that visited each nesting colony in the network during this study was investigated. All measures were calculated using the 'igraph' package in R (Csárdi and Nepusz 2006).

Movements of terns within breeding seasons and across multiple years were quantified. For this analysis, tracks were only included in the analysis if the tagged tern was tracked until the end of August for that year. The mean number of colonies that terns visited within each breeding season and across the entire tracking period was also investigated. Finally, the average number of colonies that each tern visited in single breeding seasons relative to all the colonies that tern visited over multiple years (its personal network) was evaluated; for this measure, only terns that were tracked for at least two years were included.

Timing of prospecting behavior can indicate what factors influence breeding site selection. The timing of prospecting behavior was evaluated by comparing the average number of colonies a tern would visit each month during the breeding season, April to August. The difference in the number of colonies visited each month was investigated using a generalized linear mixed effects model (GLMM). Bird identity was included as a random effect and the relationship between month and number of colonies visited was modeled using a Poisson distribution with a log link function. Model estimates were back transformed to the original scale for reporting results. The GLMM was run using the R package 'lme4' (Bates et al. 2015) and confidence intervals were calculated using the 'emmeans' package (Lenth 2016).

2.3.3 Foraging Ecology

As part of this study, data on the foraging ecology of piscivorous colonial waterbird species in the CPR was collected, with a focus on nesting Caspian terns. Initial investigations (prior to 2014) were designed to better understand the foraging behavior, diet composition, smolt consumption, and predation rates on salmonid smolts of piscivorous waterbirds to identify those waterbird species and colonies that posed the greatest risks to juvenile salmonid survival in the region. This baseline information was used to develop the *IAPMP* (see *above*; USACE 2014), which identified the two largest Caspian tern colonies in the region (i.e. the nesting colony on Goose Island in Potholes Reservoir and the nesting colony on Crescent Island in the Columbia River) as having the greatest impacts to smolt survival, as compared to other waterbird species and colonies, and which warranted management action to reduce those impacts (see *above*; USACE 2014). During the management period (2014–2019), data on foraging ecology of Caspian terns were collected to evaluate the efficacy of the *IAPMP* for reducing predation rates on juvenile salmonids at both the managed colonies (i.e. Goose and Crescent islands) and unmanaged colonies in the CPR (*Figure 2.1* above). See *below* for a brief description of these methods and *Appendix A* for more details on methodologies to measure predation rate for this study.

2.3.3.1 Foraging behavior – Foraging behavior of Caspian terns in the CPR was investigated using a variety of tracking techniques, including (1) tracking Caspian terns nesting on Goose Island in 2013 by attaching Global Positioning System (GPS) tags (BRNW 2014, Appendix C), (2) tracking Caspian terns nesting at Goose and Crescent islands during 2014–2015 by attaching satellite tags (Roby et al. 2019a), and (3) recovery of PIT tags from juvenile salmonids out-migrating in the Columbia and Snake rivers on Caspian tern colonies following the 2012 and 2014 breeding seasons (Evans et al. 2016a). These methods are presented separately *below*.

GPS tracking of Caspian terns: In 2013, prior to implementation of the *IAPMP*, GPS transmitters were used to assess how Caspian terns nesting at Goose Island in Potholes Reservoir use different locations for foraging during the breeding season (see BRNW 2014, Appendix C for a detailed description of methods). Caspian terns were captured for tagging using noose mats placed around active nests late in the incubation period. A total of 28 adult Caspian terns were GPS-tagged; 24 adults were tagged with Advanced Telemetry Systems (ATS) transmitters (44 x 22 x 16 cm, 14 g; Concord, CA) and 4 adults were tagged with Skorpa Telemetry transmitters (44 x 22 x 15 cm, 17 g; Aberfeldy, Scotland). GPS units were attached to the base of the four central rectrices (tail feathers) with glue and cable ties. Transmitters and attachments weighed an average of 2.2% of tern body mass (maximum = 2.7%).

GPS transmitters were programmed to record positions at 4-min intervals from 05:00 to 21:00 PDT. All transmitters were programmed to begin collecting data at 05:00 on 21 May. Some transmitters were turned off on 23 May and 24 May to spread data collection over a longer period. Data were downloaded remotely when terns returned to the nesting colony. GPS transmitters would fall off when terns entered the post-breeding season molt and molted their rectrices.

GPS data were filtered to remove missed locations and locations that would require flight velocities greater than 80 km/hr. 80 km/hr was chosen as a threshold for excluding location points based on visual examination of a histogram of all velocities, and by mapping locations with velocities greater than 70 km/hr. There were few locations requiring flight velocities greater than 80 km/hr. When mapped these locations appeared to represent an anomalous change in direction or velocity relative to the previous and subsequent locations.

Locations were considered active if there were three or more consecutive locations where the tern moved more than 100 m and the individual was at least 500 m from the nesting colony. Velocity and turning angle at each active location were used as measures of movement characteristics (Calenge et al. 2009). Velocity was calculated as the distance between the current location and the next location, divided by the time between locations. Turning angle was calculated as the change in direction between the previous and subsequent locations. Values of velocity ranged from 0 to 80 km/hr, and values of turning angle ranged from 0^o to 180^o.

For all active locations, k-means cluster analysis was used to identify patterns of movement that represent distinguishable behavioral states (Van Moorter et al. 2010). Cluster analysis uses multivariate data (e.g., velocity and turning angle) to identify clusters of observations with similar characteristics (Steinley 2006). Analysis was performed for all possible numbers of clusters between 1 and 10, and the gap statistic (Tibshirani et al. 2001) was used to identify the optimum number of clusters in the data set (Van Moorter et al. 2010). The gap statistic estimates the number of groups within a data set by comparing the change in within-cluster dispersion for each number of clusters to the dispersion expected from simulated reference null distributions (Tibshirani et al. 2001). Range standardization on velocity and turning angle were performed before analysis so that differences in range between variables would not affect the contribution of each variable to the clustering (Steinley 2006). The gap statistic was calculated from 50 simulated data sets and the tolerance level was set to 2; higher tolerance values increase the evidence necessary to include additional clusters (Van Moorter et al. 2010).

Foraging trips were defined as any set of five or more consecutive locations at least 500 m from the nesting colony. Within trips, foraging bouts were defined as three or more consecutive foraging locations. For each trip, the primary destination (the furthest water body visited during a trip) was identified. The proportion of terns using different foraging areas, how many foraging destinations individual terns used, and the number of trips made per day in relation to primary trip destination was also investigated. All areas within 20 km of the nesting colony were treated collectively as the Potholes Reservoir area. Four identified trips with a maximum distance from the nesting colony of less than 2.5 km were excluded, because video footage of the nesting colony during two of these trips showed that these trips did not occur. Trips were considered complete if all gaps of more than three locations (16 min) did not add up to more than 25% of total trip duration.

For each trip, the maximum distance from the nesting colony (km), total trip duration (min), the number of foraging bouts, and time spent in different behaviors (i.e. foraging, commuting, and resting) were measured. A mixed-effects models was used to compare trip duration and time spent in different behaviors as a function of primary destination. Tern identity was included as a random effect to account for lack of independence among trips made by the same individual. A variance coefficient was included to allow variance to change with maximum distance from the nesting colony. The number of foraging bouts per trip is count data; therefore, a generalized linear mixed model (GLMM) with a Poisson distribution was used in the analysis of the number of foraging bouts. Plots of standardized residuals against fitted values and normal quantile-quantile plots were used to confirm that these models met the assumptions of normality and homogeneity of variance (Zuur et al. 2009). The approximate estimate of over-dispersion in the Poisson GLMM was <1, indicating that over-dispersion was likely not an issue in this model.

Spearman’s correlation tests were used to examine the relationships between proportions of time spent in four different behaviors each day: on colony, foraging, commuting, and resting off colony. Average time spent in these four behavioral categories for each tern were calculated, based on all complete days of data collected for that individual.

Three types of core use areas (i.e. foraging, resting, and commuting) were estimated using the kernel density tool in ArcGIS (ArcMap v. 10.0). Estimates for core foraging and resting areas were each based on all recorded locations classified as foraging (n = 1,337) or resting (n = 1,771) that were at least 2.5 km from the nesting colony; we expected resting and foraging to occur adjacent to the colony but wanted to focus on identifying sites away from the nesting colony. Core commuting areas were based on all commuting locations that were at least 15 km from the nesting colony. Of primary interest were the routes terns used to reach more distant foraging areas. The search radii for each estimate were based on the north-south extent of the point file divided by 30 (3,877 m for foraging, 3,221 m for resting, and 3,763 m for commuting), and output cell size was 20 m. See BRNW 2014, Appendix C for more details on this study.

Satellite tracking of Caspian terns: The effects of management actions on the foraging behavior of adult Caspian terns satellite-tagged on Goose or Crescent islands in the Columbia Plateau region (see *above*) were investigated by dividing the river into three separate reaches of interest: (1) the Columbia River from Chief Joseph Dam downstream to the confluence with the Snake River; (2) the Snake River from Steamboat Island to the confluence with the Columbia River; and (3) the Columbia River from the confluence with the Snake River downstream to The Dalles, Oregon. Then, using ArcMap geographical information systems (GIS) software (ESRI, Redlands, CA), a 1-km buffer on either side of each river reach of interest were created and the proportion of high-quality locations for each tagged bird that fell within each buffered reach was calculated as a measure of foraging use. Using this measure of foraging behavior as the response, the proportions were logit transformed before applying linear mixed-effects models in the R statistical software package (R Core Team 2018; LME4 package [Bates et al. 2015]), with the individual as a random effect. Seven *a priori* explanatory variables were selected that were believed to influence Caspian tern use of each river reach for foraging and candidate models that included a combination of those variables as fixed effects were evaluated. The explanatory variables tested included (1) the nesting colony site where an individual tern was captured and tagged (Capture Location), (2) the sex of the individual (Sex), (3) the year when the individual was captured (Capture Year), (4) the number of years since management was initiated to dissuade terns from nesting at either Goose Island or Crescent Island (Year Post-Mgmt), (5) the ordinal date of the start of the steelhead smolt out-migration (Start of Steelhead Run), (6) the duration in days of the steelhead smolt out-migration (Duration of Steelhead Run), and (7) the interaction between the ordinal start date and duration of the steelhead out-migration (Interaction between Start and Duration of Steelhead Run). To choose between candidate models, the Akaike Information Criterion corrected for small sample sizes (AIC_c) was used, with any model with a $\Delta AIC_c \leq 2$ determined to be in support of the candidate model. Model averaging of the supported candidate models was used to calculate parameter estimates (Burnham and Anderson 2002). See *above* for more details on the tagging effort, and Roby et al. (2019a) for further details on this study.

Recovery of smolt tags from Caspian tern colonies: Some information on the foraging behavior of Caspian terns breeding at Goose, Crescent, and the Blalock islands was also provided by recovery of smolt tags on tern colonies following the breeding season (Evans et al. 2016a). The last known detections of acoustic-tagged and PIT-tagged (i.e. double-tagged) juvenile salmonids were used and subsequent recoveries of those tags on bird colonies to estimate the foraging range and behavior of Caspian terns and other piscivorous colonial waterbird species in the CPR (see *Chapter 6*). Smolts were tracked using the juvenile salmon acoustic telemetry system (JSATS; see McMichael et al. 2010), with acoustic telemetry arrays (a series of hydrophones) deployed throughout the middle Columbia River, lower Snake River, and lower Columbia River. This network of telemetry arrays resulted in multiple spatial scales or river reaches in which to evaluate where predation occurred relative to the location of individual tern colonies (see also Evans et al. 2016a for further details on this study).

2.3.3.2 Diet composition & smolt consumption – Caspian terns transport whole fish in their bills to their mates (i.e. courtship meals) and young (i.e. chick meals) at the nesting colony. Thus, taxonomic composition of the diet can be determined by direct observation of adults as they return to the nesting colony with fish in their bills (i.e. bill-load observations). Prey items in tern bill-loads were identified to the taxonomic level of family. We were confident in our ability to distinguish salmonids from non-salmonids and to distinguish among most non-salmonid taxa based on direct observations from blinds. Bill load fish and stomach contents were not collected at Caspian tern colonies in the CPR due to the potential impact of lethal sampling on small colonies; therefore, we were unable to assess the relative proportion of the various salmonid species in diets of terns nesting in the region. Furthermore, the lack of bill load and stomach content samples collected from the same individuals (see *Chapter 1*) made it impossible for us to determine whether bill-load fish were representative of what adults were consuming themselves. The percent of identifiable prey items in tern bill-loads that belonged to each prey category was calculated for each 2-week period throughout the nesting season. The diet composition of terns over the entire breeding season was based on the average of the percentages for the 2-week periods. Diet composition estimates based on bill-load identifications for Caspian terns nesting at colonies in the CPR were limited during the pre-management period to just the Crescent Island nesting colony (2000–2012) and the Goose Island nesting colony (2010–2013). See Antolos et al. 2005 for a detailed description of these methods.

A bioenergetics modeling approach was used to estimate the total number of salmonid smolts consumed by Caspian terns nesting at a particular colony during a breeding season. The bioenergetics model used in this study was based largely on the model developed to estimate smolt consumption by Caspian terns nesting in the Columbia River estuary (Roby et al. 2003b, Lyons 2010; see *Chapter 1*). The model input parameters (i.e. number of breeding adults and their young, duration of the breeding season, diet composition, average mass and energy density of prey items) for Caspian terns nesting at selected colonies in the CPR were measured directly as part of this study, while other parameters were from other studies (i.e. daily energy expenditure and metabolic efficiency of adult terns, energy requirements of young terns; see Roby et al. 2003b). To account for seasonal and annual differences in the input parameters, calculations of smolt consumption were performed separately for each 2-week period when terns were present at the nesting colony (April–July). The number of breeding adults, the number and energy requirements of their young (chicks), and diet composition varied across these 2-week periods; other parameters were assumed to be consistent across the entire breeding season. Output parameters of interest, such as estimates of biomass and numbers of prey consumed, were summed from all 2-week periods to get yearly totals. A Monte Carlo simulation procedure was used to calculate reliable 95% confidence intervals for estimates of smolt consumption by terns. Smolt consumption estimates for Caspian terns nesting at colonies in the CPR were limited to just the Crescent Island tern colony during the pre-management period (2000–2012). See Antolos et al. (2005) for details on the bioenergetics modelling framework used to estimate smolt consumption by Caspian terns nesting on Crescent Island.

2.3.3.3 Predation rates on juvenile salmonids – Passive integrated transponder (PIT) tags placed in juvenile salmonids and subsequently consumed and deposited by Caspian terns on their breeding colonies have been used to estimate avian predation rates (percentage of available fish consumed) for more than two decades in the Columbia River basin (Collis et al. 2001, Ryan et al. 2001, Antolos et al. 2005, Evans et al. 2012, Hostetter et al. 2015, Evans et al. 2016a). Recoveries of PIT tags on bird colonies can provide specific information on which fish species and populations are consumed by birds breeding at specific colonies. Predation rate studies have also been used to investigate the relative susceptibility of different groups of tagged fish to bird predation (see also *Chapter 10*). Analogous to studies of Caspian tern predation in the Columbia River estuary (see *Chapter 1*), predation rate studies involving terns in the CPR were initially conducted to determine if terns nesting at specific breeding colonies posed a significant risk to smolt survival (Ryan et al. 2001, Antolos et al. 2005, Evans et al. 2012) and then to monitor and evaluate the efficacy of management actions aimed at reducing predation rates in the CPR (Roby et al. 2015, Collis et al. 2016, Evans et al. 2019b, Collis et al. 2020).

Smolt PIT tags collected on Caspian tern colonies in the CPR were analyzed as part of this study to (1) estimate Caspian tern predation rates on ESA-listed ESUs/DPSs of anadromous salmonids and (2) assess relative differences in these predation rates prior to and during tern management actions associated with the *IAPMP*. Comparisons between current and previous predation rates were made in the context of management initiatives for terns nesting on Goose Island in Potholes Reservoir and on Crescent Island in McNary Reservoir, and relative to the management goal of achieving predation rates of less than 2% per salmonid ESU/DPS, per nesting colony (both managed and unmanaged colonies, see *above*), per year and averaged over three years (USACE 2014).

Previous research has demonstrated that juvenile steelhead are particularly susceptible to Caspian tern predation (Evans et al. 2012, USACE 2014, Roby et al. 2017, Evans et al. 2019a, Evans et al. 2009b). To ensure that adequate sample sizes of PIT-tagged steelhead from the Upper Columbia River DPS were available for predation analyses, steelhead were captured, PIT-tagged, and release at Rock Island Dam (RIS) during the 2008–2019 outmigration years. This resulted in a long-term dataset that has been used to estimate both predation rates and survival rates of steelhead during outmigration (Hostetter et al. 2012, Evans et al. 2012, Evans et al. 2014, Hostetter et al. 2018, Evans et al. 2019a, Evans et al. 2019b, Payton et al. 2020). A detailed description of the sampling methods used to capture, tag, and release steelhead smolts at RIS are presented in Evans et al. (2014). In brief, steelhead were captured, PIT-tagged, and released at the RIS juvenile fish trap throughout the smolt outmigration period of April to

June. Steelhead were randomly selected for tagging (i.e. tagged regardless of condition, origin, and size) and were tagged in concert with, and in proportion to, the run-at-large to ensure that the tagged sample was representative of the steelhead population passing the dam (tagged and untagged fish). The objective was to PIT-tag approximately 7,000 juvenile steelhead for use in predation analyses. This target sample size was selected because it was estimated that it would result in an estimate of predation rate with a minimum precision (95% credible interval) of approximately ± 2%. For all other ESUs/DPSs evaluated, we relied on groups of PIT-tagged fish from other regional tagging studies, studies where more than 7,000 tagged fish were available for most ESUs/DPSs of salmonids in most years (see *Appendix B, Tables B20-21*).

Predation rates were derived using the number of PIT tags detected on a given Caspian tern nesting colony and the number of PIT-tagged fish available as indicated by the number passing or interrogated at upstream dams. These apparent predation rates were adjusted using the proportion of consumed tags that were deposited by terns on their nesting colony (referred to as “deposition probability”) and the proportion of deposited tags subsequently detected by researchers following the nesting season (referred to as “detection probability”). A detailed description of the methods and statistical procedures used to recover smolt PIT tags from bird colonies and to estimate predation rates is provided in Evans et al. (2012), Hostetter et al. (2015), and *Appendix A.2*. A list of predation rate model assumptions and procedures used to evaluate the validity of those assumptions are also provided in Hostetter et al. (2015) and in *Appendix A.2.* Methods used to calculate estimates of predation rates were identical throughout the study period, allowing for a direct comparison of results across years and among tern colonies.

2.4 Results

2.4.1 Nesting Habitat Reduction & Active Dissuasion

2.4.1.1 Nest dissuasion in Potholes Reservoir – In 2014, the first year of management in Potholes Reservoir, a passive nest dissuasion array (stakes, ropes, and flagging) was deployed on all upland habitat deemed suitable for tern nesting on Goose Island. This was completed in late February, prior to the arrival of Caspian terns and gulls to the island to breed. Nearly 2.5 acres (1.0 ha) of habitat was covered with passive dissuasion on the main island (Roby et al. 2015). No passive dissuasion was initially placed on the rocky islets surrounding Goose Island because that habitat was deemed unsuitable for tern nesting (i.e. lacked loose soil that terns need to dig nest scrapes). Nesting gulls (both California and ring-billed gulls) were first to arrive to the island in early March, and occupied nesting territories across Goose Island, both within and outside the nest dissuasion array. Caspian terns were first observed on Goose Island during the third week in March. The passive nest dissuasion array was successful in preventing terns from attempting to nest at the former tern colony site on Goose Island, as well as in other similar upland habitat on the main island, areas that had been covered in passive dissuasion. In

2014, terns first began prospecting for nest sites on a rocky islet located just offshore of Goose Island ("Northwest Rocks"), where no passive nest dissuasion was deployed. After being informed of prospecting terns on Northwest Rocks, resource managers made the determination that passive dissuasion could not be deployed at that location during the 2014 breeding season, an adaptive management action generally allowed by the *IAPMP* (USACE 2014), because it was not known whether the rocky islet was part of Goose Island and if actions at that site were covered under the environmental analysis documents completed as part of the management plan (USACE 2014). Consequently, a small tern nesting colony became established on Northwest Rocks and nesting terns were successful in fledging young from the site in 2014 (see section *2.4.2 Nesting Ecology* for colony metrics; also see Roby et al. 2015). Caspian terns also attempted to nest along the shoreline of Goose Island where receding reservoir levels exposed bare ground nesting substrate near the waterline during the breeding season, habitat that had not been covered with passive nest dissuasion materials. Nesting attempts by terns along the shoreline of Goose Island were not successful due to in-season nest dissuasion activities, including passive and active dissuasion (i.e. human hazing) plus collection of any tern eggs discovered in those areas to prevent nesting colony formation. The level of effort required to prevent tern nesting along the shoreline of Goose Island in 2014 was fairly intense; for most of the breeding season human hazing activities were carried out by a resident field crew (i.e. field personnel living on the island) almost continuously throughout the day. Despite the passive dissuasion deployed on preferred Caspian tern nesting habitat at Goose Island and intensive active dissuasion (hazing), three pairs of Caspian terns laid eggs on Goose Island; all three tern eggs were collected under permit. Tern nest dissuasion activities (both passive and active) did not deter gulls from forming a large nesting colony on Goose Island and successfully nesting in 2014, or throughout the implementation of management on Goose Island during 2015–2019. See Roby et al. (2015) for further details on nest dissuasion activities on Goose Island in 2014.

In 2015, the second year of management in Potholes Reservoir, the existing nest dissuasion array on Goose Island was repaired and new flagging installed prior to the arrival of nesting Caspian terns and gulls to the island. In addition, a new nest dissuasion array was erected on Northwest Rocks, the site of successful tern nesting the previous year, as it was determined that management actions at that site were covered under the existing environmental documents and management plan (USACE 2014). A total of 4.1 acres (1.7 ha), or more than 85% of the upland area on Goose Island and surrounding islets, was covered in passive nest dissuasion materials in 2015, including Northwest Rocks and similar rocky habitat found on Goose Island that was not covered in passive dissuasion the previous year (Collis et al. 2016). These efforts, along with extensive human hazing conducted throughout the breeding season, were successful in preventing all but two breeding pairs of terns from successfully nesting (i.e. fledging young) on Goose Island and surrounding islets (see section *2.4.2 Nesting Ecology* for colony metrics). These two Caspian tern nests were not located together, but both were near the former tern nesting colony area on the main island and beneath passive nest dissuasion materials. A total of 43 Caspian tern eggs were found on Goose Island and nearby islets during the 2015 breeding season, mostly along the shoreline of Goose Island. Of the 43 tern eggs that were discovered in 2015, 17 were collected under permit, 23 were depredated by gulls soon after they were discovered, and three eggs produced young that subsequently fledged. See

Collis et al. (2016) for further details on nest dissuasion activities on Goose Island and the surrounding islets in 2015.

In 2016–2019, the existing nest dissuasion arrays were repaired and fixed with flagging prior to the tern nesting season as described above. In each of these four years, over 4 acres (1.6 ha) of upland habitat on Goose Island and surrounding islets was covered in passive nest dissuasion materials (Collis et al. 2017, 2018, 2019, 2020). These efforts, along with continuous (daily) human hazing and egg collection, were successful in preventing Caspian terns from nesting on Goose Island and surrounding islets during 2016–2019. In 2016, an island not previously used by nesting Caspian terns in northern Potholes Reservoir was colonized (see section *2.4.2 Nesting Ecology* for colony metrics; also see Collis et al. [2017]). Tern eggs were laid and incubated, and a few tern chicks hatched at this incipient nesting colony before the colony ultimately failed and was abandoned in early June, presumably due to predation and disturbance by American mink (Neovison vison) whose tracks were found on the island. During each subsequent year (2017–2019), nest dissuasion activities (passive/active dissuasion and tern egg collection) were conducted at sites in northern Potholes Reservoir where Caspian terns were prospecting for nest sites, thereby preventing nesting colony formation. See Collis et al. (2017, 2018, 2019, and 2020) for further details on nest dissuasion activities at Potholes Reservoir during 2016–2019.

Throughout the management period (2014–2019), the effort required to actively haze Caspian terns from Goose Island to prevent tern nesting declined in each successive year of management (Figure 2.6). Tern egg-laying on Goose Island generally declined from early in the management period (2015) to the end of the management period (2019), with a greater proportion of tern eggs laid in recent years being late in the breeding season (June and July; Figure 2.7), well beyond the peak in egg-laying observed during the pre-management period (see below). Except for three eggs laid beneath passive dissuasion in 2015, all the tern eggs laid on Goose Island and the surrounding islets during 2015–2019 (total of 93 eggs) were either collected (under permit) or depredated by gulls soon after the eggs were laid.

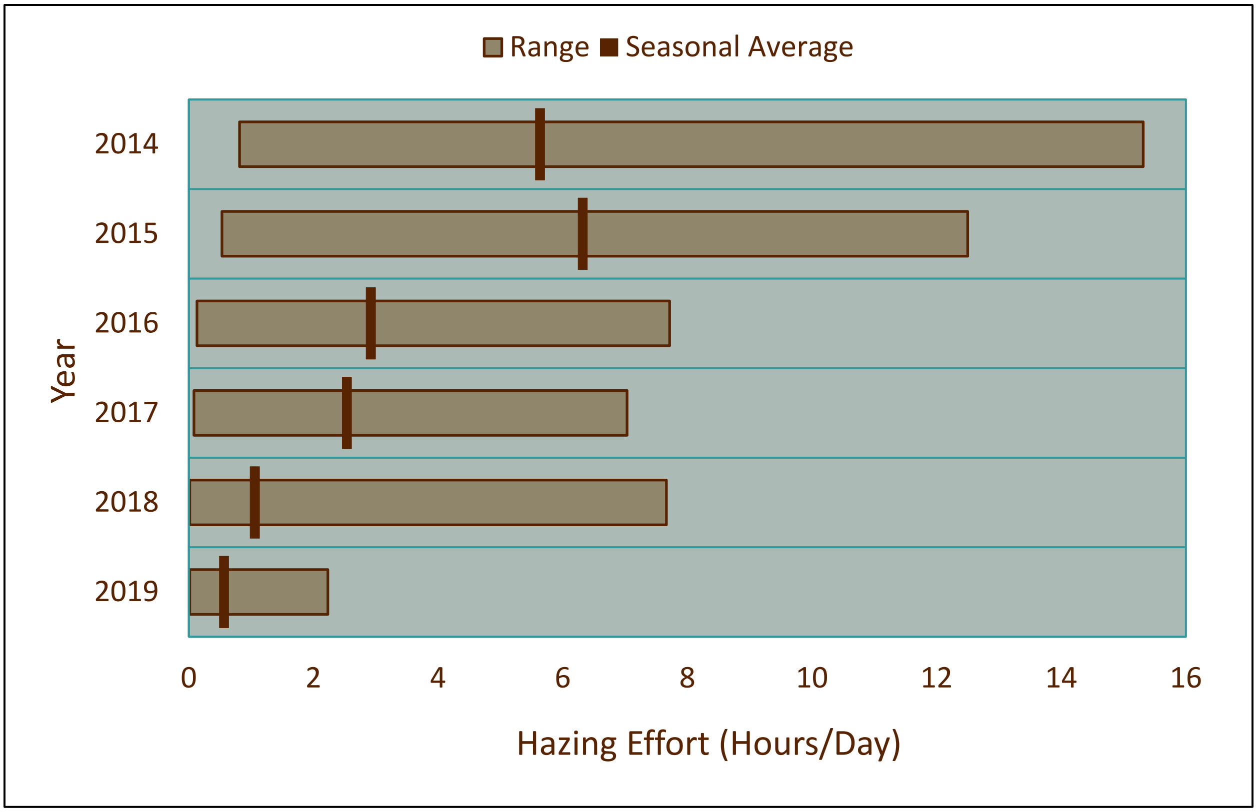

Figure 2.6. Active hazing effort (hours/day) at Goose Island in Potholes Reservoir during the management period (2014–2019).

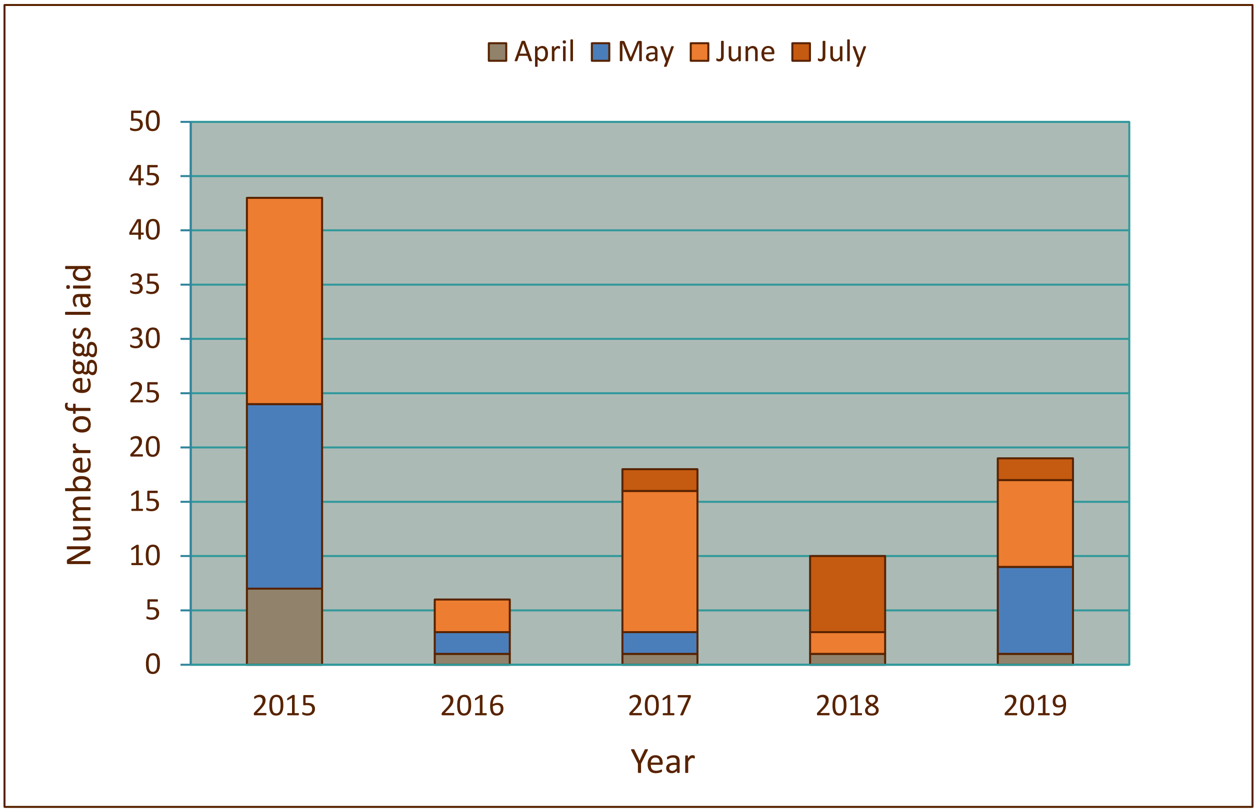

Figure 2.7. Caspian tern eggs laid on Goose Island and surrounding islets in Potholes Reservoir during the last five years of the management period (2015–2019).

2.4.1.2 Nest dissuasion on Crescent Island – In 2015, the first year of management on Crescent Island, passive nest dissuasion arrays consisting of vertical fence rows, stakes, rope, flagging, and woody debris were deployed on all upland habitat deemed suitable for tern nesting. Deployment was completed by mid-March, prior to the arrival of Caspian terns and gulls to the island to breed. A total of 2.2 acres (0.9 ha) of suitable nesting habitat for terns was covered with passive dissuasion on Crescent Island in 2015 (Collis et al. 2016). As was the case at Goose Island in Potholes Reservoir, nesting gulls (both California and ring-billed gulls) were first to arrive to the island, with large numbers (1,000s) regularly seen rafting on the water just offshore starting in late March and until late April, when the number of gulls seen near the island deceased to 100s. The decline in numbers of gulls rafting near Crescent Island was likely due to most of the gulls moving to nearby Badger Island to nest (Collis et al. 2016). Although Caspian terns were regularly seen flying over Crescent Island during the 2015 breeding season, terns were not observed landing on the island. A few gulls were observed loafing on Crescent Island during the 2015 breeding season, but nesting attempts by gulls were detected on Crescent Island.

In 2016, based on successful attempts to grow willows (Salix spp.) from whips on Crescent Island in 2014–2015, a USACE contractor planted willows and other native vegetation across the entire island, work that was completed in February 2016[1]. Maintenance of the existing passive nest dissuasion materials (see *above*) and the native vegetation planted on Crescent Island in 2016 were successful in preventing Caspian terns and gulls from establishing breeding colonies on the island in each year during the management period (2015–2019). As a result, active dissuasion (i.e. human hazing) was not required to prevent terns and gulls from loafing or nesting on Crescent Island, even in the early years of management implementation. See Collis et al. (2016, 2017, 2018, 2019, and 2020) for further details on nest dissuasion activities on Crescent Island during 2015–2019.

2.4.2 Nesting Ecology

2.4.2.1 Nesting chronology & colony attendance – During the pre-management period, the first Caspian tern egg at was detected on the Goose Island and Crescent Island colonies was detected during mid- to late April (*Figure 2.8*). At Goose Island, the median first egg date during the pre-management period was on 20 April, six days earlier than the median first egg date during the management period. Similarly, the date when the first tern chick hatched on Goose Island during the management period (26 May) was 10 days later compared to the pre-management period, although tern chicks only hatched on Goose Island during the first two years of the management period (2014–2015; *Figure 2.8*). The median first tern egg date and median first tern chick date at Crescent Island during the pre-management period was 16 April and 13 May, respectively (*Figure 2.8*). The median first fledging date did not differ greatly between Goose Island and Crescent Island or between the pre-management and management periods, ranging from 24 June to 2 July (*Figure 2.8*).

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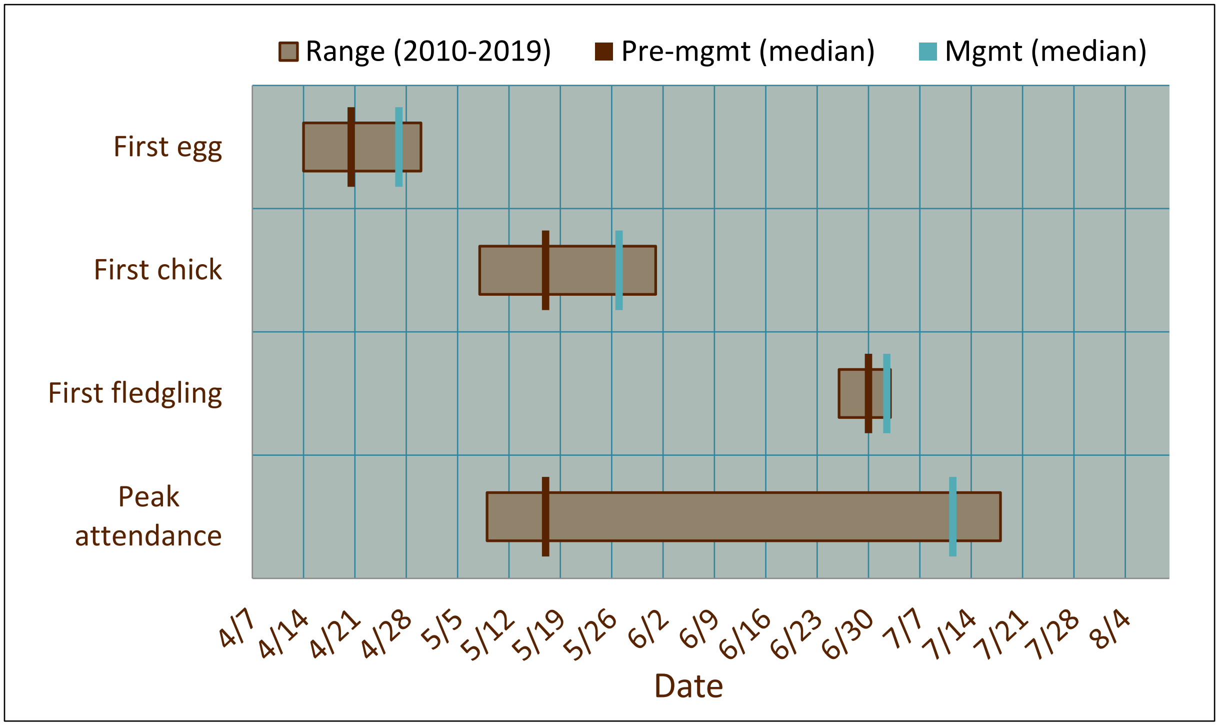
[1] In 2017, due to low survival, additional willows were planted on Crescent Island.



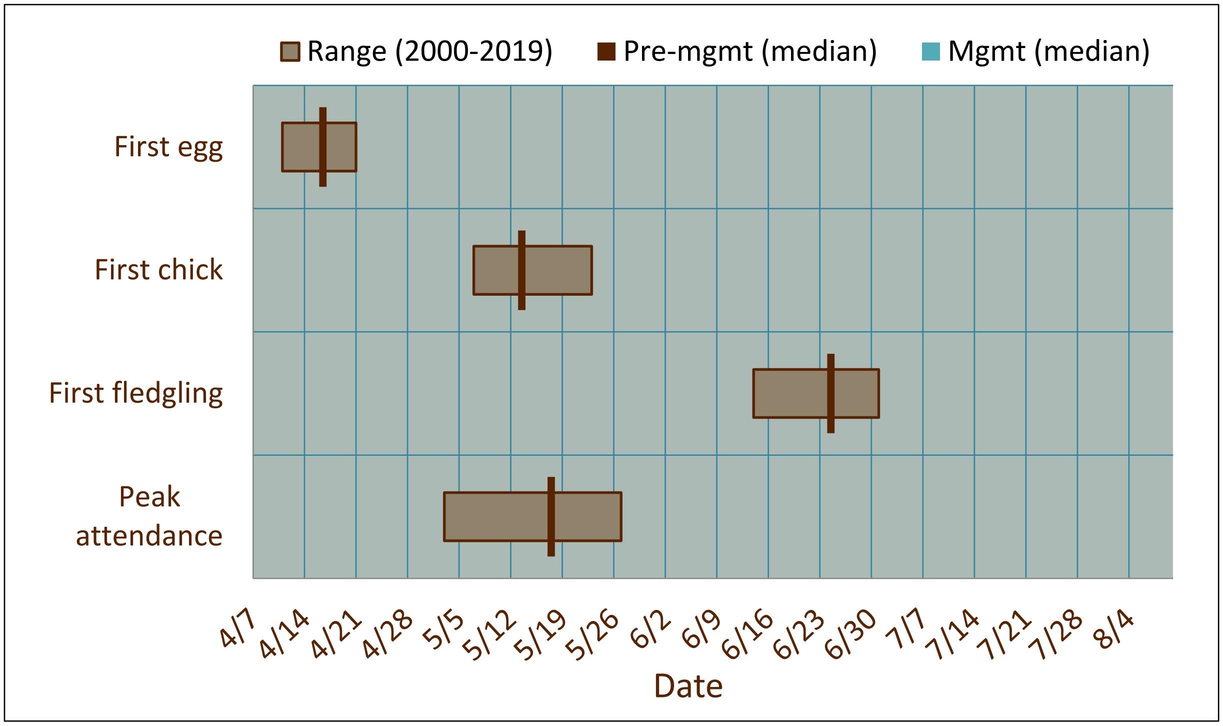
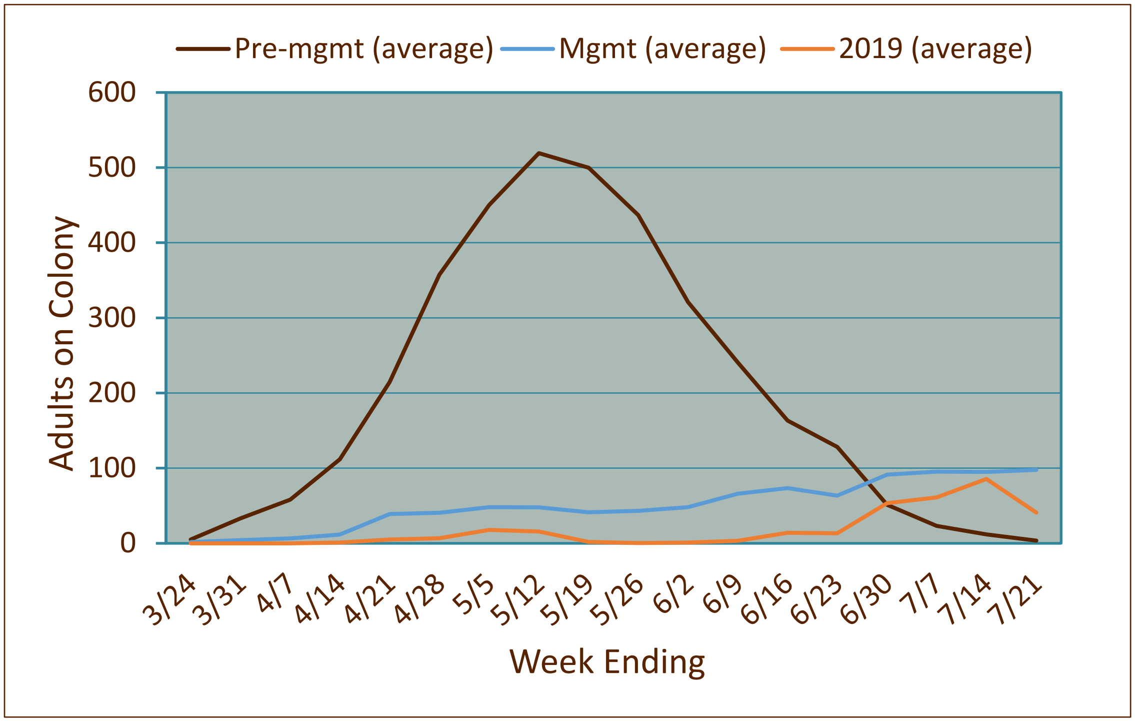


Figure 2.8. Nesting chronology for Caspian terns nesting on Goose Island in Potholes Reservoir (top) and on Crescent Island in the Columbia River (bottom) during 2000–2019. Median dates for each measure of nesting chronology are presented for both the pre-management period (2005-2013 for Goose Island; 2000-2014 for Crescent Island) and the management period (2014-2019 for Goose Island; 2015-2019 for Crescent Island). Caspian terns did not attempt to nest on Crescent Island during the management period.

The peak in nesting colony attendance for the managed colonies on Goose and Crescent islands were identified, which informed decisions on the timing of aerial photography or ground counts to estimate peak colony size (number of breeding pairs). The median date of peak nesting colony attendance differed greatly between the pre-management and management periods at the Goose Island tern colony; average date of peak nesting colony size was 16 May during the pre-management period and 11 July during the management period (*Figure 2.5* above). The much later median date of peak nesting colony attendance during the management period at Goose Island was a direct result of management to prevent nesting at that site, with most, if not all, of the terns attended the colony in July being either non-breeders, failed breeders, or post-breeding terns from other colonies. At the Crescent Island tern nesting colony, the median date of the peak in nesting colony attendance during the pre-management period (17 May) was only one day later than at Goose Island (*Figure 2.7* above).

Not only did management alter the timing of the peak in nesting colony attendance, but it also had a dramatic impact on the numbers of adult terns attending the managed colonies over the nesting season (*Figure 2.9*). During the pre-management period, the numbers of terns attending the colonies on Goose and Crescent islands increased rapidly from late March until the peak in mid-May, and then declined precipitously from late May through July (*Figure 2.9*); by August, most terns had dispersed from the two islands. At Goose Island, the numbers of adult terns counted on the nesting colony throughout the breeding season during the management period were much lower compared to the pre-management period; numbers of terns attending Goose Island decreased incrementally with each successive year of management (*Figure 2.9*). At Crescent Island, terns did not attend the site of the former nesting colony during the management period (*Figure 2.9*), presumably because all gulls had also abandoned the island as a nesting site and there were other islands nearby that were suitable for tern nesting (see *below*).

Comparable nesting chronology and attendance data at unmanaged Caspian tern nesting colonies in the CPR do not exist because these islands were not visited as frequently during the study period. Anecdotally, nesting chronology at the unmanaged tern colonies was comparable to that observed at the managed colonies during the pre-management period, while there was an increase in the numbers of terns attending most unmanaged tern colonies in the CPR region during the management period (see section *2.4.2.2 Nesting colony size*).



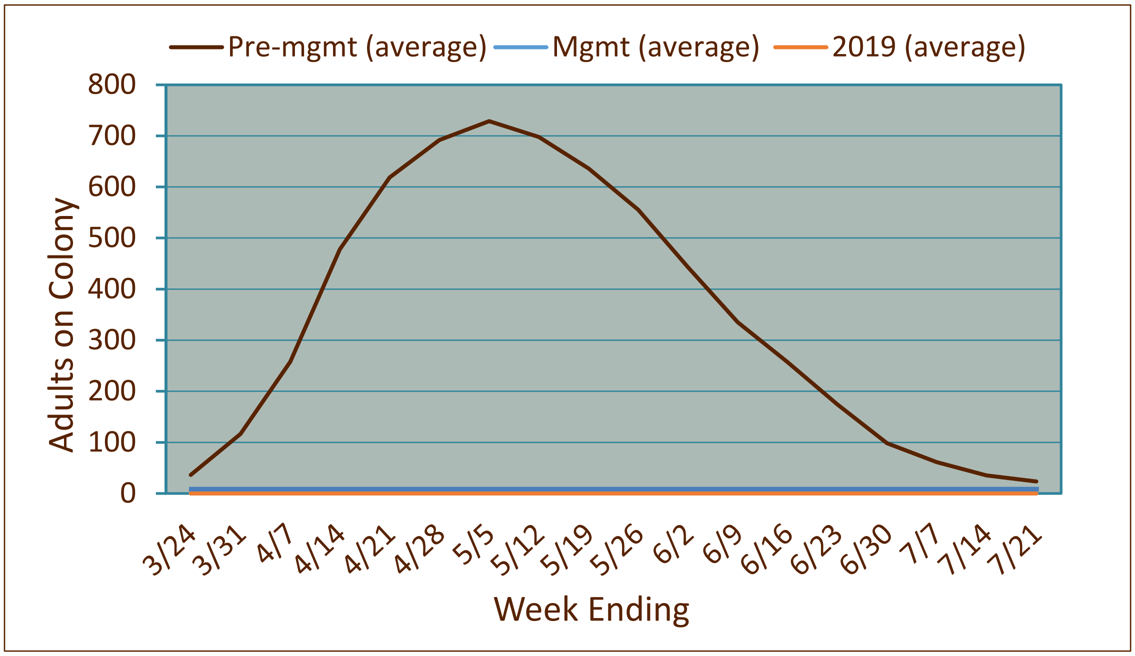
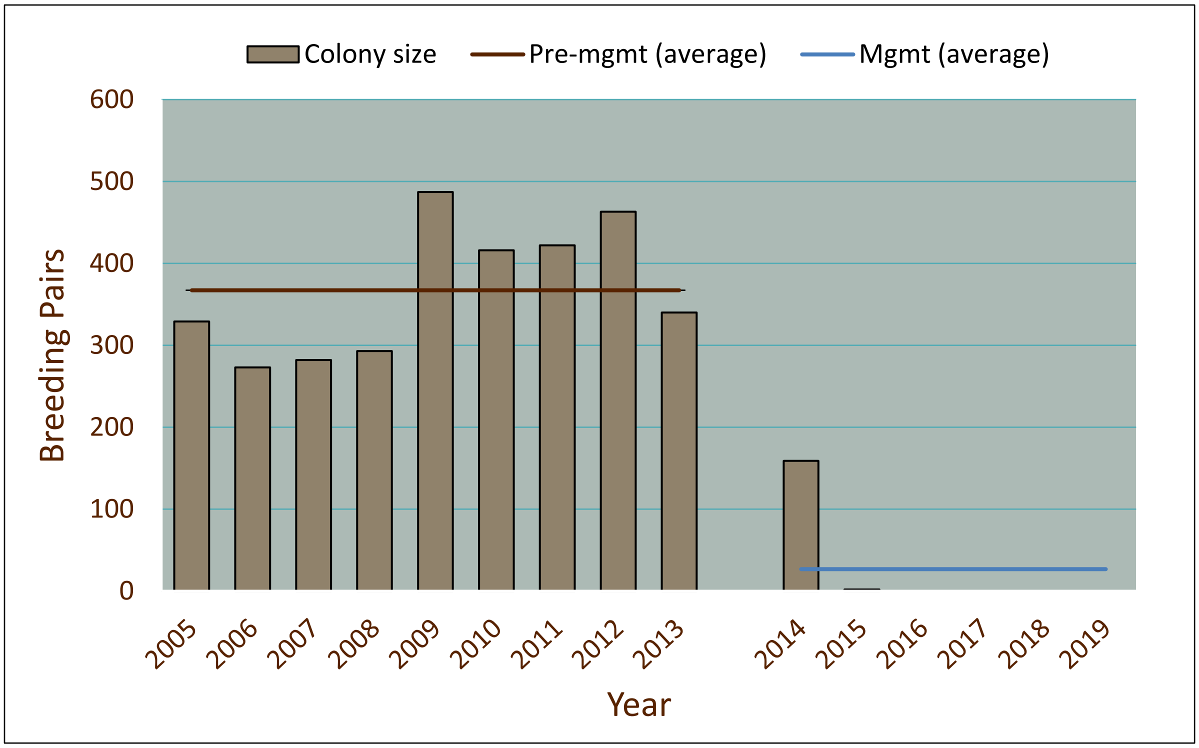


Figure 2.9. Weekly estimates from the ground of the number of adult Caspian terns on Goose Island in Potholes Reservoir (top) and on Crescent Island in the Columbia River (bottom) during the pre-management period (2005-2013 for Goose Island; 2000-2014 for Crescent Island) and the management period (2014-2018 for Goose Island; 2015-2018 for Crescent Island) and during the last year of the study (2019).

2.4.2.2 Nesting colony size – The numbers of Caspian tern breeding pairs nesting at the managed colonies on Goose and Crescent islands were estimated during 2005–2019 at Goose Island, and during 2000–2019 at Crescent Island (*Figure 2.10*). Prior to management, these two colonies were the largest for the species in the CPR (see *below*), ranging in size from 273 to 487 breeding pairs at Goose Island and from 349 to 657 breeding pairs at Crescent Island. During the management period, nesting colony size ranged from 0 to 159 breeding pairs at Goose Island, while Caspian terns did not attempt to breed (or roost) on Crescent Island during the management period (2015–2019; *Figure 2.10*). Some Caspian terns were successful in establishing a nesting colony on or adjacent to Goose Island during the first two years of management (2014–2015) due to constraints in the implementation of adaptive management actions at that site during those two years (see section *2.3.1.1 Nest dissuasion in Potholes Reservoir* above). Due to management actions implemented at each site, Caspian terns did not establish a nesting colony on Goose Island in each of the last four years of the study (2016–2019) or on Crescent Island in any year of the management period (2015–2019; *Figure 2.10*).



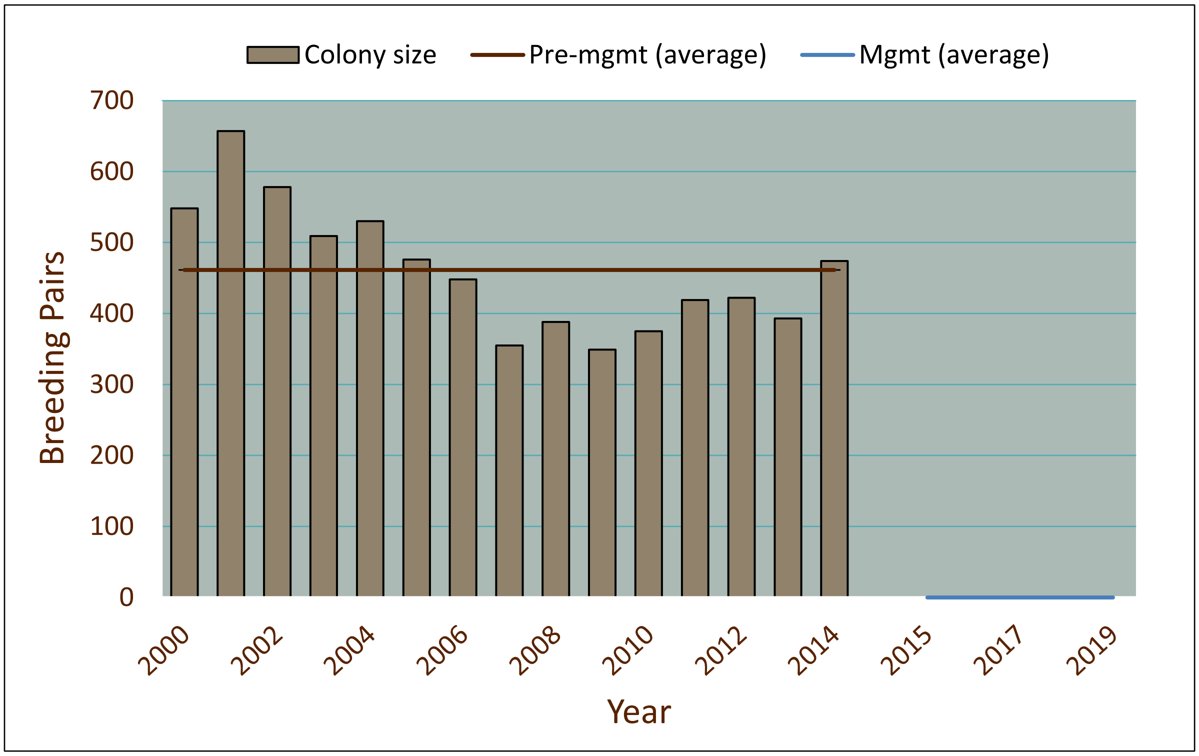


Figure 2.10. Colony size (number of breeding pairs) for Caspian terns nesting on Goose Island in Potholes Reservoir (top) and on Crescent Island in the Columbia River (bottom) during the pre-management period (2005-2013 for Goose Island; 2000-2014 for Crescent Island) and the management period (2014-2018 for Goose Island; 2015-2018 for Crescent Island). Note differences in the y-axis scales (see methods).

In 2016, Caspian terns attempted to nest at an incipient colony in northern Potholes Reservoir, where up to 144 breeding pairs were counted before the colony failed (see *below*). During 2017–2019, this nesting colony site was adaptively managed (see section *2.3.1.1 Nest dissuasion in Potholes Reservoir* above) to prevent Caspian terns from nesting, efforts that were successful in preventing nesting colony formation.

During the portion of the study period when all Caspian tern colonies in the CPR were surveyed (2005–2019), six different colonies were identified (*Table 2.2*). Only three colonies (i.e. the Blalock Islands complex on the Columbia River, Harper Island in Sprague Lake, and Twinning Island in Banks Lake) were consistently attended year-after-year during this period. Of these three colonies, the Blalock Islands complex tern nesting colony was the largest during the pre-management period (an average of 57 breeding pairs; *Table 2.2*). The Caspian tern nesting colony at Lenore Lake did not become active until management under the *IAPMP* was initiated in 2014 and grew in the number of breeding pairs in each of the subsequent three years, until nesting colony size began to decline in 2018–2019 (see *below*).

Table 2.2. Colony size (number of breeding pairs) for Caspian terns at sites in the Columbia Plateau region during 2005–2019 (years when a complete survey of all Caspian tern colonies in the Columbia Plateau region was completed). "Total" indicates the region-wide total breeding population of Caspian terns during 2005–2019.

	Columbia River				**Potholes Reservoir**	**Banks Lake**	**Sprague Lake**	**Lenore Lake**	
Year	Crescent Island	Blalock Is. complex	Foundation Island	Badger Island	Goose Is. /North Potholes	Twinning Is.	Harper Island	North Rock/ Shoal Is.	**Total**
2005	476	6	0	0	329	26	10	0	**847**
2006	448	110	0	0	273	20	7	0	**858**
2007	355	26	0	0	282	48	0	0	**711**
2008	388	104	0	0	293	27	11	0	**823**
2009	349	79	0	0	487	61	4	0	**980**
2010	375	136	0	0	416	34	4	0	**965**
2011	419	20	5	31	422	19	4	0	**920**
2012	422	6	0	60	463	22	30	0	**1003**
2013	393	26	0	0	340	13	1	0	**773**
2014	474	45	0	0	159	67	8	2	**755**
2015	0	677	0	0	2	64	10	16	**769**
2016	0	483	0	0	144	6	3	39	**675**
2017	0	449	0	41	0	0	92	123	**705**
2018	0	313	0	8	0	0	79	91	**491**
2019	0	379	0	0	0	0	18	48	**445**

Following implementation of the *IAPMP*, most of the unmanaged Caspian tern colonies increased in size (*Table 2.2* above). This was most notable at the tern nesting colony in the Blalock Islands complex, where average nesting colony size during the management period was more than eight times greater than during the pre-management period (*Figure 2.11*). Increases in nesting colony size during the management period also occurred at the unmanaged colonies on Lenore Lake and Sprague Lake, while Caspian tern nesting colony size on Twinning Island in Banks Lake decreased during the management period (*Figure 2.11*). There were also changes in the region-wide breeding population of Caspian terns following management, with 445 breeding pairs nesting in the region during 2019, the final year of this study, or nearly 50% less than the average regional breeding population size during the pre-management period (*Figure 2.12*, *Table 2.2*).

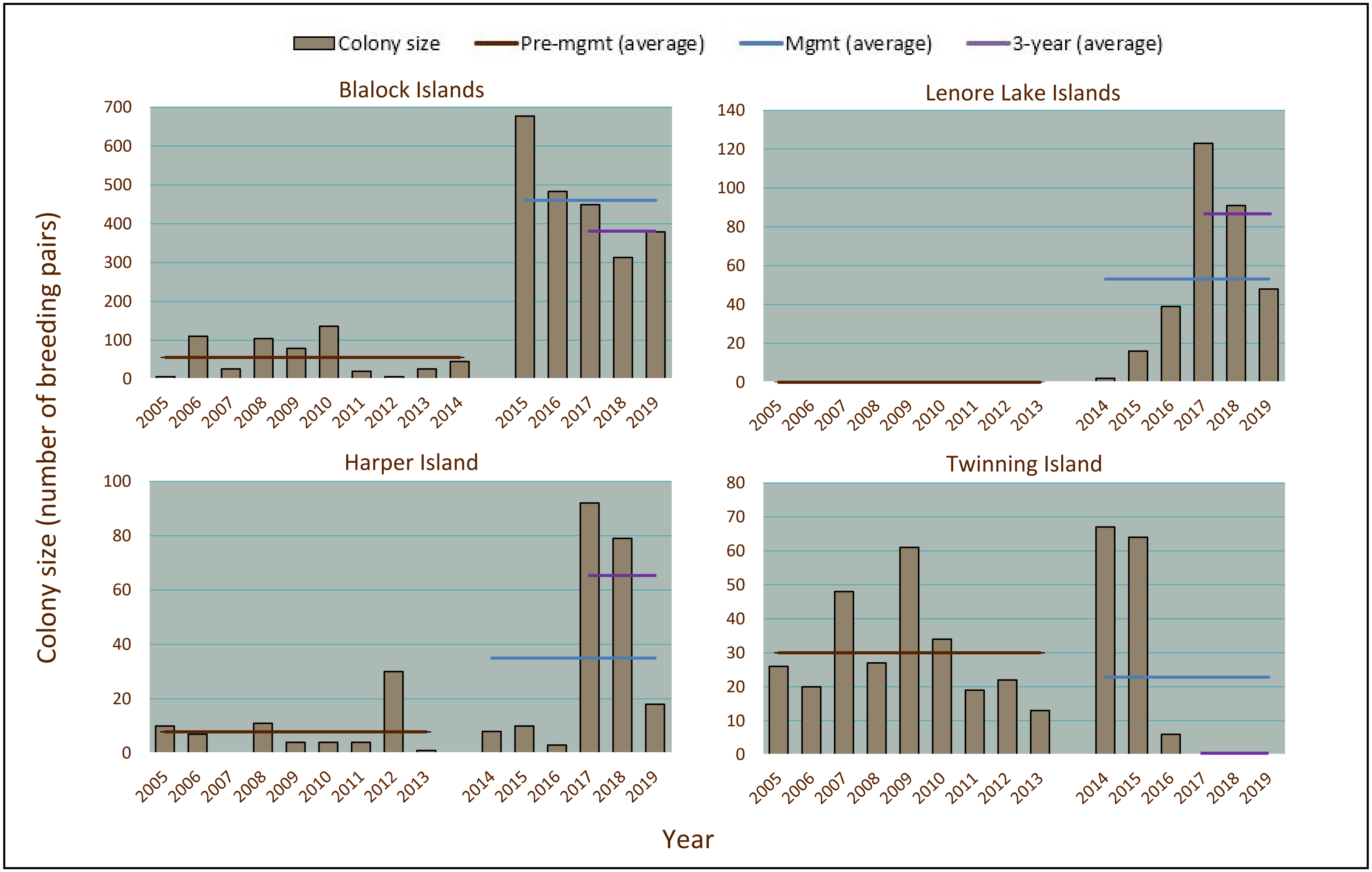


Figure 2.11. Colony size (number of breeding pairs) at the unmanaged Caspian tern colonies in the Columbia Plateau region, including the Blalock Islands complex colony, the North Rock and Shoal Island colonies in Lenore Lake, the Harper Island colony in Sprague Lake, and the Twinning Island colony in Banks Lake. Note differences in both the y-axis scales and the management periods (see methods).

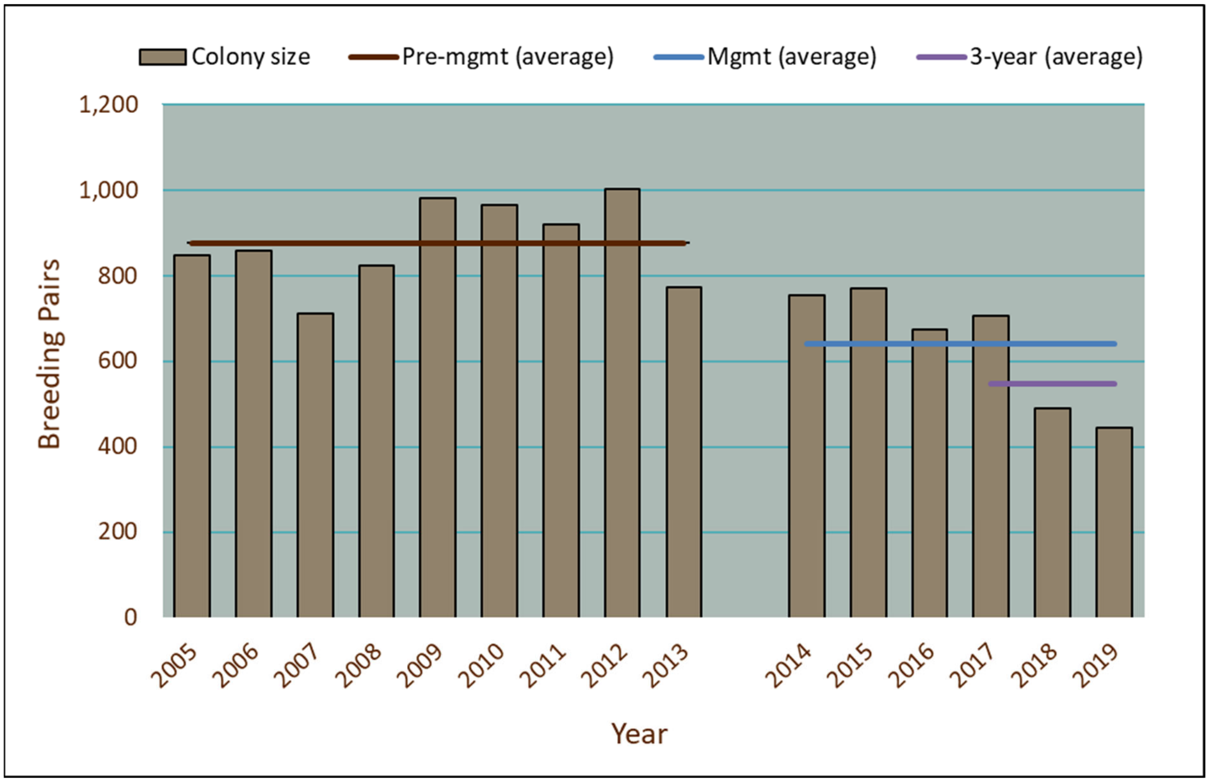
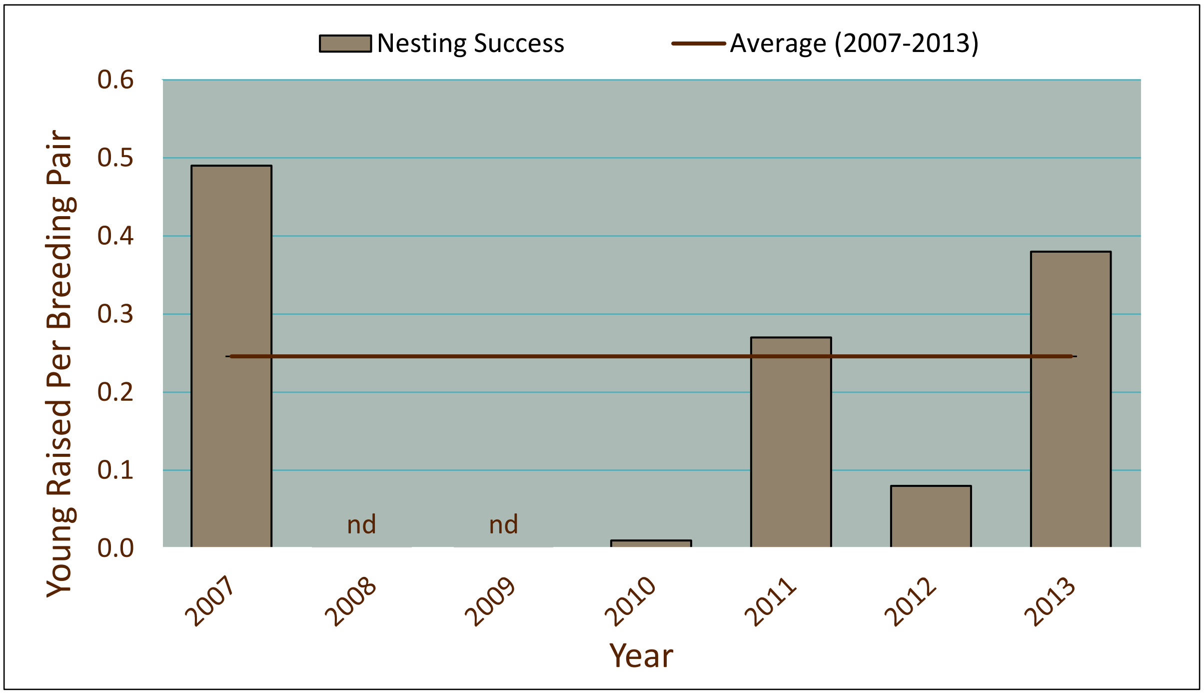


Figure 2.12. Breeding population size (total number of breeding pairs) for Caspian terns nesting at colonies in the Columbia Plateau region during the pre-management period (2005-2013, the management period (2014-2019), and the rolling 3-year average during the management period.

2.4.2.3 Nesting success – Nesting success at the Caspian tern colonies on Goose and Crescent island during the pre-management period averaged 0.25 and 0.51 young raised per breeding pair, respectively (*Figure 2.13*). This level of nesting success is considered low to moderate for Caspian terns (Cuthbert and Wires 1999) and declined gradually during the pre-management period, at least at Crescent Island. Nesting success at the Crescent Island tern colony was consistently higher than at the Goose Island colony, both on average and when comparing nesting success at the two colonies during the same year (2007–2013; *Figure 2.13*).



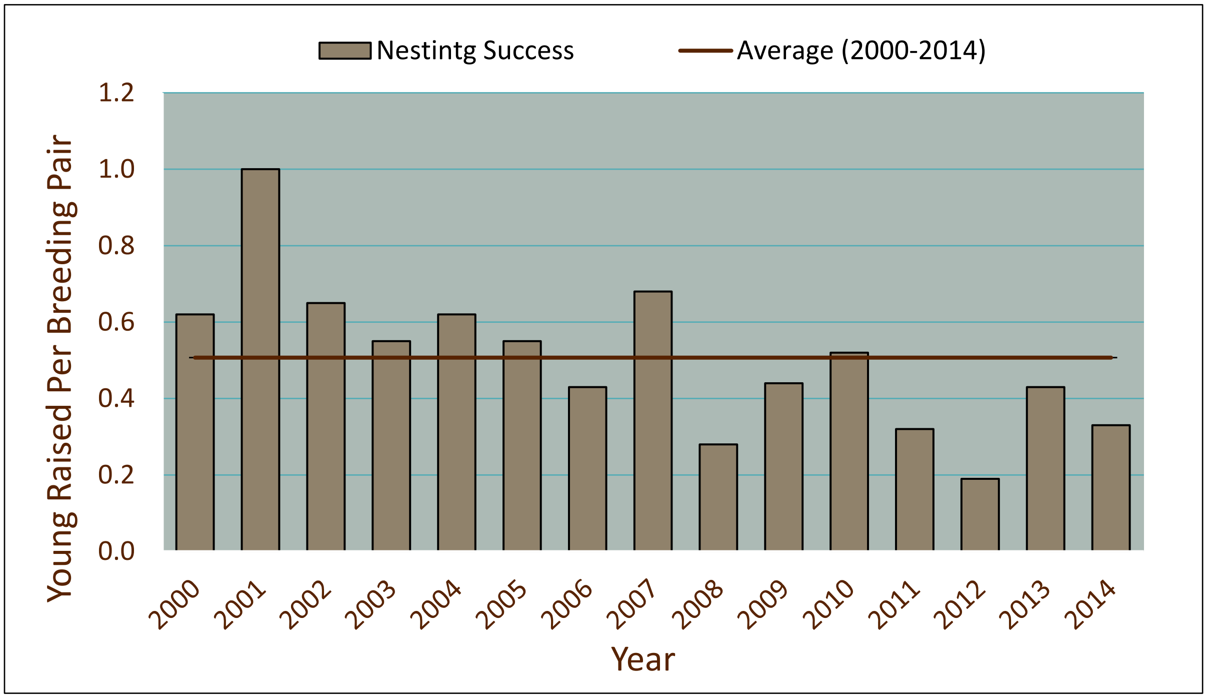


Figure 2.13. Nesting success (average number of young raised per breeding pair) for Caspian terns nesting on Goose Island in Potholes Reservoir (top) and on Crescent Island in the Columbia River (bottom) during the pre-management period. Caspian terns nested on Goose Island in 2008 and 2009 but estimates of nesting success were not available.

Prior to management, average nesting success at unmanaged Caspian tern colonies in the CPR was generally lower as compared to the colonies on Goose and Crescent islands (*Table 2.3*), ranging from 0.00 to 0.17 young raised per breeding pair and from 0.25 to 0.42 young raised per breeding pair, respectively. Following the implementation of management at Goose and Crescent islands, nesting success increased at the Blalock Islands complex tern colony and at the Harper Island tern colony in Sprague Lake, but decreased at the Twinning Island in Banks Lake (*Table 2.3*).

Table 2.3. Estimated nesting success (young raised per breeding pair) of Caspian terns at colony sites in the Columbia Plateau region from 2005–2019 (years when complete survey of the entire Columbia Plateau region completed). "nd" denotes that nesting occurred at the site, but an estimate of nesting success is unavailable and "--" denotes that no nesting occurred at the site. Although nesting occurred on Goose Island in 2014–2015 and in northern Potholes Reservoir in 2016, nesting success was not calculated (see methods).

	Columbia River				**Potholes Reservoir**	**Banks Lake**	**Sprague Lake**	**Lenore Lake**
Year	Crescent Island	Blalock Is. complex	Foundation Island	Badger Island	Goose Is. /North Potholes	Twinning Is.	Harper Island	North Rock/ Shoal Is.
2005	0.55	0.16	--	--	nd	nd	0.00	--
2006	0.43	0.00	--	--	nd	nd	0.00	--
2007	0.68	nd	--	--	0.49	0.35	--	--
2008	0.28	0.03	--	--	nd	0.33	0.00	--
2009	0.44	0.00	--	--	nd	0.33	nd	--
2010	0.52	0.00	--	--	0.01	0.00	nd	--
2011	0.32	0.00	0.00	0.00	0.27	0.16	0.00	--
2012	0.19	0.00	--	0.00	0.08	0.00	0.00	--
2013	0.43	0.12	--	--	0.38	0.00	0.00	--
2014	0.33	0.33	--	--	nd	0.00	0.00	nd
2015	--	0.37	--	--	nd	0.00	0.00	0.37
2016	--	0.43	--	--	0.00	0.00	0.00	0.59
2017	--	0.05	--	0.01	--	--	0.03	0.27
2018	--	0.18	--	0.00	--	--	0.05	0.20
2019	--	0.34	--	--	--	--	0.55	0.26

2.4.2.4 Limiting factors for colony size & nesting success – Tern colony size (number of breeding pairs) and nesting success (average number of young raised/breeding pair) at Caspian tern colonies in the CPR were highly variable across the study period (2005–2019). Factors that limit colony size and nesting success varied by colony site. Clearly, the most significant factor that limited colony size and nesting success at the managed colonies on Crescent and Goose islands during the management period were the nest dissuasion activities implemented as part of the *IAPMP*. The predominant limiting factors for the tern colony on Goose Island appeared to be predation, while the predominant limiting factor for the tern colony on Crescent Island appeared to be gull kleptoparasitism. Predation by both great horned owls and American mink was detected on numerous occasions at the Goose Island Caspian tern nesting colony prior to management. In comparison, predation on adult terns and their nest contents was not commonly detected at the Crescent Island tern nesting colony. Instead, gull kleptoparasitism (i.e., stealing bill-load fish) appeared to be the most significant limiting factor at that site. California gulls that nested adjacent to the tern nesting colony on Crescent Island affected the foraging efficiency and energetic demands of Caspian terns nesting on Crescent Island. If the fish that a tern delivered to the colony was stolen, that individual (or its mate) must compensate by spending additional time and energy foraging (Adkins et al. 2014).

Other factors also limited the size and nesting success of Caspian tern colonies in the CPR. The availability of suitable nesting substrate (i.e. bare sand or dirt) on islands devoid of mammalian predators seemed to be the greatest single limiting factor for Caspian tern colony size and nesting success in the CPR (Antolos et al. 2004, Adkins et al. 2014). Local availability of forage fish was also apparently a limiting factor for the tern colonies at Goose Island in Potholes Reservoir, Banks Lake, and Sprague Lake. Caspian terns from these colonies commuted over 100 km round trip to feed on juvenile salmonids in the Columbia River (Goose Island, Banks Lake, and Sprague Lake colonies) or in the Snake River (Banks Lake and Sprague Lake colonies; Antolos et al. 2004, Evans et al. 2012, BRNW 2014 [Appendix C]). Flooding due to fluctuating reservoir levels was observed to limit Caspian terns nesting at the Blalock Islands complex in most years. Although the USACE, the agency responsible for dam operations in the lower Columbia and Snake rivers, regulates spill at the dams, fluctuations in reservoir elevation do occur (USACE 2014). As a result, some of the low-lying islands in the Blalock Islands complex where Caspian terns have nested are subject to flooding during storm events, spring run-off, and fluctuations in spill rate and volume. Finally, encroachment by nesting gulls at Crescent Island and Goose Island and by nesting American white pelicans at Badger Island have been observed to be an important limiting factor at some Caspian tern colonies in the region (Adkins et al. 2014).

2.4.2.5 Dispersal & colony connectivity – Caspian tern dispersal and nesting colony connectivity were evaluated in the context of management at the Goose Island and Crescent Island colonies. Two data sources were used in this evaluation, namely (1) resightings of previously banded Caspian terns (Schniedermeyer 2018) and (2) results from a tracking study using satellite telemetry (Roby et al. 2019b). These results are presented separately *below*.

Band resightings: A total of 184 banded Caspian terns were classified as breeders at one of the two managed colonies prior to the initiation of management at Goose Island in 2013 or at Crescent Island in 2014. For this sample of banded terns, the proportion that transitioned from being breeders in the CPR to other status categories differed significantly between the pre-management period (2011–2012 and 2012–2013 combined) compared with the management period (2014–2015 and 2015–2016). In other words, the proportion of breeders that also bred in the region the next year, as opposed to becoming non-breeders of leaving the region altogether, declined significantly from the pre-management years to the management years. Based on post-hoc analysis of statistically significant differences in transition proportions during management, the proportion of terns that remained as breeders in the CPR between years significantly declined from 91% during pre-management to 61% and 60% during 2014–2015 and 2015–2016 in the management period, respectively. The proportion of terns that transitioned from breeders in the CPR to non-breeders in the CPR significantly increased from 4% during the pre-management period to 20% and 24% during 2014–2015 and 2015–2016 in the management period, respectively. Despite fewer terns remaining as breeders in the CPR post-management, the available band resighting data did not support the expectation that more terns emigrated to other regions to breed in 2015 and 2016, the first two years after implementation of the *IAPMP*. The proportions of terns that transitioned from breeders in the CPR to either breeders outside the region or to non-breeders outside the region were not significantly different between the pre-management period and early stages of management period (2015–2016). These results indicated that, at least for the first two years following implementation of the *IAPMP*, Caspian terns that had formerly nested at the two largest colonies in CPR primarily prospected for nesting opportunities within the CPR, rather than dispersing outside the region, but in many cases were not successful in nesting at a different colony within the CPR. See Schniedermeyer et al. (2020) for more details on study results.

Satellite telemetry: Satellite-tagged Caspian terns, like their banded counterparts, exhibited stronger philopatry (fidelity to a breeding region) to the CPR than anticipated for a species that is known to engage in long-distance dispersal from breeding sites. Most terns with actively transmitting tags were detected in the CPR during the peak of the breeding season, even in the fourth breeding season following initiation of management that successfully prevented nesting at the Goose Island and Crescent Island colonies. During the fourth breeding season after initiation of management under the *IAPMP*, 68.1% of detected locations of tagged terns were still in the CPR, and 36.3% of detected locations in the region were at former or existing breeding colonies for the species. Of the 16 tagged terns that were tracked during their fourth nesting season after tagging, 14 (87.5%) spent most of the season in the CPR. Much of the high regional philopatry of tagged terns following implementation of the *IAPMP* was due to the high proportion of terns that formerly nested at the Crescent Island colony (river km [Rkm] 510 in McNary Reservoir) shifting to nest at the colony on the Blalock Islands (Rkm 440 in John Day Reservoir; see *above*).

This satellite telemetry dataset was also used to explore the dispersal capacity and nesting colony network structure of Caspian terns in the Pacific Flyway of North America (Roby et al. 2019b). Caspian terns have the capacity to visit a significant portion (mean = 17%) of colonies in this extensive Flyway each year, traveling > 3,500 km/year to do so across annual individual networks spanning an average of > 300,000 km^2. Mapping the linkages between colonies in a variety of ways – both geographically and in various types of 'network space' allows assessment of network structure based on geographic proximity and other factors (*Figure 2.14*). At the scale of the entire Pacific Flyway (south-central Alaska to southern California), distance was related to measures of network connectivity for individual colonies. If the spatially distant colonies in Alaska are removed from the analysis of network connectivity, distance is no longer a significant effect. Thus, dispersal between colonies in Washington, Oregon, and California is well within the capacity of Caspian terns. Through these analyses, a subset of colonies that facilitate connectivity between regions were identified. Maintaining active colonies outside the Columbia River basin that are particularly well linked to colonies in the Columbia Plateau region, such as those on the Everett, Washington waterfront, in Arcata Bay, California, and at the Salton Sea of California, would likely facilitate greater dispersal away from the Columbia Plateau region and support efforts to reduce Caspian tern predation on juvenile salmonids there. See Roby et al. (2019b) for more details on results from this study.

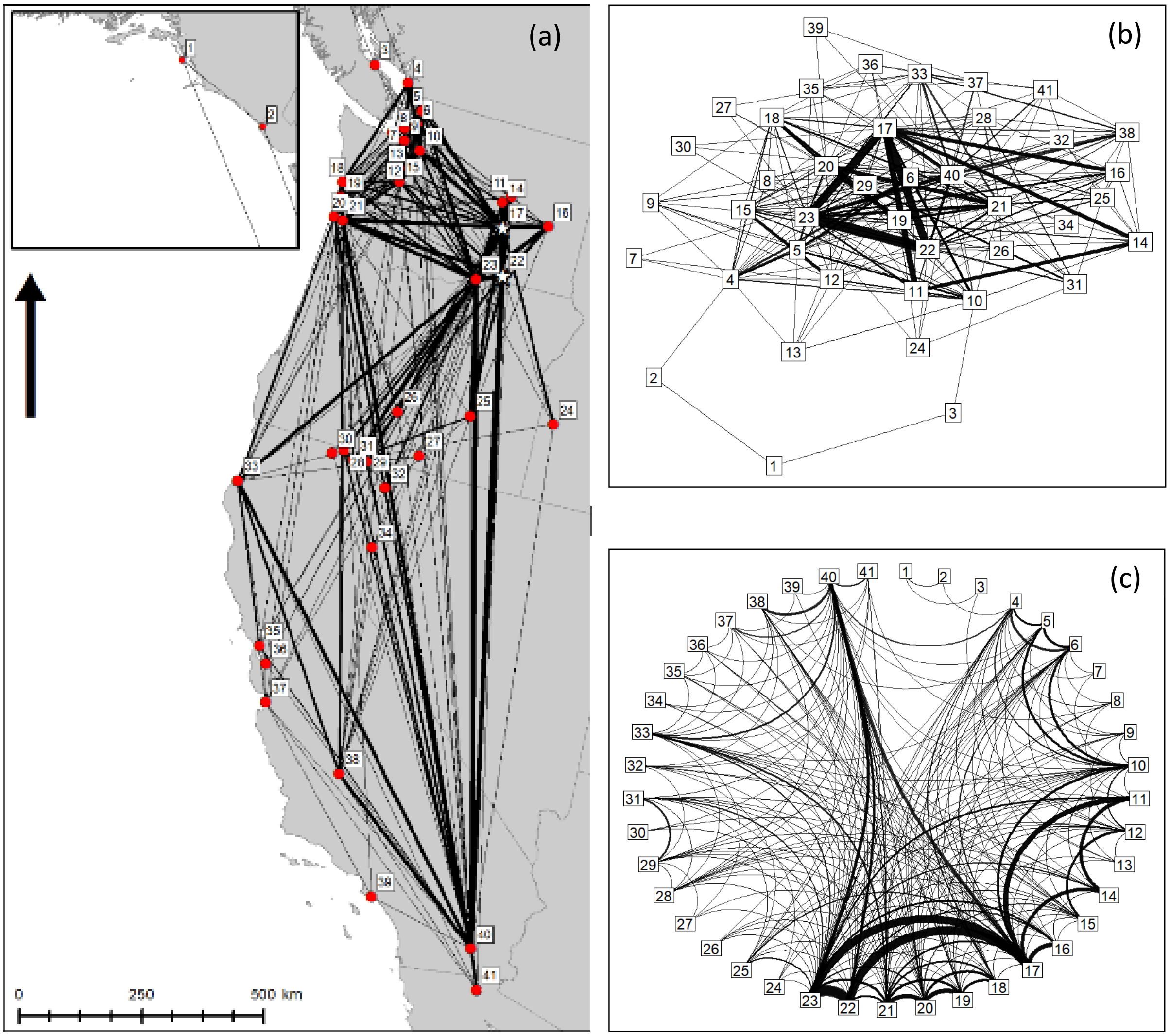


Figure 2.14. Network graphs for tern colonies in the Pacific Flyway. Colonies (nodes) and links between colonies (interconnecting lines or "edges") are plotted (a) geographically, (b) using the Kamada-Kawai algorithm, and (c) in a circular network plot with colonies ordered from north to south. Numerical colony labels in (a) correspond to colony numbers in the other two graphs. In all graphs, the line weights indicate the edge weights (number of terns linking the two colonies). Adapted from Roby et al. (2019b).

The slow pace of dispersal from the CPR by tagged Caspian terns that formerly nested at the two managed nesting colonies in the region indicates that most displaced terns persisted in prospecting for alternative nesting colony sites within the region. Although tagged terns made many apparent exploratory trips both within and outside the CPR, most terns that ventured away from the region during the nesting season returned to the region instead of permanently emigrating. This behavior has two implications: (1) tagged terns were unsuccessful in locating suitable alternative nesting habitat outside the CPR and/or (2) available alternative nesting habitat in the CPR, although less suitable than the former nesting colony sites on Goose and Crescent islands (see *above*), offered more favorable conditions for nesting and foraging (see *below*) than sites outside the region. See Roby et al. (2019b) for more detailed results from this study.

2.4.3 Foraging Ecology

2.4.3.1 Foraging behavior – The foraging behavior of Caspian terns in the CPR was investigated in the context of assessing the need for management at the Goose Island and Crescent Island colonies and evaluating the response to management in the aftermath of implementation of the *IAPMP*. Three data sources for this assessment and evaluation were used, namely (1) Global Positioning System (GPS) tagging of Caspian terns nesting on Goose Island in 2013 (BRNW 2014, Appendix C), (2) satellite tracking of Caspian terns tagged at the Goose Island and Crescent Island colonies in 2014–2015 (Roby et al. 2019a), and (3) recovery of smolt tags from Caspian tern colonies in 2012 and 2014 (Evans et al. 2016a). These results are presented separately *below*.

GPS tagging of Caspian terns: Data from 23 of the 28 GPS transmitters deployed on Caspian terns nesting at Goose Island in Potholes Reservoir in 2013 were retrieved; tracking periods for each transmitter ranged from less than 6 hours to 5 days. A total of 53 complete days of tracking data from 20 terns, between 21 May and 27 May were collected. Furthermore, 14 partial days of tracking data from tags that stopped working during the middle of the day were retrieved. Due to tag failure, we were only able to retrieve a partial day of data for three tagged individuals and we were unable to retrieve any data from transmitters on five tagged individuals. Most terns were attending nests with eggs throughout the tracking period.

A total of 97 foraging trips by 22 different Caspian terns nesting on Goose Island (*Figure 2.15*) were recorded. Half of the terns made at least one trip to the middle Columbia River and 32% of the terns exclusively made foraging trips to the Columbia River. Just over half of the terns (55%) made foraging trips within the Potholes Reservoir area and 23% of terns exclusively used the Potholes Reservoir area. Three terns (14%) made foraging trips to the Snake River. Three individuals (14%) used foraging areas that were located between Potholes Reservoir and the Columbia and Snake rivers (i.e. Eagle Lakes, Scooteney Reservoir, and Columbia National Wildlife Refuge). There was a high degree of consistency within individual terns in the primary destination during foraging trips: only 5 of 19 individuals with multiple foraging trips recorded used two different primary foraging destinations.

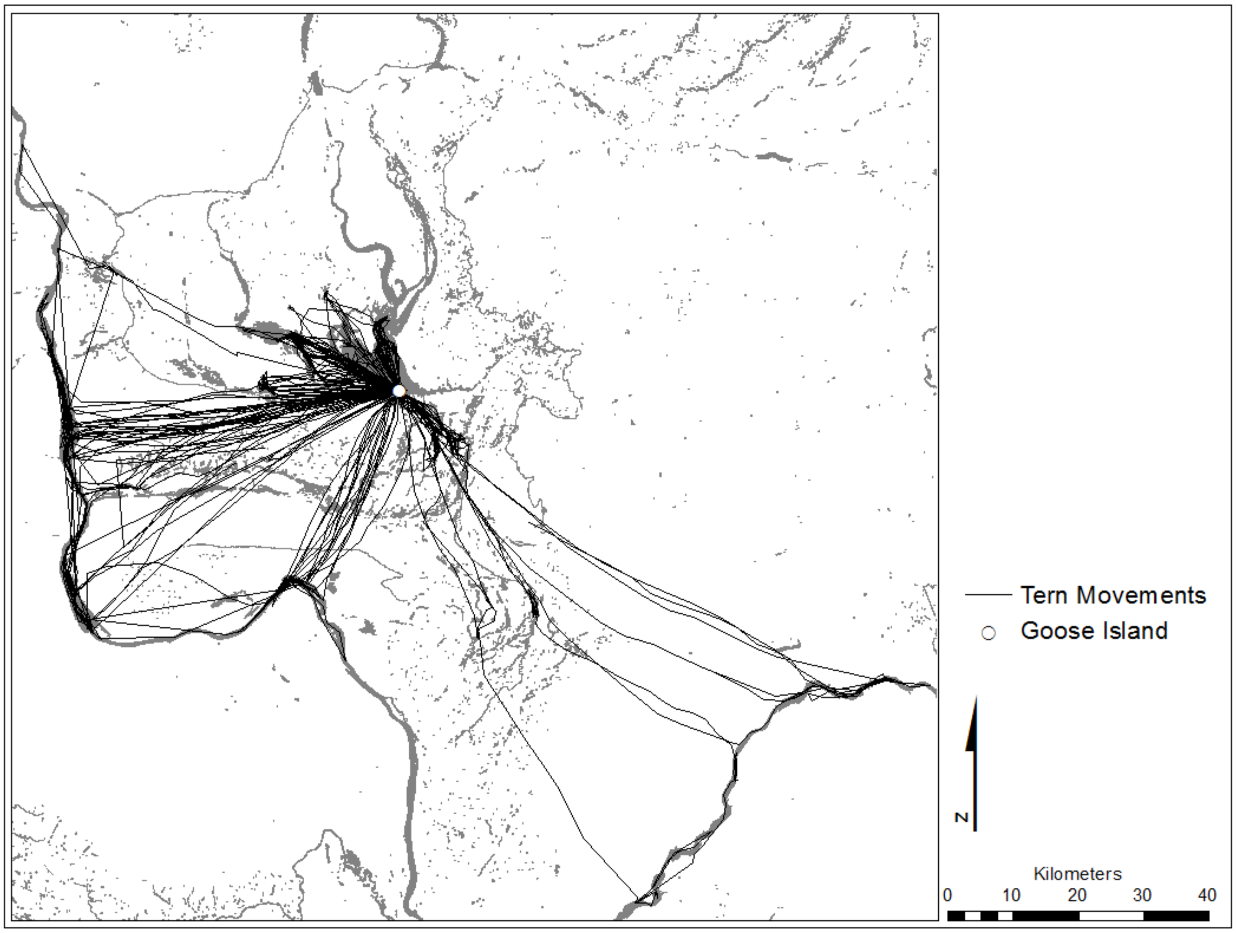


Figure 2.15. All movements recorded from Caspian terns equipped with GPS data-loggers and nesting on Goose Island in Potholes Reservoir, Washington, during the 2013 nesting season. Adapted from BRNW (2014).

There were 32 foraging trips recorded to the middle Columbia River (33%), ranging from Rock Island Dam in the north to 10 km downstream of the Hanford Reach in the south (*Figure 2.15* above). Foraging locations along the middle Columbia River were concentrated in the Hanford Reach, Priest Rapids Reservoir, and Wanapum Reservoir (up to 25 km upstream of Wanapum Dam). On average, trips to the Columbia River lasted 4.5 to 6.5 hrs (95% confidence interval = 277–391 min) and included multiple bouts of foraging activity (1.87–3.13). Foraging trips to the Columbia River included more time foraging (94–139 min) than trips within the Potholes Reservoir area. Time spent commuting (78–102 min) and resting (80–160 min) were longer for foraging trips to the Columbia River compared to foraging trips in the Potholes Reservoir area, but shorter compared to foraging trips to the Snake River.

Four foraging trips to the Snake River by three different individuals (*Figure 2.15* above) were recorded; two of the three terns were attending nests with eggs, while one tern was not actively nesting. One of the two nesting terns was tracked for two days and made one trip to the Snake River on each day. The other nesting tern was tracked for three days; on the first day it went to the Snake River and on the other two days it remained on-colony all day. Foraging trips to the Snake River extended as far as 87–93 km straight line distance from the nesting colony and took most of the day (550–912 min), longer than foraging trips to any other area. Foraging on the Snake River occurred from Lyon's Ferry Fish Hatchery in the east downstream to 36 km below Lower Monumental Dam. Foraging trips to the Snake River included more foraging bouts (3.75–7.45 bouts), more time commuting (115–191 min), and more time resting (285–524 min) than trips to other areas. The amount of time spent foraging during trips to the Snake River was similar to that of foraging trips to the Columbia River (98–253 min).

Most foraging trips recorded (60%) occurred within the area around Potholes Reservoir (within 20 km of the nesting colony). These trips took significantly less time than trips to the Columbia River or the Snake River (71–134 min) and included fewer foraging bouts (1.15–1.81 bouts).

Eight core foraging areas were identified: three areas along the middle Columbia River, one area on the lower Snake River, and four areas close to, or within, Potholes Reservoir. The most individual terns foraged in Wanapum Reservoir just north of Wanapum Dam. Priest Rapids Reservoir and the Hanford Reach were the two other core foraging areas along the middle Columbia River. One core foraging area, used by two terns, was identified on the lower Snake River, from about 15 km upstream from Lower Monumental Dam to about 8 km downstream from the Lyons Ferry Fish Hatchery. Close to the Goose Island nesting colony, most terns foraged in the Winchester Wasteway and the northeastern portion of Potholes Reservoir. Core foraging areas were also identified at Seep Lakes and Frenchman Hills. See BRNW 2014, Appendix C for more details on this study.

Satellite tracking of Caspian terns: Most Caspian terns with actively transmitting satellite tags were detected in the CPR during the peak of the steelhead outmigration, even in the fourth breeding season after the initiation of management that prevented nesting at both the Goose Island and Crescent Island colonies (*Figure 2.16*). A total of 76 satellite-tagged Caspian terns were tracked for an average of 705 days (SD = 406, range = 1 day–1,555 days). Considering terns that were satellite-tagged at either nesting colony site, the movements of 56 tagged Caspian terns were tracked during the steelhead smolt out-migration period in the initial year of management under the *IAPMP* (Year 0; 2014 for Goose Island terns, 2015 for Crescent Island terns). The movements of 53 tagged terns were tracked during the steelhead out-migration in the second breeding season following the initiation of management (Year 1), and of 36 tagged terns during the steelhead out-migration in the third breeding season following the initiation of management (Year 2). Finally, the movements of 16 tagged terns were tracked during the steelhead out-migration in the fourth breeding season following the initiation of management (Year 3; 2017 for Goose Island terns, 2018 for Crescent Island terns).

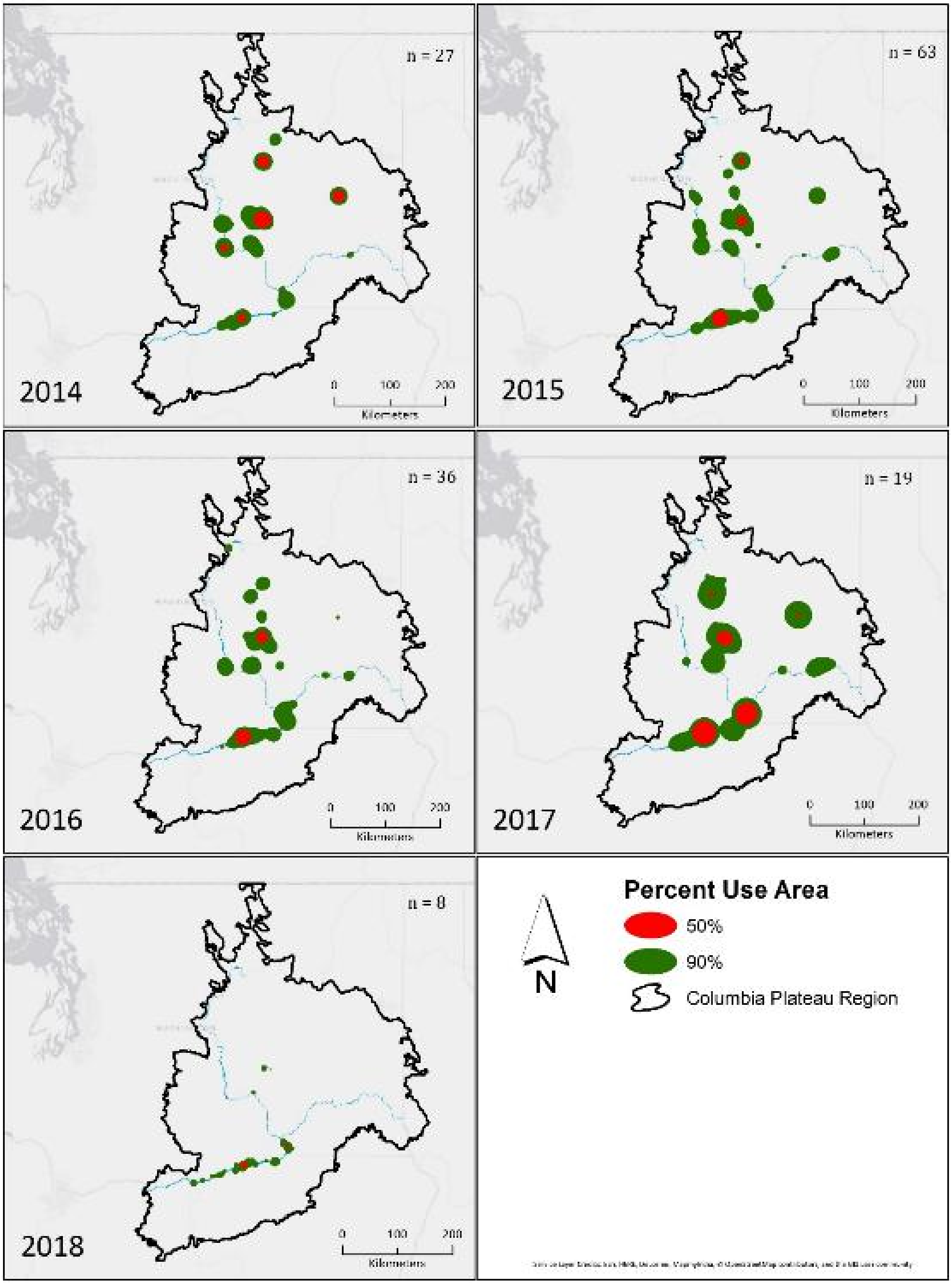


Figure 2.16. Utilization distributions of tagged Caspian terns across the Columbia Plateau region during the peak of steelhead smolt out-migration in 2014–2018. Red indicates the area where the probability of detecting an individual tagged tern was 50% and green indicates the area where the probability of detecting a tagged tern was 90% (note: areas of higher percentage use include the areas of lower percentage use that are contained within). Only locations within the Columbia Plateau region were used in calculating utilization distributions. Adapted from Roby et al. (2019a).

AIC$_c$ model selection indicated that there was differential use of foraging areas by tagged terns depending on whether the tern was tagged at the Goose Island or Crescent Island nesting colony; terns tagged at Goose Island foraged mainly along the Columbia River above the confluence with the Snake River, and terns tagged at Crescent Island foraged mainly along the Columbia River below the confluence (Roby et al. 2019a). The KDE analyses also indicated, however, that tagged terns concentrated at many of the same foraging, loafing, and nesting sites in the Columbia Plateau region throughout the 5-year study (2014–2018). Nevertheless, the proportion of terns tagged at the Goose Island nesting colony that subsequently foraged along the Columbia River above the confluence with the Snake River declined steadily and significantly following implementation of the *IAPMP*, supporting the conclusion that successfully dissuading terns from nesting at Goose Island provided benefits to smolt survival in that stretch of the Columbia River. On the other hand, the proportion of tagged terns that foraged along the Columbia River below the confluence with the Snake River did not decline significantly during the management period (2015–2018), because many Caspian terns that formerly nested at the Crescent Island colony in McNary Reservoir (Rkm 510) shifted to the nesting colony at the Blalock Islands in John Day Reservoir (Rkm 440) and continued to forage along that stretch of the Columbia River. There was little foraging use of the lower Snake River by tagged Caspian terns following the elimination of the Crescent Island tern nesting colony. See Roby et al. (2019a) for more detailed results from this study.

Recovery of smolt tags from Caspian tern colonies: Results from a study where juvenile salmonids were double tagged with JSATS and PIT tags (see section *2.3.3.1 Foraging behavior* above) indicated that Caspian terns nesting in the CPR disproportionately foraged on juvenile salmonids within reservoirs, as opposed to near hydroelectric dams, which are hotspots for gull predation on salmonid smolts in the region (see *Chapter 6*; Evans et al. 2016a). Results also indicated that Caspian terns nesting on Crescent Island disproportionately commuted upstream of their nesting colony to forage on steelhead in McNary Reservoir and in the lower Snake River, while Caspian terns nesting at the Blalock Islands commuted to forage along the lower Columbia River both upstream and downstream of the islands. Caspian tern predation on salmonids along the mid-Columbia River was primarily within Wanapum and Priest Rapids projects by terns nesting at three different colonies: Twinning Island in Banks Lake, Goose Island in Potholes Reservoir, and Crescent Island in McNary Reservoir. The foraging range of Caspian terns feeding on juvenile steelhead tended be the greatest (up to 90 km), as compared to other piscivorous colonial waterbirds nesting in the region. See Evans et al. (2016a) for more details on this study.

2.4.3.2 Diet composition & smolt consumption – The average annual proportion of juvenile salmonids (percent of identified prey items) in the diet of Caspian terns nesting on Goose Island and Crescent Island prior to management was about 21% and 69%, respectively (*Figure 2.17*). During the pre-management period (2010–2012, years when seasonal diet information at both colonies were collected), juvenile salmonids were most prevalent in the diets of Caspian tern nesting on Goose and Crescent islands in early to mid-May (*Figure 2.18*), which corresponds with the peak in the out-migration of juvenile salmonids through the CPR (FPC 2020). Juvenile salmonids were the single most prevalent prey type for terns nesting at the Crescent Island colony, while centrarchids (bass, sunfish) were the most prevalent prey type in the diet of terns nesting at the Goose Island colony (*Figure 2.19*). The annual proportion of juvenile salmonids in the diet of Caspian terns nesting at the Crescent Island colony was similar across years (*Figure 2.17*), suggesting that juvenile salmonids represent a preferred and consistently available prey type in the vicinity of that nesting colony. At the Goose Island nesting colony, the average annual proportion of juvenile salmonids in the diet was more variable compared to at the Crescent Island nesting colony (*Figure 2.17*), apparently related to inter-annual differences in the availability of forage fish (i.e. yellow perch) in Potholes Reservoir and at other foraging locations off the Columbia River. Estimates of the proportion of salmonids in the diet of Goose Island terns, however, should be interpreted cautiously. Unlike other Caspian tern colonies in the CRB where the composition of Caspian tern bill-load samples was representative of what adult terns were consuming themselves (Collis et al. 2002), this was not confirmed for Caspian terns nesting at Goose Island. It is possible, perhaps likely, that given the distance commuted by Goose Island terns to forage along the mid-Columbia River, breeding adults are consuming different fish than what they are bringing back to the colony to feed their mates and young. For this reason, diet composition data based on identification of bill-load fish were not used to estimate smolt consumption for Caspian terns nesting on Goose Island (see *below*). Bioenergetics calculations to estimate total annual consumption of juvenile salmonids by Caspian terns nesting at Crescent Island during the pre-management period (2000–2012) indicated that an average of ca. 0.46 million salmonid smolts were consumed annually (*Figure 2.20*), which is approximately an order of magnitude less than what was consumed during the pre-management period by Caspian terns nesting at the East Sand Island colony in the Columbia River estuary (ca. 5.0 million smolts; see *Chapter 1*). Data on diet composition and smolt consumption by Caspian terns nesting at Goose and Crescent islands were not collected during the management period.

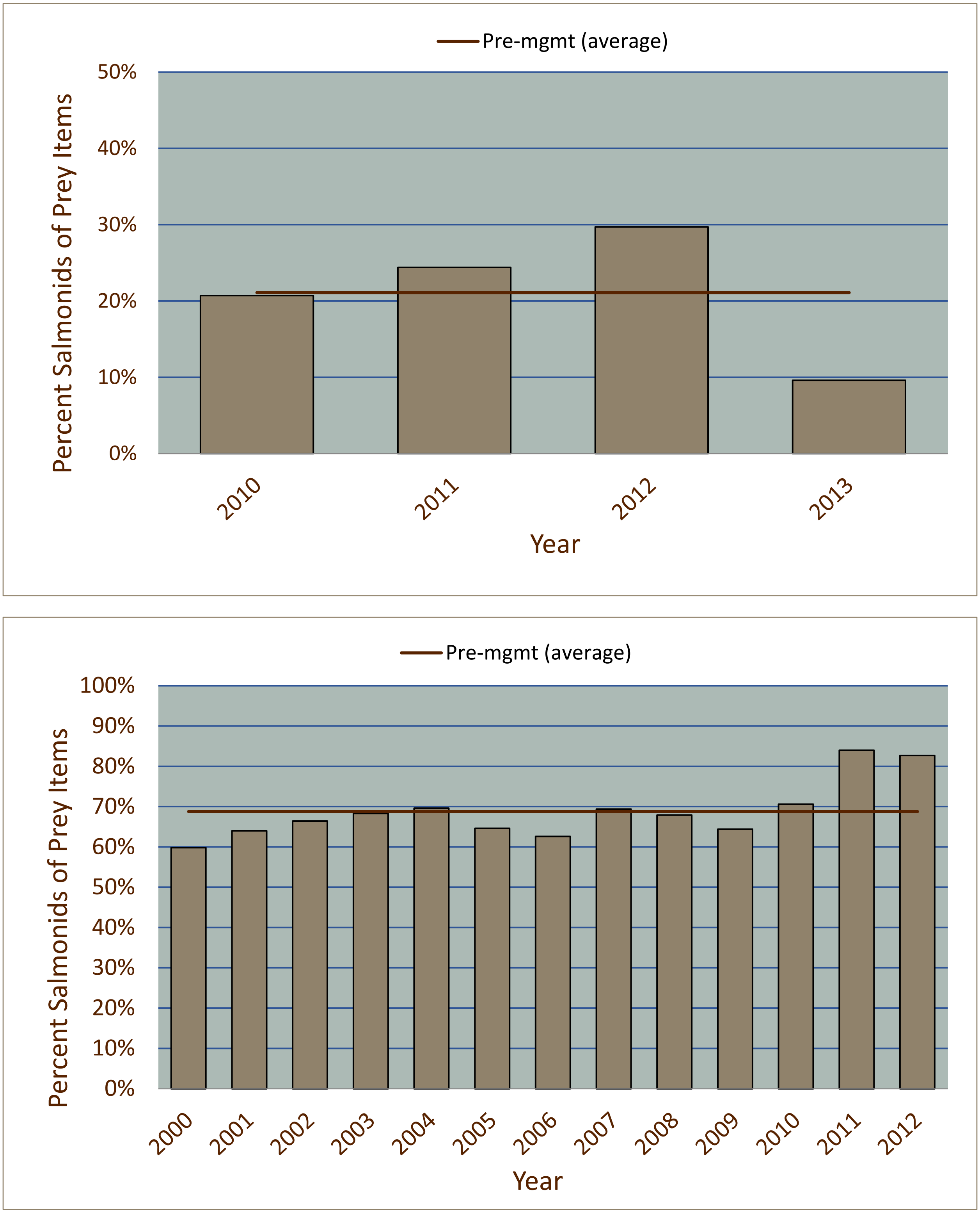


Figure 2.17. Annual proportion of juvenile salmonids in the diet (percent of prey items in bill-loads) for Caspian terns nesting on Goose Island in Potholes Reservoir (top) and on Crescent Island in the Columbia River (bottom) during the pre-management period.

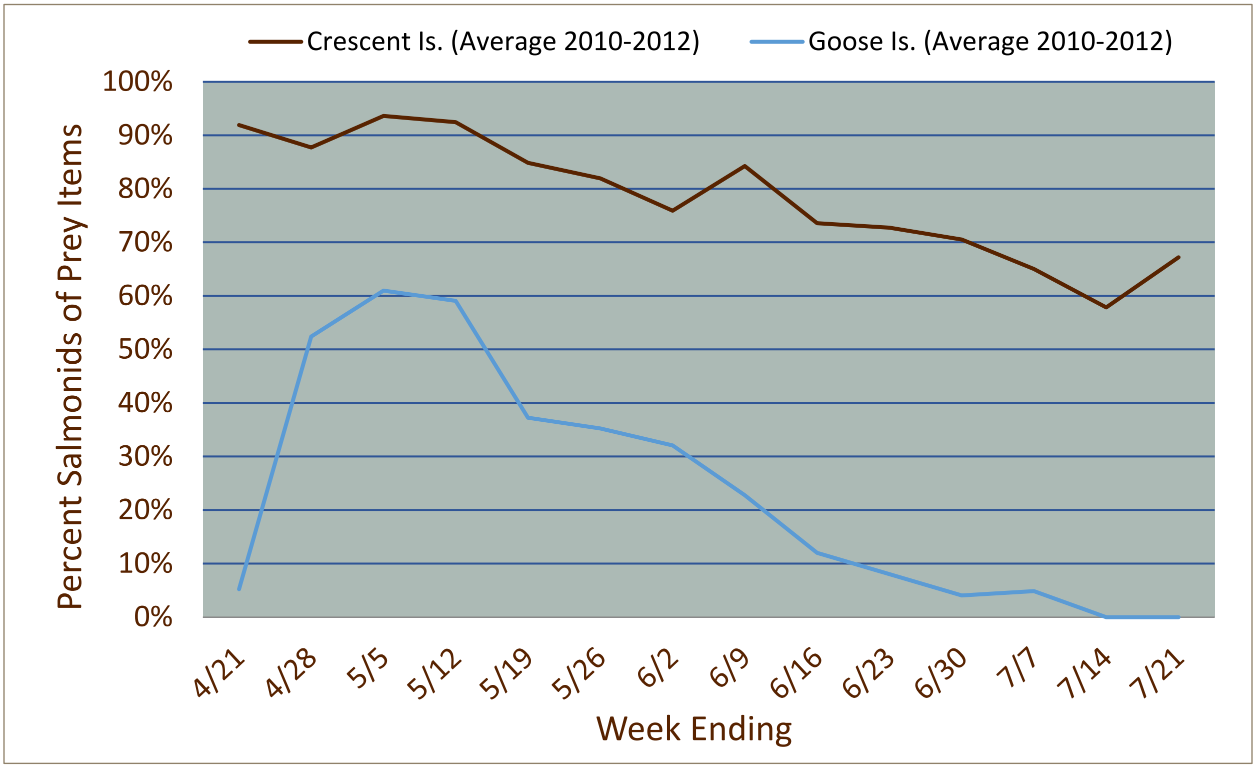
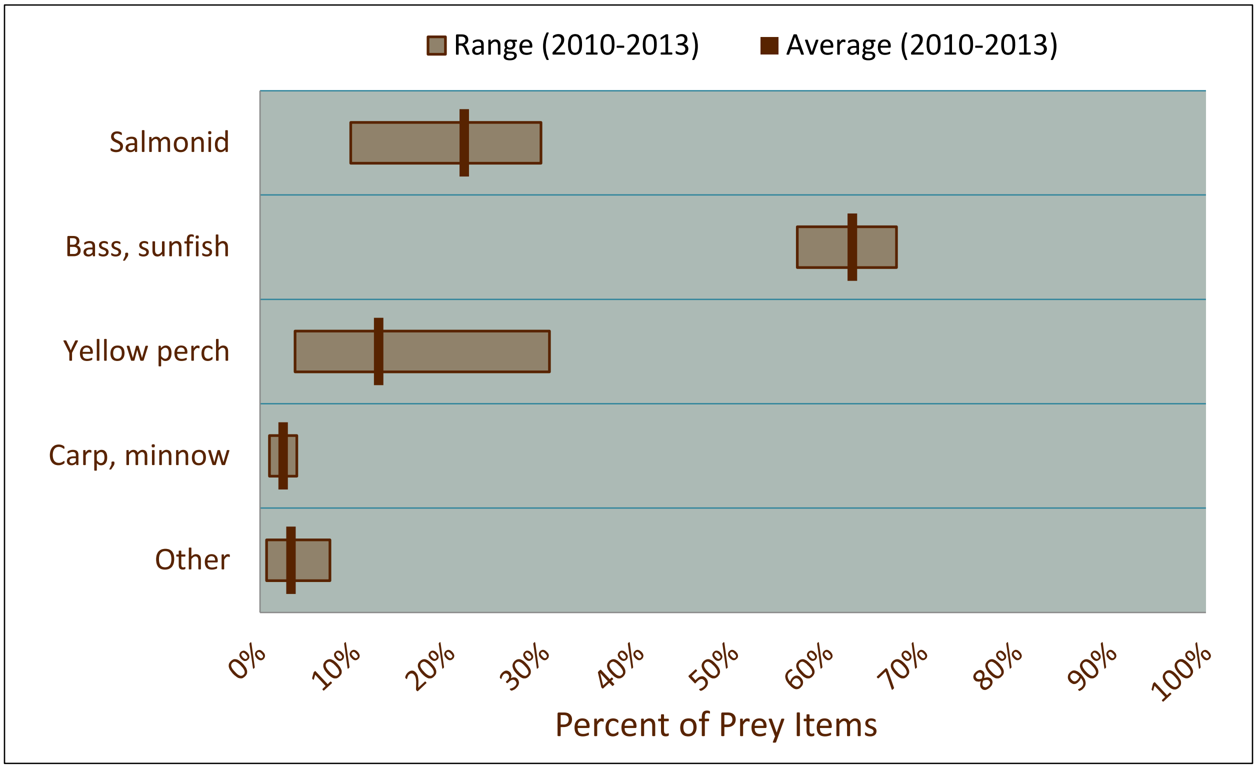


Figure 2.18. Weekly proportion of juvenile salmonids in the diet (percent of prey items in bill-loads) for Caspian terns nesting on Goose Island in Potholes Reservoir and on Crescent Island in the Columbia River during the pre-management period.



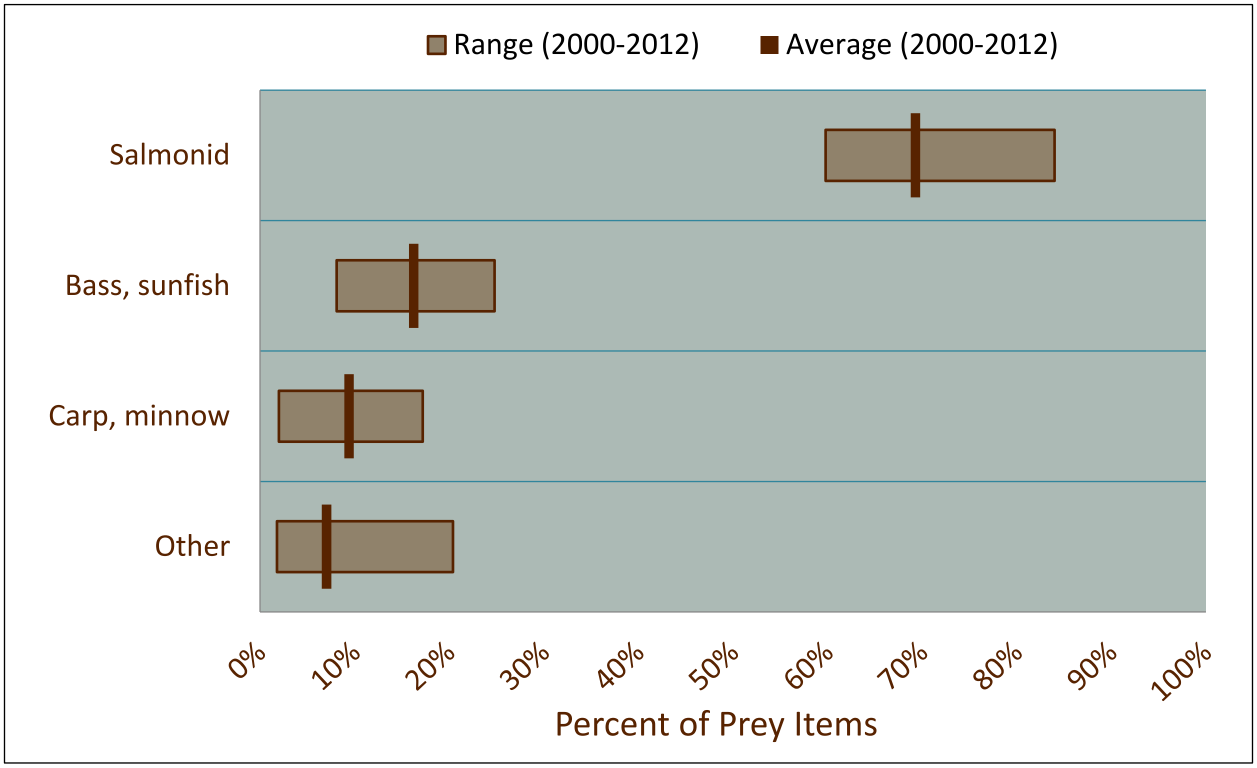


Figure 2.19. Diet composition (percent of prey items in bill-loads) for Caspian terns nesting on Goose Island in Potholes Reservoir (top) and on Crescent Island in the Columbia River (bottom) during the pre-management period.

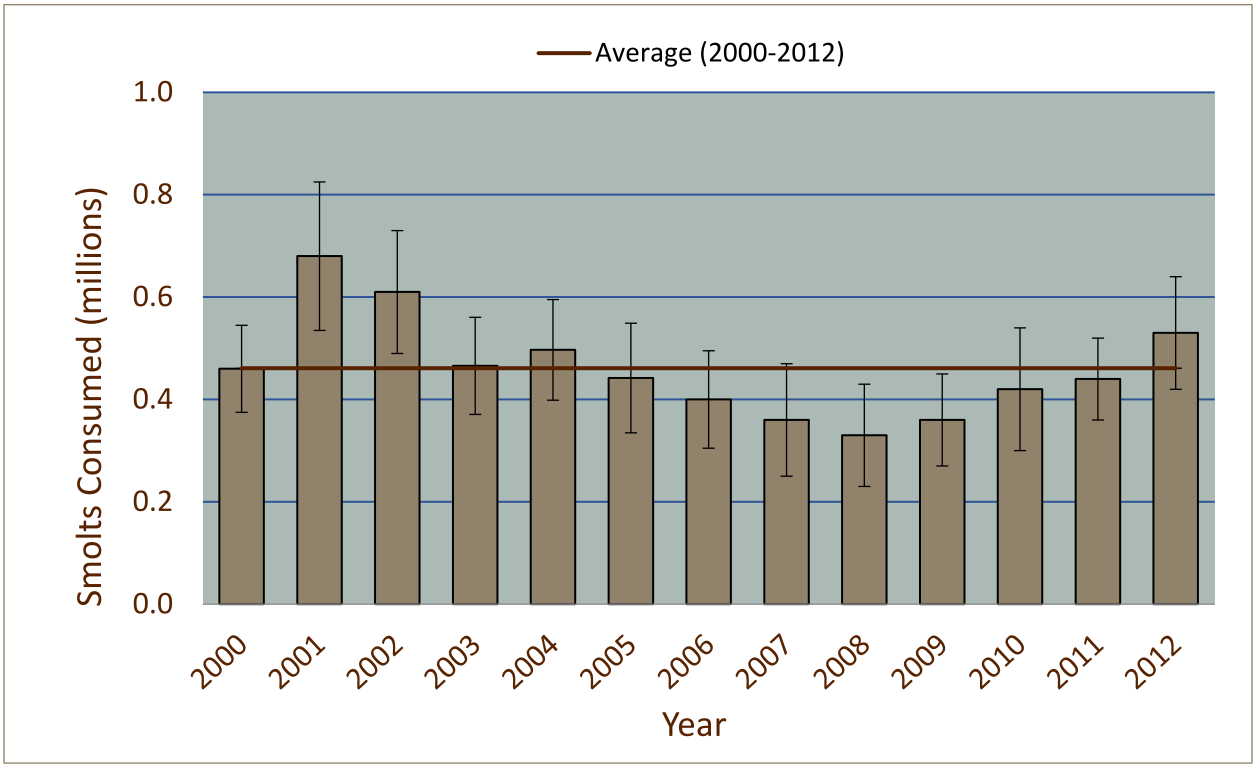


Figure 2.20. Estimated number of juvenile salmonids consumed by Caspian terns nesting on Crescent Island in the Columbia River during the pre-management period. Error bars represent 95% confidence intervals.

2.4.3.3 Colony-specific predation rates – Studies investigating predation rates on juvenile salmonids by Caspian terns in the CPR were first initiated at the Crescent Island tern nesting colony in 1999 (Ryan et al. 2001), followed by studies at the Goose Island tern nesting colony starting in 2005 (Maranto 2010), and then at several smaller tern colonies starting in 2007 (Evans et al. 2012). To date, over 250,000 PIT tags (detections of individual fish) have been recovered on tern colonies in the CPR. Most of these tagged fish were anadromous juvenile salmonids (> 99%), but small numbers of tags from other fish species (Pacific lamprey [*Entosphenus tridentatus*], bull trout [*Salvelinus confluentus*], northern pikeminnow [*Ptychocheilus oregonensis*], and smallmouth bass [*Micropterus dolomieu*]) were also recovered on tern colonies in the CPR. Standardized estimates of predation rates, those corrected for detection and deposition probabilities (see *Appendix A.2*), on ESA-listed PIT-tagged juvenile salmonids were available starting in 2007, the first pre-management year specified in the *IAPMP* (USACE 2014). A summary of annual estimates of predation rates by Caspian terns nesting at colonies in the CPR is provided below. Results are presented first for the managed colony sites in Potholes Reservoir and on Crescent Island, and then for the unmanaged tern colony sites throughout the CPR.

Goose Island in Potholes Reservoir: Prior to implementation of management actions under the *IAPMP* to reduce the size of the Caspian tern nesting colony on Goose Island in Potholes Reservoir, predation rates varied considerably by salmonid species and ESU/DPS. Predation rates were higher on smolts originating from the Upper Columbia River (UCR) relative to the Snake River (SR) due to the closer proximity of Goose Island to smolts out-migrating through the middle Columbia River compared with the lower Snake River (Evans et al. 2012). During 2007–2013, average annual predation rates on UCR salmonids ranged from 2.5% (95% credible interval = 1.7–3.6%) on UCR spring Chinook to 15.7% (14.1–18.9%) on UCR steelhead (*Table 2.4*; see *Appendix B, Table B.21* for annual estimates). Predation rates as high as 22.6% (17.2–33.7%) were observed on UCR steelhead in some years, impacts that were amongst the highest observed for any piscivorous colonial waterbird nesting colony in the Columbia Basin (*Appendix B, Table B.18-30*). Predation rates by Goose Island terns on all other ESUs/DPSs evaluated (i.e. those from the Snake River) were less than 0.1% of available fish during the pre-management period (*Table 2.4*). Following implementation of management actions on Goose Island in 2014, a nesting colony of 159 breeding pairs of Caspian terns formed on a rocky islet adjacent to Goose Island and estimates of predation rates indicated that terns nesting there consumed an estimated 0.6% (0.1–2.2%) and 2.9% (1.9–5.1%) of available PIT-tagged UCR spring Chinook and UCR steelhead, respectively (*Appendix B, Table B.21*). During 2015-2019, management actions were successful at eliminating the Goose Island tern nesting colony, resulting in estimates of predation rates less than 0.1% per ESU/DPS, per year (*Table 2.4*). Collectively, estimates of predation rates by Caspian terns nesting at Goose Island indicated that the goal of reducing predation rates to less than 2% per ESU/DPS per year was achieved in five of the six management years and during the most recent three-year management period (2017–2019; *Table 2.4*).

Table 2.4. Average annual predation rates (95% credible intervals) by Caspian terns at managed colonies during the pre-management period (2007-2013 for Goose Island and North Potholes Is.; 2007-2014 for Crescent Island) and during the management period (2014-2019 for Goose Island and North Potholes; 2015-2019 for Crescent Island). ESA-listed salmonid populations (ESUs/DPSs) from the Snake River (SR) and Upper Columbia River (UCR) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. Time periods are denoted as the average of all years with data or data from the last three years of the study period (2017–2019). NC denotes that no colony existed during that period. See Appendix B for annual estimates.

	Goose Is.			**North Potholes Is.**			**Crescent Is.**		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2019	Last 3-years 2017–2019	Pre-Management 2007–2013	Management 2016 [1]	Last 3-years 2017–2019	Pre-Management 2007–2014	Management 2015–2019	Last 3-years 2017–2019
SR Sockeye	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	1.5% (1.2–2.0)	< 0.1%	< 0.1%
SR Sp/Su Chinook	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	0.8% (0.7–1.0)	< 0.1%	< 0.1%
UCR Sp Chinook	2.5% (1.7–3.6)	< 0.1%	< 0.1%	NC	0.1% (0.1–0.3)	< 0.1%	0.5% (0.3–0.9)	< 0.1%	< 0.1%
SR Fall Chinook	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	1.0% (0.9–1.2)	< 0.1%	< 0.1%
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	4.5% (4.1–5.1)	< 0.1%	< 0.1%
UCR Steelhead	15.7% (14.1–18.9)	0.2% (0.1–0.5)	< 0.1%	NC	4.1% (2.9–6.3)	< 0.1%	2.5% (2.2–2.9)	< 0.1%	< 0.1%

[1] *Nesting colony formed in 2016 and was than actively managed during 2017–2019.*

Unnamed island in North Potholes Reservoir: Following implementation of management actions at Goose Island in Potholes Reservoir, a newly established Caspian tern nesting colony formed on an unnamed island in North Potholes Reservoir in 2016. Predation rate estimates were less than 0.2% for all ESUs/DPSs evaluated, with the notable exception of UCR steelhead at 4.1% (2.9–6.3%; *Table 2.4* above). Although steelhead are known to be particularly susceptible to Caspian tern predation (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012; see also *Chapter 10*), predation rates were higher than those anticipated based on the size of the nesting colony (144 breeding pairs) and the short duration of nesting activities (five weeks) in 2016. Results provide evidence that even a relatively small, short-lived nesting colony of Caspian terns in Potholes Reservoir can have a significant impact on steelhead survival in the mid-Columbia River. Estimates of predation rates on UCR steelhead by Caspian terns nesting at North Potholes in 2016 exceeded the 2% threshold specified in the *IAPMP*. Starting in 2017, adaptive management implemented to dissuade terns from nesting on the unnamed island and elsewhere in Potholes Reservoir were successful (see section *2.3.1.1 Nest dissuasion in Potholes Reservoir* above) and prevented the formation of new tern colonies in all subsequent years, effectively eliminating predation impacts to ESA-listed juvenile salmonids from Caspian terns nesting in the Potholes Reservoir area.

Crescent Island in McNary Reservoir: Prior to implementation of management actions to reduce the size of the Caspian tern nesting colony on Crescent Island in 2015, average annual predation rates ranged from 0.5% (0.3–0.9%) on UCR spring Chinook to 4.5% (4.1–5.1%) on SR steelhead during 2007–2014 (*Table 2.4* above; see *Appendix B, Table B.21* for annual estimates). Like Goose Island terns, predation impacts from Crescent Island terns were significantly higher on steelhead DPSs compared with salmon ESUs. Unlike terns nesting on Goose Island, however, terns nesting on Crescent Island consumed more salmonids originating from the Snake River; predation rate estimates for Crescent Island terns were consistently highest on SR steelhead, with estimates ranging annually from 3.0% (2.3–4.3%) to 6.1% (4.8–8.9%) during 2007–2014 (*Appendix B, Table B5*). Average annual predation rates were 1.5% (1.2–2.0%) on Snake River sockeye and less than 1.0% on all other salmon ESUs (*Table 2.4* above). Following implementation of management actions on Crescent Island in 2015, the nesting colony was eliminated, resulting in predation rate estimates of less than 0.1% per salmonid ESU/DPS during 2015–2019 (*Table 2.4* above). Results indicated that the management goal of reducing predation rates by Crescent Island terns to less than 2% per ESU/DPS was achieved in all years (2015–2019) following implementation of the *IAPMP* (*Table 2.4* above).

North Rock and Shoal Island in Lenore Lake: Following implementation of management actions at Goose Island in Potholes Reservoir, a new Caspian tern nesting colony formed on an island in Lenore Lake in 2014. Estimates of predation rates by Caspian terns nesting at Lenore Lake during 2014–2019 indicated that predation impacts were generally low, averaging less than 0.2% for all ESUs/DPSs of salmonids except UCR steelhead, where predation rates averaged 0.7% (0.5–1.1%; *Table 2.5*). These results suggest that at its current size, the Caspian tern nesting colony on Lenore Lake poses a minor risk to survival of UCR steelhead. Based on average annual per-capita predation rates on UCR steelhead by terns nesting at Lenore Lake (see section *2.4.3.6 Per capita predation rates*), the nesting colony would have to increase to more than 150 breeding pairs to potentially reach the 2% predation threshold identified in the *IAPMP*.

Twinning Island in Banks Lake: Prior to implementation of management actions under the *IAPMP*, average annual predation rates by Caspian terns breeding on Twinning Island were less than 0.2% per salmonid ESU/DPS (*Table 2.5*). Only small numbers of Caspian terns (less than 40 breeding pairs in most years) nested at Twinning Island during 2007–2013 (*Table 2.2*) and the small size of the colony, coupled with its distance from the middle Columbia River (at least 45 km), resulted in very low estimates of predation rates on salmonid smolts. Like other Caspian tern colonies in the region, the highest predation rates were observed on UCR steelhead (see *Appendix B, Table B.21*). Following implementation of management actions at the Caspian tern nesting colony on Goose Island in nearby Potholes Reservoir, a much larger tern nesting colony formed on Twinning Island. Predation rates in 2014 and 2015 were significantly higher than those observed prior to implementation of management actions, with rates as high as 2.6% (1.8–3.9%) on UCR steelhead in 2015 (*Appendix B, Table B.21*), rates that exceeded the *IAPMP* threshold of 2%. During the last three years of the study (2017–2019), however, very few (< 10 breeding pairs), if any, Caspian terns nested on Twinning Island and predation rates were estimated to be less than 0.1% per ESU/DPS (*Table 2.5*).

Harper Island in Sprague Lake: Prior to implementation of management actions, estimates of predation rates by Caspian terns breeding on Harper Island in Sprague Lake were only available in one year (2012). Results indicated that Caspian terns nesting on Harper Island consumed less than 0.3% of all salmonid ESUs/DPSs evaluated, with the highest estimated predation rate on SR steelhead (*Table 2.5*). Like Twinning Island, small numbers of terns nested on Harper Island in Sprague Lake (less than 40 pairs annually) during 2007–2013 (*Table 2.2*). Also, like Twinning Island, Harper Island is a considerable distance from out-migrating smolts, with the lower Snake River over 60 km away. Due to our inability to obtain permission to scan this nesting colony site on a privately-owned island during the management period, no estimates of predation rates were available. Given that estimates of predation rates were limited to just a single year, results should be interpreted cautiously, but there was no evidence that a small Caspian tern nesting colony on Harper Island poses a substantial risk to smolt survival in the lower Snake River. Collis et al. (2020) concluded that future monitoring of the Harper Island tern nesting colony may be warranted, particularly if a substantial increase in colony size occurs and permission to scan for PIT tags can be obtained.

Badger Island in McNary Reservoir: There were no efforts to recover smolt PIT tags from the small, incipient Caspian tern nesting colony on Badger Island prior to implementation of management actions at the nearby Crescent Island tern colony in 2015. In 2017, a tern nesting colony of 41 breeding pairs became established and persisted for most of the smolt out-migration season, before failing prior to fledging any young (see section *2.4.2.4 Limiting factors for colony size & nesting success* above). Predation rates by terns nesting at this colony in 2017 were estimated to be 0.4% (0.2–0.6%) and 0.5% (0.3–0.8%) of SR and UCR steelhead, respectively, with estimates for all other ESUs/DPSs less than 0.1% (*Table 2.5*).

Table 2.5. Average annual predation rates (95% credible intervals) by Caspian terns nesting at unmanaged colonies during the pre-management period (2007-2013) and during the management period (2014-2019) at Goose Island and Crescent Island. ESA-listed salmonid populations (ESUs/DPSs) from the Snake River (SR) and Upper Columbia River (UCR) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. Time periods represent the average of all years or data from the last three years of the management (2017–2019). NA denotes that predation estimates were not available. NC denotes that no colony existed during that period. See Appendix B for annual estimates.

	Twinning Island			Badger Island			Blalock Islands		
ESU/DPS	Pre-Management 2008–2013	Management 2014–2019	Last 3-years 2017–2019	Pre-Management 2007–2013	Management 2014–2019	Last 3-years 2017 [1]	Pre-Management 2007–2013	Management 2014–2019	Last 3-years 2017–2019
SR Sockeye	< 0.1%	0.1% (0.0–0.5)	NC	NA	< 0.1%	NA	0.2% (0.1–0.4)	1.6% (1.0–2.5)	1.8% (0.7–4.0)
SR Sp/Su Chinook	< 0.1%	< 0.1%	NC	NA	< 0.1%	< 0.1%	0.1% (0.1–0.2)	0.7% (0.5–0.9)	0.6% (0.4–0.9)
UCR Spr Chinook	< 0.1%	0.2% (0.0–0.7)	NC	NA	< 0.1%	< 0.1%	< 0.1%	0.6% (0.5–0.9)	0.8% (0.5–1.3)
SR Fall Chinook	< 0.1%	< 0.1%	NC	NA	< 0.1%	< 0.1%	< 0.1%	0.7% (0.6–1.1)	0.9% (0.6–1.4)
SR Steelhead	< 0.1%	< 0.1%	NC	NA	< 0.1%	0.4% (0.2–0.6)	0.5% (0.4–0.9)	3.7% (3.1–4.6)	3.1% (2.4–4.1)
UCR Steelhead	0.1%	1.1% (0.8–1.6)	NC	NA	< 0.1%	0.5% (0.3–0.8)	0.5% (0.3–0.7)	4.3% (3.6–5.6)	4.5% (3.4–6.1)

	Lenore Lake islands			Harper Island		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2019	Last 3-years 2017–2019	Pre-Management 2007–2013	Management 2014–2019	Last 3-years 2017–2019
SR Sockeye	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
SR Sp/Su Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
UCR Spr Chinook	NC	< 0.1%	0.1% (0.0–0.2)	< 0.1%	NA	NA
SR Fall Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
SR Steelhead	NC	< 0.1%	< 0.1%	0.2% (0.1–1.3)	NA	NA
UCR Steelhead	NC	0.7% (0.5–1.1)	0.9% (0.6–1.4)	< 0.1%	NA	NA

[1] *No established nesting colony existed in 2018 and 2019.*

Blalock Islands in John Day Reservoir: Prior to implementation of the management actions (2007–2013), predation rates by Caspian terns nesting on the Blalock Islands were less than 1.0% on all ESUs/DPSs evaluated, and the highest estimated predation rates were on SR and UCR steelhead (*Table 2.5* above; see *Appendix B, Table B.20*). Following implementation of management actions at Crescent Island in 2015, however, the size of the Blalock Islands tern nesting colony grew rapidly (*Table 2.2* above) and predation rates on juvenile salmonids increased significantly for all ESUs/DPSs evaluated (*Table 2.5* above and see *Appendix B, Table B.20* for annual estimates). In 2015, the first year when management occurred on both the Goose Island and Crescent Island tern colonies, predation rates on SR and UCR steelhead by Blalock Islands terns were as high as 8.0% (6.0–11.4%) and 8.2% (5.9–12.4%), respectively (*Appendix B, Table B.20*). Since 2014, average annual estimates of predation rates on UCR and SR steelhead have been 3.7% (3.1–4.6%) and 4.3% (3.6–5.6%), respectively (*Table 2.5* above). During the last three years of study (2017–2019), estimates of predation rates on UCR and SR steelhead have averaged 4.5% (3.4–6.1%) and 3.1% (2.4–4.1%), respectively, estimates that are similar to or greater than those observed at the Crescent Island tern nesting colony prior to management (*Table 2.4* above). Predation rates on SR sockeye also exceeded the 2% threshold identified in the *IAPMP* in some, but not all, years during the management period, with estimates as high as 2.3% (1.2–4.1%) in 2016 (see *Appendix B, Table B.20*). Prior to management, predation rates on SR sockeye at the Blalock Islands tern nesting colony were less than 0.5% annually (see *Appendix B, Table B.20*).

2.4.3.4 Cumulative predation rates – For most, but not all, salmonid ESUs/DPSs there was no significant decrease in the effects of all Caspian tern colonies in the CPR combined (i.e., the cumulative effects of predation) on salmonid ESUs/DPSs following implementation of the *IAPMP*. The two notable exceptions were for UCR steelhead and UCR spring Chinook, where the average annual cumulative effects of predation were reduced from 20.0% (17.3–23.0%) in UCR steelhead and 3.7% (2.7–4.8%) in UCR spring Chinook prior to management to 7.6% (6.5–9.2%) and 1.1% (0.7–1.7%) following management, respectively. Decreases in the cumulative effects of Caspian tern predation on UCR steelhead and UCR spring Chinook were directly related to the management of Caspian tern colonies in Potholes Reservoir, where terns disproportionately consumed UCR salmonids compared with SR salmonids and where tern predation impacts were eliminated or nearly starting in 2016 (*Table 2.4* above).

The finding that there was no significant change in the cumulative effects of Caspian tern predation prior to and following implementation of management actions should be interpreted cautiously because cumulative estimates of predation were based on smolt availability in the tailrace of Rock Island Dam (for UCR smolts) or Lower Monumental Dam (for SR smolts) and thus do not fully account for smolt survival to within the forage range of all tern colonies in CPR. For instances, Caspian terns breeding on the Blalock Island predominately forage in the John Day Reservoir (see section *2.4.3.1 Foraging behavior* above), which is a considerable distance downstream from Rock Island and Lower Monumental dams. As such, cumulative estimates of predation rates reported herein may underestimate the full impact of predation at some colonies. A more detailed analysis of the cumulative effects of Caspian tern predation on ESA-listed salmonid ESUs/DPSs is presented in *Chapter 7.*

Table 2.6. Average annual predation rates (95% credible intervals) by all Caspian tern colonies in the Columbia Plateau region combined during 2008-2018. ESA-listed salmonid populations (ESUs/DPSs) from the Snake River (SR) and Upper Columbia River (UCR) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. Cumulative estimates of predation were available during a six-year period prior to management (2008-2013) and a five-year (2014-2018) period during management.

ESU/DPS	Pre-Management 2008–2013	Management 2014–2018
SR Sockeye	2.2% (1.6–2.8)	3.0% (1.9–4.3)
SR Sp/Su Chinook	1.0% (0.9–1.2)	0.7% (0.6–1.0)
UCR Spr Chinook	3.7% (2.7–4.8)	1.1% (0.7–1.7)
SR Fall Chinook	1.7% (1.5–2.0)	1.5% (1.1–2.0)
SR Steelhead	5.0% (4.4–5.8)	3.6% (3.1–4.4)
UCR Steelhead	20.0% (17.3–23.0)	7.6% (6.5–9.2)

2.4.3.5 Upper Columbia River steelhead trout survival rates – Using steelhead PIT-tagged at Rock Island Dam during 2008–2019, there is evidence that survival rates of UCR steelhead smolts between Rock Island Dam and McNary Dam – the section of river where Caspian terns from the Goose Island and Crescent Island colonies forage – were significantly higher following reductions in the size of these two tern colonies during 2014–2019 (*Figure 2.21*). In 2019, Caspian tern predation rates on steelhead smolts upstream of McNary Dam were the lowest recorded since estimates of cumulative tern predation rates were first reported in 2008. Estimates of low tern predation rates in 2019 corresponded with the highest steelhead survival rates observed to date between Rock Island Dam and McNary Dam (see also Collis et al. 2020). Results indicate that Caspian tern predation rates on UCR steelhead have been significantly reduced and that steelhead smolt survival has significantly increased within the river reach where predation by Caspian terns from the Goose Island and Crescent Island colonies primarily occurred (see also Payton et al. 2020). An unintended consequence of management, however, was the dispersal of Caspian terns from the colonies on Goose and Crescent islands to the Blalock Islands in John Day Reservoir. Results of predation and survival analysis on UCR steelhead in this river reach indicated that there were significantly higher predation rates by Caspian terns and significantly lower rates of survival following the rapid increase in size of the Blalock Islands tern nesting colony during the management period (*Figure 2.21*; *Table 2.2* above for nesting colony sizes). More detailed analysis of the cumulative effects of Caspian tern predation on smolt survival during out-migration, including an investigation of additive effects of avian predation on smolt and smolt-to-adult survival, are presented in *Chapter 7* and *Chapter 8*, respectively.

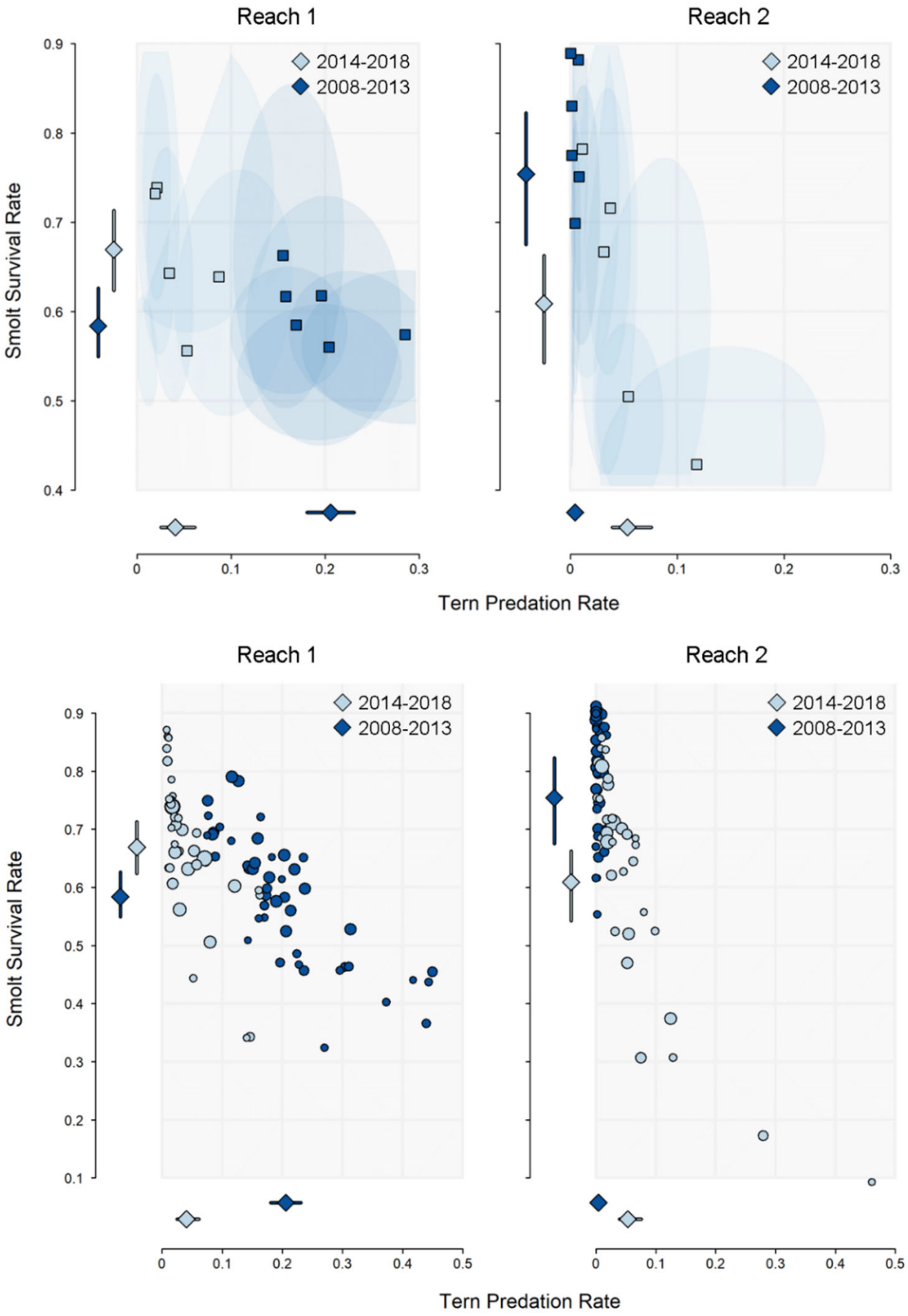


Figure 2.21. Caspian tern predation rates and survival rates of Upper Columbia River steelhead smolts during passage from Rock Island Dam to McNary Dam (Reach 1, left) and McNary Dam to Bonneville Dam (Reach 2, right) prior to (dark blue) and following (light blue) management actions that reduced the size of tern colonies at Goose Island and Crescent Island. Median annual rates (boxes, top row of graphs), weekly rates (circles, bottom row of graphs), and rates for the entire study period (diamonds) are shown. Error bars represent 95% credible intervals for annual averages, and shaded ellipses represent 95% credible regions for the joint estimation of survival rate and predation rate in each river reach and year (see also Evans et al. 2019c and Payton et al. 2020).

2.4.3.6 Per capita predation rates – Predation rates adjusted for differences in nesting colony size (per bird predation rates) were the highest on UCR steelhead by Caspian terns nesting on Goose Island or other islands in Potholes Reservoir, followed by terns nesting at Lenore Lake and on Twinning Island (Evans et al. 2012, Collis et al. 2016; see also *Table 2.7*). These are all colonies where terns forage predominately on anadromous juvenile salmonids migrating through the middle Columbia River, where UCR steelhead are particularly susceptible to predation by Caspian terns. Similarly, per capita predation rates were also the highest on SR steelhead from Caspian terns that nested on Crescent Island just below the confluence of the lower Snake River (Evans et al. 2012, Evans et al. 2016a; *Table 2.7*). Per capita predation rates on salmon ESUs were consistently lower than those on steelhead DPSs (*Table 2.7*), again demonstrating the greater susceptibility of steelhead in general to predation by Caspian terns (see also *Chapter 7*).

Per capita predation rates on salmonid smolts for Caspian tern colonies in the CPR were an order of magnitude greater than for tern colonies in the Columbia River estuary (see also *Chapter 6* and *Chapter 7).* The high per capita predation rates on steelhead DPSs by Caspian terns nesting at Goose Island and at Crescent Island was the primary reason why management actions associated with the *IAPMP* focused on reducing tern predation rates in the CPR (Lyons et al. 2011b, USACE 2014). For instance, relative to predation on smolts by California gulls and ring-billed gulls in the CPR, the highest per capita predation rates were those measured at Caspian tern colonies, especially predation rates on steelhead DPSs (see *Chapter 6*). It should be noted, however, that retrospectively analyzed predation rates – those corrected for both PIT tag detection and deposition probabilities – indicate that per capita predation rates by double-crested cormorants nesting at Foundation Island in McNary Reservoir were similar to or greater than those of Caspian terns nesting at Crescent Island, especially cormorant predation rates on salmon ESUs (see *Chapter 6*). Regardless, there is evidence from multiple colonies, both in the CPR and in the CRE, that the highest per capita avian predation rates on salmonid DPSs/ESUs in the Columbia Basin are Caspian terns preying on steelhead DPSs in the Columbia Plateau region.

Few studies have investigated the mathematical relationship between peak nesting colony size and predation rates on salmonid smolts for Caspian tern colonies in the CPR. In a study of predation on UCR steelhead by Caspian terns nesting on Goose Island, Evans et al. (2013) observed that annual predation rates on steelhead increased as the number of tern breeding pairs at the Goose Island colony increased; the highest predation rates were observed in years when the nesting colony size was greatest. Collis et al. (2016) observed a similar positive relationship between tern nesting colony size on Crescent Island and predation rates on SR steelhead, but also noted high inter- and intra-annual variation, variation that was not explained by differences in nesting colony size alone. A more detailed summary of studies investigating the influence of nesting colony size, smolt abundance, and other factors associated with intra- and inter-annual variation in Caspian tern predation rates is provided in *Chapter 9* and *Chapter 10*.

Table 2.7. Average annual per capita (per nesting pair) predation rates (95% credible intervals) by Caspian terns nesting at colonies in the Columbia Plateau region during 2007–2019. ESA-listed salmonid populations (ESUs/DPSs) from the Snake River (SR) and Upper Columbia River (UCR) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. NA denotes that estimates of predation rates were not available.

ESU/DPS	Goose Island 2007–2015	North Potholes island 2016	Crescent Island 2007–2014
SR Sockeye	0.0009% (0.0003–0.0019)	0% (0–0.0001)	0.0038% (0.0029–0.0049)
SR Sp/Su Chinook	0.0001% (0–0.0002)	0% (0–0.0001)	0.0021% (0.0018–0.0025)
UCR Sp Chinook	0.0076% (0.0048–0.0149)	0.0003% (0–0.0024)	0.0013% (0.0007–0.0020)
SR Fall Chinook	0.0003% (0.0001–0.0010)	0.0001% (0–0.0013)	0.0026% (0.0023–0.0030)
SR Steelhead	0.0002% (0.0002–0.0004)	0.0002% (0–0.0007)	0.0116% (0.0104–0.0131)
UCR Steelhead	0.0425% (0.0369–0.0497)	0.0289% (0.0205–0.0463)	0.0063% (0.0055–0.0072)

ESU/DPS	Twinning Island 2009–2015	Lenore Lake islands 2017–2019	Harper Island 2012	Badger Island 2017	Blalock Islands 2007–2019
SR Sockeye	0% (0–0.0001)	0.0031% (0.0001–0.0260)	0.0008% (0–0.0091)	NA	0.0066% (0.0039–0.0117)
SR Sp/Su Chinook	0.0002% (0–0.0008)	0.0002% (0–0.0024)	0.0001% (0–0.0061)	0.0004% (0.0001–0.0014)	0.0022% (0.0017–0.0043)
UCR Sp Chinook	0.0047% (0.0014–0.0122)	0.0017% (0.0004–0.0050)	0.0058% (0.0001–0.0878)	0.0006% (0–0.0054)	0.0018% (0.0012–0.0039)
SR Fall Chinook	0.0004% (0.0001–0.0014)	0.0006% (0–0.0057)	0.0003% (0–0.0218)	0.0010% (0.0002–0.0033)	0.0031% (0.0022–0.0059)
SR Steelhead	0.0003% (0–0.0011)	0.0002% (0–0.0030)	0.0053% (0.0009–0.0305)	0.0084% (0.0056–0.0193)	0.0105% (0.0081–0.0206)
UCR Steelhead	0.0127% (0.0094–0.0171)	0.0139% (0.0095–0.0209)	0.0007% (0–0.0149)	0.0107% (0.0058–0.0217)	0.0108% (0.0085–0.0163)

2.5 Discussion

2.5.1 Management Action Effectiveness

The *IAPMP* was implemented during 2014–2018 by the USACE and BOR, with ongoing implementation in 2019 by the Priest Rapids Coordinating Committee and the Grant County Public Utility District[2]. The goal of the management plan was to reduce predation on ESA-listed juvenile salmonids by Caspian terns nesting at colonies in the CPR, in particular the Crescent Island colony on the Columbia River and the Goose Island colony on Potholes Reservoir, the two largest Caspian tern colonies in the region (Adkins et al. 2014, BRNW 2014). The primary objective of the plan was to use various nest dissuasion techniques to prevent a Caspian tern nesting colony from forming at these two managed sites. Concomitant with management implemented as part of the *IAPMP*, the plan called for monitoring and evaluating the efficacy of management actions at both the colony- and system-level, including measuring changes in the distribution and size of Caspian tern colonies in the CPR and predation rates on ESA-listed juvenile salmonids originating from the Snake and Columbia rivers. Effectiveness monitoring was also directed at understanding the behavioral response of individual Caspian terns that formerly nested at either the Goose Island or the Crescent Island colony, including movements, dispersal, recruitment to new breeding colonies, and selection of foraging areas.

2.5.1.1 Nest dissuasion & colony size – Beginning in 2014 on Goose Island and 2015 on Crescent Island, the suitable Caspian tern nesting habitat was nearly eliminated at these two managed colony sites by using a variety of passive nest dissuasion materials (i.e. fencing, stakes, rope, flagging), vegetation plantings, and placement of woody debris (Roby et al. 2015, Collis et al. 2016, 2017, 2018, 2019, 2020) to cover all the open, bare-ground habitat that terns typically use for nesting (Quinn and Sirdevan 1998, Cuthbert and Wires 1999, Roby et al. 2003a). Once Caspian terns arrived to initiate nesting at these managed sites, active nest dissuasion (i.e. human hazing) and collection of any tern eggs laid (under permit) was carried out to prevent nesting colony formation. These efforts were successful in preventing terns from nesting on Goose Island during 2016–2019 and on Crescent Island during 2015–2019 (Roby et al. 2015, Collis et al. 2016, 2017, 2018, 2019, 2020). Prior to management, an average of 367 and 461 breeding pairs of Caspian terns nested on Goose Island and Crescent Island, respectively.

Despite the success in preventing Caspian tern nesting colony formation on Goose Island during 2015–2019, many terns that formerly nested on Goose Island continued to display high fidelity to Goose Island in particular, and the Potholes Reservoir area in general. This fidelity is likely due in part to Caspian terns having a long history of nesting in Potholes Reservoir (i.e. more than 50 years; Penland 1982) coupled with a relatively long life expectancy (some individuals live for more than 25 years; Cuthbert and Wires 1999, Suryan et al. 2004, Suzuki 2012). Furthermore, the

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[2] Implementation of facets of the *IAPMP* are continuing beyond 2019 but limited in comparison to the actions implemented during 2014-2019.

persistence of a large gull nesting colony on Goose Island throughout the management period continued to attract prospecting Caspian terns to the site; gulls are common nesting associates for Caspian terns (Cuthbert and Wires 1999). Finally, the paucity of suitable Caspian tern nesting colony sites in the vicinity of Potholes Reservoir is likely another factor that can explain the high fidelity of Caspian terns to the Goose Island nesting colony site. During the management period, seasonal attendance by Caspian terns at Goose Island was often greatest in July, while peak nesting colony attendance during the pre-management period was in early May (Collis et al. 2020). Although in some years during the pre-management period egg-laying continued into July, it is likely that the influx of Caspian terns to Goose Island in July during the management period involved either post-breeding birds or failed breeders from other colonies; therefore, some, if not most, of these birds were not visiting the site to breed.

Caspian tern use of Goose Island during the management period was largely limited to areas not covered in passive nest dissuasion, specifically shoreline habitat that became exposed as reservoir levels receded during the breeding season (Collis et al. 2017, 2018, 2019, 2020). Because of the high fidelity of terns to Goose Island and the Potholes Reservoir area, active nest dissuasion (hazing) and collection of any tern eggs laid were critical in preventing the formation of a Caspian tern nesting colony on Goose Island. Time spent actively hazing terns on Goose Island to prevent nesting colony formation was substantial (more than 6 hours/day) in the first several years of management implementation (Roby et al. 2015, Collis et al. 2016, 2017, 2018), but declined considerably to about 1 hour/day during the last two years of the study period. The large reduction in active dissuasion required to prevent tern nesting colony formation on Goose Island during 2018–2019 was due to the decline in tern activity at Goose Island during the typical peak in nesting (Collis et al. 2019, 2020). The documented number of tern eggs laid on Goose Island also declined during the management period, from 100s in 2014 to an average of 15 eggs/year during 2017–2019. Except for 2014, when nesting colony formation by terns on Goose Island could not be prevented (see *above*), the vast majority (97%) of the tern eggs laid during 2015–2019 (96 eggs) were either collected (55 eggs) or depredated by gulls immediately after they were laid (38 eggs; Roby et al. 2015, Collis et al. 2016, 2017, 2018, 2019, 2020).

Aside from the first year of management (2015) when both passive and active measures were important in preventing tern nesting, passive nest dissuasion techniques alone were successful in preventing all nesting and roosting by Caspian terns on Crescent Island (Collis et al. 2016, 2017, 2018, 2019, 2020). The complete abandonment of Crescent Island by nesting terns beginning in the first year of management was unexpected because Caspian terns had nested consistently on Crescent Island for nearly three decades prior to management (Adkins et al. 2014). One factor that likely contributed to the abandonment of Crescent Island as a nesting site by Caspian terns was the absence of nesting gulls on Crescent Island during the management period; gulls are breeding associates of Caspian terns and attract prospecting Caspian terns to nest near their colonies (Cuthbert and Wires 1999). Some of the passive nest dissuasion methods used on Crescent Island were different than those used on Goose Island, namely vertical fence rows, revegetation, and placement of woody debris, which were not only effective in dissuading nesting terns, but gulls as well. At Goose Island, gull nesting could not be prevented using the available passive and active nest dissuasion techniques; gulls were not dissuaded from nesting using only stakes, rope, and flagging and nested beneath the passive nest dissuasion array on Goose Island. Gulls on Goose Island, particularly ring-billed gulls, quickly became habituated to several active nest dissuasion methods conducted on the island prior to egg-laying by gulls (Roby et al. 2015; Collis et al. 2016, 2017, 2018, 2019, 2020). Once gull eggs were laid on Goose Island, nest dissuasion activities had to be modified to avoid causing nest failure by disturbing gulls; causing gull nest failure was prohibited under the depredation permit that was issued by the USFWS to collect tern eggs on the island during 2014–2019 (USACE 2014). This was certainly another factor that allowed the gull nesting colony on Goose Island to persist during the management period. Finally, alternative nesting habitat for gulls was readily available near Crescent Island, which was not the case at Goose Island. In the first year of management at Crescent Island, many, if not most, of the gulls that nested on Crescent Island moved to nest on Badger Island, which is located on the Columbia River just one km upriver from Crescent Island. Similarly, most of the Caspian terns displaced from Crescent Island relocated to nest at the small extant tern colony on the Blalock Islands downstream from Crescent Island, as indicated by resightings of Caspian terns previously banded on Crescent Island and tracking of Caspian terns’ satellite-tagged on Crescent Island in 2015 (Roby et al. 2016).

As part of system-level monitoring for the *IAPMP*, surveys were conducted in the CPR and outside to determine where Caspian terns displaced from the managed colonies on Goose and Crescent islands might attempt to re-nest. The desired outcome of the *IAPM*P was that terns displaced from the managed colonies in the CPR would relocate to nest outside the Columbia Basin, as opposed to renesting at new or existing colony sites within the basin. As with the *Caspian Tern Management Plan in the Columbia River Estuary* (USFWS 2005; also see *Chapter 3*), alternative nesting habitat was created for Caspian terns outside the basin, and in the case of the *IAPMP*, this included modifying a small island at Don Edwards National Wildlife Refuge (NWR) in San Francisco Bay in order to make them suitable for Caspian tern nesting (Hartman et al. 2019); Don Edwards NWR formerly supported several Caspian tern colonies whose impacts to fish of conservation concern were low (Strong et al. 2004, Collis et al. 2012b, Adrean et al. 2012). Management and monitoring of the islands enhanced for Caspian tern nesting at Don Edwards NWR during the management period were conducted by the U.S. Geological Survey – Dixon Field Station (see Hartman et al. 2016, 2017, 2018, 2019 for more details). For details on Caspian tern use of the USACE-constructed islands outside the Columbia River basin as part of Caspian tern management plans in the Columbia Basin (USFWS 2005, USACE 2014), see *Chapter 3*.

Management of Caspian terns in the CPR was successful at eliminating the tern colonies on Crescent and Goose islands, and thereby eliminating or greatly reducing predation on ESA-listed smolts from terns that nested at those two sites (see *below*). Associated with the elimination of the Crescent Island and Goose Island tern colonies, there was a 49% decline in the regional breeding population of Caspian terns, from an average of 873 breeding pairs during the pre-management period to 445 breeding pairs in 2019 (Collis et al. 2020). This suggests that roughly half of the terns that formerly nested on Goose and Crescent islands had relocated to nest outside the CPR by 2019, a hypothesis supported by resightings of Caspian terns outside the CPR that had been banded at the two colonies, as well as the satellite-tracking of terns from the two colonies that was initiated in 2014 (BRNW 2015, Roby et al. 2016, 2017, 2018, 2019b, Schniedermeyer

2018; see also *Chapter 3*). There were two new Caspian tern colonies that became established, at least briefly, during the management period, one on a small island in northern Potholes Reservoir during 2016 (which was subsequently managed and did not persist; see section *2.5.2 Adaptive Management*) and two nesting colony sites on small islands (North Rock and Shoal Island) in Lenore Lake, where impacts to juvenile salmonids were relatively low (see *below*). The only other Caspian tern colonies that were active in the CPR at the end of this study (2019) were on Harper Island in Sprague Lake (18 breeding pairs) and on two small gravel bars (Long and Middle islands) in the Blalock Islands complex on the Columbia River (379 breeding pairs). As was the case during 2015–2018, the largest Caspian tern nesting colony in the CPR during 2019 was in the Blalock Islands, where 87% of all the Caspian terns in the region nested. Compared to the average size of the Caspian tern nesting colony on the Blalock Islands prior to management (59 breeding pairs during 2005–2013), the colony was more than 6 times larger at the end of the management period (2019; Collis et al. 2016, 2017, 2018, 2019, 2020). Band resighting and satellite telemetry data suggest that the growth of the Caspian tern nesting colony at the Blalock Islands was largely due to immigration by Caspian terns that formerly nested at Crescent Island (Roby et al. 2016, 2017, 2018, 2019b, Schniedermeyer 2018). The impact on smolt survival of predation by Caspian terns nesting at the Blalock Islands continues to be high (see *below*). Analysis of data on banded terns that nested at either Goose Island or Crescent Island immediately prior to management found that the proportion of terns that returned to breed during the early years of management declined significantly; however, a significantly greater proportion of terns returned to the CPR as non-breeding birds compared to the pre-management period (Schniedermeyer 2018). Management to eliminate the tern colonies at Goose and Crescent islands reduced the number of Caspian terns nesting in the CPR, but not to the target population size of 200 breeding pairs designated in the *IAPMP* (USACE 2014) due to the growth of other extant and new colonies in the region (i.e. Blalock Islands, Lenore Lake, Sprague Lake). Furthermore, the return of many Caspian terns that formerly nested on Goose or Crescent islands to the CPR as non-breeders (Roby et al. 2019a, Schniedermeyer et al. 2020) further diminished the benefits to smolt survival associated with management under the *IAPMP* (see *below*).

The network analyses conducted on the movements of Caspian terns' satellite-tagged at Goose and Crescent islands (Roby et al. 2019b) inform the management efforts seeking to reduce Caspian tern predation on juvenile salmonids in the Columbia River basin listed as threatened or endangered under the ESA. In particular, maintaining active colonies outside the Columbia Basin that are particularly well linked to colonies in the CPR, such as the waterfront at Everett, Washington, islands in Arcata Bay, California, and islands in the Salton Sea, California, would likely facilitate greater dispersal away from the CPR. We conclude that the use of network analytical measures is a useful way to understand the importance of maintaining habitat networks and prioritizing locations (colonies) within the network for specific conservation actions. Assessing network structure using measures such as neighborhood size, betweenness, and distance centrality is a good method for identifying the appropriate scale of conservation – whether regional clusters of suitable habitats are a useful unit for conservation or if actions taken at range-wide scales are necessary.

2.5.1.2 Predation impacts on juvenile salmonids – The goal of the *IAPMP* was to reduce predation rates on ESA-listed juvenile salmonids by Caspian terns in the CPR to less than 2% per salmonid ESU/DPS, per nesting colony, per year. Estimates of Caspian tern predation rates during the management period indicated that the goal of achieving predation rates of less than 2% were met for most, but not all, tern colonies and ESA-listed salmonid ESUs/DPSs in the CPR. Since 2015, predation rates were zero or close to zero for terns nesting on Goose Island and Crescent Island due to the complete (Crescent Island) or nearly complete (Goose Island and other islands in Potholes Reservoir) elimination of nesting at these two colony sites.

Predation rates at the new, unmanaged tern nesting colony in Lenore Lake (North Rock) were also less than 2% per ESU/DPS, with the highest predation rate of 1.0% (0.6–1.7%) on Upper Columbia River steelhead. Low Caspian tern predation rates on juvenile salmonids at the Lenore Lake nesting colony is likely due to the small size of the colony and its distance from the middle Columbia River. Predation rates on juvenile salmonids at the large, unmanaged tern nesting colony on the Blalock Islands in John Day Reservoir on the Columbia River, however, exceeded the 2% threshold for two ESA-listed ESUs/DPSs; average annual predation rates during 2017–2019 (the last 3 years of the study) were 3.1% (2.4–4.1%) and 4.5% (3.4–6.1%) for Upper Columbia River steelhead and Snake River steelhead, respectively. Estimates of predation rates on juvenile salmonids were not available for Caspian terns nesting on Harper Island in Sprague Lake because access to the privately-owned island to recover smolt PIT tags was not granted; however, just 18 pairs of Caspian terns nested on Harper Island in 2019. Based on limited data collected in 2012, Caspian terns nesting on Harper Island can commute to the lower Snake River and foraging on juvenile salmonids, but predation rates were low (< 0.3% for all ESUs/DPSs), presumably due to the small size of the nesting colony and its distance from the lower Snake River (> 60 km).

In summary, comparisons of Caspian tern predation rates on juvenile salmonids prior to and during implementation of the *IAPMP* indicate there have been benefits to the Upper Columbia River steelhead DPS (see also *Chapter 7*), with predation by Caspian tern nesting on Goose Island and Crescent Island eliminated due to implementation of the *IAPMP*. There is also evidence that smolt survival for Upper Columbia River steelhead has increased in the river reach where terns that nested on Goose and Crescent islands foraged (Rock Island Dam to McNary Dam). Due to continued high rates of smolt predation by Caspian terns nesting in the Blalock Islands, however, impacts to some ESA-listed ESUs/DPSs, particularly those originating from the Snake River, remain as high or higher than those observed prior to implementation of management actions under the *IAPMP*.

2.5.2 Adaptive Management

Adaptive management was a key component of the *IAPMP* in order to maximize the likelihood of achieving the goal of the plan, both at the two managed Caspian tern colonies and system level (i.e. in the CPR; USACE 2104). At the managed colonies, a variety of methods had been identified for dissuading Caspian terns from nesting at Goose and Crescent islands (USACE 2014), and were meant to be used in different combinations, depending on the response of birds to each method. Adjustments were made not only in the combination of nest dissuasion methods used, but in the timing and duration of these methods. Adaptive changes were made in management approach both in-season and between years during the management period to maximize success in preventing formation of a Caspian tern nesting colony at either island, while at the same time complying with the conditions of permits issued by the USFWS to implement management (USACE 2014). "Best Management Practices" were developed and followed at each managed nesting colony to achieve success while complying with permits, and these practices were modified as necessary (these practices were provided as an appendix in annual reports; see Roby et al. 2015, Collis et al. 2016, 2017, 2018, 2019 for further details). At a system-level, adaptive management was deemed necessary to limit the formation of new Caspian tern colonies and the growth of existing tern colonies within the CPR during the management period (USACE 2014). If Caspian terns that were dissuaded from nesting at Goose Island or Crescent Island were to relocate to new or existing colonies within the region, and if the impacts of tern predation on survival of ESA-listed juvenile salmonids at these new or expanded colonies were deemed to be significant (i.e. > 2% per salmonid ESU/DPS, per nesting colony, per year), adaptive management actions could be implemented to mitigate for these unintended and unwanted impacts (USACE 2014). Outlined below are descriptions of the adaptive management that was practiced at both managed colonies and at a system level (i.e. in the CPR).

2.5.2.1 Managed colonies – During the first year of management at Goose Island (2014), the small rocky islets surrounding Goose Island were not covered in passive nest dissuasion materials because it was thought that the rocky islets were unsuitable for tern nesting (Roby et al. 2015). The combination of nest dissuasion methods used on Goose Island during that year were successful in preventing nesting colony formation on the main island itself, causing the dissuaded terns to seek nest sites elsewhere. Some of these terns were highly motivated to nest in the area, and a small satellite colony of Caspian terns formed on a rocky islet adjacent to Goose Island where no Caspian tern nesting activity had previously been recorded. This islet, dubbed "Northwest Rocks," eventually attracted as many as 156 breeding pairs of Caspian terns, which nested amongst earlier-nesting gulls (Roby et al. 2015). It was determined that placement of passive nest dissuasion materials and active hazing at Northwest Rocks during the 2014 breeding season was not authorized because it was unknown whether the rocky islet was part of Goose Island and whether management on the islet was covered under the environmental analysis completed as part of the *IAPMP* (USACE 2014). This issue was sorted out prior to the 2015 breeding season and passive nest dissuasion and active hazing were implemented at Northwest Rocks, and in other similar habitat on Goose Island and surrounding islets, during 2015 (Collis et al. 2016). These management actions were continued in subsequent years of the *IAPMP* and were successful in preventing the formation of a Caspian tern nesting colony on Goose Island and all nearby rocky islets during 2015–2019.

Based on the success of vegetation plantings on Crescent Island in preventing Caspian terns from nesting without the need for intensive in-season hazing, feasibility studies in 2018 and 2019 on Goose Island were conducted to try to establish native plants from seed and plugs, respectively. Efforts in both years included irrigation of test plots, but neither effort was successful in establishing vegetative cover on Goose Island (Collis et al. 2019, 2020). It is believed that the substrate on Goose Island, primarily rock with little topsoil, is unsuitable for widespread revegetation, as was successfully implemented on Crescent Island.

Caspian terns exhibited some habituation to the active dissuasion methods utilized to prevent tern nesting on Goose Island, requiring constant shifts in the methods used and the intensity of those efforts to successfully dissuade terns from nesting on the island (Roby et al. 2015, Collis et al. 2016, 2017, 2018, 2019). This habituation occurred both within and across breeding seasons, suggesting that active methods to haze terns off Goose Island will become increasingly ineffective each successive year of management. The effort currently required to prevent Caspian terns from nesting on Goose Island includes stationing a field crew on the island to conduct regular hazing throughout the breeding season (April–July). Given this level of effort, other management options aimed at preventing not only Caspian terns, but also gulls, from nesting on Goose Island may be of interest to resource managers (see *above*). As mentioned above, gulls are a common nesting associate of Caspian terns and provide a strong attraction to Caspian terns prospecting for nests sites.

2.5.2.2 System level – Changes in pool elevation in Potholes Reservoir can create or eliminate habitat that is suitable for Caspian tern nesting (Collis et al. 2017), as was the case in 2016 when a small, low-lying island in northeastern Potholes Reservoir was exposed during receding water levels and colonized by nesting Caspian terns (see *above*). This incipient Caspian tern nesting colony had grown to 144 breeding pairs, most of which were pairs dissuaded from nesting on Goose Island, before it failed due to predator disturbance (Roby et al. 2017). Both egg-laying and hatching of tern chicks were confirmed at the site prior to the nesting colony being abandoned in early June, apparently due to the presence of American mink on the small island. This was the first documented nesting by terns at the northern end of Potholes Reservoir since 2004, when a small nesting colony existed on Solstice Island (Adkins et al. 2014). Beginning with the 2017 breeding season, and continuing for the next two years, adaptive management was conducted at this incipient nesting colony site, as well as at other small, low-lying islands in northern Potholes Reservoir that were deemed suitable for tern nesting, to prevent nesting colony formation in northern Potholes Reservoir. These adaptive management actions were successful in preventing terns from reforming a nesting colony in northern Pothole Reservoir during 2017–2019 (Collis et al. 2018, 2019, 2020).

2.5.3 Management Considerations

Based on data collected during the management period (2014–2019), the *IAPMP* objective of preventing Caspian terns from nesting on Goose and Crescent islands, thereby reducing predation rates on ESA-listed salmonid ESUs/DPSs by terns nesting at these two sites to less than 2%, has been achieved. Despite this, there remain several uncertainties associated with the efficacy and long-term sustainability of these management actions in reducing the predation impacts from all Caspian tern colonies in the CPR to levels prescribed by the *IAPMP* (USACE 2014). The following is a list of management considerations with the aim of meeting these management objectives, as well as to identify any new and emergent avian predation issues worthy of management consideration both in the short- and long-term.

1. Caspian terns that formerly nested on Goose and Crescent islands continue to show high fidelity to the CPR for nesting. Many terns displaced from the Goose Island nesting colony are still attempting to nest at Goose Island and elsewhere in Potholes Reservoir, while many terns displaced from Crescent Island have relocated to nest at other unmanaged tern colonies in the region, most notably at the Blalock Islands on the Columbia River. A system-wide, adaptive management approach is needed to prevent Caspian terns that formerly nested on Goose and Crescent islands from nesting at other colonies in the region where they could continue to significantly impact survival rates of ESA-listed salmonid smolts.

2. As demonstrated at Crescent Island, revegetation is an effective and sustainable method of passive nest dissuasion for Caspian terns that can prevent nesting at that colony site without the need for perennial and persistent hazing and other methods of passive and active nest dissuasion. Stakes, rope, and flagging as a passive dissuasion technique, combined with prolonged and intensive hazing and other active dissuasion of terns prospecting for nest sites, were effective at preventing Caspian terns from nesting at Goose Island, but did not deter gulls from nesting beneath nest dissuasion arrays. It is likely that the greater and more persistent fidelity shown by Caspian terns to the Goose Island nesting colony site compared to Crescent Island was due to the continued presence of a large gull nesting colony on Goose Island. Nesting gulls are a strong social attractant for Caspian terns prospecting for nesting opportunities, because they nest communally with gulls at most Caspian tern colony sites in the Pacific Flyway. Efforts to prevent nesting effectively and sustainably by Caspian terns on Goose Island would be greatly enhanced by the elimination of the gull nesting colony at that site. It should be noted that this consideration does not consider an unintended consequence of management of this gull colony, that is if these gulls were to relocate to nest at sites along the Columbia or Snake river where their impacts to juvenile salmonid survival may be higher.

3. Feasibility studies to use vegetation as the passive nest dissuasion method on Goose Island have been unsuccessful. During 2018–2019, attempts to establish vegetation from seeds or plugs in irrigated test plots were unsuccessful. It is believed that the substrate, primarily rock with little topsoil, is unsuitable for widespread revegetation of Goose Island. Other passive nest dissuasion methods that are more sustainable compared to ropes, stakes, and flagging (e.g., placement of large woody debris, planting of mature drought-resistant vegetation) may have promise for preventing Caspian terns from nesting on Goose Island but have yet to be tested there.

4. Little potential Caspian tern nesting habitat exists along the shoreline of Goose Island when Potholes Reservoir nears peak elevation (ca. 1044’), which typically occurs in early May. In recent years, availability of tern nesting habitat during the early part of the nesting season has been limited by high water and passive dissuasion until water levels recede and expose open habitat along the shoreline that is not covered in passive nest dissuasion materials. Additionally, few potential breeding locations remain exposed in northern Potholes Reservoir at peak water levels. Based on these observations, maintaining Potholes

Reservoir levels at or near peak elevation until early June, when most juvenile salmonids have out-migrated through the mid-Columbia River, would likely reduce the number of Caspian terns using Potholes Reservoir during the peak of the smolt outmigration.

5. At Caspian tern nesting colony sites where passive nest dissuasion is less effective or where options for deploying permanent passive dissuasion are limited (e.g., Goose Island in Potholes Reservoir), some level of active hazing will likely be required each year to prevent terns from nesting. Active hazing aimed at preventing Caspian terns from nesting early in the breeding season, when availability of juvenile salmonids is greatest (April–May), will be necessary to prevent nesting colony formation and minimize predation impacts on ESA-listed populations.

6. Although predation on juvenile salmonids by Caspian terns nesting at the two managed nesting colonies in the Columbia Plateau region has been eliminated, predation rates by Caspian terns nesting at the large, unmanaged colony in the Blalock Islands complex continue to exceed the 2% threshold for several ESA-listed ESUs/DPSs. Average annual predation rates on Upper Columbia River steelhead, Snake River steelhead, and Snake River sockeye by Caspian terns nesting in the Blalock Islands are similar to or greater than those by Caspian terns nesting at the former colony on Crescent Island prior to management. Adaptive management to reduce the size of the Caspian tern nesting colony in the Blalock Islands is needed to reduce Caspian tern predation rates to levels specified in the *IAPMP*.

7. As demonstrated at the incipient nesting colony site in northern Potholes Reservoir, adaptive management using passive and active nest dissuasion techniques at tern nesting sites can quickly (in just one year) eliminate impacts from terns nesting at colony sites in the CPR. As such, adaptive management at the Blalock Islands nesting colony site could benefit ESA-listed populations originating from both UCR and SR ESUs/DPSs, but the greatest net benefit would be for SR populations, populations that have yet to receive the full benefit from Caspian tern management actions in the CPR.

8. The islands that Caspian terns use for nesting at the Blalock Islands complex are low-lying gravel bars that are prone to flooding when water levels in John Day Reservoir are relatively high (≥ 264.5’; Collis et al. 2019). If managers decide that preventing Caspian terns from nesting at this site is warranted, raising the pool level in John Day Reservoir throughout the tern breeding season (April–July) or at intervals during the breeding season (e.g., once, or twice a month) would cause nesting colony failure and likely dissuade Caspian terns from nesting in the Blalock Islands.

9. Based on PIT tag recoveries in only one year of the study, there was no evidence that a small Caspian tern nesting colony in Sprague Lake on Harper Island poses a substantial risk to smolt survival in the lower Snake River. Future monitoring of the Harper Island tern nesting colony may be warranted, particularly if a substantial increase in colony size occurs and permission to scan for PIT tags can be obtained.

10. We recommend continued monitoring of unmanaged Caspian tern colonies in the Columbia Plateau region in order to identify those tern colonies where predation rates on ESA-listed populations of juvenile salmonids are high (i.e. above the 2% threshold established by the *IAPMP*) and to help identify nesting colony sites in the region where predation impacts from Caspian terns are minimal and where management might be implemented to encourage tern nesting.

11. Caspian terns nesting at the former breeding colonies on Goose and Crescent islands exhibited higher than expected philopatry to the Columbia Plateau region, apparently because few nesting opportunities were available outside the region. The results indicate that in order to achieve the *IAPMP* objective of no more than 200 breeding pairs of Caspian terns nesting throughout the Columbia Plateau region, (1) more than five years are required for terns that formerly nested at the Crescent Island and Goose Island colonies to identify and disperse to alternative breeding colonies outside the region, (2) additional quality nesting habitat for terns may need to be provided outside the Columbia Plateau region, especially at or near former colony sites along the Washington coast, and (3) adaptive management is needed to dissuade Caspian terns from nesting at colony sites in the Columbia Plateau region where more than 40 breeding pairs currently nest and predation rates on ESA-listed populations of juvenile salmonids are above the 2% threshold stipulated in the *IAPMP*, in particular at the Caspian tern nesting colony in the Blalock Islands in John Day Reservoir.

12. Maintaining active colonies outside the Columbia River basin with high connectivity to colonies in the Columbia Plateau region, such as former and prospective colony sites along the Washington coast (e.g., Grays Harbor, Strait of Juan de Fuca, and Puget Sound), would likely facilitate greater breeding dispersal away from the Columbia Plateau region and support efforts to reduce Caspian tern predation on juvenile salmonids from the Columbia and Snake rivers.

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CHAPTER 3: Caspian Tern Management at Alternative Colony Sites Outside the Columbia River Basin

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3.1 Summary

To reduce the consumption of juvenile salmonids (*Oncorhynchus* spp.) by Caspian terns (*Hydroprogne caspia*) nesting in the Columbia River estuary, management actions during 1999–2001 were successful at shifting the large Caspian tern colony on Rice Island to restored habitat on East Sand Island, 21 km closer to the mouth of the Columbia River. Beginning in 2008, additional management actions were implemented to reduce losses of juvenile salmonids to Caspian tern predation by reducing the area of nesting habitat on East Sand Island, and thereby the size of the colony, by about 60%, while ensuring the long-term conservation of the Pacific Flyway population of Caspian terns. The Records of Decision (RODs) for the *Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (hereafter "*Estuary Caspian Tern Management Plan"*) stipulated that reduction of tern nesting habitat at East Sand Island and creation of new alternative tern nesting habitat outside the Columbia River estuary would be accomplished gradually and at a ratio of twice the area of new habitat created to the area of habitat eliminated on East Sand Island. In 2014, further management actions to reduce predation on ESA-listed salmonid smolts were implemented in the Columbia Plateau region as part of the *Inland Avian Predation Management Plan* (hereafter "*IAPMP*"). The *IAPMP* included efforts to prevent Caspian terns from nesting at Goose Island and Crescent Island, the two largest breeding colonies for the species in the Columbia Plateau region, with the goal of reducing Caspian tern predation rates on each ESA-listed evolutionarily significant unit (ESU) or distinct population segment (DPS) of salmonid in the Plateau region to less than 2% of available smolts per tern colony, per year or to less than 5% for all tern colonies combined, per year. Acknowledging that Caspian tern nesting habitat was available at other sites in the Columbia Plateau region, the *IAPMP* made concessions to allow up to 40 breeding pairs to nest at each colony site in the region and up to 200 breeding pairs to nest at all colonies in the Plateau region combined. Both the *Estuary Caspian Tern Management Plan* and the *IAPMP* were intended to redistribute the displaced Caspian tern breeding pairs to colonies outside of the Columbia River basin.

This chapter summarizes the results of monitoring efforts at 14 islands constructed or enhanced by the U.S. Army Corps of Engineers (USACE) as alternative nesting habitat for Caspian terns in compensation for the managed reduction of tern nesting habitat in the Columbia River estuary and the Columbia Plateau region as part of the *Estuary Caspian Tern Management Plan* and the *IAPMP*. Here we report the outcome of efforts to attract Caspian terns to nest on eight USACE-constructed islands in the southern Oregon and northeastern California (SONEC) region (i.e. Crump Lake, Dutchy Lake, East Link Impoundment, Gold Dike Impoundment, Sheepy Lake, Orems Unit, Tule Lake Sump 1B, and Malheur Lake) and one USACE-constructed island in the Willamette Valley of Oregon (i.e. Fern Ridge Reservoir) during 2008–2015; limited monitoring of these islands also occurred during 2016–2018. We also summarize the results of efforts to attract Caspian terns to nest at five small islands in Don Edwards San Francisco Bay National Wildlife Refuge (i.e. 3 islands in Pond SF2 and 2 islands in Pond A16) that were modified by the USACE prior to the 2015 nesting season and monitored during the 2015–2017 nesting seasons. During 2008–2015, we collected data on tern colony size (number of breeding pairs), breeding chronology, nesting success (average number of young raised/breeding pair), diet, factors limiting colony size and

nesting success, and efforts to control predators of terns and their nests at new colony sites in the SONEC region. We also assessed the effects of these new colony sites on the Pacific Flyway population of Caspian terns, as nesting opportunities for terns were reduced in the Columbia River estuary and in the Columbia Plateau region. During the 2016–2018 breeding seasons, we conducted aerial surveys of USACE-constructed islands and used aerial photography to estimate tern colony sizes.

The USACE completed construction of the first artificial islands as alternative nesting habitat for Caspian terns in 2008 as part of the *Estuary Caspian Tern Management Plan* to relocate Caspian terns nesting in the Columbia River estuary and thereby reduce their impacts on survival of ESA-listed salmonid smolts. By the start of the 2015 nesting season, the USACE had physically built or enhanced a combined total of 10.25 acres (4.15 ha) of island nesting habitat in the SONEC region, the Willamette Valley of Oregon, and the San Francisco Bay Area as potential nesting habitat for about 6,000 breeding pairs of Caspian terns, the total number of tern pairs that were displaced by implementation of both the *Estuary Caspian Tern Management Plan* and the *IAPMP*. Out of this total area of constructed island nesting habitat, the USACE estimated that a total of 8.3 acres (3.36 ha) of "effective" Caspian tern nesting habitat had been permanently provided by the beginning of the 2018 nesting season. To maximize the likelihood of successfully establishing and maintaining Caspian tern nesting colonies on these newly created colony sites, we (1) used social attraction techniques (i.e. tern decoys and audio playback of tern vocalizations), (2) closely monitored tern colony sites to identify potential factors limiting colony formation and nesting success, (3) managed any tern predators that could cause failure of incipient colonies, and (4) managed vegetation encroachment in order to provide terns with suitable bare-ground nesting habitat at each site.

Caspian terns initiated nesting on 10 of the 11 islands or island groups (91%) during the first nesting season after each island was constructed by the USACE, surrounded by water, and social attraction was deployed. The only USACE-constructed island that was not colonized by nesting Caspian terns during the first nesting season of availability was the one island that was not situated in a region with a history of Caspian tern nesting, the tern island built on Fern Ridge Reservoir in the Willamette Valley of Oregon. In the SONEC region, where eight separate islands were built in areas with a history of Caspian tern nesting, the number of breeding pairs using each island varied substantially among islands and among years (range = 0–690 breeding pairs). Subsequently, inter-annual variability in the total number of Caspian tern breeding pairs nesting on all eight USACE-constructed islands in the SONEC region during 2012–2018 was also high (range = 258–1,441 breeding pairs). On average, however, the breeding population of Caspian terns in the SONEC region during 2012–2018 was 870 breeding pairs (n = 7 years), nearly three times that of the decade preceding construction of the new tern islands (ca. 330 breeding pairs). Following island construction, the majority of the regional breeding population was concentrated at four of the eight USACE-constructed islands in the SONEC region, each of which supported tern colonies in excess of 500 breeding pairs in at least one year; the other four islands did not support colonies in excess of 70 breeding pairs.

Caspian tern nesting success (i.e. average number of young raised/breeding pair) at the USACE-constructed islands in the SONEC region also varied greatly among colony sites and among years (range = 0.00–1.00 young raised/breeding pair). Five of the eight islands supported nesting success in excess of 0.50 young/breeding pair during at least one nesting season; this level of nesting success is considered moderately good and capable of supporting a stable population of Caspian terns. On two of the eight islands, however, no young terns were raised to fledging during the study period.

Poor colony attendance and/or low nesting success at the USACE-constructed islands in the SONEC region during some years were caused by a variety of factors, usually associated with periodic drought cycles in the SONEC region, including (1) loss of open water foraging habitat within foraging distance of tern islands; (2) low availability of forage fish within foraging distance of tern islands; (3) low water levels that resulted in land-bridging of island colony sites, allowing access for terrestrial predators; and (4) intense predation pressure on tern nests by gulls nesting on the tern islands. At certain islands, however, factors seemingly un-related to periodic drought cycles appeared to limit colony size and nesting success. For example, at the Sheepy Lake tern island in Lower Klamath National Wildlife Refuge competition for nesting space with a variety of other colonial waterbirds nesting on the island was a major limiting factor. And at the Tule Lake Sump 1B tern island in Tule Lake National Wildlife Refuge nocturnal disturbance by great horned owls (*Bubo virginianus*) and raccoons (*Procyon lotor*) was a chronic limiting factor for colony size and nesting success. Because of climatic factors, nest site competition, and nest predation, complete nesting failures or low nesting success were not uncommon at most colony sites in the SONEC region. Further, we found that active management to remove encroaching vegetation and control predators on USACE-constructed tern islands was important for maintaining Caspian tern colonies on most islands in the SONEC region.

Resightings of Caspian terns marked with field-readable leg-bands indicated that adult Caspian terns banded at other colonies in the Pacific Flyway quickly located the new tern islands constructed by the USACE in the SONEC region, and these sites were soon integrated into the wider network of potential colony sites used by the breeding population of Caspian terns in the Pacific Flyway. Resightings of banded Caspian terns on the USACE-constructed islands in the SONEC region that were banded at colonies in the Columbia River basin indicated that most terns that initially recruited to these islands were immigrants from the Columbia River estuary, ca. 500 km away. Later in the study period, and coinciding with implementation of the *IAPMP*, more immigrants to the USACE-constructed islands in the SONEC region were from the Columbia Plateau region than from the Columbia River estuary. Conversely, the terns that recruited to the islands constructed at Don Edwards National Wildlife Refuge in south San Francisco Bay were mostly immigrants from other colonies in the Bay Area and not terns displaced from managed colonies in the Columbia River basin, as evidenced by few resightings of banded Caspian terns from outside the San Francisco Bay Area.

The diet of Caspian terns nesting on USACE-constructed islands in the SONEC region consisted mainly of native and introduced minnow species (Cyprinidae) and there were minimal predation impacts to ESA-listed species (i.e. suckers in the family Catostomidae) or other native fishes of conservation concern (i.e. salmon and trout in the family Salmonidae or lamprey in the family Petromyzontidae). Consequently, management concerns over Caspian tern consumption of threatened and endangered fishes in the Columbia River basin were not simply transferred to the SONEC region by the effort to restore Caspian tern nesting colonies and enhance the breeding population of Caspian terns in the Upper Klamath, Summer Lake, Harney, and Warner basins.

The efficacy of the parts of the *Estuary Caspian Tern Management Plan* and the *IAPMP* that sought to provide alternative nesting habitat for Caspian terns outside the Columbia River basin relied on attracting Caspian terns to nest at the newly provided colony sites outside of the basin. Successfully restoring Caspian tern nesting sites required providing islands that were free of predators and retained open and sparsely vegetated nesting substrates. Islands that are periodically or frequently land-bridged, surrounded by only shallow water, or are overgrown with vegetation do not provide suitable nesting habitat for Caspian terns. In general, the techniques employed by the *Estuary Caspian Tern Management Plan* and the *IAPMP* proved effective at attracting moderately sized Caspian tern nesting colonies to new colony sites in the SONEC region and the San Francisco Bay Area. Our results, however, suggest that the nesting colony sites created as part of those plans are unlikely to ever support a large proportion of the total number of Caspian terns that were displaced from the managed colonies in the Columbia Basin, and reinforces the need for additional Caspian tern colony sites in coastal Washington, as was initially planned under the *Estuary Caspian Tern Management Plan*. Anthropogenic climate change and over-allocation of water resources in the SONEC region, which is naturally prone to drought cycles, was and is likely to remain a persistent challenge for maintaining large and productive nesting colonies of Caspian terns at the USACE-constructed islands. Sustaining the management of vegetation and predators on these islands, which was financed by the USACE during 2008–2016, will likely require the sustained support of the state and federal land management agencies that assume ownership of the tern islands, because the management plans to construct and maintain the islands as nesting habitat for Caspian terns have expired. Finally, there appears to have been little breeding dispersal from the Columbia River basin to the nesting habitat constructed in the San Francisco Bay Area to date; the cause for this limited breeding connectivity is unclear but is likely related to the distance (ca. 1,000 km) between the managed colonies in the basin and the alternative colony sites in the Bay Area. Nevertheless, the newly constructed tern islands will help reverse declines in the local Caspian tern breeding population in the Bay Area and help conserve the overall Pacific Flyway population of the species.

3.2 Introduction

Caspian terns (*Hydroprogne caspia*) have a nearly cosmopolitan distribution throughout the globe, occurring on all continents except for Antarctica. In western North America at the start of the 20th century, much of the Pacific Flyway population of Caspian terns nested in small colonies at inland lakes and marshes. The southern Oregon and northeastern California region (hereafter referred to as the SONEC region) of western North America historically supported a number of Caspian tern colonies (Bailey 1902, Finley 1907, Whetmore 1919, Willett 1919; *Figure 3.1*). The SONEC region is a large geographic area that includes parts of south-central and southeastern Oregon, the northeastern corner of California, and small portions of the northwestern corner of Nevada (Fleskes and Yee 2007; *Figure 3.1*). The region contains relatively shallow lakes and wetlands fed mainly by spring runoff of winter snowpack, and the amount of available surface water fluctuates greatly based on drought cycles. Agriculture (e.g., cattle ranching and alfalfa production) is important economically in the region and the allocation of scarce water resources is complicated, contentious, and tightly controlled throughout much of the region (Braunworth et al. 2003, Mayer 2005, ODFW 2007).

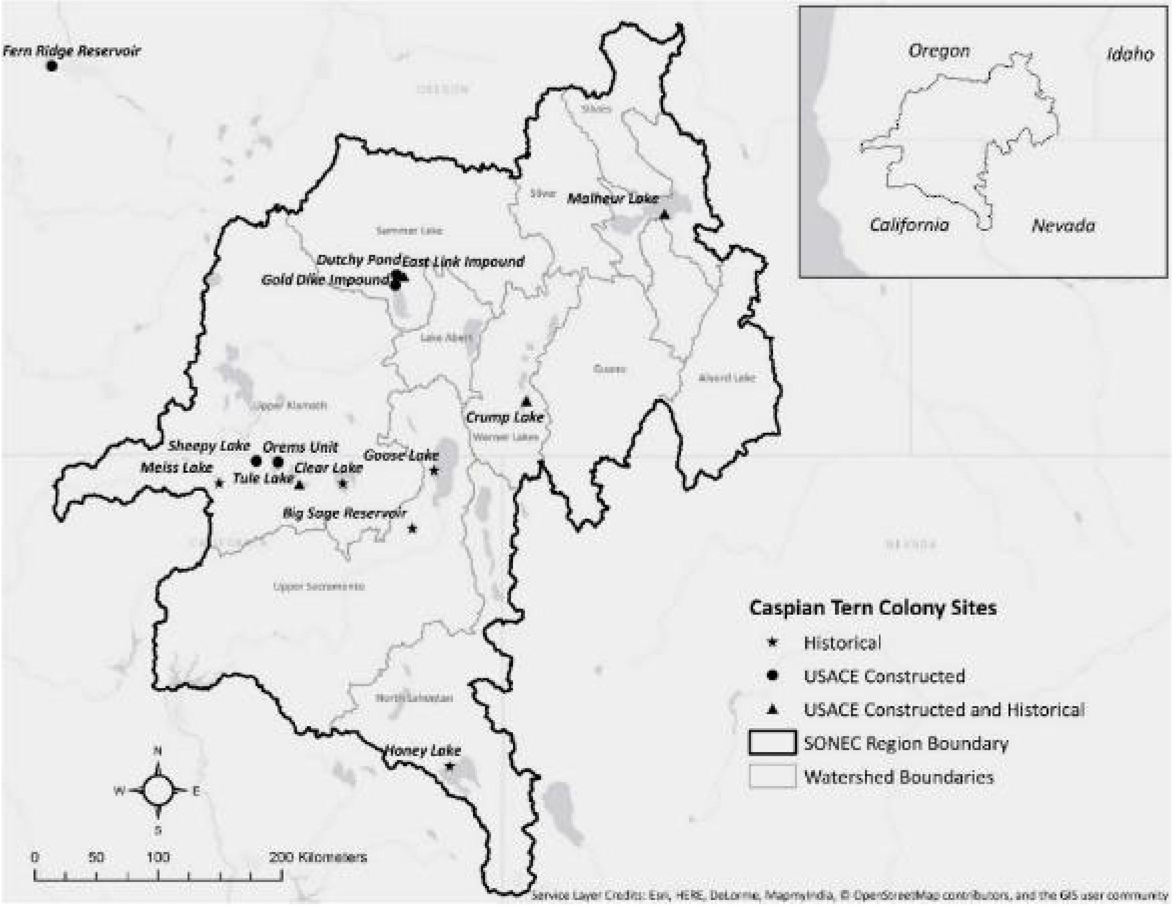

Figure 3.1. Map of the southern Oregon and northeastern California (SONEC) region showing locations of historical Caspian tern nesting islands and islands constructed for tern nesting by the U.S. Army Corps of Engineers as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary.

Within the SONEC region, much of the historical Caspian tern nesting habitat was lost over time due to land use and water use changes associated with agricultural development. The remaining nesting locations in the region are ephemeral due to the tendency for frequent drought cycles in the SONEC region (Malevich et al. 2013). At a time when nesting opportunities in the SONEC region were declining, Caspian terns began colonizing coastal and estuarine habitats in central/northern California and Washington State, and the size of the Pacific Flyway-wide population increased (Gill and Mewaldt 1983). In 1984, Caspian terns began nesting at a dredged material disposal site on an island in the Columbia River estuary, on the border of Oregon and Washington states. The breeding population of Caspian terns in the estuary quickly increased to about 8700 pairs by 1998 (Roby et al. 2003b), by far the largest colony for the species in the world and peaked in 2008 at over 10,600 pairs (Roby et al. 2009). By 1998, consumption of out-migrating salmonid (*Oncorhynchus* spp.) smolts by the large numbers of Caspian terns breeding in the Columbia River estuary was identified as a potential limiting factor in the recovery of threatened or endangered salmonids listed under the U.S. Endangered Species Act (USFWS 2005, 2006). Prior to establishing very large colonies on dredged material disposal sites in the Columbia River estuary (i.e. on Rice and East Sand islands), Caspian terns in the Pacific Flyway mainly formed small breeding colonies, nested in ephemeral habitats, and moved among breeding sites in response to nesting habitat loss or disturbances (Collis et al. 2002, Suryan et al. 2004). These nesting ecology traits were exploited during 1999–2001 to relocate the terns nesting on Rice Island to East Sand Island (21 km closer to the mouth of the Columbia River) as a means to shift the diet composition of Caspian terns nesting in the Columbia River estuary away from a preponderance of juvenile salmonids. Relocating the tern colony from Rice Island to East Sand Island resulted in a large increase in the proportion of marine forage fish in the diet and reduced the consumption of juvenile salmonids by Caspian terns nesting in the estuary by more than half (Roby et al. 2002; see *Chapter 1*). However, Caspian terns nesting at East Sand Island continued to consume an average of about 5 million out-migrating salmonid smolts each year (see *Chapter 1*).

Beginning in 2008, additional management actions were implemented to reduce losses of juvenile salmonids to Caspian tern predation in the Columbia River estuary, while ensuring the long-term conservation of the Pacific Flyway population of Caspian terns. The Records of Decision (RODs) for the *Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (hereafter referred to as the *“Estuary Caspian Tern Management Plan”*), signed in November 2006 and updated in 2015, sought to reduce the size of the tern colony on East Sand Island by reducing the area of tern nesting habitat, while creating or restoring tern nesting habitat elsewhere as compensation. The area of tern nesting habitat on East Sand Island was to be gradually reduced as alternative nesting habitat was provided at the ratio of one acre of reduction per two acres of alternative habitat created. The goal of the *Estuary Caspian Tern Management Plan* was to redistribute approximately 60% of the East Sand Island tern colony by creating alternative colony sites outside the Columbia River basin in Oregon and California, thereby reducing tern predation rates on steelhead smolts (*O. mykiss*) in the Columbia River estuary by a similar percentage (USFWS 2005, USFWS 2006, USACE 2006, USACE 2015a, USACE 2015b). At the time when the RODs were issued, a 60% reduction in the size of the Caspian tern colony on East Sand Island would have displaced about 5,400 breeding pairs. The U.S. Army Corps of Engineers – Portland District (USACE-Portland District) gradually reduced the area of suitable nesting habitat for Caspian terns on East Sand Island from a baseline of 5 acres (2.0 ha) in 2008 to 1.0 acre (0.4 ha) in 2015 and implemented nest dissuasion measures to prevent Caspian terns from establishing new nesting colonies elsewhere in the Columbia River estuary. The reduction in nesting habitat at East Sand Island as described in the *Estuary Caspian Tern Management Plan* was completed by

2015, and the area of nesting habitat prepared for terns at East Sand Island remained 1.0 acres (0.4 ha) during 2016-2019.

Management actions to reduce tern predation on ESA-listed salmonid smolts in the Columbia River basin were also implemented in the Columbia Plateau region as part of the *Inland Avian Predation Management Plan* (*IAPMP*; USACE 2014). Starting in 2014, management was implemented to dissuade nesting activity and displace Caspian terns from the two historically largest breeding colonies in the Columbia Plateau region (Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir; *Figure 3.2*). The goal of the *IAPMP* was to reduce Caspian tern predation rates in the Plateau region on each ESA-listed ESU/DPS of salmonids to less than 2% of available smolts per year (USACE 2014). Acknowledging that Caspian tern nesting habitat was available at other sites in the Columbia Plateau region, the management objective of the *IAPMP* was to reduce the number of Caspian terns to fewer than 200 breeding pairs throughout the Columbia Plateau region and redistribute the remaining breeding pairs to colonies outside of the Columbia Basin.

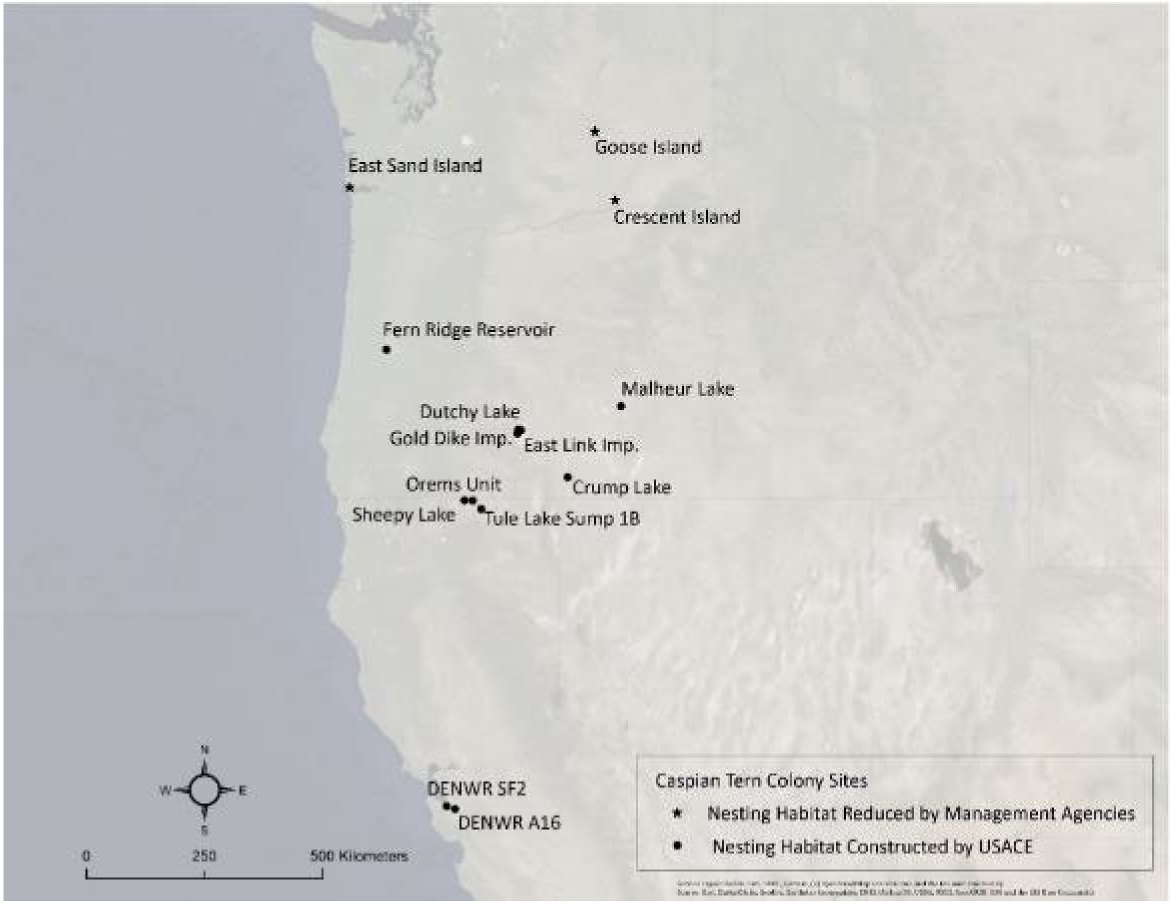

Figure 3.2. Locations of managed Caspian tern colonies in Oregon, Washington, and California states, and locations of nesting islands constructed or enhanced as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary and the Inland Avian Predation Management Plan.

By 2015, the first year that tern nesting habitat at East Sand Island was reduced to 1.0 acre (0.4 ha), the USACE had physically built or enhanced a total of 10.25 acres (4.15 ha) of islands in Oregon and California as potential alternative nesting habitat for about 6,000 pairs of Caspian terns, the total number of displaced tern pairs if both the *Estuary Caspian Tern Management Plan* and the *IAPMP* achieved their objectives. Islands built as potential tern nesting habitat in the Upper Klamath Basin, however, were not expected to be available in every year following construction due to planned flooding regimes. For example, of the 2-acre (0.82-ha) island constructed in Tule Lake Sump 1B of Tule Lake National Wildlife Refuge (NWR), only 1.35 acres (0.55 ha) was considered as "effective" Caspian tern nesting habitat on an annual basis, as the island was likely to be land-bridged one out of every three years (USACE 2015; *Table 3.1*). Regardless, to maximize the likelihood of successfully establishing tern nesting colonies, the strategy involved creating or enhancing nesting habitat (i.e. island construction or improvement), using social attraction techniques (i.e. decoys and audio playback systems), and closely monitoring tern islands to identify and manage potential factors limiting Caspian tern nesting success at the newly created nesting habitat. Once Caspian tern colonies were established, on-going (annual or periodic) management of nesting substrate (e.g., vegetation control and erosion abatement) and other factors limiting tern colony size and nesting success (e.g., predators) needed to be addressed to ensure the long-term sustainability of each site as tern nesting habitat (Kress 2000, Kress and Hall 2002, Roby et al. 2002).

Table 3.1. Location, area of provided tern nesting habitat, and year of construction for the 14 islands built or enhanced as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary and the Inland Avian Predation Management Plan.

General Location	Colony Location	State	Area of Nesting Habitat (acre)	Area of Effective Nesting Habitat (acre)	Year Constructed
Willamette Valley	Fern Ridge Reservoir	Oregon	1.0	1.0	2008
Warner Valley	Crump Lake	Oregon	1.0	1.0	2008
Summer Lake Wildlife Area	East Link Impoundment	Oregon	0.5	0.5	2009
Summer Lake Wildlife Area	Dutchy Lake	Oregon	0.5[a]	0.5[a]	2009
Tule Lake NWR	Tule Lake Sump 1B	California	2.0	1.35	2009
Summer Lake Wildlife Area	Gold Dike Impoundment	Oregon	0.5	0.5	2009
Lower Klamath NWR	Orems Unit	California	1.0[b]	0.2[b]	2009
Lower Klamath NWR	Sheepy Lake	California	0.8[c]	0.8[c]	2010
Malheur NWR	Malheur Lake	Oregon	1.0	1.0	2012
Don Edwards NWR	Pond A16	California	0.69[d]	0.69[d]	2015
Don Edwards NWR	Pond SF2	California	1.26[e]	1.26[e]	2015

[a] Floating island removed prior to the 2013 breeding season; [b] Removed from effective service in 2017; [c] 0.8-acre (0.3-ha) floating island removed and replaced with a 1.0-acre (0.4-ha) rock-core island prior to the 2018 breeding season; [d] Tern nesting habitat consisted of two small islands in close proximity; [e] Tern nesting habitat consisted of three small islands in close proximity.

This study summarizes the results of monitoring the 14 different tern islands that were constructed or enhanced by the USACE as alternative Caspian tern nesting habitat in compensation for the managed reduction of tern nesting habitat in the Columbia River estuary and in the Columbia Plateau region as part of the *Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* and the *IAPMP*. Full implementation of these two management plans was expected to displace approximately 6,000 breeding pairs of Caspian terns from their respective former breeding colonies. Our monitoring efforts sought to address several questions: (1) were artificial islands constructed as Caspian tern nesting sites used by breeding terns; (2) if artificial islands were occupied by nesting Caspian terns, were nesting attempts successful in raising young; (3) what factors limited the size and nesting success of Caspian tern breeding colonies on artificial islands; (4) if Caspian tern breeding colonies formed on artificial tern islands, were those colonies sustainable at or near full occupancy over the longer term; (5) did shifting Caspian tern breeding pairs from colonies in the Columbia River basin to colonies on USACE-constructed islands create new impacts on fish species of conservation concern.

3.3 Study Sites

A total of nine tern islands were constructed from 2008 to 2012 as part of the *Estuary Caspian Tern Management Plan,* and an additional five small islands were modified and enhanced as tern nesting habitat in Don Edwards San Francisco Bay National Wildlife Refuge (DENWR) in 2015 as part of the *Estuary Caspian Tern Management Plan* and the *IAPMP*. As part of the two management plans, the USACE constructed Caspian tern nesting habitat at Fern Ridge Reservoir (OR), Crump Lake (OR), East Link Impoundment (OR), Dutchy Lake (OR), Tule Lake Sump 1B (CA), Gold Dike Impoundment (OR), Orems Unit (CA), Sheepy Lake (CA), Malheur Lake (OR), DENWR (CA; *Table 3.2*).

The first island constructed by the USACE-Portland District was a 1-acre (0.4-ha) rock core island that was built in Fern Ridge Reservoir (a USACE owned property; 44.07° N, -123.28° W) near Eugene, Oregon, in the southern Willamette Valley. The island was built near the western end of Royal Avenue during the winter of 2007–2008 (*Table 3.3*), when reservoir levels were low enough so that heavy equipment could access the construction site on old roadbed that is normally submerged when the reservoir is full. The shoreline of the island was protected with large rock revetment and the surface of the island was covered with fine gravel. There was no history of Caspian terns nesting at Fern Ridge Reservoir or other nearby water bodies when the tern island was built in early 2008. Caspian terns had been recorded foraging at the reservoir and passing through during spring and fall migration, however, and the site offered an opportunity to provide habitat in the migration path of Caspian terns heading to or returning from the Columbia River basin (Seto et al. 2003).

Table 3.2. Area of available tern nesting habitat (acre) during 2008–2018 on islands constructed or enhanced by the U.S. Army Corps of Engineers as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary and the Inland Avian Predation Management Plan. Rock-core, silt-core, or floating islands were constructed in the Willamette Valley (OR), the southern Oregon and northeastern California (SONEC) region (OR/CA), and existing small islands in Don Edwards San Francisco Bay National Wildlife Refuge (CA) were enhanced to accommodate nesting Caspian terns.

Colony Location (State)	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Fern Ridge Reservoir (OR)	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
Crump Lake (OR)	1.0	1.0	1.0	1.0	1.0	1.0[a]	1.0[a]	1.0[a]	1.0	1.0	1.0
East Link Impoundment (OR)	--	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
Dutchy Lake (OR)	--	0.5	0.5	0.5	0.5	NA[c]	NA[c]	NA[c]	NA[c]	NA[c]	NA[c]
Tule Lake Sump 1B (CA)	--	--	0.0[b]	2.0	2.0	2.0	2.0	2.0	2.0	2.0	2.0
Gold Dike Impoundment (OR)	--	--	0.0[b]	0.0[b]	0.5	0.5	0.5	0.5	0.5	0.5	0.0[a]
Orems Unit (CA)	--	--	0.0[b]	1.0[a]	0.0[b]	0.0[b]	0.0[b]	0.0[b]	0.0[b]	NA[d]	NA[d]
Sheepy Lake (CA)	--	--	0.8	0.8	0.8	0.8	0.8[a]	0.8[a]	0.8	0.8	1.0[e]
Malheur Lake (OR)	--	--	--	--	1.0	1.0	1.0	1.0[a]	1.0	1.0	1.0
Don Edwards NWR (CA)	--	--	--	--	--	--	--	1.95	1.95	1.95	1.95
Total Area (acre)	2	3	3.8	6.8	7.3	6.8	6.8	8.75	8.75	8.75	8.45

[a] Nesting habitat completely or nearly land-bridged during part of the breeding season, [b] Nesting habitat completely land-bridged throughout the entire breeding season, [c] Island decommissioned and removed from the site, [d] Island structure remains in place but removed from service, [e] Floating Island removed and replaced with a rock-core island.

The second island to be built was another 1.0-acre (0.4-ha) rock core island that was built at the site of a historical island and Caspian tern colony site in Crump Lake (42.284° N, -119.844° W) in the Warner Valley of southcentral Oregon, near the small town of Adel in the SONEC region. All islands and seasonally flooded wetlands associated with Crump Lake are owned and managed by the Oregon Division of State Lands. Historically, there was a low-lying rocky island near the center of Crump Lake, which, when exposed, was used by a variety of colonial nesting waterbirds, including Caspian terns. The natural island was destroyed in the 1950s when looters bulldozed the island searching for Native American artifacts (Roby et al. 2003a). A small number of Caspian terns were documented to be nesting at Crump Lake in 1966 (9 breeding pairs; Marshall 1969) and in 1987 (6 breeding pairs; Stern 1988). In the early 1990s, an effort was made to improve the natural island as perennial nesting habitat for waterbirds by the Oregon Department of Fish and Wildlife (ODFW), but wind and wave erosion quickly reduced island elevation such that it continued to be inundated during high-water periods (G. Keister, ODFW, pers. comm.). Caspian terns would periodically attempt to nest on the site, but fluctuating lake levels caused by seasonal input and withdrawal of water led to situations where nesting birds could lose their nests to flooding or terrestrial predators midway through the breeding season. In 2000, the remnant Crump Lake island was exposed throughout the breeding season and approximately 150 Caspian tern breeding pairs nested there, but the island was submerged during 2001-2002 and unavailable as nesting habitat. In 2003, as water levels in Crump Lake rose and inundated the remnant nesting island, a temporary wooden nesting platform was constructed adjacent to the remnant island to test the feasibility and outcome of providing artificial Caspian tern nesting habitat at the site (Roby et al. 2003a). A total of 49 breeding pairs of Caspian terns successfully nested on the artificial platform, and an additional 22 breeding pairs nested on the remnants of the natural island following a drop in lake levels that exposed part of the island. There is no published record of Caspian terns nesting at Crump Lake during the 2004–2007 breeding seasons. The USACE completed construction of an artificial tern island in Crump Lake prior to the 2008 breeding season (*Table 3.3*). When the island was built during February, Crump Lake was nearly completely dry, so a temporary 1-km road could be built to the construction site so that heavy equipment could access the site. As with the Fern Ridge tern island, the shoreline was protected from wave action by large rock revetment and the surface was covered with fine gravel. Once island construction was completed, the temporary road to the island on the lakebed was removed.

The USACE constructed three additional tern islands, each 0.5-acre (0.2-ha), in Summer Lake Wildlife Area in south-central Oregon and part of the SONEC region (*Table 3.3*). The Summer Lake Wildlife Area is located near the town of Summer Lake and is owned and managed by ODFW. The Summer Lake Wildlife Area consists of an extensive area of spring-fed wetlands, moist soil units, and freshwater impoundments associated with the Ana River, which flows into the northern end of Summer Lake. The lake itself is extremely alkaline and supports no fish life. Prior to 2002, up to a few dozen Caspian terns occasionally nested on a small, unnamed island at the north end of Summer Lake, near the southern boundary of the Wildlife Area, in association with a much larger mixed colony of ring-billed gulls (*Larus delawarensis*) and California gulls (*L. californicus*; Shuford and Craig 2002). After 2001, nesting by Caspian terns or gulls was not recorded on the unnamed island in Summer Lake because the island had been land-bridged due to low water levels in Summer Lake (M. St. Louis, ODFW, pers. comm.). During 2002–2005, Caspian terns nested in low numbers on a small artificial island made from mounded soil in the East Link Impoundment of the Wildlife Area, but suitable nesting habitat was very limited (Roby et al. 2003a). Caspian terns were documented nesting in the East Link Impoundment in 2002 (1 breeding pair), 2003 (5 breeding pairs), and 2005 (3 breeding pairs). No Caspian terns nested at Summer Lake during the 2004–2008 breeding seasons.

The first artificial island constructed at the Summer Lake Wildlife Area by the USACE, completed prior to the 2009 nesting season, was a rock core island built in the middle of East Link Impoundment on the eastern side of the Wildlife Area (42.956^o N, -120.715^o W), where small numbers of Caspian terns had previously nested on mounded soil islands when the water levels in the impoundment were low. The East Link tern island was constructed when the impoundment was nearly dry, allowing access to the site by heavy equipment. The second tern island built in the Summer Lake Wildlife Area, completed just prior to the 2009 nesting season, was a floating island in Dutchy Lake built from recycled plastic and reinforced with steel cables (Floating Islands West, Mokelumne Hill, CA). The floating island was built in modules that were then towed out to the center of Dutchy Lake and assembled. Dutchy Lake (42.967^o N, -120.769^o W) is a small lake in the northern part of the Wildlife Area that cannot be drained and has not been known to dry out, even during severe drought. The rectangular floating island was anchored at all four corners and top-dressed with light-weight pumice gravel as nesting substrate. Following the conclusion of the 2009 nesting season, a third 0.5-acre (0.2-ha) island was completed at Summer Lake Wildlife Area; a rock core island similar to the one built in East Link Impoundment was built in Gold Dike Impoundment near the southern boundary of the Wildlife Area (42.913^o N, -120.779^o W) when the impoundment was nearly dry; however, this impoundment was not reflooded and the island surrounded by water until the 2012 nesting season.

Prior to the 2010 nesting season, the USACE completed construction of three additional tern islands in northeastern California, two in Lower Klamath National NWR and one in Tule Lake NWR (*Table 3.3*). Lower Klamath NWR and Tule Lake NWR are located south of Klamath Falls, Oregon, and are owned and managed by the US Fish and Wildlife Service (USFWS). Tule Lake NWR is located entirely in California, whereas Lower Klamath NWR straddles the Oregon-California border. These two Refuges were set aside to protect waterfowl nesting habitat in the years following the start of the Klamath Reclamation Project (1905) by the U.S. Bureau of Reclamation (BOR) in order to convert the lakes and marshes of the Lower Klamath Lake and Tule Lake areas into agricultural lands. Nesting by Caspian terns was first documented in 1899 in the area that would later become the Tule Lake NWR (Bailey 1902) and in 1905 in the area that would become the Lower Klamath NWR (Finley and Bolhman 1907). All nesting habitat for Caspian terns in the vicinity of these two NWRs was subsequently lost as lands both within and outside the refuges were converted to agriculture (USBR 1998, USFWS 2009, Shuford 2010), but small nesting colonies of Caspian terns continued to form in some years at nearby locations within the Upper Klamath Basin where gull colonies were found (e.g., Clear Lake NWR and Meiss Lake, [Shuford and Craig 2002]). Following the 2009 nesting season, the USACE completed construction of a 2-acre (0.8-ha) rock-core island in the southwestern part of Tule Lake Sump 1B (42.842^o N, -121.477^o W) in Tule Lake NWR near Tulelake, California. Tule Lake Sump 1B was drained late in 2009, allowing heavy equipment to access the construction site, but Sump 1B was not flooded again until just before the 2011 nesting season. Also, following the 2009 nesting season, in nearby Lower Klamath NWR, a 1.0-acre (0.4-ha) silt core island was built in Orems Unit, an impoundment near the eastern boundary of the Refuge (41.962^o N, -121.634^o W). The Orems Unit impoundment was dry during island construction, and substrate from the impoundment was used to build the island once a rock revetment perimeter had been constructed to retain the silt core. Early in 2010 a second tern island was constructed in Lower Klamath NWR; a 0.8-acre (0.3-ha) floating island (Floating Islands West, Mokelumne Hill, CA) was built in the middle of Sheepy Lake (41.968^o N, -121.792^o W), a small lake in the western part of the Refuge that cannot be drained and rarely dries out completely. The floating island in Sheepy Lake was decommissioned and removed following the 2017 nesting season, however, and replaced with a 1.0-acre (0.4-ha) rock core island prior to the 2018 nesting season.

The final tern island to be built by the USACE in the SONEC region was completed in early 2012, when a 1.0-acre (0.4-ha) rock-core island was constructed in the southern part of Malheur Lake (43.293^o N, -118.817^o W) within Malheur NWR (*Table 3.3*). Malheur NWR is located in eastern Oregon about 348 km south of the town of Burns and is owned and managed by the USFWS.

Malheur NWR is within one the largest freshwater wetland areas in the United States, and there is a long history of waterbird use of this area when conditions are favorable. Caspian terns were noted as nesting at Malheur Lake in the early 1900s (Willet 1919), but Caspian tern nesting colonies at Malheur NWR were uncommon by the 1930s (Jewett 1936) and were completely absent from the early 1960s through 1979 (Thompson and Littlefield 1980). During the 1984–1986 breeding seasons, an estimated 300–400 Caspian tern breeding pairs nested on a natural island on the north side of Malheur Lake (Paullin et al. 1988), and between 25–192 breeding pairs nested on an island in the northcentral part of the lake during the 1997–2001 breeding seasons (Shuford and Craig 2002). Data on nesting by Caspian terns at Malheur Lake are lacking during 2002–2010, but Caspian terns were confirmed nesting on an island located on private property on the northern edge of the lake in 2011 and 2012. The USACE completed construction of an artificial tern island in Malheur Lake near the mouth of the Donner-und-Blitzen River shortly before the 2012 breeding season. Water levels in the lake were fairly high during construction, so a modular barge was used to ferry materials out to the site of the island. Like other rock core islands built as tern nesting habitat, the tern island in Malheur Lake had a rock revetment shoreline and was surfaced with fine gravel. The Malheur Lake tern island was the eighth and final island that the USACE built in the SONEC region.

Prior to the 2015 nesting season a total of five small islands in two different salt pond impoundments in Don Edwards San Francisco Bay NWR (DENWR), located at the southern end of San Francisco Bay, California were enhanced by the USACE-Walla Walla District as tern nesting habitat (*Table 3.3*). Previously, a number of small islands of mounded soil had been created as waterbird nesting islands by the Refuge when the two salt pond impoundments, Pond A16 (37.447^o N, -121.971^o W) and Pond SF2 (37.491^o N, -122.133^o W), were dry. DENWR is owned and managed by the USFWS and is comprised of salt ponds, intertidal salt marsh, and intertidal mudflats, as well as upland habitats. Historically, Caspian terns nested at salt ponds on vegetation-free levees (Gill 1977), like those found at DENWR, in the San Francisco Bay Area, but the locations and sizes of breeding colonies were highly dynamic from year to year (Strong et al. 2004). In 1979, before the major expansion of Caspian tern nesting in the Columbia River estuary, over half of the Pacific Flyway breeding population of Caspian terns (ca. 3,000 breeding pairs) was located in the San Francisco Bay Area (Wires and Cuthbert 2000, Shuford and Craig 2002, Roby et al. 2002, Suryan et al. 2004). By the early 1980s, around 25% of the breeding population of Caspian terns in the Pacific Flyway (ca. 1,500 breeding pairs) was still located in the San Francisco Bay Area (Gill and Mewaldt 1983). By 2001, that proportion had decreased to less than 10% (ca. 830 breeding pairs: Shuford and Craig 2002). During 2003–2009 the number of Caspian terns nesting in the San Francisco Bay Area averaged ca. 1,075 breeding pairs (range: 830–1,372 breeding pairs), with the largest colonies located at Brooks Island and Knight Island in the central and the northern parts of San Francisco Bay, respectively (Collis et al. 2012). In 2011, less than 325 Caspian tern breeding pairs (ca. 3% of the Pacific Flyway breeding population) nested in the San Francisco Bay Area, with ca. 98% of those terns nesting at Brooks Island (Peck-Richardson et al. 2019).

Before the USACE enhanced the existing soil mound islands in Pond A16 and Pond SF2, there had been no prior history of nesting by Caspian terns in either of these salt ponds (Collis et al. 2002), although some Caspian tern nesting had occurred on small soil mound islands in nearby salt ponds. These islands were, however, subject to erosion along the shoreline due to wave action and the surface of the islands turned to sticky muck after a rain that subsequently developed large cracks after drying that posed hazards to tern chicks. The USACE placed rock revetment along the shorelines of the five islands in order to prevent shoreline erosion and top-dressed each island with a shell and gravel mix in order to provide a suitable substrate for tern nesting. Each island averaged 0.39 acres (0.16-ha), for a total area of tern nesting habitat of 1.95 acres (0.79-ha).

Not all tern islands built by the USACE were available as tern nesting habitat through the 2018 breeding season, when our study ended. In 2014, the USACE-Portland District removed the 0.5-acre (0.2-ha) artificial floating island from Dutchy Lake in Summer Lake Wildlife Area following failures of the anchoring system and some island construction materials. Similarly, in early 2018, the USACE-Portland District removed the 0.8-acre (0.3-ha) artificial floating island from Sheepy Lake in Lower Klamath NWR following the failure of some island construction materials; however, the floating island in Sheepy Lake was replaced with a 1.0-acre (0.4-ha) rock-core island before the 2018 nesting season. The 14 islands constructed or enhanced as alternative nesting habitat for Caspian terns by the USACE were located between ca. 250 km and ca. 1,000 km from the Caspian tern colonies on East Sand Island, Crescent Island, and Goose Island, which were managed to reduce or eliminate tern nesting habitat (*Table 3.3*). For additional details on the criterion used to select sites for colony restoration for Caspian terns outside the Columbia Basin as part of the *Estuary Caspian Tern Management Plan* and the *IAPMP*, see Seto et al. (2003) and Collis et al. (2012a).

Table 3.3. Straight-line distances between tern islands constructed or enhanced by the U.S. Army Corps of Engineers in the Willamette Valley (OR), the southern Oregon and northeastern California (SONEC) region (OR/CA), and the San Francisco Bay Area (CA) and the three Caspian tern colonies in the Columbia River basin that were managed as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary and the Inland Avian Predation Management Plan. Also provided is the year of island construction, and the first year that each colony location was available as nesting habitat for terns.

Island Location	State	Distance from East Sand Is. (km)	Distance from Crescent Is. (km)	Distance from Goose Is. (km)	Island Construction Complete	First Season Available
Fern Ridge Res.	OR	250	410	448	Feb-2008	2008
Crump Lake	OR	552	429	523	Mar-2008	2008
East Link Imp.	OR	449	377	461	Jan-2009	2009
Dutchy Lake	OR	446	377	461	Mar-2009	2009
Tule Lake Sump 1B	CA	530	515	597	Aug-2009	2011
Gold Dike Imp.	OR	451	383	467	Sept-2009	2012
Orems Unit	CA	513	508	588	Sept-2009	2011[a]
Sheepy Lake	CA	508	513	591	Mar-2010	2010
Malheur Lake	OR	525	312	412	Feb-2012	2012
DENWR A16 Pond	CA	993	993	1082	Feb-2015	2015
DENWR SF2 Pond	CA	986	992	1080	Feb-2015	2015

[a] *Nesting habitat completely or nearly land-bridged for much of the breeding season.*

3.4 Methods

We constructed a permanent wooden frame and plywood observation blind on the edge of each tern island soon after island construction was completed so that colony monitors could collect data on tern colony size, nesting success, diet composition, and factors limiting tern colony size and nesting success without disturbing birds nesting on the islands. The Tule Lake Sump 1B tern island was an exception; because the island had a large surface area (0.81 ha), the observation blind was built in the center of the island, and an above-ground tunnel of landscape fabric was used to access the blind without disturbing birds nesting on the island. For the two floating tern islands in Sheepy Lake and Dutchy Lake, an observation blind was built on a small floating platform that was then lashed to the side of the floating island. At each of the monitored USACE-constructed islands, we recorded the date when the first Caspian tern egg was laid, the date of peak breeding colony size, and the date when the first young fledged based on monitoring from the observation blind using binoculars and spotting scopes. In order to document the date that the first egg was laid, we identified, photographed, mapped, and tracked up to the first 50 tern nest scrapes that were formed; these nest scrapes were monitored during each colony visit from the initiation stage (i.e. regularly attended nest scrape) until the nest failed or chicks became too mobile to reliably assign to a particular nest scrape. If a nest failed early in the breeding season, then another nest was selected for monitoring to maintain sample size. Once chicks began to hatch, we also monitored and counted chicks by plumage development class (i.e. downy, mid-sized, black-masked, and black-capped) during each colony visit. We considered chicks to be in the downy plumage class while they lacked juvenile feathers on the back or wings (i.e. ca. < 10 days post-hatch), to be in the mid-sized plumage class once juvenile feathers developed on the back and wings but prior to the appearance of a black mask (i.e. ca. 10–24 days post-hatch), to be in the black-masked plumage class once black juvenile feathers developed across the eyes and forming a “mask” (i.e. ca. 25–30 days post-hatch), and to be in the black-capped plumage class once black juvenile feathers developed across the top of the head forming a “cap” (i.e. ca. > 30 days post-hatch). We used regular chick monitoring to identify the first date when young fledged during each breeding season for each occupied island. We also counted the number of active tern nests during each visit in order to estimate the size of the breeding colony on any given day and used the date with the largest average count of active nests as the date of peak colony attendance (see *below*).

We deployed 250–300 Caspian tern decoys (Mad River Decoys; Waitsfield, VT; and Dave Smith Decoys, Lebanon, OR) and two audio playback systems (Murremaid Music Boxes, Bremen, ME) at each island deemed suitable for tern nesting (i.e. surrounded by water) prior to the start of each breeding season (date range: 07 April – 30 April) in 2008–2015. The only exception to this social attraction protocol was the Crump Lake tern island, where social attraction was only deployed during the first two breeding seasons (2008 and 2009) and not deployed thereafter. We secured decoys on the surface of each island by driving PVC tubing into the substrate and placing each decoy, which had a 3/8”-diameter hole drilled in the bottom, over a tube. We setup the decoys to represent breeding pairs and single attending adults spaced about 1-m apart and placed decoys in both sitting and standing postures to better simulate an active Caspian tern breeding colony. Caspian tern decoys were arranged in a colony-like formation beginning 15–20m in front of the observation blind (*Figure 3.3*). We used two independent audio playback systems at each island to increase the volume of playback and provide system redundancy in case of equipment failure. Each audio playback system broadcast through two outdoor Omni speakers (TIC Corp., La Puente, CA), which were placed within the decoy array. A continuous loop of Caspian tern vocalizations was broadcast throughout the entire breeding season. Social attraction was only deployed during the 2008 and 2009 breeding seasons at Crump Lake tern island because after the 2009 nesting season ODFW requested that social attraction no longer be deployed due to concerns over the size of the Crump Lake tern colony and the potential impact of Caspian tern predation on the ESA-listed Warner sucker (*Catostomus warnerensis*). In addition to data collection, we also used the observation blinds to facilitate predator control activities.

Figure 3.3. Image showing the Caspian tern decoy array at the island constructed by the U.S. Army Corps of Engineers in Gold Dike Impoundment, Summer Lake Wildlife Area at the start of the 2012 nesting season.

We counted all adult terns, active tern nests (i.e. breeding pairs attending eggs or chicks), and tern chicks twice during each colony visit, once shortly after arrival and again shortly before leaving the observation blind. We varied the timing of visits to each colony in order to capture the range of daily colony attendance. Prior to the beginning of each breeding season, we installed or repaired a 10m X 10m rope grid over the colony surface (marked by the decoy array; 50m long X 40m wide total) to assist with counting efforts throughout the breeding season. We conducted colony counts using binoculars by starting at one side of the colony and scanning across the colony and counting all birds. Caspian tern colonies at the USACE-constructed islands were small enough that all adult terns, attended nests, and chicks could be counted individually, but at the largest of these colonies we used mechanical tally counters to assist with obtaining accurate counts. All counts were conducted by two observers simultaneously. These counts were then averaged as an estimate of the total number, but only simultaneous counts within 10% of each other were retained as valid. All counting sessions where counts differed by more than 10% were immediately repeated.

We used the estimate of the peak number of breeding pairs, obtained late in the incubation period, and the peak number of fledglings produced, obtained early in the fledging period, at each of the monitored USACE-constructed islands to calculate an annual estimate of average nesting success (average number of young raised/breeding pair) for each active tern colony during 2008–2015. We considered a tern chick to have successfully fledged if it was observed in flight or reached an age of 37 days post-hatching. In most years, we also conducted post-breeding season surveys to enumerate the number of dead fledglings or near fledging-age chicks to correct estimates of nesting success.

Under permit (Federal Bird Banding Permit #05271), we captured and banded adult and pre-fledged Caspian terns in order to measure movement rates among colonies and connectivity between tern colonies on USACE-constructed islands in the SONEC region and other colonies throughout the Pacific Flyway. We captured adult Caspian terns using monofilament noose mats or walk-in dome traps placed around active nests (i.e. nest scrapes containing ≥ 1 egg) during the late incubation period. We captured pre-fledged Caspian tern chicks by herding flightless young from the colony into holding pens just prior to the expected date of first fledging. Once captured, terns were immediately transferred to holding crates until they were banded and released. We banded terns with a federal numbered metal leg band and two colored plastic leg bands on one leg, and a colored plastic leg band engraved with a unique alpha-numeric code that allowed for the identification of individual terns at a distance (i.e. at colonies or roost sites). Pre-fledged Caspian tern chicks that were too small for the full complement of bands were fitted solely with a federal numbered metal leg band. We conducted banding operations during early morning hours when temperatures were moderate to reduce the risk of heat stress in captive birds.

As part of our overall monitoring effort, we also banded Caspian terns at colonies in the Columbia River estuary, in the Columbia Plateau region, and elsewhere in the Pacific Flyway in order to monitor vital rates of the Flyway-wide tern population and assess inter-colony movements and dispersal patterns of terns to and from colonies in the SONEC region through resightings of banded individuals. At the USACE-constructed islands in the SONEC region, we resighted banded Caspian terns during each visit to the colony using binoculars and/or a spotting scope from observation blinds situated at the periphery of the nesting habitat. During each resighting effort, the entire colony was systematically scanned by a trained observer using a spotting scope, and each banded bird observed was recorded with its complete color band combination and alpha-numeric code, when possible. We also attempted to determine the breeding status of each banded tern that was observed.

From the observation blinds, we also monitored factors that can limit the size and nesting success of each Caspian tern breeding colony (e.g., colony disturbances, availability and quality of nesting habitat, avian and mammalian predator activities, forage fish delivery rates) on USACE-constructed tern islands. We also used continuous video monitoring to assist with detection of predator activity at some of the islands. Using data collected from the observation blinds, we quantified disturbance rates at each tern colony by noting all disturbances that occurred during data collection for diet composition and counts of adults and young on-colony. We also counted the number of gulls in close association with each active tern colony twice daily, monitored tern chick meal delivery rates, and made qualitative assessments of the available tern nesting habitat throughout the breeding season. We also enumerated all gulls present at each island twice a week using a combination of blind-based and boat-based counts. At sites where disturbances by nocturnal avian predators (e.g., great horned owls [*Bubo virginianus*]) and mammalian predators (e.g., raccoons [*Procyon lotor*]) caused Caspian tern nest failure and colony abandonment, we selectively removed problem individuals using non-lethal and lethal methods under depredation permits. In addition, to aid in the persistence of new tern colonies, we selectively removed gulls that were observed repeatedly depredating tern nests to consume eggs and small chicks. Further, we attempted to reduce space competition between Caspian terns and California and ring-billed gulls at the Sheepy Lake floating island by placing netting over a designated tern nesting area and hazing gulls from that area. Finally, prior to the 2011 nesting season, we installed small wooden structures and shelters on each USACE-constructed tern island in the SONEC region to provide shade and protection from predators. We used padded leg-hold traps to live-capture great horned owls, which were released alive at a distance of >100 km from the capture site. Raccoons and gulls were removed using a scoped .22 caliber long rifle. All predator control activities were carried out under permit.

We used the Caspian tern habit of transporting whole fish (i.e. bill-loads) to the colony to feed their mates and young to determine the taxonomic composition of the diet (i.e. proportional contribution of different forage fish species) of terns nesting at each tern colony. We identified fish prey in bill-loads to the lowest taxonomic level possible and estimated the total length (mm) based on the length of the fish relative to the length of the tern's bill. We randomly selected terns returning to the colony with a bill-load by looking out from the blind and selecting the first tern in flight with a fish that was within ~50 m of the blind. We used binoculars to initially spot, track, and identify prey items to determine diet composition. The number of bill-load fish identified per day varied by the size of the colony, and weekly bill-load identification quotas varied accordingly. At colonies with < 50 individuals we attempted to collect 25 bill-load identifications per week; at colonies with 50 to 100 individuals we attempted to collect 50 bill-load identifications per week; at colonies with 101 to 200 individuals we attempted to collect 100 bill-load identifications per week; at colonies with 201 to 1000 individuals we attempted to collect 150 bill-load identifications per week.

As part of MSc thesis research at Oregon State University, Patterson (2012) investigated the foraging behavior of Caspian terns nesting at the USACE-constructed islands in Sheepy Lake and in Tule Lake Sump 1B using remotely downloadable GPS data loggers (Telemetry Solutions, Concord, California) during the 2011 breeding season (n = 10 tagged individuals/colony). From the GPS location data, Patterson (2012) calculated the time spent on-colony, the time spent foraging, the time spent commuting, and the proportion of time spent resting off-colony for each GPS-tagged tern. Then, using mixed-effects models, she examined the influences of breeding colony, breeding status, sex, and the interaction of breeding colony x breeding status on daily activity rates. Lastly, she delineated foraging area(s) used by each GPS-tagged tern based on the biased random bridge approach (Benhamou and Cornelis 2010, Benhamou 2011), and generated utilization distributions for each tagged breeding individual based on these derived foraging locations. See Patterson (2012) for detailed methods.

3.5 Results

3.5.1 Nesting Ecology

3.5.1.1 Nesting chronology – Caspian tern nesting chronology at the USACE-constructed islands in the SONEC region varied considerably among years during the study period. Despite the close geographic proximity of those Caspian tern colonies that formed on USACE-constructed islands, nesting chronology also varied considerably between colony sites (*Table 3.4*). At these colonies, we observed the first Caspian tern eggs as early as the last week of April and as late as the third week of June, the first fledged young (i.e. hatch year young capable of sustained flight) as early as the second week of July and as late as the last day of August, and peak colony attendance occurred as early as the last week of May or as late as the last week of July. For all five tern colony sites in the SONEC region combined, the median date when the first egg was observed was 24 May, the median date when the first fledged young was detected was 05 August, and the median date of peak colony attendance was 23 June (*Figure 3.4*). The date of peak colony attendance for Caspian terns nesting at each of the five sites where the USACE constructed islands in the SONEC region varied by just over a month (date range = 23–41 days; *Table 3.4*).

At the Summer Lake Wildlife Area, Sheepy Lake, and Malheur Lake tern colonies, the date when the first tern eggs were observed preceded the median date for all five tern colony sites because the first egg dates at the Crump Lake and Tule Lake Sump 1B tern colonies occurred later in some years. At the Malheur Lake tern colony, although the range of dates when peak colony attendance occurred was similar to other Caspian tern colonies in the SONEC region, the first Caspian tern eggs were consistently observed earlier in the season and chicks began fledging at earlier dates compared to other Caspian tern colonies in the region. We typically observed the earliest peak colony attendance at the tern colonies in the Summer Lake Wildlife Area. The latest dates of peak colony attendance were typically observed at the Tule Lake Sump 1B tern colony due to early season disturbances by avian and mammalian predators (see section *3.5.1.4 Factors limiting colony size & nesting success*). Along with a delayed peak in colony attendance, we observed Caspian tern chicks fledging from the colony at Tule Lake Sump 1B at later dates than young reared at the other USACE-constructed islands.

Table 3.4. Earliest and latest observation date and range (# of days) for nesting chronology of Caspian terns breeding at islands constructed in the southern Oregon and northeastern California (SONEC) region of North America by the U.S. Army Corps of Engineers as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary. Data were collected at Crump Lake during 2008–2015, at Summer Lake Wildlife Area during 2009–2015, at Sheepy Lake during 2010–2015, at Tule Lake Sump 1B during 2011–2015, and at Malheur Lake during 2012–2015.

Location	Earliest Observation (Date)	Latest Observation (Date)	Observation Date Range (# of days)
First Egg			
Crump Lake	May 12, 2013	June 14, 2011	33
Summer Lake Wildlife Area[a]	May 7, 2015	May 23, 2012	16
Sheepy Lake	May 5, 2013	May 20, 2011	15
Tule Lake Sump 1B	May 9, 2015	June 19, 2011	41
Malheur Lake	April 28, 2015	May 3, 2012	5
First Fledgling			
Crump Lake	July 19, 2012	August 14, 2011	26
Summer Lake Wildlife Area[a]	July 10, 2015	August 8, 2010	29
Sheepy Lake	July 16, 2013	August 12, 2011	27
Tule Lake Sump 1B	August 3, 2014	August 31, 2011	28
Malheur Lake	July 16, 2015	July 28, 2014	12
Peak Colony Attendance			
Crump Lake	June 8, 2015	July 18, 2011	40
Summer Lake Wildlife Area[a]	May 24, 2011	June 27, 2012	34
Sheepy Lake	June 13, 2014	July 6, 2010	23
Tule Lake Sump 1B	June 22, 2012	July 24, 2014	32
Malheur Lake	June 12, 2015	July 23, 2012	41

[a] *Includes the USACE-constructed islands in Dutchy Lake, East Link Impoundment, and Gold Dike Impoundment at the Summer Lake Wildlife Area*.

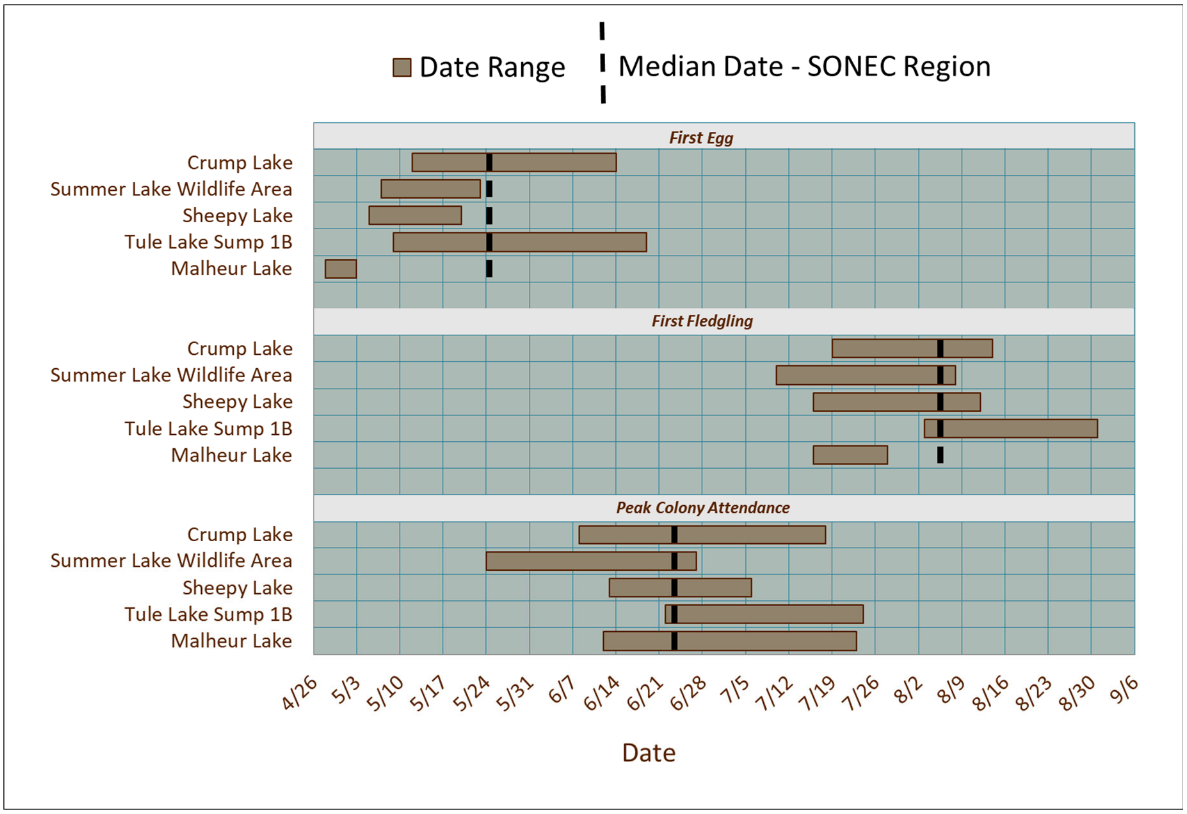

Figure 3.4. Nesting chronology of Caspian terns breeding at islands constructed in the southern Oregon and northeastern California (SONEC) region of North America by the U.S. Army Corps of Engineers as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary. Data were collected at Crump Lake during 2008–2015, at Summer Lake Wildlife Area during 2009–2015, at Sheepy Lake during 2010–2015, at Tule Lake Sump 1B during 2011–2015, and at Malheur Lake during 2012-2015.

3.5.1.2 Colony size – The USACE-constructed island at Crump Lake in the Warner Valley, Oregon, was first available as nesting habitat for Caspian terns during the 2008 nesting season, when ca. 430 Caspian tern breeding pairs used the island. The following year (2009) ca. 690 breeding pairs of Caspian terns nested at the island. During the 2010 nesting season, however, only ca. 70 Caspian tern breeding pairs attempted to nest at the Crump Lake tern island. The size of the Crump Lake tern colony declined further to ca. 35 breeding pairs during the 2011 nesting season. Nesting activity by Caspian terns at the Crump Lake tern island rebounded somewhat during the 2012 and 2013 nesting seasons when 115 breeding pairs and 223 breeding pairs, respectively, attempted to nest at the island. The Crump Lake tern island was nearly unused by nesting Caspian terns during the 2014 and 2015 nesting seasons, with only one breeding pair and three breeding pairs, respectively, attempting to nest. During the 2016 and 2017 nesting seasons, approximately 30 breeding pairs of Caspian terns attempted to nest at the Crump Lake tern island in each year. In 2018, ca. 90 breeding pairs were counted at the Crump Lake tern island. See *Table 3.5* for a summary of Caspian tern nesting use of the USACE-constructed island at Crump Lake.

Table 3.5. Estimated peak number of breeding pairs of Caspian terns nesting at colony sites in the southern Oregon and northeastern California (SONEC) region and in the Willamette Valley, Oregon, during 2008–2019. Unless noted, the U.S. Army Corps of Engineers constructed each colony site as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary.

	Estimated Peak Number of Breeding Pairs											
	Willamette Valley	Warner Valley	Summer Lake Wildlife Area			Lower Klamath NWR		Tule Lake NWR	Malheur NWR	Clear Lake NWR	Goose Lake Valley	
Year	Fern Ridge Res.	Crump Lake	Dutchy Lake	East Link Impoundment	Gold Dike Impoundment	Sheepy Lake	Orems Unit	Tule Lake Sump 1B	Malheur Lake /Singhus Ranch[a]	Clear Lake[a]	Goose Lake[a]	**Annual Total**
2008	0	430	NA	NA	NA	NA	NA	NA	NA	58	246	**734**
2009	0	690	8	7	NA	NA	NA	NA	NA	32	0	**737**
2010	0	71	0	29	NA	258	NA	NA	NA	0	0	**358**
2011	0	35	0	2	0	188	2	34	150[b]	12	0	**423**
2012	0	115	0	10	4	212	NA	207	317[b]	59	0	**924**
2013	0	223	NA	21	0	316	NA	79	530[c]	0	0	**1,169**
2014	0	1	NA	22	0	520	NA	109	134[c]	0	0	**786**
2015	0	3	NA	27	0	336	NA	545	148[c]	0	0	**1,059**
2016	--	27	NA	22	0	95	NA	100	14[c]	0	0	**258**
2017	--	26	NA	27	0	245	NA	274	21[c]	0	0	**593**
2018	--	86	NA	70	0	238	NA	627	420[c]	0	0	**1,441**
2019	--	--	NA	--	--	~220	--	~205	--	0	--	**--**

[a] Naturally occurring island; [b] Combined number of breeding pairs for both colonies on Malheur Lake; [c] All nesting by Caspian terns in the Harney Basin was on the USACE-constructed island in Malheur Lake; NA = site not available as nesting habitat; -- = data not available

The USACE-constructed island at Fern Ridge Reservoir in the southern Willamette Valley, Oregon, was also first available as nesting habitat for Caspian terns during the 2008 nesting season. While small numbers of Caspian terns were occasionally observed flying over the island or landing on it in 2008 and subsequent years, no nesting by Caspian terns occurred at the site during 2008–2015 (*Table 3.5* above). After the 2015 nesting season all efforts to monitor the island for evidence of tern nesting ceased.

The USACE-constructed islands at Dutchy Lake and at the East Link Impoundment in Summer Lake Wildlife Area first became available as nesting habitat for Caspian terns in 2009, and a total of 15 breeding pairs of terns nested on the two islands that year, 8 breeding pairs at Dutchy Lake and 7 breeding pairs at East link Impoundment. During the 2010 and 2011 nesting seasons, breeding activity by Caspian terns at the Summer Lake Wildlife Area was limited to the island in East Link Impoundment, where ca. 30 breeding pairs attempted to nest in 2010 and only 2 breeding pairs attempted to nest in 2011. During the 2012 nesting season a total of 14 Caspian tern breeding pairs established nests on the USACE-constructed islands in Summer Lake Wildlife Area, with 10 breeding pairs attempting to nest on the East Link island and 4 breeding pairs attempting to nest on the island in Gold Dike Impoundment. During the 2013–2017 nesting seasons, ca. 20–30 Caspian tern breeding pairs nested each year on the island in East Link Impoundment, but no tern nesting attempts occurred on the island in Gold Dike Impoundment. In 2018, ca. 70 breeding pairs of Caspian terns nested on the island in East Link Impoundment, by far the greatest number of Caspian tern breeding pairs documented to have nested at the Summer Lake basin in the past 30 years. See *Table 3.5* above for a summary of Caspian tern use of the USACE-constructed islands at Summer Lake Wildlife Area for nesting.

In Lower Klamath NWR, the floating island at Sheepy Lake and the silt-core island in Orems Unit were completed before the 2010 nesting season, when ca. 260 Caspian tern breeding pairs nested on the Sheepy Lake tern island. During the following nesting season, 2011, the number of Caspian terns that attempted to nest at the Sheepy Lake tern island declined to ca. 190 breeding pairs. Two Caspian tern breeding pairs also attempted to nest at the Orems Unit island in 2011, when the island was briefly surrounded by water for the first time in May of that year. During 2012–2019, all Caspian tern nesting at Lower Klamath NWR occurred on the island in Sheepy Lake. During the 2012 nesting season, just over 200 breeding pairs of Caspian terns attempted to nest on the Sheepy Lake floating island. The size of the tern colony on the Sheepy Lake island increased to ca. 320 breeding pairs in 2013 and to ca. 520 breeding pairs in 2014. The number of terns nesting on the Sheepy Lake floating island then decreased to ca. 340 breeding pairs in 2015, and down to just ca. 95 breeding pairs in 2016. During the 2017 nesting season, ca. 250 breeding pairs of Caspian terns attempted to nest on the floating island at the Sheepy Lake. Following the 2017 nesting season, however, the condition of the floating island was severely degraded and structurally compromised and the USACE replaced it with a 1-acre (0.4-ha) rock core island in the same location prior to the 2018 nesting season (USACE 2017). During the 2018–2019 nesting seasons, between ca. 220 and 240 breeding pairs of Caspian terns attempted to nest on the rock core island at Sheepy Lake. See *Table 3.5* above for a summary of Caspian tern use of the USACE-constructed islands at Lower Klamath NWR for nesting.

The USACE-constructed tern island at Tule Lake Sump 1B in Tule Lake NWR was first available as Caspian tern nesting habitat in 2011, and in that year a small number of Caspian terns (ca. 35 breeding pairs) nested on the island. The number of Caspian tern breeding pairs that used the Tule Lake Sump 1B island increased to ca. 210 breeding pairs during the 2012 nesting season. During the 2013 and 2014 nesting seasons, the size of the tern colony on Tule Lake Sump 1B declined to ca. 80–110 breeding pairs, but colony size increased substantially in 2015 to nearly 550 breeding pairs. The size of the tern colony then dropped again to about 100 breeding pairs during the 2016 nesting season. We estimated that ca. 275 breeding pairs attempted to nest on the Tule Lake Sump 1B island in 2017, followed by a large increase in colony size to ca. 630 breeding pairs in 2018. The size of the Caspian tern colony at Tule Lake Sump 1B dropped again in 2019, down to approximately 200 breeding pairs. See *Table 3.5* above for a summary of Caspian tern use of the USACE-constructed islands at Tule Lake NWR for nesting.

The USACE-constructed tern island in Malheur Lake, Malheur NWR, Oregon, was completed prior to the 2012 nesting season. In the previous year (2011), high water levels in Malheur Lake created a natural island on private property (Singhus Ranch) just north of the northern boundary of the Refuge, and ca. 150 breeding pairs of Caspian terns nested at the site. During the 2012 nesting season, ca. 85 Caspian tern breeding pairs nested on the island at the Singhus Ranch, and ca. 230 breeding pairs established nests on the USACE-constructed island on the southern side of the lake. During the 2013 nesting season, the tern colony on the USACE-constructed island in Malheur Lake increased substantially to ca. 530 breeding pairs. The number of Caspian terns nesting on the tern island in 2014 and 2015, however, was much lower at ca. 130–150 breeding pairs. There was even less nesting activity by Caspian terns at the Malheur Lake tern island during the 2016–2017 nesting seasons, and only ca. 15–20 breeding pairs were present at the island. The number of terns nesting on the island increased dramatically in 2018, however, when ca. 420 breeding pairs nested at the colony. See *Table 3.5* above for a summary of Caspian tern use of the USACE-constructed island at Malheur NWR for nesting.

The breeding population of Caspian terns in the SONEC region nearly tripled from the decade before the initiation of island construction by the USACE compared to the 7-year period after all eight tern islands had been built as part of the *Estuary Caspian Tern Management Plan*. The average number of Caspian terns that nested in the region during 2012–2018, following the completion of the USACE-constructed islands, was about 870 breeding pairs, compared to an average of about 330 breeding pairs during 1997–2007 (Shuford and Craig 2002, Roby et al. 2003a, Shuford 2010, USFWS unpubl. data; *Figure 3.5*). Nevertheless, there was a high degree of inter-annual variability in the number of breeding pairs of Caspian terns nesting in the SONEC region, both before the initiation of island construction (range = 50–700 breeding pairs) and after all eight USACE-constructed islands were completed (range = 258–1441 breeding pairs). Also, monitoring efforts for Caspian terns in the SONEC region prior to the initiation of island construction were less frequent and less thorough. A small proportion of the Caspian tern breeding population in the SONEC region during 2008–2012 occurred at historical colony sites but beginning in 2013 all known nesting by Caspian terns in the SONEC region was at the USACE-constructed islands (*Table 3.5* above).

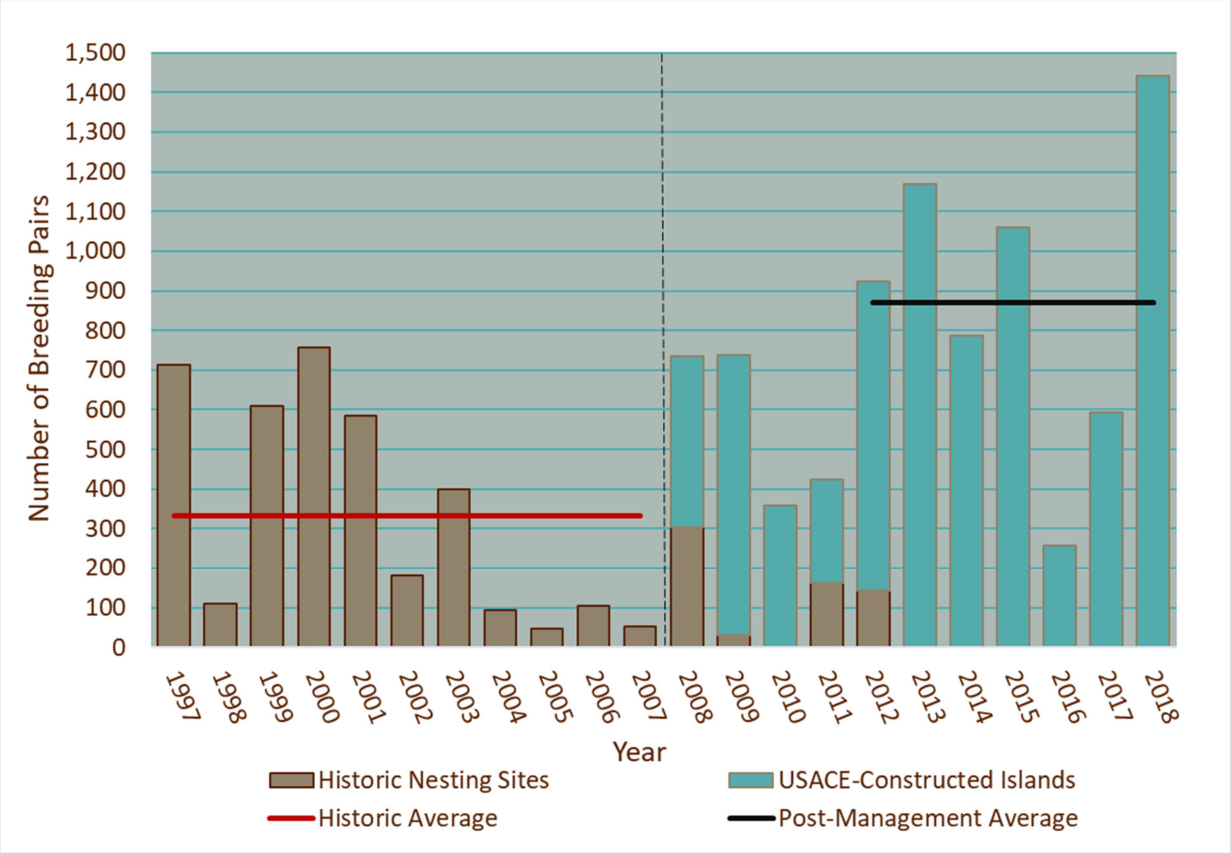

Figure 3.5. Number of Caspian tern breeding pairs at colonies in the southern Oregon/northeastern California (SONEC) region during 1997–2018. Horizontal lines represent the average number of Caspian tern nesting pairs during the pre-management period (1997–2007; red line) and the post-management period following completion of all 8 USACE-constructed tern islands (2012–2018; black line). The dashed vertical line indicates the start of tern island construction in the SONEC region as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary*. Additional pairs of Caspian terns nested at the islands enhanced by the U.S. Army Corps of Engineers in San Francisco Bay at Don Edwards National Wildlife Refuge in 2015–2019 (see below for details). Note: Estimated numbers of Caspian terns during the pre-management period are from Shuford and Craig (2002), Roby et al. (2003a), Roby et al. (2009), Shuford (2010), and USFWS unpubl. data.*

The U.S. Geological Survey-Western Ecological Research Center at Dixon Field Station, California, monitored Caspian terns nesting on five small islands in Don Edwards San Francisco Bay NWR (DENWR) during the 2015–2017 nesting seasons. These five islands were located in two separate salt ponds and had been modified and enhanced for Caspian tern nesting by the USACE. Detailed results can be found in Hartman et al. (2019) and are only briefly summarized here. During the 2015 nesting season, Hartman et al. (2019) reported a total of 245 breeding pairs nesting on islands in Pond A16 and Pond SF2 (80 breeding pairs and 165 breeding pairs, respectively). The number of Caspian terns nesting on the five islands in DENWR increased during the 2016 nesting season to a total of 434 breeding pairs (Pond A16: 169 breeding pairs; Pond SF2: 265 breeding pairs). During the 2017 nesting season, the final year of monitoring by Hartman et al. (2019), a total of 664 breeding pairs nested on two of the five tern islands enhanced by the USACE at DENWR, 227 breeding pairs in Pond A16 and 437 breeding pairs in Pond SF2). The islands were again surveyed in 2018 as part of the USFWS's *Pacific Flyway Caspian Tern Population Monitoring Strategy*, and a total of 272 breeding pairs of Caspian terns were counted, 190 pairs in Pond A16 and 282 pairs in Pond SF2 (Peck-Richardson et al. 2019). Thus, an average of ca. 450 breeding pairs of Caspian terns (range = 245–664 breeding pairs) nested at the islands enhanced by the USACE in DENWR in southern San Francisco Bay during the 2015–2018 nesting seasons (Hartman et al. 2019, Peck-Richardson et al. 2019).

3.5.1.3 Nesting success – Caspian tern nesting success (i.e. average number of young raised/breeding pair) at the Crump Lake tern island averaged 0.19 young raised/breeding pair during the 2008–2015 nesting seasons. In 2008, the tern colony produced 145 fledglings for an average nesting success of 0.34 young raised/breeding pair. During the 2009 nesting season the size of the Caspian tern breeding colony at Crump Lake tern island was substantially larger, but only ca. 60 fledglings were produced, for an average nesting success of 0.09 young raised/breeding pair. Caspian terns failed to produce any young at the Crump Lake tern island during the 2010 nesting season, but nesting success in both 2011 and 2012 was 0.43 young raised/breeding pair (ca. 15 fledglings and 50 fledglings produced, respectively). In 2013, Caspian tern nesting success at the Crump Lake tern island was 0.27 fledglings/breeding pair (61 fledglings produced); however, no fledglings were produced during either the 2014 or the 2015 nesting seasons. See *Table 3.6* for a summary of Caspian tern nesting success at the USACE-constructed island in Crump Lake.

Table 3.6. Estimated Caspian tern nesting success (average number of young raised/breeding pair) at eight different islands constructed in the southern Oregon and northeastern California (SONEC) region by the U.S. Army Corps of Engineers (USACE) as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary. Nesting success for terns nesting at the USACE-enhanced islands in Don Edwards NWR are presented in the text below.

	Estimated Nesting Success (young raised/breeding pair)							
	Warner Valley	Summer Lake Wildlife Area			Lower Klamath NWR		Tule Lake NWR	Malheur NWR
Year	Crump Lake	Dutchy Lake Impoundment	East Link Impoundment	Gold Dike	Sheepy Lake	Orems Unit	Tule Lake Sump 1B	Malheur Lake
2008	0.34	NA	NA	NA	NA	NA	NA	NA
2009	0.09	0.63	1.00	NA	NA	NA	NA	NA
2010	0.00	NA	0.10	NA	0.65	NA	NA	NA
2011	0.43	NA	0.00	NA	0.11	0.00	0.03	NA
2012	0.43	NA	0.00	0.00	0.68	NA	0.00	0.84
2013	0.27	NA	0.14	NA	0.37	NA	0.00	0.14
2014	0.00	NA	0.55	NA	0.07	NA	0.60	0.11
2015	0.00	NA	0.41	NA	--	NA	0.17	0.01
2016[a]	0.00	NA	0.00	NA	--	NA	C	--

[a] monitoring by Turnstone Environmental (results from 2016 Anadromous Fish Evaluation Program presentation); NA = site either not available as tern nesting habitat or no terns attempted to nest; C = some young confirmed to have fledged, but no estimate of nesting success available; -- = data not available

No nesting attempts by Caspian terns were detected at the USACE-constructed island at Fern Ridge Reservoir in the southern Willamette Valley of Oregon during the first eight nesting seasons after the island was built, so no fledgling terns were produced at the island.

The 2009 nesting season was the first and only season when Caspian terns nested on the floating island in Dutchy Lake at Summer Lake Wildlife Area, and in that season 8 breeding pairs of Caspian terns raised 5 fledglings, or an average of 0.63 young raised/breeding pair. In that same season, 7 breeding pairs of Caspian terns nested for the first time on a different USACE-constructed island in East Link Impoundment and raised 7 fledglings, or an average of 1.0 young raised/breeding pair. The number of terns nesting at the East Link island increased in 2010, but only 3 young were fledged, resulting in an average nesting success of 0.1 young raised/breeding pair. During the 2011 and 2012 nesting seasons, very few Caspian terns attempted to nest at the USACE-constructed islands in Summer Lake Wildlife Area and no young were successfully fledged from any of the islands. During the 2013 nesting season, 3 Caspian tern fledglings were raised at the tern colony on East Link island, or an average of 0.27 young raised/breeding pair. Nesting success during the 2014 and 2015 nesting seasons improved, however, and an average of 0.55 young raised/breeding pair (11 fledglings produced) and 0.41 young raised/breeding pair (12 fledglings produced), respectively, were raised at the East Link tern colony. During 2009–2015, Caspian terns raised an average of 0.31 young/breeding pair at the USACE-constructed island in East Link Impoundment. See *Table 3.6* above for a summary of Caspian tern nesting success at the USACE-constructed islands in Summer Lake Wildlife Area.

Caspian terns raised an average of 0.38 young/breeding pair at the Sheepy Lake floating island in Lower Klamath NWR during 2010–2014; nesting success was not estimated at this colony in 2015. During the 2010 nesting season, Caspian terns successfully fledged about 168 young from the Sheepy Lake colony, or an average of 0.65 young raised/breeding pair. In 2011, the number of fledglings produced at the Sheepy Lake colony declined to just 20, or an average of 0.11 young raised/breeding pair. In 2012, nesting success at the Sheepy Lake floating island rebounded to an average of 0.68 young raised/breeding pair (145 fledglings produced). Another 118 young were fledged from the Sheepy Lake colony in 2013, or an average of 0.37 young raised/breeding pair. Nesting success declined again in 2014, however, when only 35 young were successfully fledged from the Sheepy Lake island, or an average of 0.07 young raised/breeding pair. During the 2010–2018 nesting seasons, no Caspian tern young were fledged from the Orems Unit tern island in Lower Klamath NWR, but the island was only surrounded by water during the early part of the 2011 nesting season, and only 2 pairs of Caspian terns attempted to nest on the Orems Unit island in that year, and both failed. See *Table 3.6* above for a summary of Caspian tern nesting success at the USACE-constructed tern islands in Lower Klamath NWR.

Nesting success at the Caspian tern colony on the USACE-constructed island in Tule Lake Sump 1B averaged 0.16 young raised/breeding pair during the 2011–2015 nesting seasons. In 2011, the first year that the island in Tule Lake Sump 1B was available as nesting habitat, only one chick was successfully fledged from the colony, or an average of 0.03 young raised/breeding pair. During the 2012 and 2013 nesting seasons, Caspian terns failed to fledge any young at the island in Tule Lake Sump 1B. During the 2014 and 2015 nesting seasons, however, 65 fledglings and 90 fledglings, respectively, were produced, or an average of 0.60 young raised/breeding pair and 0.17 young raised/breeding pair, respectively. See *Table 3.6* above for a summary of Caspian tern nesting success at the USACE-constructed island in Tule Lake NWR.

Nesting success at the tern colony on the USACE-constructed island in Malheur Lake averaged 0.28 fledglings/breeding pair during the 2012–2015 nesting seasons. In 2012, the first nesting season when the new island was available for tern nesting, an estimated 195 Caspian tern fledglings were produced at the island, or an average of 0.84 young raised/breeding pair. We did not collect nesting success data at the smaller Caspian tern colony that formed on a natural island at the northern edge of Malheur Lake in 2012. During the 2013 and 2014 nesting seasons, nesting success at the Malheur Lake tern island declined to an average of just 0.14 young raised/breeding pair (76 fledglings produced) and 0.11 young raised/breeding pair (15 fledglings produced), respectively. Only one Caspian tern chick successfully fledged from the USACE-constructed island in Malheur Lake during the 2015 nesting season, or an average of 0.01 young raised/breeding pair. See *Table 3.6* above for a summary of Caspian tern nesting success at the USACE-constructed island in Malheur NWR.

The US Geological Survey-Western Ecological Research Center at Dixon Field Station, California, monitored Caspian terns nesting on five small islands in Don Edwards San Francisco Bay NWR (DENWR) during the 2015–2017 nesting seasons. Detailed information on nesting success can be found in Hartman et al. (2019) and are only briefly summarized here. Nesting success of Caspian terns at the tern islands in DENWR was not monitored after 2017. During the 2015 nesting season, the first season that the islands in DENWR were available for nesting Caspian terns, Hartman et al. (2019) reported that nesting success averaged 0.68 young raised/breeding pair on one of the two tern islands in Pond A16 and averaged 1.0 young raised/breeding pair and 0.69 young raised/breeding pair at two of the three tern islands in Pond SF2. During the 2016 nesting season, Hartman et al. (2019) reported that nesting success averaged 0.3 young raised/breeding pair at one tern island in Pond A16 and averaged 0.27 young raised/breeding pair at one of the three tern islands in Pond SF2. In 2017, Hartman et al. (2019) reported that nesting success averaged 0.47 young raised/breeding pair at one of two tern islands in Pond A16 and averaged of 0.3 young raised/breeding pair at one of three tern islands in Pond SF2.

Nesting success at the USACE-constructed islands in the SONEC region was highly variable among years and among colonies (*Table 3.5* above). Per year, Caspian terns produced an average of 0.27 fledglings/breeding pair (range of average annual nesting success values = 0.11–0.57 young raised/breeding pair) at the SONEC region colonies from 2008 to 2015. In comparison, Caspian terns nesting at the USACE-enhanced islands at DENWR in San Francisco Bay produced an average of 0.44 fledglings/breeding pair (range of average annual nesting success values = 0.28–0.69 young raised/breeding pair) during the 2015–2017 nesting seasons, the only years when nesting success was estimated (Hartman et al. 2019). A general pattern observed at nearly all USACE-constructed or enhanced tern islands was for relatively high Caspian tern nesting success in the first nesting season, followed by lower nesting success in year two. Also, Caspian tern nesting success averaged higher at the newly established tern colonies in the San Francisco Bay area compared to those in the SONEC region.

3.5.1.4 Factors limiting colony size & nesting success – A suite of biological and environmental factors may act to limit the colony size and nesting success of Caspian terns. In the SONEC region, those factors included regional drought and water allocation practices, forage fish availability, vegetation encroachment on nesting substrates, competition for nesting space with and nest predation by colony associates (i.e. gulls), and nocturnal disturbances by and nest losses to avian and mammalian predators. We found that colony size and nesting success of the Caspian tern colonies that formed at the USACE-constructed tern islands in the SONEC region were impacted by a combination of one or more of these limiting factors during 2008–2016.

Water limitations and allocation practices in the drought prone SONEC region can have a severe impact on Caspian tern colony size and nesting success for several reasons. First, drought can lower water levels in lakes and impoundments to the point where nesting islands are land-bridged and accessible to terrestrial predators. On average, during 2008–2016, around a quarter of the islands constructed by the USACE as Caspian tern nesting habitat in the SONEC region were completely or nearly land-bridged during all or part of the nesting season due to regional drought, with over half of the islands land-bridged in some years (*Table 3.7*). Second, drought can reduce or eliminate foraging habitat within short commuting distance of colony sites, thereby forcing breeding terns to commute long distances in search of food. Third, drought can have a direct, negative impact on stocks of forage fishes that are the primary prey resources for nesting Caspian terns, forcing them to disperse in search of alternative nesting opportunities. For example, at Malheur Lake, lake levels can become so low during drought years that adult common carp (*Cyprinus carpio*), an introduced and invasive species, are so concentrated in the lake that the availability of forage fish may be reduced to a level that can no longer sufficiently support tern reproduction. On average, during 2008–2016, Caspian tern nesting success at the USACE-constructed islands in the SONEC region was limited by low availability of forage fish at about half of the active Caspian tern colonies, with up to 80% of the active colonies impacted in some years (*Table 3.8*).

Beginning in 2010, vegetation encroachment on the prepared Caspian tern nesting substrates at the USACE-constructed islands in the SONEC region was a potential limiting factor to colony size at 50%–86% of active colonies (*Table 3.7*). Once invasive weeds became established on the islands, vegetation encroachment was a persistent and recurring issue at six of the eight islands (i.e. Crump Lake, Sheepy Lake, Tule Lake Sump 1B, Dutchy Lake, Gold Dike Impoundment, and East Link Impoundment). As part of social attraction efforts, we proactively reduced the potential impacts from vegetation encroachment by mechanically removing vegetation prior to the nesting season in most years and herbicides were occasionally applied by local land managers or USACE contractors to suppress vegetative regrowth at some sites. Without these annual vegetation removal efforts, substantial areas of some islands would have been overgrown and thereby rendered unsuitable as nesting habitat for Caspian terns.

Large colonies of California gulls and ring-billed gulls became established around the Caspian tern colonies at three of eight USACE-constructed islands in the SONEC region (i.e. Crump Lake, Sheepy Lake, and East Link Impoundment), and smaller gull colonies were present at two others (i.e. Tule Lake Sump 1B and Malheur Lake). Only at islands where Caspian tern colonies failed to form or persist did gull colonies also not become established (i.e. Dutchy Lake, Gold Dike Impoundment, and Orems Unit). At all sites where nesting gulls were present, some individual gulls turned to predation on Caspian tern eggs and chicks to provision their own young. In an effort to reduce predation on Caspian tern eggs and chicks by gulls, a total of 141 California gulls and 86 ring-billed gulls were lethally removed (under permit) at five of the eight USACE-constructed islands in the SONEC region during 2008–2016 (*Table 3.9*). Despite gull control measures, gull predation on Caspian tern eggs and chicks likely remained the major limiting factor to nesting success at 59% of the active tern colonies at the USACE-constructed islands in the SONEC region during 2008–2016 (*Table 3.8*).

In addition to limiting nesting success of Caspian terns through predation on eggs and chicks, California and ring-billed gulls arrived and initiated nesting earlier than Caspian terns at the islands in Sheepy Lake and East Link Impoundment, and thereby interfered with the formation of a cohesive colony by Caspian terns. As such, once the construction of all islands was completed, competition for nesting space with gulls was an apparent limiting factor to colony size for a third or more of the active Caspian tern colonies on USACE-constructed islands in the

SONEC region in most years (*Table 3.7*). Our attempts to reduce space competition between Caspian terns and gulls at the Sheepy Lake floating island by suspending netting over a designated tern nesting area and hazing gulls prior to the arrival of prospecting Caspian terns were mostly unsuccessful; gulls established breeding territories on top of the netting, which they then transferred to the ground once we removed the netting and ceased our hazing activities.

Table 3.7. Annual and average annual proportion of Caspian tern colonies on islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region, 2008–2016 where colony size was limited by water availability due to allocation or regional drought, vegetation encroachment on nesting substrates, or competition with co-nesting gull species.

	Year									
Limiting Factor	2008 (n=1)	2009 (n=3)	2010 (n=7)	2011 (n=7)	2012 (n=8)	2013 (n=7)	2014 (n=7)	2015 (n=7)	2016 (n=7)	**Average**
Water Allocation or Drought	0.00	0.00	0.43	0.29	0.13	0.14	0.57	0.57	0.29	**0.24**
Vegetation Encroachment	0.00	0.00	0.50	0.60	0.86	0.83	0.67	0.50	0.50	**0.50**
Space Competition w/ Gulls	0.00	0.33	0.50	0.33	0.29	0.33	0.33	0.33	0.33	**0.31**

Table 3.8. Annual and average annual proportion of Caspian tern colonies on islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region, 2008–2016 where nesting success (i.e. average number of young raised/breeding pair) was limited by either nocturnal avian predators, nocturnal mammalian predators, egg or chick predation by co-nesting gull species, or insufficient prey resources.

	Year									
Limiting Factor	2008 (n=1)	2009 (n=3)	2010 (n=7)	2011 (n=7)	2012 (n=8)	2013 (n=7)	2014 (n=7)	2015 (n=7)	2016 (n=7)	**Average**
Nocturnal Avian Predators	0.00	0.00	0.25	0.33	0.29	0.50	0.50	0.33	0.00	**0.24**
Nocturnal Mammalian Predators	0.00	0.00	0.00	0.00	0.14	0.17	0.33	0.50	0.17	**0.15**
Gull Predation on Eggs or Chicks	1.00	0.67	0.75	0.50	0.43	0.50	0.50	0.50	0.50	**0.59**
Low Forage Fish Availability	0.00	0.67	0.50	0.40	0.43	0.40	0.80	0.80	0.20	**0.47**

Table 3.9. Number of predators removed (under permit) from islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region, 2008–2016. Predators were removed to protect Caspian tern adults, eggs, and chicks from predation in order to help establish breeding colonies on USACE-constructed islands in the SONEC region as part of the Caspian Tern Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary.

Colony Location	Species Removed	2008	2009	2010	2011	2012	2013	2014	2015	2016[a]	**Total**
Crump Lake	California Gull	7				23	8			1	**39**
	Ring-billed Gull	3				13	4			4	**24**
East Link Impoundment	California Gull	--						4	4		**8**
	Ring-billed Gull	--						9	11		**20**
Gold Dike Impoundment	Raccoon	--	--	--	--				5		**5**
Sheepy Lake	California Gull	--	--	2	42	9	8	9			**70**
	Ring-billed Gull	--	--		3	1		1			**5**
Tule Lake Sump 1B	California Gull	--	--	--		1	1		1		**3**
	Ring-billed Gull	--	--	--		1	2	1	6	6	**16**
	Great Horned Owl	--	--	--	2		2	2	1		**7**
	Raccoon	--	--	--			1	8	6	8	**23**
Malheur Lake	California Gull	--	--	--	--	4	8	6	3		**21**
	Ring-billed Gull	--	--	--	--	5	5	3	8		**21**

[a] *Predator removal performed by Turnstone Environmental Consultants, Inc/Anchor QEA, LLC.*

Nearly all of the USACE-constructed tern islands in the SONEC region were constructed within swimming range of some mammalian predators (e.g., American mink [*Neovison vison*], raccoons) or flight range of avian predators (e.g., great horned owls) that are known to cause colony disturbances resulting in overnight, prolonged, or permanent nest site abandonment by Caspian terns. On average, nocturnal avian and mammalian predators were responsible for limiting Caspian tern nesting success at 24% and 15%, respectively, of the islands constructed by the USACE during 2008–2016 (*Table 3.8*). In some years, however, nesting success at half of the colonies was negatively affected due to disturbances by nocturnal avian and/or mammalian predators (*Table 3.8*). In order to reduce predation on Caspian tern eggs and chicks from nocturnal mammalian predators, a total of 28 raccoons were lethally removed (under permit) at two of the eight USACE-constructed islands in the SONEC region during 2008–2016 (*Table 3.9* above). Also, we live-trapped and relocated (under permit) a total of 7 great horned owls in an effort to reduce nocturnal avian predation on Caspian tern adults, eggs, and chicks at the USACE-constructed island in Tule Lake NWR during the 2011–2015 breeding seasons (*Table 3.9* above). We observed bald eagles (*Haliaeetus leucocephalus*) landing on the USACE-constructed island at Fern Ridge Reservoir in video recordings made during the 2009–2011 nesting seasons, and in one instance killing Caspian terns that were roosting in amongst tern decoys. Avian predators and the lack of a history of Caspian terns nesting in the Willamette Valley were likely the primary factors responsible for the lack of any nesting attempts by Caspian terns at the Fern Ridge Reservoir tern island.

Hartman et. al (2019) reported no nesting by gulls on the USACE-enhanced islands in DENWR where Caspian terns nested. Hartman et. al (2019) did report some disturbances by gulls to the Caspian tern nesting colonies; however, few tern eggs or chicks were apparently lost due to gull predation and this factor was unlikely to have limited colony size or nesting success of Caspian terns at those sites.

3.4.1.5 Dispersal & colony connectivity – We captured and banded adult (n = 59) and pre-fledging (n = 1,053) Caspian terns at colonies in the SONEC region from 2003 to 2015 (*Table 3.10* and *Table 3.11*) in order to monitor dispersal and colony connectivity. In addition, we banded over 7,000 Caspian terns at colonies in the Columbia River estuary, over 3,500 Caspian terns at colonies in the Columbia Plateau region, and over 1,500 Caspian terns elsewhere in the Pacific Flyway from 1997 to 2016 as part of our overall monitoring effort for the Pacific Flyway population of Caspian terns (see *Chapter 1* and *Chapter 2*). We resighted Caspian terns at the USACE-constructed islands in the SONEC region that had been banded at colonies as far north as coastal Alaska and as far south as the San Francisco Bay Area (*Table 3.12*), and we observed a subset of these banded individuals displaying nesting behaviors, such as attending nest scrapes, incubating eggs, brooding chicks, and/or attending chicks (*Table 3.13*). Conversely, we observed terns that were banded at the USACE-constructed islands as pre-fledged young at colonies located in coastal areas and in the Columbia Plateau region, in the Columbia River estuary, and at the Salton Sea in southeastern California (*Table 3.14*).

Table 3.10. Number of adult Caspian terns banded at islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region, 2009–2011. For each year, values indicate the number of Caspian terns fitted with field-readable alphanumeric leg bands on one leg and with two colored plastic bands and a metal U.S. Geological Survey (USGS) band on the other leg.

	Colony Location					
Year	Crump Lake (OR)	Summer Lake (OR)	Sheepy Lake (CA)	Tule Lake Sump 1B (CA)	Malheur Lake (OR)	**Yearly Total**
2009	30	--	--	--	--	**30**
2010	--	3	4	--	--	**7**
2011	--	--	11	11	--	**22**
Grand Total	**30**	**3**	**15**	**11**	**0**	**59**

-- Indicates no banding effort

Table 3.11. Number of Caspian terns banded as pre-fledged young at islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region, 2003–2015. For each year, values indicate the number of Caspian terns fitted with field-readable alphanumeric leg bands on one leg and with two colored leg-bands and a metal U.S. Geological Survey (USGS) band on the other leg; the number of tern chicks fitted with a metal USGS band only is presented in parentheses.

	Colony Location					
Year	Crump Lake (OR)	Summer Lake (OR)	Sheepy Lake (CA)	Tule Lake Sump 1B (CA)	Malheur Lake (OR)	**Yearly Total**
2003[a]	31 (14)	2	--	--	--	**31 (14)**
2008	132 (12)	--	--	--	--	**132 (12)**
2009	63 (20)	--	--	--	--	**63 (20)**
2010	--	--	64 (66)	--	--	**64 (66)**
2011	--	--	19	4 (2)	--	**23 (2)**
2012	37 (5)	--	128 (7)	--	180 (10)	**345 (22)**
2013	--	--	72 (35)	--	40 (16)	**112 (51)**
2014	--	--	--	--	--	**0**
2015	--	--	--	89 (5)	--	**89 (5)**
Grand Total	**263 (51)**	**2**	**283 (108)**	**100**	**246**	**859 (192)**

[a] Banding scheme consisted of three colored plastic bands on one leg and two colored plastic bands and a metal USGS band on the other leg; -- indicates no banding effort.

Table 3.12. Number of banded Caspian tern individuals resighted during each nesting season on islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region, 2008-2016. Table includes individuals resighted during multiple years at the SONEC region tern colonies, and includes terns identified from colored leg-bands and field readable alphanumeric leg-bands. Blank cells indicate no banded individuals from the region were resighted in the SONEC region colonies in that year.

	Banding Location						
Year	Coastal (AK)	Coastal (WA)	Columbia Plateau (WA)	Columbia River Estuary (OR)	SONEC (OR/CA)	SF Bay (CA)	**Yearly Total**
2008[a]			22	6	4		**32**
2009		1	28	22	5	2	**58**
2010		4	41	32	28	2	**107**
2011	1	2	91	96	58	5	**253**
2012		9	327	140	97	9	**582**
2013	1	11	323	132	107	7	**581**
2014	3	17	229	112	114	8	**483**
2015[a]		9	145	55	104	6	**319**
2016[a]		2	15	9	23		**49**

[a] Reduced resighting effort at Caspian tern colonies in the SONEC region.

Table 3.13. Number of banded Caspian tern individuals resighted and displaying nesting activities during each monitoring year at islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region by regional banding location, 2008–2016. Table includes individuals resighted while attempting to nest during multiple years at SONEC region tern colonies, and includes terns identified from colored leg-bands and field readable alphanumeric leg-bands. Blank cells indicate no banded individuals from the region were resighted at SONEC region colonies in that year.

	Banding Location					
Year	Coastal (WA)	Columbia Plateau (WA)	Columbia River Estuary (OR)	SONEC (OR/CA)	SF Bay (CA)	**Yearly Total**
2008[a]		2		1		**3**
2009		2	2			**4**
2010		3	3	4		**10**
2011		8	1	12		**21**
2012		67	24	35		**126**
2013	1	76	23	43	2	**145**
2014	1	29	16	26	1	**73**
2015[a]	4	37	22	34	4	**101**
2016[a]	1	4	3	11		**19**

[a] Reduced resighting effort at SONEC region Caspian tern colonies; Note: Nesting activities included attending a nest scrape, incubating egg(s), brooding chick(s), observed with a chick, or observed feeding a chick.

Table 3.14. Number of Caspian terns fitted with field readable alphanumeric leg-bands as pre-fledged young at islands constructed by the U.S. Army Corps of Engineers in the southern Oregon and northeastern California (SONEC) region and subsequently resighted elsewhere in the Pacific Flyway, 2011–2016. Table includes individuals resighted in multiple regions within a year and resighted during multiple years. Blank cells indicate no banded individuals from the SONEC region colonies were observed in that year.

	Regional Resighting Location within the Pacific Flyway				
Year	Coastal (WA)	Columbia Plateau (WA)	Columbia River Estuary (OR)	Salton Sea (CA)	**Yearly Total**
2011				2	**2**
2012				1	**1**
2013	--			--	
2014	--	3	2	3	**8**
2015	3	17	3	--	**23**
2016[a]	7	23	2	--	**32**

[a] Reduced resighting effort at the Caspian tern colonies located in the Pacific Flyway; Note: -- indicates no resighting effort within the region during the nesting season.

In collaboration with the U.S. Geological Survey-Western Ecological Research Center at Dixon Field Station, California, we identified 48 banded Caspian terns at the USACE-enhanced islands in DENWR during the 2015–2017 breeding seasons. Of those 48 banded Caspian terns, 29 individuals (60%) were banded in the San Francisco Bay Area, 11 individuals (23%) were banded at East Sand Island in the Columbia River estuary, 2 individuals (4%) were banded at the Goose Island colony in the Columbia Plateau region of Washington State, 2 individuals (4%) were banded at a colony on the Bellingham waterfront in coastal Washington State, and 4 individuals (8%) were banded in the SONEC region. During the three years of monitoring at the USACE-enhanced tern islands in DENWR, Dixon Field Station researchers confirmed nesting activity for 26 of the 48 banded terns. Of these 26 banded terns, 20 individuals (77%) were banded in the San Francisco Bay Area, 3 individuals (12%) were banded at East Sand Island in the Columbia River estuary, 1 individual (4%) was banded at the Goose Island colony in the Columbia Plateau region of Washington State, 1 individual (4%) was banded at a colony on the Bellingham waterfront in coastal Washington State, and 1 individual (4%) were banded in the SONEC region (Hartman et. al 2019).

Although resighting summaries of banded Caspian terns demonstrate the existence of connectivity within the network of Caspian tern colony sites in the Pacific Flyway, they do not directly provide movement rates of Caspian terns between regions of the Flyway, rates of natal and breeding philopatry, or rates of natal and breeding dispersal. To that end, Suzuki et al. (2018) developed a spatially explicit demographic model for the Pacific Flyway population of Caspian terns using the population simulation modeling software HexSim (Schumaker 2016). The demographic model was built on our database of resightings of Caspian terns banded with field-readable alphanumeric leg-bands and for which vital rates, movement probabilities between regions of the Pacific Flyway, and natal philopatry/dispersal rates were estimated using Program MARK (White and Burnham 1999). Results of this modeling indicated that there was a 0.11 probability of a “Pre-breeding” tern (i.e. sub-adult without prior breeding history), a 0.007 probability of a “Colony Attending” tern (i.e. breeding adult), and a 0.027 probability of a “Non-colony Attending” tern (i.e. non-breeding adult) moving from the Columbia River estuary to the SONEC region. In comparison, the probabilities of movement from the SONEC region to the Columbia River estuary were 0.08 for a “Pre-breeding” tern, 0.046 for a “Colony Attending” tern, and 0.155 for a “Non-colony Attending” tern. Further, the probability of a tern moving from the Columbia Plateau region to the SONEC region was 0.17 for “Pre-breeding” terns, 0.039 for “Colony Attending” terns, and 0.035 for “Non-colony Attending” terns. In comparison, the probabilities of movement from the SONEC region to the Columbia Plateau region were 0.27 for “Pre-breeding” terns, 0.116 for “Colony Attending” terns, and 0.308 for “Non-colony Attending” terns. Lastly, modeling indicated that terns fledged at the USACE-constructed islands in the SONEC region had very high rates of natal philopatry (i.e. returning to the natal region during the first visit to the breeding range) compared to terns fledged from other colonies throughout the Pacific Flyway. Caspian terns fledged from colonies in the Columbia River basin exhibited relatively high rates of natal dispersal to colonies in the SONEC region (Suzuki et al. 2018).

3.5.2 Foraging Ecology

3.5.2.1 Diet composition – Several ESA-listed fish species and other fishes of conservation concern could potentially be negatively affected by predation from Caspian terns nesting on USACE-constructed islands in the SONEC region; these include the ESA-listed shortnose sucker

(*Chasmistes brevirostris)* and Lost River sucker (*Deltistes luxatus*) in the Upper Klamath Basin of Oregon and California; the Great Basin redband trout (*Oncorhynchus mykiss gibbsi*) and the Lahontan cutthroat trout (*Oncorhynchus clarkia henshawi*) in Harney County, Oregon; and the ESA-listed Warner sucker (*Catostomus warnerensis*) in the Summer Lake basin and the Warner Valley, Oregon; and several species of lamprey (Petromyzontidae) in the Warner, Klamath, and Modoc basins. Populations of each of these species have been documented within the foraging range of Caspian terns nesting at each of the USACE-constructed islands in the SONEC region; however, there are also abundant native and introduced fish species within the region that are available as forage for Caspian terns.

We identified a total of ca. 11,800 prey items to the family- or species-level being delivered by Caspian terns to the colony in their bills at the USACE-constructed island at Crump Lake in the Warner Valley, Oregon, during 2008–2013. Based on our identification of these bill-loads, the diet of Caspian terns nesting at Crump Lake consisted of minnows (i.e. Cyprinidae), including tui chub (*Gila bicolor)*, common carp (*Cyprinus carpio*), and speckled dace (*Rhinichthys osculus)*; catfish (i.e. Ictaluridae), including *Ameiurus* and *Ictalurus* species; sunfish (i.e. Centrarchidae), including crappie (*Pomoxis* spp.), sunfish (*Lepomis* spp.), and bass (*Micropterus* spp.); salmonids (i.e. Salmonidae), including trout (*Oncorhynchus* spp.); suckers (i.e. Catostomidae); and lamprey (i.e. Petromyzontidae). Of these identified bill-loads, over 95% consisted of a combination of tui chub, crappie, and catfish in each year (*Table 3.15*).

Table 3.15. Diet composition of Caspian terns nesting on the island constructed by the U.S. Army Corps of Engineers in Crump Lake, Warner Valley, Oregon, based on identified prey in tern bill-loads (2008–2013).

	Prey Type				
Year	Tui Chub	Catfish spp.	Crappie spp.	Unidentified Minnow	Other
2008 (n = 2,915)	59.9%	22.8%	15.5%	0.0%	1.8%
2009 (n = 3,611)	81.3%	4.6%	13.4%	0.0%	0.7%
2010 (n = 455)	11.6%	9.0%	77.8%	0.0%	1.5%
2011 (n = 1,617)	26.2%	14.0%	56.5%	0.3%	3.3%
2012 (n = 1,671)	18.9%	38.7%	39.5%	0.0%	2.9%
2013 (n = 1,542)	48.4%	21.9%	25.7%	0.1%	4.0%
Average (2008–2013)	**41.0%**	**18.5%**	**38.1%**	**0.1%**	**2.4%**

We identified a total of ca. 2,200 prey items to the family- or species-level being delivered by Caspian terns to the colonies on USACE-constructed islands at the Summer Lake Wildlife Area, Oregon, during 2009–2014. Based on our identification of these bill-loads, the diet of Caspian terns nesting at the Summer Lake Wildlife Area consisted mainly of tui chub and salmonids. On average, ca. 85% of the prey items delivered to the Summer Lake tern colonies were tui chub and ca. 11% were salmonids (*Table 3.16*). Few native salmonids reside in the Summer Lake basin, but hatchery rainbow trout (*O. mykiss*) are reared and released into several rivers and small lakes within foraging distance of terns nesting on islands in the Summer Lake Wildlife

Area. We also identified catfish (*Ameiurus* spp.), centrarchids, suckers, and occasionally goldfish (*Carassius auratus*) in the bill-loads of terns nesting at the colonies in Summer Lake Wildlife Area.

Table 3.16. Diet composition of Caspian terns nesting on the islands constructed by the U.S. Army Corps of Engineers at Summer Lake Wildlife Area, Oregon, based on identified prey in tern bill-loads (2009–2014).

	Prey Type			
Year	*Gila* spp.	Salmonid spp.	Unidentified Minnow	Other
2009 (n = 828)	91.7%	6.5%	0.0%	1.8%
2010 (n = 359)	68.8%	25.9%	0.6%	4.7%
2011 (n = 258)	81.0%	17.8%	0.0%	1.2%
2012 (n = 57)	84.2%	7.0%	0.0%	8.8%
2013 (n = 130)	96.2%	3.1%	0.0%	0.8%
2014 (n = 568)	87.1%	6.0%	4.6%	2.3%
Average (2009–2014)	**84.8%**	**11.1%**	**0.9%**	**3.3%**

We identified a total of ca. 6,600 prey items being delivered by Caspian terns to the colony on the USACE-constructed island in Sheepy Lake, Lower Klamath NWR, California, during 2009–2014. On average, ca. 82% of identified bill-loads were in the minnow family (Cyprinidae) with species identified as tui chub, blue chub (*Gila coerulea*), speckled dace, or fathead minnow (*Pimephales promelas*). Of these, the two chub species accounted for an average of ca. 67% of the identified bill-loads (*Table 3.17*). Centrarchids, such as crappie, sunfish, Sacramento perch *(Archoplites interruptus*), yellow perch (*Perca flavescens*), and bass, accounted for ca. 15% of the identified bill-loads on average. Other prey species delivered to the Sheepy Lake tern colony included catfish (both *Ameiurus* and *Ictalurus* spp.), lamprey, common carp, salmonids, goldfish, and sculpin (Cottidae).

Table 3.17. Diet composition of Caspian terns nesting on the island constructed by the U.S. Army Corps of Engineers in Sheepy Lake, Lower Klamath National Wildlife Refuge, California, based on identified prey in tern bill-loads (2010–2014).

	Prey Type					
Year	*Gila* spp.	Fathead Minnow	Sunfish and Bass	Perch spp.	Unidentified Minnow	Other
2010 (n = 1,351)	67.7%	10.7%	9.1%	4.4%	3.6%	4.5%
2011 (n = 1,945)	50.1%	17.6%	11.8%	14.5%	3.1%	2.8%
2012 (n = 1,208)	72.4%	8.5%	12.8%	3.9%	0.0%	2.4%
2013 (n = 1,469)	74.6%	6.9%	2.0%	10.8%	4.4%	1.3%
2014 (n = 657)	71.2%	17.2%	3.0%	3.5%	2.3%	2.7%
Average (2010–2014)	**67.2%**	**12.2%**	**7.8%**	**7.4%**	**2.7%**	**2.7%**

We identified a total of ca. 5,500 prey items being delivered by Caspian terns to the colony at the USACE-constructed island in Tule Lake Sump 1B, Tule Lake NWR, California, during 2011–2014. On average, ca. 74% of the identified bill-loads were minnows, including tui chub, blue chub, and fathead minnow. Of these, the two chub species accounted for an average of ca. 68% of identified bill-loads (*Table 3.18*). Centrarchids, such as crappie, sunfish, Sacramento perch, yellow perch, and bass, accounted for ca. 25% of identified bill-loads on average. Other prey species delivered to the Tule Lake Sump 1B tern colony included speckled dace, catfish (*Ameiurus* spp.), lamprey, common carp, goldfish, and suckers.

Table 3.18. Diet composition of Caspian terns nesting on the island constructed by the U.S. Army Corps of Engineers in Tule Lake Sump 1B, Tule Lake National Wildlife Refuge, California, based on identified prey in tern bill-loads (2011–2014).

	Prey Type					
Year	*Gila* spp.	Perch spp.	Sunfish and Bass	Fathead Minnow	Unidentified Minnow	Other
2011 (n = 1,785)	50.6%	33.5%	7.8%	6.0%	1.0%	1.1%
2012 (n = 783)	78.7%	3.6%	7.5%	9.7%	0.0%	0.5%
2013 (n = 977)	73.9%	19.4%	2.4%	1.4%	1.8%	1.0%
2014 (n = 1,941)	68.6%	12.8%	11.8%	1.9%	3.3%	1.5%
Average (2011–2014)	**68.0%**	**17.3%**	**7.4%**	**4.7%**	**1.5%**	**1.0%**

We identified a total of ca. 7,300 prey items being delivered by Caspian terns to the colony on the USACE-constructed island in Malheur Lake, Oregon, during 2012–2015. On average, ca. 93% of identified bill-loads were in the minnow family, including tui chub and common carp. Of these, tui chub accounted for an average of ca. 38% of identified bill-loads and common carp accounted for an average of ca. 50% of identified bill-loads (*Table 3.19*). Catfish (*Ameiurus* spp.) accounted for ca. 4% of identified bill-loads on average, while salmonids, on average, made up ca. 1% of identified bill-loads. Other prey species delivered to the Malheur Lake tern colony included redside shiner (*Richardsonius balteatus*), chiselmouth (*Acrocheilus alutaceus)*, northern pikeminnow (*Ptychocheilus oregonensis*), yellow perch, suckers, and various other centrarchids.

Table 3.19. Diet composition of Caspian terns nesting on the island constructed by the U.S. Army Corps of Engineers in Malheur Lake, Malheur National Wildlife Refuge, Oregon, based on identified prey in tern bill-loads (2012–2015).

	Prey Type					
Year	Tui Chub	Common Carp	Bullhead spp.	Salmonid spp.	Unidentified Minnow	Other
2012 (n = 2,738)	54.3%	31.0%	7.3%	0.6%	3.2%	3.5%
2013 (n = 2,456)	56.4%	30.0%	7.5%	0.7%	2.2%	3.3%
2014 (n = 1,520)	27.7%	68.1%	0.7%	1.6%	0.2%	1.6%
2015 (n = 618)	14.4%	68.3%	0.5%	1.0%	14.6%	1.3%
Average (2012–2015)	**38.2%**	**49.4%**	**4.0%**	**1.0%**	**5.0%**	**2.4%**

No data on the diet composition of Caspian terns nesting on the USACE-enhanced islands at DENWR in the San Francisco Bay Area were collected during the 2015–2017 post-construction monitoring period.

3.5.2.2 Upper Klamath Basin telemetry study – Although tui chub and blue chub were the dominant prey types in the diet of Caspian terns nesting at USACE-constructed islands in the Upper Klamath Basin, there were substantial differences in the foraging behavior and time-activity budgets between terns nesting at the Tule Lake Sump 1B colony and those nesting at the Sheepy Lake colony. Terns from the Tule Lake Sump 1B colony spent a greater proportion of time at the colony site (i.e. colony attendance) than did those nesting at the Sheepy Lake colony. Conversely, GPS-tagged terns from the Sheepy Lake colony spent a greater amount of time commuting to foraging locations further from the colony than did GPS-tagged terns from the Tule Lake Sump 1B colony (*Table 3.20*). GPS-tagged Caspian terns that were breeding at the Sheepy Lake colony (n = 5) foraged within a 54-km radius of the colony, whereas those from the Tule Lake Sump 1B colony (n = 7) foraged within a 28-km radius of the colony. However, 80% of the foraging trips made by terns tagged at the Tule Lake Sump 1B colony were within 10 km of that colony. There was some overlap in foraging locations, with terns tagged at the Sheepy Lake colony choosing to forage near the Tule Lake Sump 1B colony. Kernel density estimates of the foraging locations of GPS-tagged terns from each colony are presented in *Figure 3.6*. Although travel distances to foraging areas were greater for terns tagged at the Sheepy Lake colony, the incidence of resting away from the colony was similar between terns tagged at the two colony sites during the incubation period, but during the chick-rearing period terns from the Sheepy Island colony rested at non-colony locations more than those from the Tule Lake Sump 1B colony. Regardless of colony, however, female and male terns in the study spent a similar amount of time present on the colony, with tagged individuals during the incubation period having higher rates of colony attendance than those caring for chicks. Lastly, colony attendance by GPS-tagged terns dropped off substantially following failed nesting attempts (i.e. loss of eggs or chicks) and the amount of time spent resting and/or foraging at locations away from the colony increased regardless of their respective colony. See Patterson (2012) for detailed results.

Table 3.20. Summary of central-place foraging trips of Caspian terns breeding at the islands constructed by the U.S. Army Corps of Engineers at Sheepy Lake and Tule Lake in the Upper Klamath Basin, California. Values for trip duration, foraging time, commuting time, and maximum distance from colony are medians (± 95% confidence intervals) as estimated by mixed-effects models. (From Patterson [2012], with permission.)

	Sheepy Lake	Tule Lake Sump 1B
Number of trips	8	35
Proportion of trips with commuting	1.00	0.69
Proportion of trips with resting	0.63	0.09
Duration of trip (min)	186 (105 – 329)	55 (42 – 72)
Commuting time (min)	48 (18 – 122)	5 (3 – 10)
Foraging time (min)	72 (39 – 134)	39 (29 – 52)
Maximum distance (km)	27 (15 – 47)	6 (4 – 8)

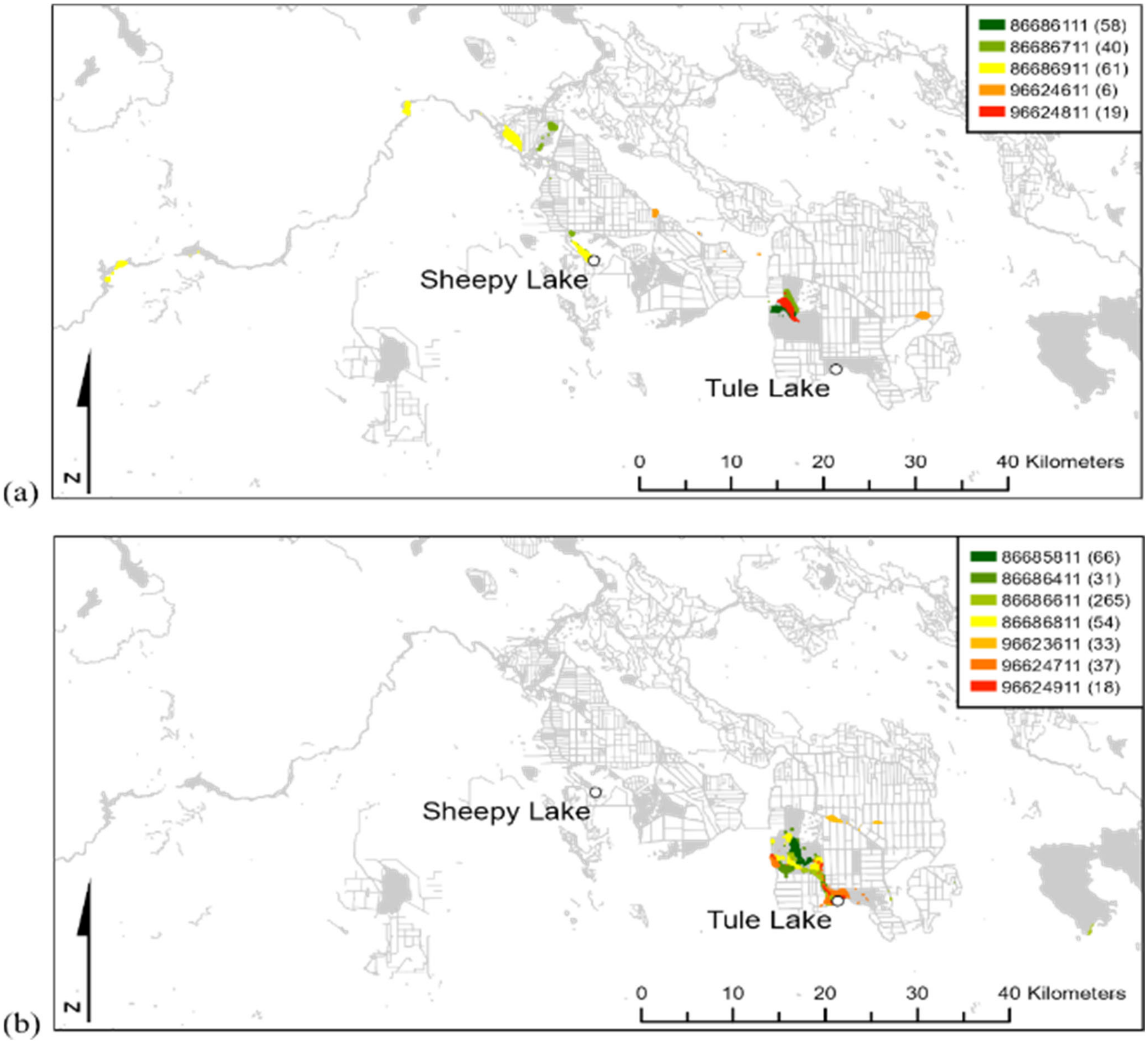

Figure 3.6. Foraging areas of Caspian terns breeding at colonies on islands constructed by the U.S. Army Corps of Engineers in (a) Sheepy Lake and (b) Tule Lake in the Upper Klamath Basin of Oregon and California. Foraging areas are based on the 50% utilization distributions of all foraging locations for each individual tern, estimated using the biased-random bridge approach. Legend indicates tern identity, and the number of foraging locations used to estimate foraging area is in parentheses. Adapted from Patterson (2012).

3.5.2.3 Consumption of forage fish of conservation concern – We found limited evidence of mortality for forage fish species of conservation concern from predation by Caspian terns nesting at the USACE-constructed islands in the SONEC region. We recovered one passive integrated transponder (PIT) tag from a juvenile Warner sucker tagged in 2008 on the Crump Lake tern island following the 2008 nesting season (0.8% of the 130 Warner suckers tagged in that year; Scheerer et al. 2008). Also, there was a very low percentage of suckers in tern bill-loads (< 0.1% of identified bill-loads) at the Caspian tern colony on Crump Lake island during the

2008–2013 nesting seasons. In addition, we detected a single PIT tag from a juvenile sucker at a mixed-species loafing area outside of the Caspian tern nesting area on the USACE-constructed island in Sheepy Lake following the 2013 nesting season, which represented 0.2% of the juvenile suckers PIT-tagged at Upper Klamath Lake during 2009–2012 (Evans et al. 2016). We did not detect any other PIT tags from suckers at USACE-constructed tern islands in the Upper Klamath Basin following the 2010–2014 breeding seasons. We did, however, observe a very small number of juvenile suckers in Caspian tern bill-loads at the USACE-constructed islands in the Upper Klamath Basin (< 0.1% of identified bill-loads), indicating that Caspian terns nesting at those sites did occasionally consume suckers during those years. Because the suckers observed in tern bill-load were juveniles, it was not determined whether they were ESA-listed or non-listed suckers (i.e. Klamath largescale suckers [*Catostomus snyderi*]); juvenile suckers in this region cannot be identified to the level of species based on morphology (Markle et al. 2005). When combined, the small number of PIT tags recovered, and the limited number of suckers identified in tern bill-loads suggest that Caspian terns nesting at USACE-constructed islands in the SONEC region rarely consumed Warner suckers at Crump Lake or shortnose and Lost River suckers in the Upper Klamath Basin.

We were unable to visually distinguish between wild redband trout and hatchery-raised rainbow trout in Caspian tern bill-loads at the USACE-constructed tern island at Malheur Lake, but we did detect PIT tags from wild redband trout from the Donner und Blitzen River on the Malheur Lake tern island. Concurrent with the Caspian tern breeding seasons, ODFW tagged redband trout in the Donner und Blitzen River system during 2013–2015, tagging ca. 3,100–3,600 juvenile redband trout within the size classes available to foraging Caspian terns (i.e. fork length measurement of 6.5–30 cm) each year (total = 10,325 PIT tags: ODFW data). Following the 2013 breeding season, we recovered 15 PIT tags from redband trout on the USACE-constructed tern island at Malheur Lake, or 0.5% of the redband trout PIT-tagged in that year. We recovered additional PIT tags from redband trout following the 2014 (n = 53 tags), and 2015 (n = 14 tags) breeding seasons. The exact number of PIT-tagged redband trout available in the system as prey for Caspian terns nesting on the USACE-constructed Island in Malheur Lake was unknown in 2014 and 2015, however, because annual survival rates were not determined and the estimated average growth rate of 3.2 cm/year for redband trout in the Malheur Lake river system (Meeuwig and Ramirez 2018) would have resulted in some tagged trout exceeding the maximum size limit for Caspian tern prey. Nevertheless, the proportion of recovered redband trout tags at the Malheur Lake tern colony following the 2014–2015 breeding seasons was likely very low. In 2013 and 2015, tags were recovered from an area of the island occupied solely by Caspian terns during the breeding season, but in 2014 redband trout tags were found on parts of the island occupied by multiple piscivorous waterbird species during the breeding season, including Caspian terns, American white pelicans (*Pelecanus erythrorhynchos)*, double-crested cormorants (*Phalacrocorax auritus*), and several species of gull. The relatively small numbers of redband trout tags, coupled with the low percentage of salmonids in the diet of Caspian terns nesting on the USACE-constructed island in Malheur Lake, suggest that Caspian tern predation did not substantially impact redband trout survival during the 2013–2015 breeding seasons.

3.6 Discussion

During 2008–2016, we monitored and evaluated the response of Caspian terns to the enhancement of nesting habitat in the SONEC region and in the Willamette Valley of Oregon as part of the *Estuary Caspian Tern Management Plan* (USACE 2006, USACE 2015a). That plan called for construction of alternative colony sites for Caspian terns outside of the Columbia Basin concurrent with reductions in tern nesting habitat at East Sand Island in an effort to reduce tern predation on ESA-listed salmonid smolts in the Columbia River estuary through the redistribution of approximately 60% of the Caspian terns nesting at the East Sand Island colony (USFWS 2005, USFWS 2006, USACE 2006, USACE 2015a, USACE 2015b). To this end, the USACE-Portland District constructed eight islands with a final total area of 5.35 acres (2.17 ha) as effective alternative nesting habitat for Caspian terns in the SONEC region (USACE 2017; Memorandum for the Record: Caspian Tern Nesting Habitat), a region where the historical breeding population of Caspian terns had declined. The USACE-Portland District also constructed a 1.0-acre (0.4-ha) island in the Willamette Valley of western Oregon, a region with no history of Caspian tern nesting. In addition, the USACE–Walla Walla District, in collaboration the U.S. Fish and Wildlife Service (USFWS), modified and improved five small soil mound islands in two separate salt ponds in Don Edwards San Francisco Bay National Wildlife Refuge (DENWR) as part of the *IAPMP* (USACE 2014). The goal of the *IAPMP* was to reduce predation on ESA-listed juvenile salmonids by Caspian terns nesting at colonies in the Columbia Plateau region, namely colonies on Crescent Island in the mid-Columbia River and on Goose Island in Potholes Reservoir, the two largest Caspian tern colonies in the region (Adkins et al. 2014). The *IAPMP* also called for the enhancement or construction of alternative Caspian tern nesting habitat in the San Francisco Bay Area, where the historical breeding population of Caspian terns had declined and suitable nesting habitat for terns appeared to be limiting (Strong et al. 2004, Collis et al. 2012, Adrean et al. 2012), in compensation for nesting habitat reductions in the Columbia Plateau region. Towards this goal, the USACE-Walla Walla District enhanced 1.95 acres (0.79 ha) of tern nesting habitat on five islands distributed between two reclaimed salt ponds at DENWR in southern San Francisco Bay (Hartman et. al 2016). During the 2015–2017 nesting seasons, the USGS-Western Ecological Research Center at Dixon Field Station (USGS-Dixon) monitored and evaluated the response of Caspian terns to the provisioning of this new nesting habitat at DENWR (Hartman et al. 2016, 2019).

3.6.1 Management Action Effectiveness

The efficacy of the parts of the *Estuary Caspian Tern Management Plan* and the *IAPMP* that sought to provide alternative nesting habitat for Caspian terns outside the Columbia River basin was dependent on attracting Caspian terns to nest at the newly created or enhanced colony sites outside of the basin, and then maintaining those islands as nesting habitat for Caspian terns such that they were predator-free, and the nesting substrate remained open and sparsely vegetated. To that end, a comprehensive restoration program that included a combination of habitat enhancement (island building and maintenance), social attraction (decoys and acoustic playback of vocalizations), intensive colony monitoring, and predator control (if necessary; Kress 1983, 2000; Kress and Hall 2002) was implemented to maximize the likelihood of successfully establishing perennial nesting colonies of Caspian terns on the USACE-constructed islands.

The USACE built islands as Caspian tern nesting habitat in the SONEC region starting in 2008 completed that component of the management plan just before the 2012 nesting season. Afterward, during 2013–2018, all Caspian tern nesting colonies in the SONEC region were located on USACE-constructed islands and the average number of Caspian tern breeding pairs that nested in the region nearly tripled compared to the estimated average during 1997–2007. The rapid colonization of the USACE-constructed islands by Caspian terns, and the subsequent increase in the number of breeding pairs in the SONEC region, suggests that availability of nesting habitat was a limiting factor for the population of Caspian terns nesting in this portion of the Pacific Flyway prior to the implementation of the *Estuary Caspian Tern Management Plan*. The USACE-constructed islands in the SONEC region attracted immigrants from managed Caspian tern colonies in the Columbia River basin, as well as elsewhere in the Pacific Flyway. Population modeling indicated, however, that Caspian terns banded as fledglings in the SONEC region generally had a higher probability of moving to a colony in the Columbia River basin than the reverse. Nonetheless, in 2018, a year when the number Caspian tern breeding pairs in the Columbia River basin was much reduced by management actions (see *Chapter 1* and *Chapter 2*), the USACE-constructed islands in the SONEC region supported a total of ca. 1,450 breeding pairs of Caspian terns, the largest number of breeding pairs ever recorded in the region during a single year (Shuford and Craig 2002, USFWS 2005). If the nesting habitat for Caspian terns that has been provided on USACE-constructed islands was eliminated, only an estimated average of 200 Caspian tern breeding pairs would nest in the SONEC region (Suzuki et al. 2018). This regional population estimate is somewhat lower than that observed during 1997–2007 due to further losses of historical colony sites caused by prolonged drought conditions in the region.

Following efforts to enhance Caspian tern nesting habitat at DENWR in the San Francisco Bay Area during 2015 as part of the *IAPMP*, Caspian terns began nesting on small islands in both of the two reclaimed salt ponds where tern nesting habitat was enhanced (i.e. Pond A16 and Pond SF2). Caspian tern nesting colonies formed on just three of the five islands where social attraction was deployed in 2015, however, and then in 2016 and 2017 there were Caspian tern breeding colonies on just one island in each pond (Hartman et al. 2019). In 2015, ca. 1,010 breeding pairs of Caspian terns nested in the San Francisco Bay Area (Peterson et al. 2017), and of this breeding population approximately 25% nested at the USACE-enhanced islands in DENWR the first year that they were available as nesting habitat (Hartman et al. 2019). During the 2016–2019 breeding seasons Caspian terns continued to nest at the USACE-enhanced islands in Pond SF2 and Pond A16 at DENWR (Rachel Tertes, USFWS, pers. comm.). During the 2018 nesting season there was an estimated 1,050 breeding pairs of Caspian terns nesting at colonies in the San Francisco Bay Area, and despite the absence of social attraction materials on the USACE-enhanced islands in DENWR, ca. 45% of the breeding Caspian terns in that region nested on those islands (Peck-Richardson et al. 2019). Resighting of banded terns suggested that most of the Caspian terns immigrating to the USACE-enhanced islands at DENWR came from other colonies in the San Francisco Bay Area, with limited recruitment of terns from the Columbia River basin. In addition, results from a 5-year satellite tracking study of Caspian terns displaced from colonies in the Columbia Plateau region that was initiated in 2014 did not indicate that Caspian terns moved in large numbers from those managed colonies to the USACE-enhanced islands at DENWR (Roby et al. 2019, see *Chapter 2*).

Of the nine islands constructed as alternative Caspian tern nesting habitat outside of the Columbia River estuary as part of the *Estuary Caspian Tern Management Plan,* the Fern Ridge Reservoir site in the Willamette Valley, Oregon, was the only island where a tern nesting colony did not form. We deployed social attraction materials and recorded a small number of Caspian terns landing on the island, but no nesting activity (i.e. egg-laying) was detected at the site. Unlike other sites selected for construction of islands as alternative nesting habitat for Caspian terns as part of the *Estuary Caspian Tern Management Plan*, Fern Ridge Reservoir and the rest of the Willamette Valley had no prior history of a Caspian tern nesting colony, but terns had been recorded foraging at the reservoir and passing through during spring and fall migration (Seto et al. 2003). Similarly, other piscivorous colonial waterbirds (i.e. gulls, double-crested cormorants, and American white pelicans) can be found at Fern Ridge Reservoir at various times throughout the year, but do not nest at the reservoir (ODFW 2009). Given this lack of a previous nesting history by Caspian terns and other piscivorous colonial waterbirds at Fern Ridge Reservoir, management agencies recognized that social attraction might be required at the site for an extended period of time before Caspian terns would initiate nesting, but the site was selected in large part due to the low potential for conflicts between nesting Caspian terns and fish species of conservation concern (USFWS 2005). No nesting by Caspian terns at the USACE-constructed island at Fern Ridge Reservoir, however, was detected during the first 12 nesting seasons (2008–2019) after the island was built.

The *Estuary Caspian Tern Management Plan* required that nesting habitat be created outside of the Columbia River estuary at a ratio of 2:1 (area:area) as nesting habitat was reduced on East Sand Island, resulting in the objective of creating ca. 8.0 acres (3.24 ha) of islands suitable as nesting habitat for Caspian terns. Each constructed colony site was expected to support ca. 350–2,000 breeding pairs, or a total of about 5,400 breeding pairs (USACE 2006). Nesting islands that are land-bridged or surrounded by only shallow water, inundated or over washed by water, and/or covered in dense vegetation, however, do not provide suitable nesting habitat for Caspian terns. As such, although a sufficient area of island habitat was built during implementation of the management plan, the plan fell short of providing suitable nesting habitat for Caspian terns at a 2:1 ratio during most years following the initiation of habitat reductions at East Sand Island. First, the tern island that was built on Fern Ridge Reservoir 2008 never supported nesting by Caspian terns, yet despite this the 1.0-acre (0.4-ha) island continued to be included as part of the total area of nesting habitat provided for Caspian terns as compensation for lost nesting habitat at East Sand Island. Second, the efficacy of management to establish persistent and successful nesting colonies in the SONEC region as alternative nesting habitat for terns displaced from colonies in the Columbia River basin was necessarily dependent on water resources in this arid, high desert region. But due to the persistent over-allocation of water resources in the region and the increasing frequency and intensity of drought with climate change, the waterbodies surrounding and proximate to the USACE-constructed tern islands are less and less reliable. In hindsight, the frequency at which the constructed islands would become land bridged and nearby foraging habitat would dry out due to water shortages and/or allocation priorities was underestimated. Although the area of Caspian tern nesting habitat constructed in the Upper Klamath Basin was adjusted in recognition of periodic water shortages that would impact the suitability of some islands as tern nesting habitat in some years, a similar approach was not applied to other USACE-constructed islands in the SONEC region. Consequently, the area of suitable alternative nesting habitat that was provided for Caspian terns outside of the Columbia River estuary in compensation for the area of habitat reduction on East Sand Island did not consistently meet the objective of a 2:1 ratio until the islands at DENWR were modified and enhanced in 2015. The USACE-constructed islands in DENWR were also intended to provide alternative nesting habitat for ca. 600 pairs of Caspian terns displaced from managed colonies on the Columbia Plateau region as part of the *IAPMP*.

Social attraction techniques had previously been shown to be effective at enticing Caspian terns to initiate nesting at new or abandoned colony sites (Kress 1983, Roby et al. 2002, Collis et al. 2002). Decoys and audio playback of vocalizations provide the appearance of nesting conspecifics and may indicate to prospecting terns that the site is safe from predators. The presence of other colonial waterbird species that frequently serve as colony associates of Caspian terns (i.e. gulls; Quinn and Sirdevan 1998, Cuthbert and Wires 1999) can also be indicators of suitable nesting habitat for Caspian terns. Social attraction techniques also proved to be a highly effective method for establishing Caspian tern breeding colonies on newly constructed islands in the SONEC region and the San Francisco Bay Area. In the SONEC region, Caspian terns attempted to nest on all eight USACE-constructed islands during the first breeding season when each island was surrounded by water and social attraction was deployed. The majority of tern nesting activity, however, was concentrated at just four of the eight artificial islands in the SONEC region during the study period. Nesting colonies of Caspian terns also formed on several of the islands in DENWR during the first nesting season following island enhancement and deployment of social attraction. Our findings indicated, however, that the 8.3 acres (3.36-ha) of effective alternative nesting habitat constructed by the USACE for Caspian terns in the Willamette Valley, SONEC region, and San Francisco Bay Area were only able to attract and reliably support a fraction of the ca. 6,000 breeding pairs that were displaced from the East Sand Island colony and the two largest colonies for the species in the Columbia Plateau region, as part of the *Estuary Caspian Tern Management Plan* and the *IAPMP*. Nevertheless, construction of additional nesting habitat in the SONEC region and in San Francisco Bay was an important step towards restoring Caspian terns in two areas where breeding populations and nesting habitat had declined dramatically from historical levels.

3.6.2 Limiting Factors for Colony Size & Nesting Success

Predation and disturbance from avian and mammalian predators can be major limiting factors for Caspian tern colony size and nesting success (i.e. average number of young raised/breeding pair; Collar 2017, Schniedermeyer 2018, Bailey 2018). A primary selective force for Caspian terns using islands as nesting sites is security from predation by terrestrial mammals, which are proficient predators on the eggs and young of ground-nesting birds (Martin 1993). Terrestrial mammalian predators (e.g., foxes [*Vulpes* spp.], raccoons, coyotes [*Canis latrans*], mink) that access or are introduced to islands supporting breeding colonies of ground-nesting waterbirds generally lead to colony failure and dispersal of breeding adults to alternative colony sites (Erwin et al. 2001, Shuford and Craig 2002, Antolos et al. 2005, Adkins et al. 2014). Predatory birds (e.g., owls, eagles, hawks, falcons) are also known to cause nesting colony failure and subsequent colony abandonment by terns (Nesbit 1975, Collar et al. 2017). At Caspian tern colonies in the SONEC region of western North America, we observed predation on tern eggs and chicks by California gulls, ring-billed gulls, great horned owls, black-crowned night-herons (*Nycticorax nycticorax*), and raccoons. Coyotes were observed near Crump Lake and Malheur Lake tern islands during low water years, but we did not document any predation on Caspian tern eggs or chicks by coyotes at the USACE-constructed islands in the SONEC region. We attempted to reduce raccoon predation at the USACE-constructed islands in Tule Lake Sump 1B, Tule Lake NWR, and in Gold Dike Impoundment, Summer Lake Wildlife Area using live-trapping methods, but we found live-trapping of raccoons at those sites to be ineffective and were only able to reliably remove raccoons from those islands using lethal control methods (i.e. firearms). Conversely, once a great horned owl was detected regularly visiting a tern island at night, we were usually able to live-trap and relocate it away from the colony within a few days. Actively removing predators to protect ground-nesting waterbirds during the breeding season can promote nesting success in the short term but requires repeated efforts (annual) to remove most or all nest predators to sustain a successful colony (Côté and Sutherland 1997). Removing all predators in the area of each USACE-constructed island that are capable of accessing the island is neither a desirable, practical, or ethical goal. Consequently, continued monitoring of Caspian tern colonies on USACE-constructed islands in the SONEC region and targeted removal of mammalian and avian predators that are significantly impacting colony size and nesting success will remain crucial for ensuring USACE-constructed islands continue to support Caspian tern breeding colonies.

Nighttime disturbances by great horned owls and raccoons were responsible for overnight abandonment of the Caspian tern breeding colony at the tern island in Tule Lake Sump 1B during multiple breeding seasons. Nocturnal abandonment, when combined with low nighttime air temperatures and precipitation, resulted in substantial tern chick mortality due to exposure, in addition to direct mortality due to predation. In years when numerous raccoons and/or great horned owls were removed from the Tule Lake Sump 1B island, Caspian terns were able to raise appreciable numbers of young at that colony. At the tern island in Gold Dike Impoundment, Summer Lake Wildlife Area, we documented very little use by Caspian terns and other waterbirds during each year when the island was monitored, whereas the tern island in East Link Impoundment, Summer Lake Wildlife Area regularly supported about 20 breeding pairs of Caspian terns and a few hundred breeding pairs of California and ring-billed gulls. We were not permitted to trap and remove the great horned owls that were observed visiting either of these USACE-constructed tern islands in Summer Lake Wildlife Area, but we did remove raccoons from the Gold Dike tern island during the 2015 breeding season. Despite nocturnal disturbances from great horned owls, Caspian terns nesting on the East Link tern island were able to raise some young in most years. It is possible that the lack of waterbird use of the tern island in Gold Dike Impoundment and the consistently small number of Caspian tern breeding pairs at the tern island in East Link Impoundment are not the result of nocturnal predators, but rather the result of limited carrying capacity to support piscivorous waterbirds due to limited foraging habitat in the Summer Lake valley and a limited prey base where foraging habitat does exist.

Caspian tern colonies in the SONEC region, including those on USACE-constructed islands, are usually associated with breeding colonies of gulls, and frequently the gull colonies are far larger than the associated Caspian tern colony. Nesting gulls may indicate nest site suitability for Caspian terns, but kleptoparasitism (i.e. theft of tern bill-load fish) and predation of tern eggs and chicks by gulls have been identified as limiting factors for nesting success at a number of Caspian tern breeding colonies (Antolos et al. 2006, Adkins et al. 2014, Collar et al. 2017, Bailey 2018). We found that gull predation was a factor limiting both the size and nesting success of some Caspian tern colonies in some years at USACE-constructed islands in the SONEC region. Because selectively culling gulls that depredate tern nests can be an effective method of enhancing nesting success at some tern colonies and can help with establishing new tern colonies (Kress 2000, Guillemette and Brousseau 2001, Kress and Hall 2002, Roby et al. 2002), we employed targeted removal of predatory gulls as needed. Gulls nesting at the USACE-constructed islands in Sheepy Lake and Crump Lake were the most likely to resort to predation on the nest contents of Caspian terns, and we culled a relatively large number of gulls at those two colony sites compared to the others. Our gull removal efforts were reactionary and aimed at removing gulls that habitually targeted tern eggs or chicks as prey, which typically followed the loss of several tern chicks or eggs to a predatory gull before that individual was culled. At the Sheepy Lake colony, in addition to culling predatory individuals, we attempted to proactively reduce overlap in space use between the tern colony and the gull colony using netting and gull dissuasion techniques to protect nesting terns from the negative effects of being surrounded by nesting gulls.

The *Estuary Caspian Tern Management Plan* acknowledged that periodic drought would likely impact the availability of both nesting habitat and prey fish populations for Caspian terns nesting in the SONEC region (USFWS 2005). During the last two decades, however, the region has experienced one of the most severe and sustained drought periods on record (Williams et al. 2020). Although the waterbodies in the SONEC region that were selected by the USACE for construction of tern islands were more insulated from the effects of drought than some historical nesting sites, we observed substantial reductions in the extent and level of water around most of the USACE-constructed islands resulting in some islands becoming accessible to terrestrial predators (i.e. Crump Lake, Gold Dike Impoundment, Sheepy Lake, Orems Unit, and Malheur Lake). These dry lakebed conditions not only provided increased access to tern colonies by land-based predators, but also reduced or eliminated the aquatic habitat of forage fish and forced resident fish to retreat to habitat refugia (Scheerer et al. 2010). As central-place foragers, nesting Caspian terns also experienced reduced foraging prospects within commuting distance of their nest site. Above-average snowpack during the winter of 2010–2011 provided some temporary drought relief to the SONEC region, but by 2014 approximately half of the waterbodies within foraging distance of tern colonies on USACE-constructed islands, waterbodies where we had previously observed foraging by Caspian terns, were dry (BRNW, unpubl. data).

Insufficient runoff from snowpack in the Warner Valley basin during the winters proceeding the 2013–2015 nesting seasons resulted in the USACE-constructed island at Crump Lake being land-bridged for the latter part of the 2013 nesting season, completely land-bridged during all of the 2014 nesting season, and land-bridged for most of the 2015 nesting season. In 2013, Caspian terns did successfully raise a few young at the Crump Lake island, and despite access to the island by land-based predators and reduced forage fish availability during the 2014 and 2015 breeding seasons, a small number of Caspian terns attempted to nest on the island but were unsuccessful. During the 2014 nesting season, there was also a high level of motorized human activity on the dry lakebed around Crump Lake island, which may have caused disturbance to prospecting Caspian terns at the site. In 2016, lake levels in the Warner Valley somewhat rebounded and the USACE-constructed island in Crump Lake was again surrounded by water and a few Caspian terns again attempted to nest at the site. However, forage fish populations apparently had not recovered sufficiently to support the colony in 2016, and the terns again failed to successfully raise any young.

We did occasionally observe disturbances by avian predators to Caspian terns nesting at the Malheur Lake tern island, but those impacts appeared negligible and limits to colony size and nesting success at that site appeared to be mainly due to drought and associated effects on the availability and quality of forage fish. During 2012 and 2013, years when water levels were relatively high at Malheur Lake, approximately half of Caspian tern bill-loads identified at the Malheur Lake tern colony were tui chub, similar to other tern colonies in the SONEC region. As lake levels declined, however, the diet switched to predominately juvenile common carp. Further, as lake levels around the Malheur Lake tern island decreased, so did nesting success at the tern colony. Common carp have been identified as having the greatest negative impact on the health of the aquatic ecosystem at Malheur Lake, but unfortunately, due to the life history and fecundity of common carp, predation by Caspian terns is not likely to contribute significantly to control of carp populations in Malheur Lake. The USFWS has, however, implemented an ongoing program to remove large numbers of adult common carp from Malheur Lake in order to address the issues of poor water quality and limited submergent and emergent vegetation in the lake due to the abundance of carp (USFWS 2012). Efforts to reduce densities of adult common carp in Malheur Lake will have long-term benefits for native fish populations, waterfowl, and nesting piscivorous waterbirds, including Caspian terns (Ivey et al. 1998). By enhancing the size of native fish populations in Malheur Lake through removal of invasive carp, the size and nesting success of the Caspian tern breeding colony at the Malheur Lake tern island will likely improve.

Predation and disturbance from avian and mammalian predators was reportedly low at the Caspian tern colonies on USACE-enhanced island in DENWR during the 2015–2017 breeding seasons, as were the number of nesting and loafing gulls present at those colonies (Hartman et al. 2019). Consequently, reported average nesting success at the DENWR colonies was similar to that observed at other monitored Caspian tern colonies in the Pacific Flyway during seasons with moderate to good nesting success (Antolos et al. 2004); the exception was the East Sand Island colony during 1999–2004, where average nesting success was roughly twice that observed at the DENWR colonies. Although the combined Caspian tern colonies on the USACE-constructed islands in Pond A16 and Pond SF2 supported nearly half of the breeding population of Caspian terns in the San Francisco Bay Area in some years, only two of the five small islands enhanced by the USACE at DENWR were used as nesting habitat by Caspian terns. Further, at the islands that were used, only a small proportion of the total surface area was occupied by nesting Caspian terns. As such, the available acreage of nesting habitat for Caspian terns at DENWR was not likely a limiting factor for the size of breeding colonies at those sites. Rather, the size of the Caspian tern colonies on artificial islands in DENWR may have been limited by the size of the local pool of potential immigrants from other colonies the San Francisco Bay Area and an apparent low immigration rate by terns displaced from managed colonies in the Columbia Basin.

3.6.3 Foraging Ecology

The results from a study in 2011 using GPS telemetry indicated that the foraging areas used by Caspian terns nesting at the Sheepy Lake and Tule Lake Sump 1B colonies differed greatly despite their close proximity in the Upper Klamath Basin. Caspian terns are central-place foragers, so the distance between the two colonies may partially explain the differences in foraging behavior documented at the two colonies; however, limited prey availability in the region during the 2011 nesting season may have also contributed to the differences. The Caspian tern colonies at Sheepy Lake and Tule Lake Sump 1B were relatively small in 2011 (188 breeding pairs and 34 breeding pairs, respectively) and competition for prey should have been low if prey fish were abundant nearby. Yet, despite some overlap in foraging areas about 10 km north of the Tule Lake Sump 1B colony, the GPS-tagged terns breeding at the two colonies appeared to segregate their foraging locations and thereby partition spatially the available prey resources. Terns from the Sheepy Lake colony made foraging trips to relatively distant locations, and the increased foraging effort (i.e. longer trip duration, reduced colony attendance, increased flight distances) by terns nesting at Sheepy Lake was associated with lower body mass in adults and lower size-adjusted body mass of pre-fledged chick at that colony. Nevertheless, the increased foraging effort of terns nesting at Sheepy Lake was not reflected in lower nesting success (i.e. average number of young raised/breeding pair) compared to terns nesting at Tule lake Sump 1B. Any potential benefits to the nesting success of terns at the Tule Lake Sump 1B colony from lower foraging effort were apparently offset by nest losses due to colony disturbances and predation from gulls, raccoons, and great horned owls.

Long-distance foraging trips, similar to those documented for GPS-tagged terns from the Sheepy Lake colony, have also been associated with limitations in forage fish availability at other Caspian tern colonies within the Pacific Flyway (Lyons et al. 2005, Anderson et al. 2007, Adrean et al. 2012). Water availability, and consequently the availability and distribution of fish prey for Caspian terns, can exhibit high inter-annual variability in the SONEC region. As such, the foraging conditions experienced by Caspian terns in the Upper Klamath Basin during 2011 only represents a small portion of the range of potential foraging conditions in the region. Nevertheless, the study by Patterson (2012) highlighted the importance of providing predator-free Caspian tern nesting habitat with access to a reliable prey base for the success of efforts at establishing and sustaining nesting colonies of Caspian terns in the SONEC region.

Based on observations of bill-load prey deliveries to the colonies, we found that the diet of Caspian terns nesting on USACE-constructed islands in the SONEC region consisted mainly of native and introduced species of minnows (Cyprinidae). Further, PIT tag recovery efforts indicated that there were minimal predation impacts to ESA-listed fish species and other fishes of conservation concern by Caspian terns nesting at the USACE-constructed islands in the SONEC region. As such, conflicts over tern predation and protected fishes were not transferred from the Columbia River basin to the SONEC region by the effort to attract terns to alternative colony sites.

3.6.4 Adaptive Management

The management agencies used an adaptive management approach to facilitate success in achieving the objective of the *Estuary Caspian Tern Management Plan* to provide alternative nesting habitat for terns displaced from the Columbia River estuary. These adaptive management efforts fell into three broad categories: (1) monitoring USACE-constructed islands in the SONEC region, the Willamette Valley, and the San Francisco Bay Area for at least three breeding seasons after island completion to identify and mitigate factors limiting the establishment, size, and nesting success of any Caspian tern colonies that formed; (2) implementing predator control at Caspian tern colonies as needed to aid in colony formation and persistence; and (3) monitoring the USACE-constructed islands themselves for ways to enhance these structures as secure nesting habitat for Caspian terns.

Monitoring is a key component of the adaptive management approach (Stankey et al. 2005). During the 2008–2016 breeding seasons, the USACE-Portland District funded our efforts to monitor and evaluate the response of Caspian terns to the enhancement of nesting habitat in the SONEC region through the construction of eight tern islands as part of the *Estuary Caspian Tern Management Plan* (USACE 2006, USACE 2015a). The USACE also funded limited monitoring at the tern island on Fern Ridge Reservoir in the Willamette Valley, Oregon, during the 2008–2011 nesting seasons. The USACE-Portland District conducted limited monitoring of Caspian tern colonies on six of the eight USACE-constructed islands in the SONEC region during the 2017–2018 breeding seasons using aerial surveys flown by the U.S. Civil Air Patrol. During the 2015–2017 breeding seasons, the USACE-Walla Walla District funded USGS-Dixon to monitor and evaluate the response of Caspian terns to the enhancement of nesting habitat at DENWR as part of the *IAPMP* (USACE 2014). In addition, we conducted aerial surveys of the Caspian tern colony sites at DENWR during the 2018 breeding season as part of the *Monitoring Strategy for Caspian Terns within the Pacific Region* (USFWS 2015), and the San Francisco Bay Bird Observatory initiated a citizen science program to monitor the tern islands in 2019 (Tarjan et al.

2020). Nesting chronology of Caspian terns at colonies in the SONEC region, however, varied considerably among colonies despite their geographic proximity. The variability in nesting chronology throughout the SONEC region and elsewhere may impact the accuracy of colony size estimates derived from monitoring strategies that rely on infrequent monitoring or a snapshot approach and this variability should be considered when designing a monitoring strategy.

Information gathered through colony monitoring provided the basis for all adaptive management actions taken at the USACE-constructed tern islands. Where implemented, close colony monitoring allowed for the timely detection of Caspian tern nest predators on the USACE-constructed islands. As a result, when permits were in place, the management agencies were able to quickly respond by authorizing the removal of nest predators that impacted Caspian tern colonies. Caspian terns were attracted to nest at nearly all the USACE-constructed islands during the first nesting season following island completion, and during the first nesting season Caspian tern nesting success was generally moderate to good. Colony monitors usually reported declining nesting success in subsequent nesting seasons, however, as predators increasingly keyed in on the tern colonies as a food source. Close colony monitoring not only allowed the identification of the predators most responsible for declines in tern nesting success, but it also provided opportunities to develop more efficient and humane methods for controlling the impact of predators on tern nesting success. At USACE-constructed islands in the SONEC region we were able to develop a variety of non-lethal predator deterrence methods, including the use of netting, hazing, and trap and transfer techniques, in addition to traditional methods of lethal predator control.

Monitoring at the Crump Lake tern island revealed that in the second nesting season after island completion the size of the Caspian tern colony peaked at 690 breeding pairs, considerably more than the 500 breeding pairs that was expected as the maximum tern colony size the island would support. The Oregon Department of Fish and Wildlife (ODFW) was concerned that such a large tern colony on Crump Lake might negatively affect survival of Warner suckers, a fish species listed as threatened under the ESA. Consequently, during subsequent breeding seasons no social attraction (decoys or audio playback systems) were deployed on the Crump Lake tern island. The size of the Caspian tern colony on Crump Lake island did not exceeded 500 breeding pairs after 2009.

Monitoring of Caspian tern colonies that formed on USACE-constructed islands in the SONEC region indicated that shelter from the sun and from aerial predators might enhance tern chick survival and therefore nesting success. After several years of monitoring, it appeared that the bare gravel surfaces of the constructed islands did not offer adequate protection from the sun or from aerial predators for Caspian tern chicks. Starting in 2011, we installed small wooden structures and shelters on each of the islands to provide shade and protection from predators at each tern island in the SONEC region, shelters that were readily used by young terns, especially in hot weather. The effects of these shelters on chick survival, however, were not quantified.

In 2009, when a 0.5-acre (0.2-ha) floating island was constructed in Dutchy Lake, Summer Lake Wildlife Area, Oregon, it was the largest structure of its kind to have been built. After the first nesting season following island construction, the Dutchy Lake floating island did not support any successful tern nesting attempts in the subsequent three nesting seasons. The following year, a 0.8-acre (0.3-ha) floating island was constructed by the same contractor in Sheepy Lake, Lower Klamath NWR, California. The floating island in Dutchy Lake was plagued by issues related to the anchoring system and construction materials starting in 2009. Attempts to correct the anchoring issues were made in 2010, but the island was taken out of service and removed from Dutchy Lake in 2013; no substitute tern nesting habitat was constructed in Summer Lake Wildlife Area or elsewhere in the SONEC region after the Dutchy Lake island was removed. Improvements to the anchoring system and construction materials were added to the design of the larger floating island deployed in Sheepy Lake. As such, the floating island in Sheepy Lake remained structurally intact for a longer period of time than did the floating island in Dutchy Lake, but it was also eventually decommissioned and removed from service. In 2018, a 1.0-acre (0.4-ha) rock-core island was constructed in Sheepy Lake as Caspian tern nesting habitat following the removal of the floating island structure after the 2017 nesting season.

3.6.5 Adaptive Management Considerations

There are several key uncertainties that may impact future action effectiveness and the necessity for adaptive management to fully achieve the objectives of the *Estuary Caspian Tern Management Plan* and the *IAPMP* as they relate to providing alternative nesting habitat for Caspian terns outside of the Columbia River basin. A primary uncertainty is the lack of a comprehensive management plan that ensures at least minimal annual maintenance and monitoring of the five USACE-constructed islands in the SONEC region that have so far shown promise as long-term colony sites for Caspian terns: Sheepy Lake in Lower Klamath NWR, Tule Lake Sump 1B in Tule Lake NWR, East Link Impoundment in Summer Lake Wildlife Area, Crump Lake in the Warner Valley, and Malheur Lake in Malheur NWR. If managed and monitored conscientiously, these five islands could support a regional Caspian tern breeding population of up to 2,000 breeding pairs, providing the region is not experiencing drought.

As of late 2020, the ownership, management, and maintenance of the islands constructed by the USACE as nesting habitat for Caspian terns are in the process of being transferred from the USACE to the respective land management agencies on whose land the islands were constructed (i.e. Lower Klamath NWR, Tule Lake NWR, Malheur NWR, Don Edwards NWR, Oregon Department of Fish and Wildlife, and Oregon Division of State Lands). However, no funding has been set aside or allocated to these management agencies for maintenance or monitoring of the islands, nor have any fund-raising plans been developed. With the exception of monitoring conducted every three years as part of the *Monitoring Strategy for Caspian Terns within the Pacific Region* (USFWS 2015), any effort to monitor breeding birds at the USACE-constructed islands will officially become the responsibility of the respective land management agencies. It is encouraging that there is interest by the resource managers overseeing the Upper Klamath National Wildlife Refuge Complex, Malheur NWR, Don Edwards NWR, and

Summer Lake Wildlife Area in continuing to monitor waterbird colonies on islands on their respective properties. Nevertheless, the level at which budgets and resource priorities will allow for continued monitoring and adaptive management (i.e. predator and/or vegetation control) of the nesting habitat created as part of the *Estuary Caspian Tern Management Plan* remains uncertain. The future of the USACE-constructed island in Crump Lake, which will be owned and managed by the Oregon Division of State Lands, is especially uncertain because the island is not in a state or federal wildlife conservation area.

The construction and enhancement of Caspian tern nesting habitat in the SONEC and San Francisco Bay regions was a positive step toward providing alternative nesting habitat for Caspian terns displaced by management actions to reduce nesting habitat in the Columbia River basin. Five years after the completion of efforts to provide alternative nesting habitat for Caspian terns as part of the *Estuary Caspian Tern Management Plan* and the *IAPMP*, however, less than half the number of breeding pairs displaced by management (ca. 6,000 breeding pairs) have occupied the new nesting sites. Caspian terns have shown greater than expected philopatry to the managed colony sites in the Columbia River basin, suggesting that comparable nesting and/or foraging opportunities outside the basin are lacking.

Anthropogenic climate change and the impact of frequent and persistent drought in the western United States are likely to continue to limit the availability of water throughout the SONEC region (Williams et al. 2020). During the last decade, following the creation of artificial islands as nesting habitat as part of the *Estuary Caspian Tern Management Plan*, the availability of those islands in the SONEC region as suitable tern nesting habitat has been variable, and the frequency at which Caspian terns will be able to find suitable nesting habitat and adequate forage fish resources in the future is uncertain.

Recent modeling of the Caspian tern population in the Pacific Flyway (Suzuki et al. 2018) has highlighted the importance of the nesting habitat on USACE-constructed islands in the SONEC region for maintaining the Flyway-wide population following the intentional elimination or reduction of nesting habitat in the Columbia River basin through active management. Nevertheless, in 2018, a year when the largest number of Caspian tern breeding pairs was recorded nesting in the SONEC region and the breeding population in the San Francisco Bay Area remained relatively stable, the overall Pacific Flyway breeding population declined (Peterson et al. 2017, Peck-Richardson et al. 2019). Additionally, as salinity levels in the Salton Sea increase beyond the biological tolerance of fish, the future availability of this important migratory stopover and breeding site for Caspian terns in the Pacific Flyway and the resultant population-level impacts are uncertain (Lyons et al. 2018). The highly variable climatic conditions in the increasingly arid SONEC region and the apparent limited carrying capacity for nesting Caspian terns in the southern part of San Francisco Bay highlights the need for additional Caspian tern nesting sites in coastal areas of the Pacific Flyway to help ensure the long-term sustainability of the Pacific Flyway population of Caspian terns.

Coastal Washington formerly supported the largest colony of Caspian terns in the Pacific Flyway (Penland 1981), but no colonies of comparable size (ca. 4,000 breeding pairs) currently exist in coastal Washington. While alternative colony sites for Caspian terns in coastal Washington were evaluated for suitability (Seto et al. 2003) and potential sites were proposed in the Draft EIS, no nesting habitat in coastal Washington was constructed or enhanced as part of the *Estuary Caspian Tern Management Plan.* In the absence of a designated colony site in coastal Washington, Caspian tern colonies nevertheless formed temporarily on warehouse rooftops, unused barges, and abandoned, fenced-off industrial sites in coastal Washington. These colony sites are frequently disturbed by human activity or intentionally hazed, however, and are not suitable as long-term colony sites. To facilitate greater and sustained breeding dispersal away from the Columbia River basin and support efforts to reduce Caspian tern predation on juvenile salmonids, we recommend that managers consider providing alternative nesting sites for Caspian terns in Grays Harbor, the Strait of Juan de Fuca, or Puget Sound. These regions have a history of supporting large Caspian tern colonies, have high connectivity with the Caspian tern colonies in the Columbia River basin, and alternative colony sites are available that do not pose a risk to salmonid stocks of conservation concern.

3.7 Conclusions

1. The primary objective of the plan *Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary*, which was implemented starting in 2008, was to reduce the size of the large Caspian tern breeding colony on East Sand Island by reducing the area of suitable nesting habitat, while creating or restoring alternative tern nesting habitat elsewhere in the Pacific Flyway as compensation.

2. The primary objective of the *Inland Avian Predation Management Plan*, which was implemented starting in 2014, was to reduce Caspian tern predation rates in the Columbia Plateau region on each ESA-listed ESU/DPS of salmonids to less than 2% of available smolts per year by displacing the two largest tern colonies in the Plateau region, while restoring tern nesting habitat in the San Francisco Bay Area as compensation.

3. By 2018, the USACE had built or enhanced a total of 8.3 acres (3.4 ha) of islands in Oregon and California as effective nesting habitat for about 6,000 pairs of Caspian terns, the total number of terns displaced by both the *Estuary Caspian Tern Management Plan* and the *Inland Avian Predation Management Plan*.

4. Social attraction techniques (decoys and audio playback of tern vocalizations) were deployed on all USACE-constructed islands, and Caspian terns colonized and attempted to nest on 10 of 11 islands or island groups during the first nesting season after island completion.

5. The average size of the breeding population of Caspian terns in the SONEC region nearly tripled after the construction of eight new tern nesting islands in the region, but inter-annual variation in the size of the regional breeding population remained high.

6. Caspian tern nesting success (average number of young raised/breeding pair) at the USACE-constructed islands in the SONEC region was highly variable among islands and among years due to drought and colony disturbances by avian and mammalian predators.

7. Perennial active management to remove vegetation and control nest predators were important components for ensuring the persistence of Caspian tern colonies at USACE-constructed islands in the SONEC region and may be required in the future to maintain the islands as suitable tern nesting habitat.

8. Resightings at USACE-constructed tern islands in the SONEC region of Caspian terns banded at the large breeding colony in the Columbia River estuary indicated that initially most terns recruiting to the new colony sites in the SONEC region were immigrants from the Columbia River estuary, 450–550 km from islands in the SONEC region.

9. Resightings at USACE-constructed tern islands in the SONEC region of Caspian terns banded at Crescent Island in the mid-Columbia River and Goose Island in Potholes Reservoir indicated that terns immigrated to the colony sites in the SONEC region from the Columbia Plateau region following the implementation of the *Inland Avian Predation Management Plan*, 300–600 km from islands in the SONEC region.

10. Caspian terns that recruited to islands modified and enhanced by the USACE in Don Edwards San Francisco Bay National Wildlife Refuge were mostly immigrants from other colonies in the San Francisco Bay Area, rather than those displaced from managed colonies in the Columbia River basin.

11. The diet of Caspian terns nesting on USACE-constructed islands in the SONEC region consisted mainly of native and introduced species of minnows (Cyprinidae); predation impacts to ESA-listed fish species and other fishes of conservation concern were minimal, indicating that management concern over Caspian tern predation on protected fishes was not transferred from the Columbia Basin to the SONEC region by the effort to attract terns.

12. Nesting chronology of Caspian terns at colonies in the SONEC region varied considerably among colonies despite their close geographic proximity, which may impact the accuracy of colony size estimates derived from monitoring strategies that rely on infrequent monitoring or a snapshot approach.

13. The long-term success of the plan *Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* and the *Inland Avian Predation Management Plan* depends on the availability of a reliable network of suitable breeding colony sites outside of the Columbia Basin. Anthropogenic climate change and the impact of frequent and persistent drought in the western United States are likely to continue to impact water resources throughout the SONEC region and elsewhere. As such, additional alternative colony sites in coastal Washington (i.e. Grays Harbor, Strait of Juan de Fuca, and/or Puget Sound) may be needed to support the permanent relocation of Caspian terns to colony sites outside of the Columbia Basin and limit the impact of avian predation on imperiled salmonids, while simultaneously maintaining a robust population of Caspian terns in the Pacific Flyway.

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CHAPTER 4: Double-crested Cormorant Management in the Columbia River Estuary

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4.1 Summary

The double-crested cormorant (*Phalacrocorax auritus*) population in the western United States began to increase in the mid-1970s following a ban on the pesticide DDT and protection of the species under the Migratory Bird Treaty Act in 1972. Nesting double-crested cormorants were first detected on East Sand Island at river km 8 in the Columbia River estuary in 1988 and there were 91 breeding pairs nesting on the island in 1989, the first year the colony was enumerated. Subsequently, the colony rapidly increased in size, with initial immigration likely coming from declining colonies along the coasts of Washington and British Columbia. The East Sand Island colony soon became the largest cormorant colony in the region and by 2006 it had increased to over 13,000 breeding pairs and was the largest known breeding colony for the species in the Pacific Flyway. The large size of the colony was made possible by an abundance of forage fish near the mouth of the Columbia River and an absence of terrestrial predators and human disturbance on East Sand Island. Further, the large size of the colony apparently provided safety in numbers from bald eagle (*Haliaeetus leucocephalus*) predation, a limiting factor implicated in the declines of nesting colonies in the Salish Sea to the north that likely contributed to the initial emigration to East Sand Island.

From 1997 until 2014, we investigated the nesting and foraging ecology of double-crested cormorants at the large colony on East Sand Island and other smaller breeding colonies in the Columbia River estuary as part of an effort to understand the impact of cormorant predation on the survival of juvenile salmonids (*Oncorhynchus* spp.) that are listed under the U.S. Endangered Species Act (ESA). Initially, double-crested cormorants on East Sand Island nested on the rock jetty at the extreme western end of the island. Gradually, and as the colony grew in size, nesting double-crested cormorants shifted further east along the rocky revetment and onto the sandy upland habitat to the north. By the end of the study period in 2014, the rock jetty, which was the first habitat occupied by nesting cormorants on East Sand Island, was no longer occupied and all double-crested cormorant nesting activity was east of the rock jetty.

Bald eagle disturbance appeared to be an important factor influencing nesting success of double-crested cormorants at East Sand Island. During 2008–2012 we found that eagle disturbances to the cormorant colony were positively correlated with Columbia River discharge. In addition to direct mortality of cormorants from predation by eagles, eagle disturbances at the colony were associated with heightened predation on cormorant eggs and nestlings by glaucous-winged/western gulls (*Larus glaucescens* X *L. occidentalis*). During 2006–2013, two factors emerged that explained much of the inter-annual variation in cormorant nesting success: river discharge and the proportion of juvenile salmonids in the diet. As river discharge increased, the proportion of juvenile salmonids in the cormorant diet also increased, apparently because the availability of marine forage fish, the main food source for cormorants nesting at East Sand Island, declined with increasing discharge. As such, nesting success of double-crested cormorants at East Sand Island was negatively correlated with both increased river discharge and the proportion of salmonid smolts in the diet. Double-crested cormorants were resilient to these limiting factors, however, and nesting success averaged nearly two young raised per breeding pair throughout the pre-management study period (1997–2013), above the average documented in other studies of the species. Presumably, the large colony size effectively swamped the effects of avian predators on cormorant nesting success but did not often cause local depletion of prey resources. It is also likely that the regular presence and public visibility of researchers working on the island during the study period deterred intentional human disturbance and persecution, a not uncommon limiting factor for breeding colonies of double-crested cormorants. Overall, breeding performance was consistent with that seen elsewhere for cormorant populations experiencing intrinsic growth.

The diet of double-crested cormorants nesting on East Sand Island consisted of a wide variety of marine, estuarine, and freshwater forage fishes, including juvenile salmonids belonging to 11 of the 13 populations of anadromous salmonids from the Columbia River basin that are listed under the ESA. The diet of double-crested cormorants nesting on East Sand Island consisted of an average of 12.5% juvenile salmonids (% of total prey biomass) but ranged from 1.7% to 27.9% of the diet, depending on the year. In contrast, the diet of double-crested cormorants nesting at the colony on Rice Island in the upper estuary consisted of 35% to 54% juvenile salmonids, apparently due to the lower availability of marine forage fish associated with nesting in the freshwater sector of the estuary. Despite the relatively low proportion of juvenile salmonids in the diet of double-crested cormorants nesting at East Sand Island, cormorants from this large colony annually consumed an estimated average of 10.9 million juvenile salmonids during 1998–2013. Estimates of the number of juvenile salmonids consumed annually ranged from a low of about 2.9 million smolts in 2005 to a high of about 20.9 million smolts in 2011, two years when the size of the double-crested cormorant colony on East Sand Island differed by just 6%. Although millions of salmonid smolts were consumed annually by cormorants nesting at East Sand Island during the study period, on average they typically consumed less than 10% of all available smolts migrating through the Columbia River estuary in any given year.

An investigation of predation rates on juvenile salmonids (percentage of available fish consumed) by double-crested cormorants nesting on East Sand Island indicated that impacts were highly variable depending on the salmonid population and year. Estimated predation rates by East Sand Island cormorants were consistently the highest on ESA-listed Lower Columbia River Chinook salmon (*O. tshawytscha*), with an average annual predation rate of 27.5% (95% credible interval = 24.3–30.7%). At the other extreme, the estimate of average annual predation rate on Upper Willamette River spring Chinook salmon smolts during 2003–2014 was 1.8% (1.3–2.6%). Average annual estimates of cormorant predation rates on juvenile salmonids in the Columbia River estuary that originated from Upper Columbia River (UCR), Snake River (SR), or Middle Columbia River (MCR) populations ranged from 2.7% (2.3–3.2%) for SR fall Chinook salmon to 7.5% (6.3–9.3%) for MCR steelhead (*O. mykiss*) during 2003–2014. There was no evidence that predation rates differed based on the rear-type (hatchery vs. wild) or out-migration history (in-river vs. transported) of smolts, suggesting that East Sand Island cormorants indiscriminately preyed on smolts within a population, despite high variation in predation rates among populations.

Dispersal of double-crested cormorants from the East Sand Island colony and the connectivity of the East Sand Island colony with other cormorant colonies in the Pacific Flyway was investigated using mark-recovery and satellite telemetry techniques. Cormorants that were banded on East Sand Island were resighted/recovered across an extensive area of the Pacific Flyway from Vancouver Island, BC, to the U.S.-Mexico border. The region with the highest number of resightings/recoveries was the Salish Sea of coastal Washington and British Columbia, with secondary hot spots in the San Francisco Bay Area and the Southern California Bight. Cormorants that were satellite-tagged on East Sand Island were subsequently tracked as far north as northern Vancouver Island and as far south as the Colorado River delta in northwestern Mexico. Despite extensive dispersal of tagged cormorants from East Sand Island to the north and south along the Pacific Flyway, there was limited dispersal inland and across the Cascade-Sierra Nevada mountain ranges. The colonies with the highest connectivity to the East Sand Island colony were the Astoria-Megler Bridge colony (15 km up-river in the Columbia River estuary), the former colony site on Rice Island (26 km up-river in the Columbia River estuary), and the former colony site on No-Name Island in Grays Harbor, Washington (78 km to the north). There was surprisingly little connectivity with cormorant colonies along the Oregon coast or in the Columbia Plateau region of eastern Washington.

Pilot studies to test the feasibility of non-lethal methods to reduce the size of the double-crested cormorant colony on East Sand Island, and thereby its impact on survival of juvenile salmonids, without causing colony abandonment were conducted during 2007–2013. A combination of human hazing of prospecting cormorants and destruction of partially built cormorant nests on one side of a large privacy fence transecting the colony proved highly effective at limiting the area of nesting habitat available for cormorant nesting. Social attraction techniques (decoys and audio playback of cormorant vocalizations) and nesting substrate enhancement were demonstrated to be effective in encouraging double-crested cormorants to nest in previously unused habitats on East Sand Island and on other islands in the Columbia River estuary with previous histories of cormorant nesting attempts. Double-crested cormorants, however, were slow to respond to social attraction and nesting substrate enhancement at sites with no previous history of cormorant nesting attempts, such as the artificial islands constructed by the U.S. Army Corps of Engineers (USACE) as alternative colony sites for Caspian terns (*Hydroprogne caspia*). Additional confounding factors possibly inhibited cormorant colonization of these sites, including (1) a comparative lack of nearby active colonies to act as sources of colonizing individuals, (2) smaller cormorant populations in the regions where islands were constructed, and (3) researcher visitation to these sites to monitor tern restoration efforts in the initial years following island construction. Nevertheless, the artificial islands constructed by the USACE were eventually colonized by double-crested cormorants within 4–8 years of Caspian terns and gulls (*Larus* spp.) initiating nesting on the islands.

Starting in 2015, the USACE, the U.S. Fish and Wildlife Service, and USDA-Wildlife Services implemented a management plan with the goal of substantially reducing predation on ESA-listed juvenile salmonids by double-crested cormorants in the Columbia River estuary. The *Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the*

Columbia River Estuary (hereafter *Cormorant Management Plan*; USACE 2015) consisted of two phases. Phase I was primarily lethal in approach and consisted of culling up to 10,912 adult double-crested cormorants and oiling the eggs in up to 15,184 cormorant nests on the East Sand Island colony over a 4-year period, with the objective of reducing the size of the colony to no more than 5,380–5,939 breeding pairs, or at most 42% of the average pre-management colony size. Phase II was primarily nonlethal in approach and consisted of reducing the available cormorant nesting habitat on East Sand Island to an area capable of supporting no more than 5,380–5,939 breeding pairs of double-crested cormorants in the long term.

During Phase I (2015–2017), USDA-Wildlife Services reported culling 5,576 adult double-crested cormorants near the East Sand Island colony and oiling the eggs in 6,181 cormorant nests on the colony. The reported take was far less than intended because, beginning in May 2016, the cormorant colony on East Sand Island was repeatedly abandoned and most nesting cormorants dispersed from the colony site. The East Sand Island double-crested cormorant colony had fewer than 5,380 active nests during the 2017 breeding season, and adaptive management provisions of the *Cormorant Management Plan* triggered the initiation of Phase II in 2018 (USACE 2015). During Phase II (2018–2019), most of the former habitat used by nesting cormorants on East Sand Island was converted to intertidal mudflats and a privacy fence was erected to separate the designated cormorant colony area from parts of the island where cormorant nesting would be dissuaded by hazing. In 2018, a double-crested cormorant colony of about 3,670 breeding pairs formed in the designated colony area and nested successfully. During 2019, after the conversion of much of the former cormorant colony area to intertidal mudflats, only a few hundred pairs of cormorants attempted to nest on East Sand Island; the colony experienced repeated partial and total abandonment events during the 2019 nesting season, and no nesting attempts were successful. In 2020 there were no sustained nesting attempts by double-crested cormorants on East Sand Island and the colony was apparently completely abandoned.

The factors responsible for the repeated and unprecedented colony abandonment events by cormorants attempting to nest on East Sand Island during 2016–2017 are not completely understood, but the primary factor appears to be the cumulative effects of disturbance and stress on cormorants from bald eagles and the activities of USDA-Wildlife Services technicians engaged in culling and egg oiling. It is likely that these two factors acted synergistically to dissuade cormorants from nesting on East Sand Island during Phase I of the *Cormorant Management Plan*. The conversion of much of the former cormorant colony area on East Sand Island to inter-tidal mudflats and other modifications on the west end of East Sand Island during the winter of 2018-2019 as part of Phase II of the Plan apparently resulted in nearly a complete dispersal of cormorants from the East Sand Island colony. Concurrent with these repeated abandonment and dispersal events at East Sand Island, the formerly small breeding colony (< 400 breeding pairs until 2015) of double-crested cormorants on the Astoria-Megler Bridge, 15 km up-river, became a heavily used roosting site for non-breeding cormorants, and the number of double-crested cormorants nesting on the bridge increased to over 5,000 breeding pairs by the 2020 nesting season.

To evaluate the efficacy of the *Cormorant Management Plan* for reducing predation rates on ESA-listed salmonid populations by double-crested cormorants nesting on East Sand Island to levels specified by NOAA Fisheries (USACE 2015), we compared average annual estimates of predation rates prior to and following implementation of the Plan. Results indicated that predation rates by East Sand Island cormorants on most, but not all, salmonid populations were lower following implementation of management actions associated with Phase I of the *Cormorant Management Plan*. For example, average annual predation rates by East Sand Island cormorants on eight of the 11 ESA-listed ESUs/DPSs of salmonids that were evaluated decreased during Phase I of the Plan. Average annual predation rates by East Sand Island cormorants on all salmonid populations evaluated were significantly lower during Phase II (2018), with average annual predation rates decreasing by 70 to 90%, depending on the salmonid ESU/DPS. During 2016–2019, however, attendance by double-crested cormorants at the East Sand Island colony was intermittent, with multiple episodes of dispersal from the colony and partial or complete colony abandonment, resulting in large numbers (several thousands) of loafing/roosting cormorants in the upper estuary during the peak smolt out-migration period and the rapid expansion of the nesting colony on the Astoria-Megler Bridge. Predation rates on juvenile salmonids by double-crested cormorants loafing/roosting in the upper estuary or nesting on the Astoria-Megler Bridge were not measured but were presumably greater on a per capita basis because these sites are in or near the freshwater zone of the estuary, where juvenile salmonids are relatively more prevalent compared to marine forage fish. Thus, the total or net impact on survival of juvenile salmonids from predation by all double-crested cormorants in the Columbia River estuary during the management period is unknown.

The future of the East Sand Island cormorant colony is uncertain. In 2019 few cormorants attempted to nest on East Sand Island and those attempting to nest failed to raise young, while in 2020 no persistent cormorant nesting attempts were detected on the colony. Currently, the largest double-crested cormorant colony in the Pacific Flyway is the Astoria-Megler Bridge colony, 15 km up-river from the East Sand Island colony. Like the former Rice Island cormorant colony, the cormorant colony on the Astoria-Megler Bridge is upriver from the marine zone of the estuary and double-crested cormorants nesting on the bridge are likely consuming more juvenile salmonids per capita than cormorants nesting on East Sand Island. Although smolt consumption rates have not been estimated for the double-crested cormorant colony on the Astoria-Megler Bridge, with the growth of the colony on the bridge concurrent with the near abandonment of the colony at East Sand Island, it is unlikely that cormorant predation on ESA-listed juvenile salmonids throughout the estuary has been reduced to the level intended by the *Cormorant Management Plan*. As such, the objectives of the Plan would be more likely achieved if the breeding colony on the Astoria-Megler Bridge were to be relocated back to East Sand Island. Colony relocation can likely be achieved by dissuasion and hazing of cormorants nesting on the Astoria-Megler Bridge combined with nesting substrate enhancement and social attraction at the former colony site on East Sand Island. Colony relocation would reduce impacts on survival of ESA-listed juvenile salmonids to levels intended by the *Cormorant Management Plan*, minimize damage to the bridge from cormorant guano deposition, and help ensure the long-term conservation of the double-crested cormorant population in the Pacific Flyway.

4.2 Introduction

4.2.1 Western Population of the Double-crested Cormorant

The double-crested cormorant (*Phalacrocorax auritus*) is limited in its distribution to North America. The western population of the species warrants consideration as a separate management unit from the populations east of the Continental Divide because of differences in population status and biology (Anderson et al. 2004b, Wires and Cuthbert 2006, Mercer et al. 2013, Wires 2014). The breeding range of the western population includes all nesting colonies west of the Continental Divide in British Columbia, Washington, Oregon, Idaho, California, Nevada, Utah, Arizona, and portions of Montana, Wyoming, Colorado, and New Mexico (Adkins et al. 2014). The last full census and status assessment for the western population of double-crested cormorants was conducted during 2009 (Adkins and Roby 2010, Adkins et al. 2014). The Adkins et al. (2014) status assessment was motivated in part by concern over the potential impact of predation by double-crested cormorants on juvenile salmonids (*Oncorhynchus* spp.), particularly in the Columbia River basin and along the Pacific coast where some salmonid populations are listed for protection under the U.S. Endangered Species Act (ESA).

Colony size data for the western population of double-crested cormorants indicated that there were approximately 31,200 breeding pairs in the western population during ca. 2009 (Adkins et al. 2014). The total estimated number of double-crested cormorant breeding pairs in the Pacific Region (i.e. the coastal portion of the range of the western population in British Columbia, Washington, Oregon, and California) increased by 72% from 1987–1992 to ca. 2009, despite the estimated number of breeding pairs located in coastal British Columbia and Washington declining by ca. 66% during this same period (Adkins et al. 2014). During this apparent southward distributional shift in the cormorant breeding population and based on the best available data from that period, the average annual growth rate (λ) of the number of breeding double-crested cormorants in the Pacific Region was 1.03. This compares to a λ of about 1.07 for the double-crested cormorant population east of the Continental Divide during recent decades. Most of the increase in cormorant numbers within the Pacific Region from 1987 to 2009 was attributable to an increase in the size of the nesting colony on East Sand Island in the Columbia River estuary (Adkins et al. 2014). Disturbance at breeding colonies by bald eagles (*Haliaeetus leucocephalus*) and humans were likely the primary limiting factors on the growth of the western population through 2014 (Adkins et al. 2014).

The substantial increase in the size of the western population from the late 1980s to the late 2000s was a continuation of the population growth noted by Carter et al. (1995), and was primarily caused by the following two events, both in 1972: the protection of double-crested cormorants for the first time under the Migratory Bird Treaty Act and the banning of the persistent chlorinated hydrocarbon pesticide DDT (Wires 2014). These two events not only resulted in gradual population recovery in the West, but substantial increases throughout North America. Because double-crested cormorants had been rare or absent from large portions of their former range in North America for nearly a century, the population recovery was perceived by some publics as an invasion by a non-native species (Wires and Cuthbert 2006, Wires 2014).

4.2.2 Double-crested Cormorant Nesting in the Columbia River Estuary

Prior to 1980 very few, if any, double-crested cormorants were recorded nesting in the Columbia River estuary (Carter et al. 1995). At the time of European settlement in the region, it is likely that double-crested cormorants nested opportunistically in the Columbia River estuary, but records are lacking. Starting in 1980, double-crested cormorants were noted nesting on navigational aids and the old railroad trestles. A breeding colony of double-crested cormorants on East Sand Island in the Columbia River estuary was first noted in 1988 (Carter et al. 1995), and 91 nests were counted on the ground at the west end of the island in 1989 (D. Bell *pers. comm.*, Naughton et al. 2007). Concurrent with the colonization of East Sand Island in the late 1980s, Rice Island, a large dredged-material disposal island in the upper estuary (*Figure 4.1*), was also colonized by ground-nesting double-crested cormorants (Carter et al. 1995, Adkins and Roby 2010). As early as 1991–1992, the East Sand Island colony had grown to 2,026 breeding pairs and the Rice Island colony had grown to 1,211 breeding pairs (Carter et al. 1995). During the subsequent 15 years, the East Sand Island cormorant colony continued to grow while the Rice Island colony was abandoned. Early on, the dramatic increase in numbers of double-crested cormorants nesting in the Columbia River estuary stood in contrast to declines at other colonies along the coast of Washington and British Columbia (Adkins et al. 2014). Increases in the number of double-crested cormorants nesting in the estuary during the late 1980s and early 1990s were so rapid that they clearly were not primarily intrinsic (i.e. recruitment of young raised in the estuary) and involved considerable immigration from colonies in coastal regions to the north. Band recovery data in support of large-scale immigration to the Columbia River estuary by double-crested cormorants during the 1980s and 1990s are, however, lacking.

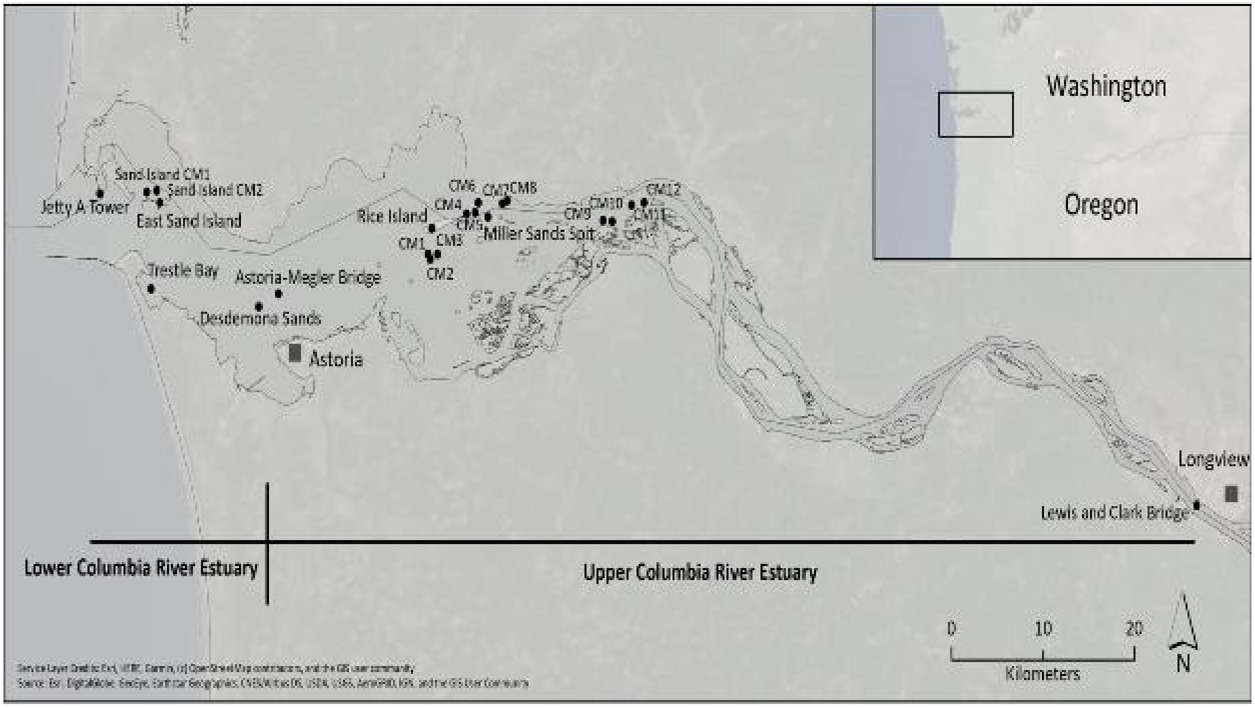

Figure 4.1. Columbia River estuary on the border of Oregon and Washington states, showing the location of sites where double-crested cormorants are known to have nested, including Jetty A Tower (river km 4); East Sand Island (river km 8); Trestle Bay (river km 10); Desdemona Sands (river km 20); Astoria-Megler Bridge (river km 23); Rice Island (river km 34) and Miller Sands Spit (river km 38), two large dredged material disposal islands; navigational channel markers (river kms 38–48); and the Lewis and Clark Bridge (river km 106) at Longview, Washington.

Aside from the ground-nesting colonies on East Sand Island, Rice island, and Miller Sands Spit, several colonies of double-crested cormorants have formed on artificial structures over water in the Columbia River estuary. These include the pilings at Desdemona Sands (just downstream of the Astoria-Megler Bridge), abandoned railroad trestles in Trestle Bay in Fort Stevens State Park, navigational aids in the vicinity of Miller Sands Spit (*Figure 4.1* above), navigational aids further up-river near Pillar Rock Sands (*Figure 4.1* above), and the Astoria-Megler Bridge (*Figure 4.1* above). The Desdemona Sands pilings supported a colony of at least 120 breeding pairs in the 1990s, but the last year when double-crested cormorants were recorded nesting at Desdemona Sands was 2000, when 61 pairs nested there (Adkins et al. 2014). The abandoned railroad trestles in Trestle Bay supported about 32 breeding pairs as late as 1992 (Carter et al. 1995). Collectively, the Miller Sands Spit navigational aids usually supported between 50 and 200 breeding pairs during the 1990s and early 2000s but starting in 2008 between 200 and 350 breeding pairs were recorded nesting there (Adkins and Roby 2010). The navigational aids further up-river have generally supported fewer than 100 breeding pairs. Double-crested cormorants were first detected nesting on the Astoria-Megler Bridge in 2004, when a small number of breeding pairs were counted nesting among pelagic cormorants (*P. pelagicus*). After 2004, the Astoria-Megler Bridge colony grew substantially (see section *4.4.1 Colony size, Nesting Habitat Use, & Nesting Density*).

4.2.3 Double-crested Cormorant Nesting on East Sand Island

The double-crested cormorant colony on East Sand Island first became established at the extreme western end of the island, where a rock jetty was constructed during World War II to protect the island from further erosion. The colony was first detected in 1988, and 91 breeding pairs of double-crested cormorants were counted on East Sand Island in 1989 (Naughton et al. 2007). Immigration from other colonies and subsequent intrinsic growth occurred to the extent that the East Sand Island colony became the largest for the species on the Pacific coast of North America (Anderson et al. 2004b). From 1989 to 1991 the East Sand Island colony of double-crested cormorants increased from 91 breeding pairs to over 2,000 breeding pairs (Carter et al. 1995), indicating large scale immigration of breeding birds to East Sand Island from outside the Columbia River estuary. By 1997, the first year of the present study, the East Sand Island colony had increased to over 5,000 breeding pairs, suggesting that immigration from other cormorant colonies both within and outside the Columbia River estuary remained a factor in colony growth at East Sand Island during the mid-1990s.

4.2.4 Double-crested Cormorant Nesting on Rice Island

In 1988, a ground-nesting colony of double-crested cormorants was first detected at Rice Island, a large dredged-material disposal island at river km 34 in the upper Columbia River estuary (Carter et al. 1995; *Figure 4.2*). By 1990, the Rice Island colony had grown to 761 breeding pairs (A. Clark, Willapa NWR, unpubl. data; Naughton et al. 2007). During 1991–1992 over 1,200 pairs of double-crested cormorants were reported on Rice Island (A. Clark, Willapa NWR, unpubl. data; Carter et al. 1995). The Rice Island cormorant colony persisted until 1997, the first year of the present study, when the colony was located near the large colony of Caspian terns at the downstream end of Rice Island (Roby et al. 1998). By 1999, however, the Rice Island colony of double-crested cormorants had been abandoned, apparently because of human disturbance related to the management of the Rice Island colony of Caspian terns (see *Chapter 1*). The concurrent increase in the size of the East Sand Island cormorant colony suggested that most of the cormorants that formerly nested on Rice Island emigrated to the East Sand Island cormorant colony, 26 km closer to the mouth of the river (*Figure 4.2*). In 2001 a smaller double-crested cormorant colony reformed on Rice Island but was abandoned again in 2004.

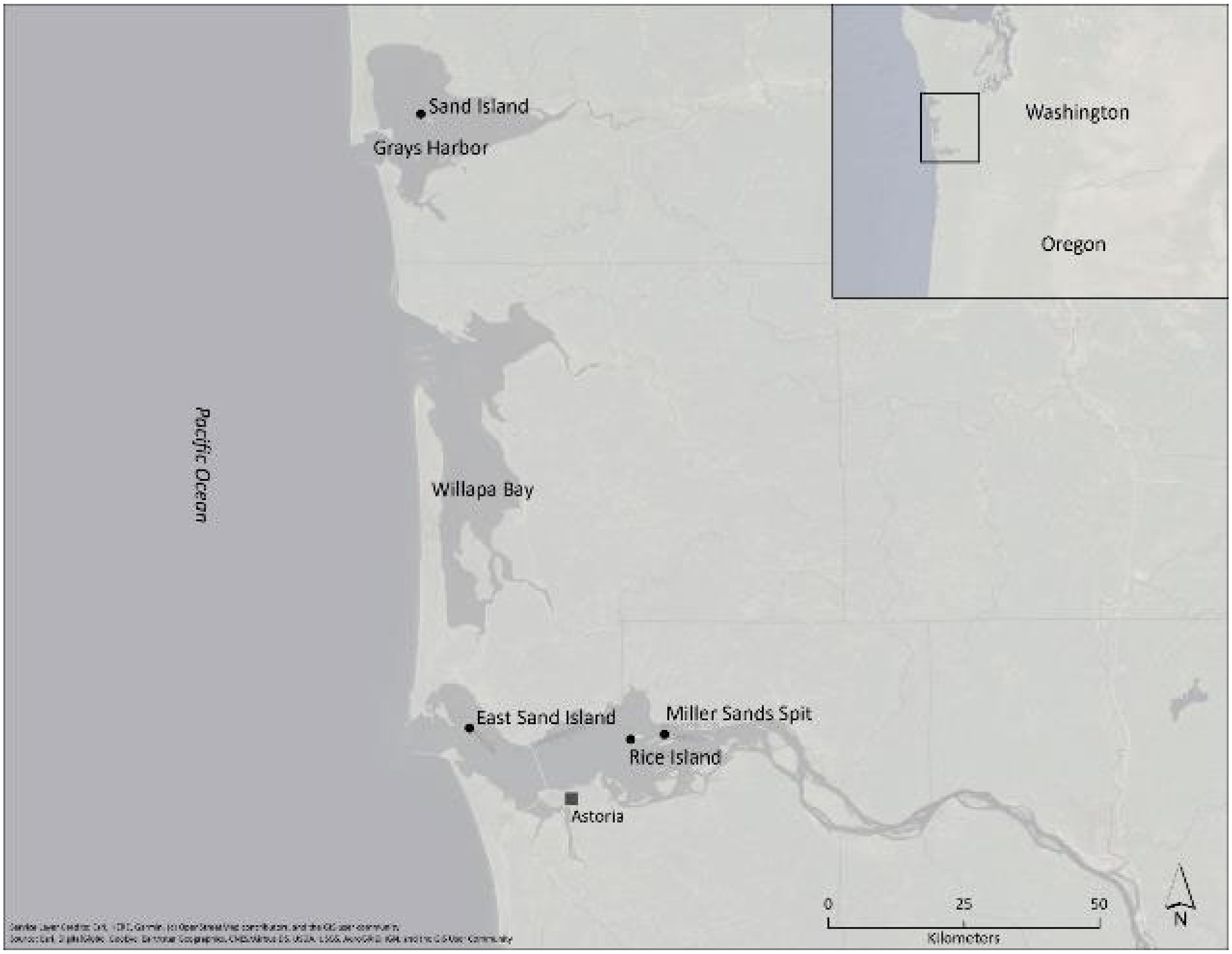

Figure 4.2. Columbia River estuary on the border of Oregon and Washington states, and Willapa Bay and Grays Harbor in Washington State, showing the locations of Rice Island (river km 34) and Miller Sands Spit (river km 38) in the upper estuary, East Sand Island (river km 8) in the lower estuary near Astoria, Oregon, and Sand Island in Grays Harbor, Washington.

During the 1997 and 1998 breeding seasons, diet samples were collected from double-crested cormorants nesting on Rice Island and on channel markers and navigational aids located nearby. These diet data indicated that the proportion of salmonids in the diet (% of prey biomass) of cormorants nesting on Rice Island was about 35% and 54% in 1997 and 1998, respectively, whereas the percent biomass of salmonids in the diet of cormorants nesting at East Sand Island was about 16% in both years (Roby et al. 1998, Collis et al. 2000). Thus, per capita consumption of juvenile salmonids by double-crested cormorants nesting at Rice Island was 2-3 times greater than that of cormorants nesting on East Sand Island. Similar to the effects of intentionally moving the colony of Caspian terns from Rice Island to East Sand Island during 1999–2001 (see *Chapter 1*), the inadvertent shift of the double-crested cormorant colony from Rice Island to East Sand Island in 1999 probably resulted in a significant reduction in per-capita consumption of juvenile salmonids by double-crested cormorants nesting in the Columbia River estuary.

4.2.5 Double-crested Cormorant Management Plan for the Columbia River Estuary

The *Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary* (hereafter referred to as the "*Cormorant Management Plan*") was developed by the U.S. Army Corps of Engineers (USACE)-Portland District with input and cooperation from the U.S. Fish and Wildlife Service, USDA-Wildlife Services, Oregon Department of Fish and Wildlife, and the Washington Department of Fish and Wildlife (USACE 2015). The goal of the *Cormorant Management Plan* was to reduce consumption of juvenile salmonids by double-crested cormorants in the Columbia River estuary by reducing the size of the large cormorant breeding colony on East Sand Island and preventing new cormorant colonies from forming in the estuary, especially further up-river where the proportion of salmonids in the diet of cormorants would likely be greater than at East Sand Island.

The *Cormorant Management Plan* did not target a specific reduction in predation rates on juvenile salmonids by cormorants in the estuary, but instead sought to reduce the size of the double-crested cormorant colony on East Sand Island to less than 5,380–5,939 breeding pairs in order to achieve reductions in predation rates (USACE 2015). NOAA Fisheries, the agency responsible for the recovery of ESA-listed anadromous salmonids in the Columbia River basin, determined that a cormorant colony on East Sand Island of 5,380–5,939 breeding pairs would restore survival of juvenile steelhead trout (*O. mykiss*) to baseline or base period levels, which occurred prior to 2002 (NMFS 2008). These objectives were identified as reasonable and prudent alternative (RPA) action 46 in the 2008 Biological Opinion and associated 2010 and 2014 Supplemental Biological Opinions issued by the National Marine Fisheries Service for operation of the Federal Columbia River Power System (FCRPS; NMFS 2008, NMFS 2010). The *Cormorant Management Plan* then adopted the colony size range of 5,380–5,939 breeding pairs that was specified in RPA 46 as the upper limit for the size of the East Sand Island colony following management (USACE 2015).

The *Cormorant Management Plan* was designed for implementation in a two-phase approach, with Phase I involving actions to directly reduce the numbers of double-crested cormorants nesting at the colony on East Sand Island using primarily lethal techniques (culling and egg-oiling). The 4-year lethal strategy included annually culling 13.5% of the adult double-crested cormorants nesting on East Sand Island on or near the colony, with a total cull of 10,912 individuals by the end of Phase I. Also during Phase I, the eggs in 46% of the double-crested cormorant nests would be oiled to prevent hatching in the first three years, with egg-oiling destroying the eggs in a total of 15,184 cormorant nests. Following the implementation of Phase I, the East Sand Island colony of double-crested cormorants was expected to be within the size range of 5,380–5,939 breeding pairs, the targeted upper limit for colony size, and the *Cormorant Management Plan* would transition to Phase II. The objective of Phase II was to reduce the availability of cormorant nesting habitat on East Sand Island so that the size of the double-crested cormorant colony would remain less than 5,380–5,939 breeding pairs indefinitely. The plan was to convert much of the former upland habitat on the western half of East Sand Island where double-crested cormorants had previously nested to inter-tidal mudflats, which would be unsuitable for cormorant nesting. Any residual colony of less than 5,380–5,939 breeding pairs of double-crested cormorants would nest in designated habitat on the rock jetty that forms the western tip of East Sand Island.

4.2.6 Evaluating the Efficacy of the Double-crested Cormorant Management Plan

Thirteen of 20 evolutionarily significant units (ESUs) or distinct population segments (DPSs) of Columbia Basin salmonids are currently listed as either threatened or endangered under the ESA, whereas double-crested cormorants are protected under the Migratory Bird Treaty Act. Studies of double-crested cormorant predation rates on juvenile salmonids have been conducted at breeding colonies located throughout much of the Columbia River basin, both in the Columbia River estuary and in the Columbia Plateau region (see *Chapter 6*). Avian predation in the Columbia River estuary is of particular concern to fisheries managers because it affects fish belonging to all 13 ESA-listed populations of anadromous salmonids in the Columbia River basin (NMFS 2019); smolts that have survived freshwater out-migration to the estuary have a higher probability of survival to adulthood compared to smolts that have yet to complete out-migration (Roby et al. 2003, USFWS 2005, Evans et al. 2019b). As part of this chapter, we summarize and synthesize predation rate studies based on smolt PIT (Passive Integrated Transponder) tag recoveries on double-crested cormorant colonies in the Columbia River estuary, with a focus on the East Sand Island cormorant colony. These data were used to assess the impacts of double-crested cormorants nesting on East Sand Island on the survival of ESA-listed juvenile salmonids. More specifically, we summarized predation rates on ESA-listed populations (ESUs or DPSs) of PIT-tagged juvenile salmonids by double-crested cormorants breeding in the estuary.

PIT tags placed in juvenile salmonids and subsequently recovered on the double-crested cormorant colony on East Sand Island were also used to monitor the efficacy of the *Cormorant Management Plan* for reducing predation rates on juvenile salmonids. Recovery of smolt PIT tags on piscivorous waterbird colonies has been used to estimate predation rates (percentage of available fish consumed) in the Columbia River basin for over two decades (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012). Analysis of salmonid PIT tag recoveries were used to (1) quantify smolt losses from specific salmonid ESUs/DPSs due to piscivorous waterbirds from particular breeding colonies, (2) investigate the relative susceptibility of different groups of salmonid smolts to avian predation, and, more recently, (3) evaluate the efficacy of management plans aimed at reducing predation by piscivorous waterbirds by reducing the number of breeding birds at specific colony sites (USFWS 2005, USACE 2014, USACE 2015). As part of this chapter, we summarize and synthesize results of analyses to estimate predation rates by double-crested cormorants on populations of juvenile salmonids in the Columbia River estuary, with a focus on the efficacy of management actions aimed at reducing predation rates by double-crested cormorants nesting at the East Sand Island colony.

In addition to summarizing our predation rate studies, we also summarize the results of our studies of double-crested cormorant nesting ecology, foraging ecology, diet composition, colony connectivity, and factors limiting colony size and nesting success at the colony on East

Sand Island in the Columbia River estuary during 1997–2014. During the management period (2015-2019), few or no data were collected on cormorant nesting ecology, foraging ecology, diet composition, colony connectivity, and factors limiting colony size and nesting success at the East Sand Island colony, and the data collected on colony size were collected primarily by other groups. We include the available data from the management period here in order to present results that represent the most comprehensive synthesis of data on double-crested cormorants available for the Columbia River estuary.

4.3 Methods

4.3.1 Colony Size, Nesting Habitat Use, & Nesting Density

Georeferenced, high-resolution, vertical aerial photography was used to estimate the peak size of the breeding colony (number of breeding pairs), nesting habitat use, and nesting density of double-crested cormorants on East Sand Island in the Columbia River estuary. In nearly all years during the pre-management period (1997-2014) colony size peaked late in the incubation period. The size of the cormorant colony on East Sand Island during each breeding season was based on at least three independent counts of the number of attended nests visible in aerial photography; standard errors from the independent counts were used to calculate confidence intervals for the yearly estimate of colony size during the 1997–2014 breeding seasons (pre-management period) and the 2017–2018 breeding seasons (management period).

Point files generated from double-crested cormorant nest counts were used to delineate colony boundaries and calculate the area of used nesting habitat on East Sand Island; nesting density was calculated as the peak number of breeding pairs divided by the total area of occupied nesting habitat (nests/m^2). Although photogrammetry technologies (e.g., aircraft type, film and digital cameras, image resolution, and photogrammetry software) changed and evolved during the more than two-decade period when the cormorant colony at the western end of East Sand Island was monitored, either desktop-based or cloud-based Geographic Information System (GIS) platforms were used throughout the study to count occupied cormorant nests visible in vertical aerial images. Beginning in 2006, Brandt's cormorants (*Phalacrocorax penicillatus*) began nesting within the much larger double-crested cormorant colony on East Sand Island, and ground-based observations were incorporated to identify and enumerate Brandt's cormorant nests in GIS-based delineations of habitat used by double-crested cormorants and estimates of double-crested cormorant colony size and nesting density.

During the 1997–2004 breeding seasons, staff in the Survey, Mapping, and Photogrammetry Department at Bonneville Power Administration (BPA) delineated the nesting habitat occupied by double-crested cormorants on East Sand Island, and final estimates of peak colony size and nesting density were derived from nest counts performed by staff at Oregon State University and BPA. We delineated the East Sand Island cormorant colony and estimated colony size and nesting density in-house during the 2005–2014 and 2017–2018 breeding seasons. During the 2015 and 2016 breeding seasons, two different USACE contractors delineated the cormorant colony area and estimated colony size and nesting density based on our established methodologies (David Smith and Associates [2016] and Anchor QEA [2017]). Finally, the USACE, in collaboration with USDA-Wildlife Services (APHIS), estimated the size of the double-crested cormorant colony on East Sand Island during the 2019 breeding season.

4.3.2 Nesting Chronology, Colony Attendance, & Nesting Success

Colony monitors observed the double-crested cormorants nesting at the west end of East Sand Island from elevated observation blinds. Blinds were accessed via above-ground tunnels to prevent disturbance to nesting or roosting cormorants, gulls, and California brown pelicans (*Pelecanus occidentalis californicus*). In each year, we began data collection activities soon after the first double-crested cormorants were observed landing and loafing on the colony area. Colony monitors recorded the date when they detected the first double-crested cormorant eggs, the first hatchling chicks, and the first hatch-year young that were capable of sustained flight (fledglings).

During 2008–2013, high resolution aerial photography of the colony was taken approximately every two weeks throughout the breeding season from early May to early September, rather than just once during peak colony attendance. Attended double-crested cormorant nests were counted in photography to assess colony size through late July, when active nests contain eggs or nestlings, and adult double-crested cormorants were counted to assess colony attendance for the remainder of the nesting season, when most cormorant nests had been vacated. During 2015–2018, the management period, aerial photography was taken, and cormorant nests and adult cormorants were counted each week when cormorants were present on the colony between April/May and August/September with the exception of a 3-week gap in data collection in June 2015 and two 1-week gaps in July 2015 (David Smith and Associates 2016, Anchor QEA 2017, Turecek et al. 2018, Turecek et al. 2019). No colony attendance data were available for the 2019 or 2020 breeding seasons on East Sand Island.

With few exceptions, multiple surveys were completed in each nesting season during 1997–2014 to ascertain when the peak number of breeding pairs were present at cormorant colony sites in the Columbia River estuary other than East Sand Island. The cormorant colony on the Astoria-Megler Bridge (46.21°N, 123.86°W) was surveyed by boat each month from April to July during 1997–2008 and during 2011–2014. In 2009 and 2010, one survey of the Astoria-Megler Bridge was completed during late May. Due to the vantage provided by boat-based surveys, attended double-crested cormorant nests on the bridge were discernable, but eggs and small chicks were not visible. Twelve navigational markers near Miller Sands Spit (located between the Astoria-Megler Bridge and Fitzpatrick Island (river km [Rkm] 51; *Figure 4.1* above) were surveyed by boat or by fixed-wing aircraft multiple times during the cormorant nesting season. During 1997–2010, the peak number of active double-crested cormorant nests on all these channel markers was estimated by adding together the maximum number of attended nests counted at each marker in each season. During 2011–2019, the peak number of breeding pairs on these markers was the maximum number of attended nests counted on a single date for all markers combined. Periodic boat-based, ground-based, and aerial surveys were conducted each year at other potential colony locations within the Columbia River estuary to detect and enumerate any additional double-crested cormorant nests.

During 2015 and 2016, cormorant colony sites within the Columbia River estuary other than East Sand Island were surveyed from the air weekly during the cormorant nesting season. During the 2017–2019 nesting seasons, these colonies were surveyed approximately every 1–3 weeks, except for the Astoria-Megler Bridge. The cormorant colony on the Astoria-Megler Bridge was surveyed by boat weekly during the 2018 nesting season, and three surveys of the bridge colony were conducted during the 2019 nesting season by boat, with aerial imagery obtained using an unmanned aerial vehicle (UAV).

Nesting success (average number of young raised/breeding pair) at the double-crested cormorant colony on East Sand Island was estimated based on observations from blinds of cormorant nest contents from mid-April through July each year during 1997–2013 and in 2018; no data on nesting success were collected during 2014–2017 and in 2019. Colony monitors recorded the number of chicks that survived each week until 28 days of age for a minimum sample of 129 nests located in at least five separate areas of the cormorant colony. Chicks can leave the nest when older than 28 days post-hatching.

4.3.3 Limiting Factors for Colony Size & Nesting Success

Colony size and nesting success of double-crested cormorants nesting on East Sand Island can be limited by a variety of bottom-up and top-down factors, such as availability and quality of forage fish, predation on adults and nest contents, disturbance from predators, human disturbance, availability and quality of nesting habitat, extreme weather events, and tidal flooding. The level of predation pressure on nesting cormorants and their nest contents, a major top-down factor limiting colony size and nesting success, was evaluated through two metrics: (1) the number of bald eagles counted in the vicinity of the double-crested cormorant colony (within 300 m) and (2) the frequency and cause of disturbance events that flushed more than about 5% of the cormorants attending the colony or more than 50% of cormorants attending a satellite colony. The availability and quality of forage fish was measured through diet studies (see *below*) and the availability of nesting habitat was assessed through measurements of nest density (see *above*). Anecdotal data on limiting factors were also collected at the double-crested cormorant colony on East Sand Island, including observations of bald eagles depredating adult cormorants, widespread failure of nests during storms or disturbance events, and flooding of nests in low-lying areas of the colony during high high tide events.

4.3.4 Foraging Behavior

We investigated the foraging behavior of double-crested cormorants nesting on East Sand Island using cormorants tagged with VHF-radio transmitters (n = 85) during the 2000–2001 breeding seasons (Anderson et al. 2004a). We also used cormorants tagged with GPS data loggers (GPS-TDLog; n = 24) to investigate foraging habitat use during the 2014 breeding season (Peck-Richardson et al. 2018). Anderson et al. (2004a) captured cormorants at night using hand-held spotlights and aluminum fish-landing nets during mid- to late May, when cormorants were attending nests with eggs. They tagged cormorants with 10-g VHF radio transmitters manufactured by Advanced Telemetry Systems (ATS; Insanti, Minnesota) and tracked them from fixed-wing aircraft. Based on aerial VHF-telemetry detections, Anderson et al. (2004a) estimated the proportional use of three salinity zones in the Columbia River estuary (*Figure 4.3*) by tagged cormorants during June and July of 2000 and 2001. Peck-Richardson et al. (2018) captured cormorants by hand at night during the late incubation and early chick-rearing periods. Breeding adults were accessed through removable panels on the sides of above-ground tunnels that traversed the colony area, using methods developed by Courtot et al. (2016). Cormorants fitted with GPS data loggers were subsequently recaptured on their nests (n = 21) from the tunnels to retrieve the data loggers for downloading. Peck-Richardson et al. (2018) tagged cormorants with a ca. 41-g GPS-TDLog tag manufactured by Earth and Ocean Technologies (Kiel, Germany) and with an additional 1-g VHS radio transmitter (ATS; Insanti, Minnesota), and retrieved movement data by directly downloading data from the tags. Using geolocated dive records, Peck-Richardson et al. (2018) generated foraging area utilization distributions for GPS-TDLog tagged cormorants throughout the 2014 nesting season using kernel density estimates (KDEs) in ArcMap 10.2.2, based on methods developed by Macleod (2014). Detailed descriptions of capture, tagging, and statistical analysis methods can be found in Anderson et. al (2004a), Courtot et al. (2016), and Peck-Richardson et al. (2018).

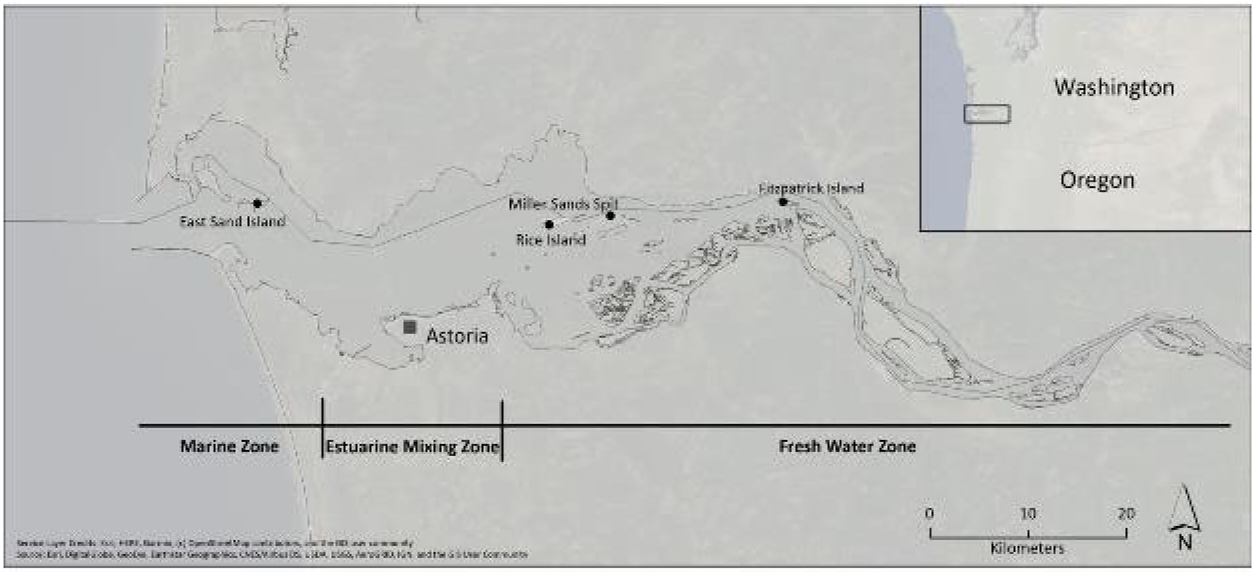

Figure 4.3. Map of the Columbia River estuary showing the three major salinity zones (after Simenstad et al. 1990 as modified by Anderson et al. 2004a). Map shows the locations of East Sand Island (river km 8) in the marine zone of the estuary, the Astoria-Megler Bridge in the estuarine mixing zone of the estuary near Astoria, Oregon, and Rice Island (river km 34) and Miller Sands Spit (river km 38) in the freshwater zone of the estuary.

4.3.5 Diet Composition

We estimated the diet composition (% of prey biomass) of double-crested cormorants nesting at East Sand Island in the Columbia River estuary during 1999–2013 based on the soft-tissue of prey items in the foreguts of lethally collected cormorants. We collected ca. 10 stomach samples from cormorants per week during each breeding season, from late April to the end of July. We identified all undigested soft tissue present in the foreguts of collected cormorants to the taxonomic level of family, and when possible to the genus and/or species level. Although some sampled stomachs contained identifiable bones, only cormorant stomachs that contained soft tissue from prey items were included in the quantitative analysis of diet composition. We identified soft tissue using external features when possible or, when necessary, by artificially digesting the soft tissue to reveal diagnostic bones. Unidentifiable soft tissue lacking diagnostic bones was excluded from analysis.

When possible, we identified salmonids (Chinook salmon [*Oncorhynchus tshawytscha*], coho salmon [*O. kisutch*], chum salmon [*O. keta*], sockeye salmon [*O. nerka*], steelhead trout [*O. mykiss*], and cutthroat trout [*O. clarki clarki*]) in the cormorant diet to species using the morphology of external soft tissues or diagnostic bones. More frequently, however, salmonids were identified to species using genetic techniques. Selected soft tissue samples identifiable as salmonids from collections during 1998–2005 were identified to species using PCR-based genetic techniques (Greig et al. 2002), whereas during 2006–2013 salmonids in foregut samples were identified to species using PCR amplified genetic material following procedures outlined in Purcell et al. (2004).

During the 1998–2005 breeding seasons, we classified Chinook salmon in the cormorant diet as either yearlings or sub-yearlings based on date of sample collection; during the 2006–2013 breeding seasons, however, this classification was based on microsatellite DNA analysis of soft tissue samples (after Seeb et al. 2007) and inferred from the adult run timing of the identified stock. For diet composition analysis and prey consumption calculations, salmonids identified to species were grouped into 4-week periods across the season and, due to small sample sizes for most years, identifiable salmonid prey items within each 4-week period were pooled across all years of the study.

Our cormorant collection procedures and protocols were reviewed and approved by the Institutional Animal Care and Use Committee at Oregon State University. Scientific collection permits were obtained for all procedures involving handling or collection of cormorants from the U.S. Fish and Wildlife Service (Migratory Birds and Habitats, Region 1, Portland, Oregon), the U.S. Geological Survey (Bird Banding Lab, Patuxent, Maryland), and the Oregon Department of Fish and Wildlife (Salem, Oregon).

4.3.6 Smolt Consumption Estimates Based on Bioenergetics Calculations

We constructed bioenergetics models to obtain supportable estimates of the number of juvenile salmonids consumed by double-crested cormorants in the Columbia River estuary (Madenjian and Gabrey 1995, Glahn and Brugger 1995, Derby and Lovvorn 1997). Estimates of annual smolt consumption by double-crested cormorants nesting at the East Sand Island colony were calculated using a bioenergetics modeling approach modified from that described by Roby et al. (2003; Lyons 2010). Briefly, the model begins with estimates of the energy expenditure rates of individual adult double-crested cormorants, expanded by the estimated number of adult cormorants present at the colony. Multiplying by metabolic efficiency produced the total energy requirements of the colony. Estimates of the numbers of juvenile salmonids and other prey types consumed to meet energy demands of the cormorant colony were then derived from measurements of the proportion (% biomass) of salmonids and other prey categories in the diet, the mean mass of individual prey items in each prey category, and the mean energy density of items in each prey category.

We collected data on abundance of adult cormorants present at the East Sand Island colony across the breeding season in all study years from late March to late summer or early fall and collected data on diet composition from late April until the end of July. Energy expenditure rates of adult double-crested cormorants were measured using the doubly-labeled water technique during the chick-rearing stage of the nesting period (Lyons 2010). Energy requirements of chicks were scaled from published values for cormorants and other birds (Weathers 1992, Gremillet et al. 2000, 2003). Prey energy densities were derived from samples collected from Caspian terns nesting in the Columbia River estuary using proximate composition analysis (Reynolds and Kunz 2001). We obtained prey mass from whole fish recovered from Caspian terns and, for larger prey types, from minimally digested samples removed from the stomachs of collected cormorants. We estimated prey consumption for discrete 2-week periods across the cormorant breeding season. We used a Monte Carlo simulation procedure to calculate reliable 95% confidence intervals for estimates of smolt consumption by cormorants (Furness 1978). By estimating the proportion of each salmonid species/run type (i.e. coho salmon, sockeye salmon, yearling Chinook salmon, sub-yearling Chinook salmon, steelhead trout) in the salmonid smolt portion of the cormorant diet based on collected stomach samples (*Table 4.1*), it was possible to estimate using the bioenergetics model the numbers of each salmonid species/run type consumed by double-crested cormorants nesting at East Sand Island on an annual basis.

Table 4.1. Proportional breakdown (by frequency) of salmonids by species/age-class in stomachs of double-crested cormorants collected near East Sand Island in the Columbia River estuary during 2000 – 2013. Salmonid species/age class was identified using molecular genetic techniques. Table modified from Lyons et al. (2014), with permission.

	Time Period			
Species/Age-class	3/27–5/7	5/8–6/4	6/5–7/2	7/3–7/30
Sub-yearling Chinook Salmon	0.03	0.18	0.87	0.91
Yearling Chinook Salmon	0.10	0.19	0.03	0.00
Coho Salmon	0.54	0.32	0.83	0.01
Sockeye Salmon	0.00	0.02	0.00	0.00
Steelhead Trout	0.33	0.29	0.07	0.08
Number of Identified Juvenile Salmonids (n)	68	94	35	25

4.3.7 Predation Rates on Juvenile Salmonids

Methods to estimate predation rates (percentage of available fish consumed) on PIT-tagged juvenile salmonids by double-crested cormorant nesting at colonies in the Columbia River estuary are provided in *Appendix A.* In brief, PIT-tagged smolts determined to be available as prey to cormorants in the estuary, coupled with detections (recoveries) of salmonid tags on cormorant colonies, were used to estimate predation rates via a hierarchical Bayesian model. The model integrated multiple factors of uncertainty in the tag recovery process, including imperfect detection of PIT tags on cormorant colonies, on-colony PIT tag deposition probabilities, and temporal (weekly) changes in fish availability (see *Appendix A*). As noted in *Chapter 1* and *Chapter 2*, accurate predation rate estimates based on PIT tag recoveries on bird colonies in the Columbia River basin have not always been available and some of the estimates reported in earlier technical reports, management plans, and peer-reviewed manuscripts, particularly those published prior to 2014, were reported as minimum estimates of predation rate (e.g., Collis et al. 2001, Ryan et al. 2003, USFWS 2005, Evans et al. 2012, Sebring et al 2013). Herein we report accurate estimates of predation rates by using recently developed models that account for the proportion of consumed PIT tags that were deposited by cormorants on their breeding colony (referred to as deposition probabilities) and the proportion of deposited PIT tags subsequently recovered by researchers following the breeding season (referred to as detection probabilities; Hostetter et al. 2015; see also *Appendix A*).

To evaluate the efficacy of the *Cormorant Management Plan* and of NOAA Fisheries RPA 46 to reduce predation rates on ESA-listed salmonid populations, average annual cormorant predation rates prior to and during management actions on East Sand Island were compared. Management time periods were defined as 2003–2014 (pre-management), 2015–2017 (management, Phase I), and 2018 (management, Phase II). Although Phase I management was implemented starting in 2015, management had little effect on the size of the East Sand Island cormorant colony in 2015. Also, while double-crested cormorants have nested on East Sand Island since 1988, 2003 was the first year that NOAA Fisheries considered predation rates by cormorants to be part of "current" levels associated with Biological Opinions regarding operation of the Federal Columbia River Power System (FCRPS; NMFS 2008). As such, for the purposes of evaluating the efficacy of the *Cormorant Management Plan* to reduce predation rates, the pre-management time series begins in 2003, but predation rate results dating back to 1998 were available and are reported in this chapter and elsewhere in this Synthesis Report (see *Appendix B, Table B23*). The time series ends in 2018, the last year that accurate, standardized estimates of predation rates were available (Evans et al. 2019a).

4.3.8 Pilot Studies of Cormorant Nest Dissuasion Techniques

If resource managers had chosen to reduce cormorant predation on juvenile salmonids in the Columbia River estuary by non-lethally reducing the number of double-crested cormorants nesting on East Sand Island, techniques to effectively discourage nesting on a portion of the East Sand Island colony would have been required. Such techniques could also be used to limit growth of the East Sand Island colony once a target colony size had been achieved. Starting in 2007 and continuing through the 2013 nesting season, we tested the feasibility of several non-lethal methods to discourage nesting by cormorants on East Sand Island without causing the entire colony to abandon. These nest dissuasion methods included (1) erecting artificial perches for bald eagles, (2) erecting visual barriers (i.e. landscape fabric fencing) or physical obstacles (i.e. stakes, ropes, and flagging) within the cormorant colony, (3) covering portions of nesting habitat with pond liner, (4) hazing prospecting cormorants with a green laser, (5) localized on-colony human disturbance (hazing), and (6) human hazing of a significant portion of the cormorant colony on one side of a large visual barrier ("privacy" or "dissuasion" fence).

Bald eagles are known predators of adult double-crested cormorants nesting at the East Sand Island colony. Prior to the 2007 nesting season, two artificial perches made of large driftwood (approximately 4-m high) were erected at the west and east ends of the satellite cormorant colony next to the tidal pond, located in the northwest section of the cormorant colony (*Figure 4.4*). Eagle use of these perches and other areas both within and near the colony was monitored every 30 minutes from the observation tower in the cormorant colony during extended observation periods. These eagle perches were removed after the 2007 nesting season. We also monitored the frequency, magnitude, and duration of all disturbances to nesting cormorants from eagles and other causes (Roby et al. 2008).

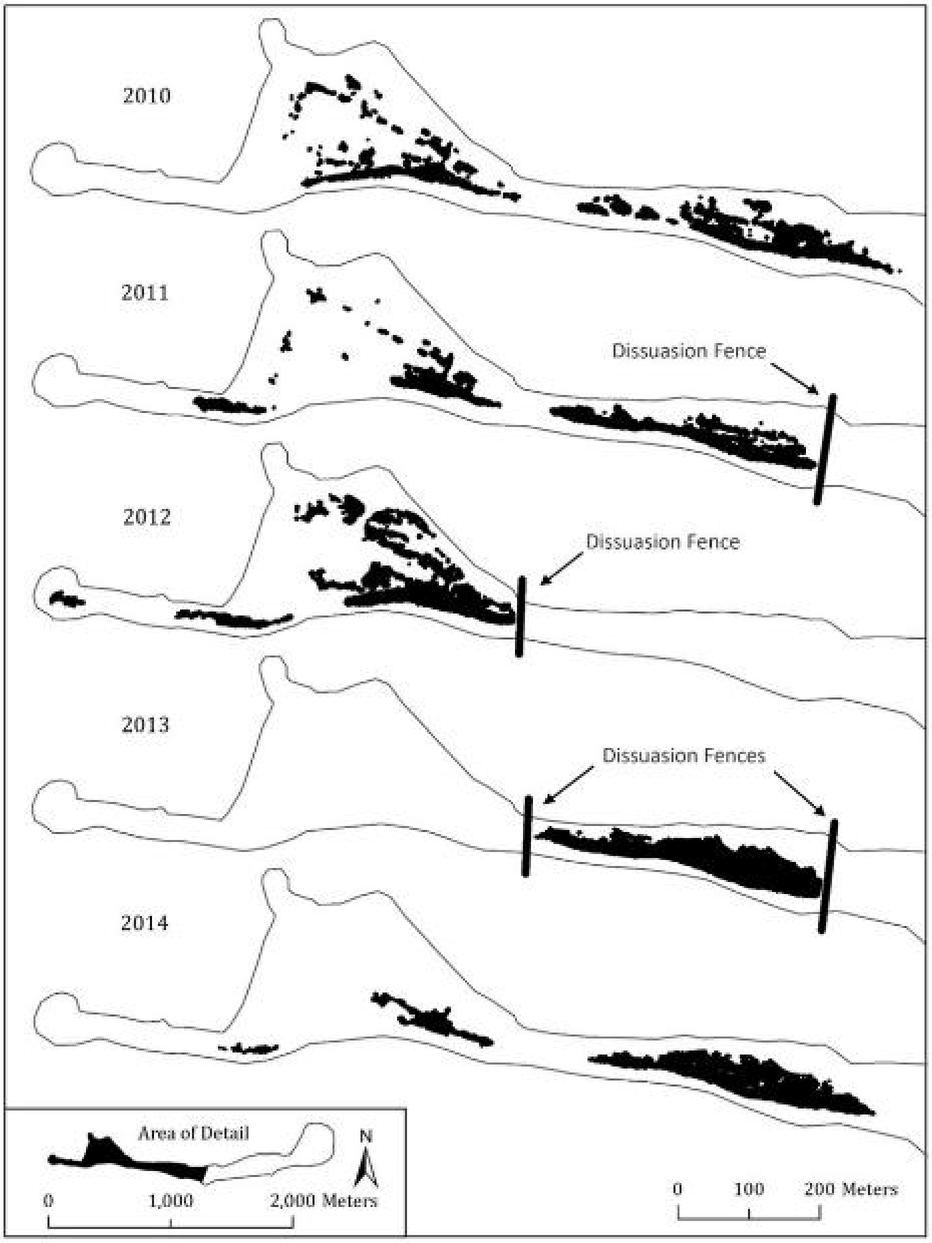

Figure 4.4. The distribution of double-crested cormorant nests (shaded area) on the western half of East Sand Island in the Columbia River estuary from 2010 through 2014. Dissuasion fences were erected within the available nesting habitat during 2011–2013 as part of an effort to test the efficacy of non-lethal techniques to limit the size and distribution of the cormorant colony on East Sand Island.

In order to assess whether rows of landscape fabric fencing would deter cormorants from building nests between or near the fencing, landscape fabric fencing (10-m length x 1.2-m height) was erected in parallel rows on rocky revetment within the cormorant colony prior to the 2007 nesting season. The landscape fabric fencing was erected in four rows spaced at 5-m, 7.5-m, and 10-m intervals. In a separate experimental plot on bare ground in the colony interior, stakes, ropes, and flagging were used to create a net-like pattern between fence stakes within the plot. This passive dissuasion technique did not catch the wind as much as landscape fabric fencing, but still provided a physical barrier to movements of nesting cormorants. These two passive dissuasion plots were monitored for the presence of cormorants both between and immediately adjacent to the plots by direct observation from a nearby observation tower situated within the cormorant colony (Roby et al. 2008).

During the 2009 and 2010 nesting seasons, we tested the efficacy of spreading pond liner material (45 mil thick) over sections of the cormorant colony on East Sand Island in order to limit the area of suitable nesting substrate for cormorants. Pond liner was selected because it is a smooth, durable, and heavy material that would not only eliminate the structure that cormorants prefer for nest-building, but is resistant to cracking or tearing, and does not catch the wind when draped over the ground surface and weighted at the edges. In 2009, an area of about 140 m^2 of the portion of the cormorant colony area where the nesting substrate was rocky revetment was covered with pond liner (Roby et al. 2010). In 2010, an area of about 300 m^2 of the cormorant colony on rocky revetment was covered with pond liner (Roby et al. 2011).

During the 2008 and 2009 nesting seasons, we tested the efficacy of using a 50 milliwatt (mW) green laser (LEM50 laser torch) for dispersing targeted double-crested cormorants from roosting and nesting areas on East Sand Island (Collis et al. 2009, Roby et al. 2010). The laser was acquired in the first week of May 2008, after double-crested cormorants had initiated egg-laying; therefore, testing of the laser for hazing cormorants in 2008 was limited to roosting cormorants and flocks of cormorants that were encountered off-colony. Technicians attempted to haze roosting cormorants daily and to vary the time of day, weather, range to target birds, and light conditions under which the laser was tested. The responses of target individuals and flocks were recorded, and tests of the green laser that resulted in a flushing response by some or all the target cormorants were considered successful. In 2009, prior to the initiation of any nesting on the cormorant colony, an observation blind was built at the terminus of an above-ground tunnel that would allow researchers access to a designated laser hazing area without disturbing nesting cormorants. The selected laser hazing area was adjacent to other nesting cormorants so that the efficacy of the laser to dissuade selected pre-nesting individuals could be compared to other pre-nesting individuals nearby. Beginning on April 15, twice daily (one hour prior to sunrise and at civil twilight), technicians entered the blind, recorded the number of cormorants in the target area, and noted their behavior using a Bushnell Night Vision 26-4050 – Monocular 4 x 50. At 30 min prior to sunrise and 30 min after civil twilight, technicians directed the laser onto the target area of the colony and swept the beam back and forth at the feet of the targeted birds, until the maximum number of cormorants in the target area were flushed. This process was repeated until the target area remained free of birds for one hour (or, in the case of the pre-sunrise dissuasion, until daylight prevented the laser from effectively dissuading any birds from the area). Laser dissuasion trials ceased as soon as evidence of egg-laying was detected in the immediate area. See Roby et al. (2010) for details of the methodology.

Isolated, localized human disturbance of cormorants prospecting for nest sites was tested in 2008 and 2009 as a potential method to discourage double-crested cormorants from nesting on a portion of the East Sand Island colony. In early April, prior to the initiation of any breeding, a visual barrier (a fence of black landscape fabric, 1.5-m tall) was erected to isolate a small section of the eastern portion of the double-crested cormorant colony. In 2007, this eastern-most section of the cormorant colony occupied an area of roughly 1,150 m^2 and included approximately 1,000 cormorant nests (ca. 7% of the 2007 colony size). An above-ground tunnel was built prior to the nesting season in 2008 and 2009 to allow researchers access to this area of the colony without disturbing nesting cormorants (Collis et al. 2009, Roby et al. 2010). Beginning 1 May, and on multiple occasions during the week immediately prior to the expected date of first egg-laying, a single researcher emerged from the tunnel onto that section of the cormorant colony, thereby flushing cormorants from the area. The researcher remained in view of cormorants for a short period, initially less than 3 min, before withdrawing into the tunnel. During these disturbances, additional researchers situated at three different vantage points observed the reaction of the cormorants, recording the number of cormorants affected (including any non-target individuals) and the duration of cormorant absence from the disturbed area. Because this was a pilot study, the length and frequency of the disturbances were varied in order to achieve the desired effect of dissuading prospecting cormorants. Disturbances ceased as soon as evidence of egg-laying by cormorants was detected in that part of the colony.

In 2011, 2012, and 2013, we tested the feasibility of dissuading double-crested cormorants from nesting on a large portion of the former colony area on East Sand Island using privacy (“dissuasion”) fences, nest destruction, and targeted human hazing (Roby et al. 2012, 2013, 2014; *Figure 4.4* above). This active dissuasion approach built on our experiences testing localized human disturbance on the East Sand Island colony during 2008 and 2009. In 2011, we tested the feasibility of dissuading cormorants from nesting on East Sand Island in an area where approximately 15% of the colony had nested in 2010. In 2010, approximately 1,500 double-crested cormorant nests were in the 2011 dissuasion area. Prior to the arrival of breeding cormorants, a 2.4-m high by 65-m long dissuasion fence was erected across the cormorant colony in 2011 (*Figure 4.4* above). Afterward, we attempted to prevent cormorants from nesting on the east side of the fence, while minimizing disturbance to cormorants nesting west of the visual barrier that was created by the dissuasion fence. In addition to the dissuasion fence, an observation blind and tunnel system were constructed to provide researchers access to an enclosed researcher camp without disturbing nesting cormorants outside of the targeted dissuasion area. The enclosed camp concealed all researcher activity from cormorants within the dissuasion area, as well as those cormorants nesting west of the dissuasion fence, and provided an elevated vantage point for observations on either side of the dissuasion fence. To augment the effectiveness of dissuasion efforts, all nest structures that were constructed by cormorants in the dissuasion area prior to the start of hazing were destroyed by scattering the nesting materials (Roby et al. 2012).

In 2012, we repeated and expanded efforts to test the feasibility of dissuading double-crested cormorants from nesting on a portion of their nesting colony on East Sand Island. The dissuasion area was increased in 2012 to 62% of the area used by nesting cormorants in 2010. A dissuasion (privacy) fence (2.4 m high by 25 m long) was erected across the cormorant colony (*Figure 4.4* above) and an attempt was made to prevent cormorants from nesting to the east of the fence, while minimizing the disturbance to cormorants nesting to the west of the fence. The cormorant nest dissuasion area encompassed an area of approximately 6.5 acres (2.6 ha), and the linear distance from the dissuasion fence east to where the eastern-most cormorants nested in 2010 was 0.65 km. Two techniques to dissuade cormorants from nesting on the east side of the dissuasion fence were investigated in concert: human disturbance and destruction of existing cormorant nests (i.e. scattering of sticks used to form nests; Roby et al. 2013).

In 2013, two dissuasion fences were erected in the same locations as the fences erected in 2011 and 2012 (2.4 m high by 25 m long and 65 m long, respectively; *Figure 4.4* above), and hazing was used to prevent cormorants from nesting east of the eastern fence and west of the western fence. In 2013, double-crested cormorants were dissuaded from nesting in ca. 75%, or about 12 acres (4.8 ha), of the available nesting habitat used in 2010, leaving them with 4 acres (1.6 ha) of habitat for nesting between the two dissuasion fences (*Figure 4.4* above). Based on the amount of habitat occupied by nesting cormorants on East Sand Island over the previous 16 years, generally less than 3 acres (1.2 ha: BRNW *unpubl. data*), the dissuasion experiment in 2013 still provided ample nesting habitat to accommodate all the cormorants that had previously nested on East Sand Island. Two camps, six observation blinds, and an above-ground tunnel system were constructed in addition to the two dissuasion fences to provide researchers access to the area without disturbing nesting cormorants outside of the targeted dissuasion areas. The camps concealed all non-hazing researcher activity from cormorants within the two dissuasion areas, as well as from those cormorants nesting between the two dissuasion fences. Observation blinds adjacent to each dissuasion fence provided elevated vantage points for observations on both sides of the dissuasion fences (Roby et al. 2014).

To help evaluate the efficacy of dissuasion efforts and to determine whether hazed double-crested cormorants dispersed from East Sand Island and nested elsewhere, we captured and marked cormorants in the dissuasion areas prior to the 2011, 2012, and 2013 nesting seasons (see section *4.3.10 Dispersal & Colony Connectivity*).

4.3.9 Pilot Studies of Social Attraction & Colony Relocation

Studies to explore the feasibility of using techniques of social attraction and habitat enhancement to relocate nesting double-crested cormorants to alternative colony sites within the Columbia River estuary were conducted during 2004–2007. These studies sought to determine whether social attraction and habitat modification techniques could be used to induce double-crested cormorants to nest in areas where they have not previously nested and, if so, whether these techniques could be used to redistribute some of the double-crested cormorants nesting in the Columbia River estuary to alternative colony sites outside the estuary. Whereas these techniques had been demonstrated effective for relocating or re-establishing breeding colonies of Caspian terns (see *Chapter 1* and *Chapter 3*), no previous studies had demonstrated the efficacy of social attraction and habitat enhancement techniques for double-crested cormorants (Feldman 2011, Wires 2014).

Feasibility studies were first conducted on East Sand Island adjacent to the active double-crested cormorant colony on the western half of the island. We created two test plots in 2004, three test plots in 2005, and six test plots in both 2006 and 2007 in the interior of East Sand Island to evaluate whether nesting habitat enhancement and social attraction techniques would induce cormorants to nest in areas where cormorants had not previously nested (see Suzuki et al. 2015). Briefly, we deployed 12 double-crested cormorant decoys (Mad River Decoys, Waitsfield, Vermont) in sitting and incubating postures and an audio system with one or two speakers that broadcast cormorant vocalizations in each plot where social attraction techniques were tested. Vocalizations of double-crested cormorants were digitally recorded from actively breeding birds using waterproof sound-recording equipment during the incubation period at the East Sand Island colony (Alaska's Spirit Speaks, Fairbanks, Alaska) and broadcasted using solar-charged sound systems (Murremaid Music Boxes, South Bristol, Maine).

During the 2004 breeding season, we tested two types of habitat enhancement techniques. We set up a 41-m^2 plot using pieces of driftwood that were moved to the plot to create structure similar to natural habitat used by nesting double-crested cormorants elsewhere on the island. This driftwood plot was filled with small woody debris to supply abundant nest-building material for prospecting cormorants. A few dozen old cormorant nests from previous breeding seasons were also added to the plot. We created the second test plot with a square array of 49 old truck tires laid out in a 99-m^2 area. Each tire was filled with sand and topped with a cormorant nest used in the previous year. The two plots were approximately 20 m apart and separated by a gully and sparse herbaceous vegetation (Suzuki et al. 2015).

During the 2005 breeding season, we tested whether cormorants could be induced to nest on artificial structures by constructing two wooden platforms measuring 5 m × 5 m and elevated approximately 0.7 m above ground level, in two different areas of the East Sand Island colony where cormorants had not previously nested (Collis et al. 2006). Thirty-six old truck and car tires, each filled with sand and topped with a single cormorant nest, were placed on each platform. The distance between the platforms was approximately 30 m, and the platforms were separated by a gully and sparse herbaceous vegetation. Details on how these test plots and control plots within the main cormorant colony were monitored are provided in Suzuki et al. (2015).

During the 2006 and 2007 nesting seasons we established and monitored six habitat modification and social attraction plots in the interior of East Sand Island outside of the area previously used by nesting cormorants (*Figure 4.5*). Four of the six plots were on wooden platforms elevated slightly above the sandy substrate, while two were situated on the ground. Two of the four plots on elevated wooden platforms were equipped with social attraction (decoys and audio playback of vocalizations) and two were not. All six plots had a 6 x 6 array of old tires with fine woody debris on top as nesting substrate (*Figure 4.5*).

Figure 4.5. Three test plots for double-crested cormorant habitat enhancement and social attraction on East Sand Island in the Columbia River estuary during 2006. These plots were adjacent to the active double-crested cormorant colony, but in areas not previously used by nesting cormorants. Each plot consisted of a 6 x 6 array of old tires with a previously used cormorant nest on top of each tire. A dozen cormorant decoys and one speaker continuously broadcasting cormorant vocalizations were deployed on the nearest and furthest plots.

We also established test plots for social attraction and habitat enhancement on other islands in the Columbia River estuary. In 2006, we created a test plot at the downstream end of Rice Island (*Figure 4.1* above), near where double-crested cormorants had nested as recently as 2003. In a test plot that measured 4.0 m × 4.3 m we placed a total of 36 old tires filled with fine woody debris, 40 cormorant decoys, and an audio system with four speakers that broadcasted recordings of cormorant vocalizations (*Figure 4.6*). The plan was to remove the test plot after the 2006 breeding season, regardless of whether cormorants attempted to nest there or not. If the plot was used by nesting cormorants in 2006, removal of the habitat enhancement and social attraction after the 2006 breeding season would assess whether cormorants would nest at the same site in subsequent years without habitat enhancement and social attraction (see Suzuki et al. 2015 for a detailed description of data collection techniques).

Figure 4.6. Social attraction test plot for double-crested cormorants at the downstream end of Rice Island in the upper Columbia River estuary during 2006, before colonization of the site by nesting cormorants (top pane) and after colonization by 35 pairs of double-crested cormorants (bottom pane). The test plot consisted of 36 old tires with fine woody debris filling the centers, 40 cormorant decoys, and an audio playback system broadcasting double-crested cormorant vocalization from four patio speakers.

We also created a test plot at the downstream end of Miller Sands Spit (*Figure 4.1* above) each year prior to the 2004–2008 breeding seasons; the test plot was located about 25 km east of the East Sand Island cormorant colony, at a site where a few double-crested cormorants had attempted to nest in 2001 but were unsuccessful. In 2004, we gathered driftwood into a 10 m × 8 m plot and filled the plot with fine woody debris suitable as nest material. A total of 62 cormorant decoys and an audio system with two speakers that broadcasted cormorant vocalizations were placed in the plot. In 2005, we created a smaller driftwood plot (8 m × 5 m) with 24 cormorant decoys and two speakers in the same area of the island, but with 25 old tires filled with fine woody debris to simulate nesting material. In the subsequent two years (2006–2007), we repeated the test at the same site using larger numbers of decoys (40), tires (36), and speakers (4) placed in an even smaller plot (4.5 m × 4.5 m; *Figure 4.7*). Finally, in 2008, the fifth and final year of deploying cormorant social attraction at the downstream end of Miller Sands Spit, decoys and old tires were deployed, but no audio playback system.

Figure 4.7. Social attraction test plot for double-crested cormorants at the downstream end of Miller Sands Spit in the upper Columbia River estuary during 2007, before colonization of the site by nesting cormorants (top panel) and after colonization by 90 pairs of double-crested cormorants (bottom panel). The test plot consisted of 36 old tires with fine woody debris filling the centers, 40 cormorant decoys, and an audio playback system broadcasting double-crested cormorant vocalization from four patio speakers.

In 2005, a test plot was created on top of a small rocky islet at the mouth of Trestle Bay (*Figure 4.1* above; Collis et al. 2006), a site with no prior history of cormorant nesting, although small numbers of double-crested cormorants had previously nested on nearby railroad trestles in Trestle Bay as recently as 1992 (Carter et al. 1995). The plot, which was located about 5 km south of the cormorant colony on East Sand Island, measured approximately 10 m × 20 m in size and was set up with 26 cormorant decoys, 24 old car and truck tires, and an audio system with two speakers that broadcasted cormorant vocalizations. The center of each tire was filled with fine woody debris (Suzuki et al. 2015).

In 2007, 2008, and 2009 we attempted to employ habitat enhancement and social attraction techniques to induce double-crested cormorants to nest at a site where cormorant nesting attempts had not been previously recorded. We selected Fern Ridge Wildlife Area near Eugene, Oregon because it supported significant numbers of cormorants during the non-breeding season and we were able to obtain permission to launch a floating platform in one of the impoundments in the Fisher Butte Unit, where access by the public was restricted. A floating platform, about 9 m (30 feet) long by 4.5 m (15 feet) wide, was constructed from sections of floating dock material. Plywood sideboards about 0.3 m (1 foot) high were attached to the sides of the floating platform to retain material on the platform. Forty-eight old tires were placed on the platform and fine woody debris was placed in each tire for nesting material. Forty hand-painted double-crested cormorant decoys (Mad River Decoy, Waitsfield, Vermont) were then secured on the platform. Finally, two audio playback systems (Murremaid Music Boxes, South Bristol, Maine), each with two speakers, were placed on the platform, along with solar panels and deep cycle batteries for powering the playback systems. Before the beginning of April the floating platform was towed out into Fisher Butte cell #2 and anchored in about 1 meter of water, about 150 m from the nearest dike. The platform was checked from the dike weekly during the first two months of the nesting season for signs of cormorant nesting and to verify that the audio playback system was functioning (Roby et al. 2008).

In 2010, we moved the floating platform with associated nest substrate enhancement and social attraction to Summer Lake Wildlife Area in south-central Oregon. The floating platform was deployed on Dutchy Lake, a permanent water body near the northern boundary of the Wildlife Area. Double-crested cormorants had occasionally nested in small numbers in Summer Lake Wildlife Area prior to 2009. When the floating platform was deployed on Dutchy Lake, however, there were no established cormorant nesting colonies in or near the Wildlife Area, although a few double-crested cormorants were summer residents at the Wildlife Area (Roby et al. 2011).

In 2011, the floating platform and associated social attraction was again deployed on Dutchy Lake in Summer Lake Wildlife Area, Oregon. In addition, a cormorant social attraction plot was set up on the 2-acre (0.8-ha) Caspian tern island constructed by the USACE in Tule Lake Sump 1B, Tule Lake National Wildlife Refuge, near Tulelake, California (Roby et al. 2012). The 2011 nesting season was the first following island construction when Sump 1B was re-watered sufficiently to create an island (see *Chapter 3*). There was a small cormorant colony on a small natural island in Tule Lake Sump 1B prior to 2011. In early April, we placed 30 old tires, 22 hand-painted decoys of adult double-crested cormorants (Mad River Decoys, Vermont), two audio playback systems with two speakers each (Murremaid Music Boxes, Maine), and associated solar panels and batteries on or near the rocky rip-rap located on the southwest side of the tern island in Tule Lake Sump 1B. In early August, we added 34 decoys representing double-crested cormorant chicks (Dave Smith Decoys, Lebanon, Oregon) within the tires. The cormorant social attraction plot was checked upon arrival and departure from the island, and, during Caspian tern colony monitoring, the plot was scanned hourly for 3 min for any signs of cormorant presence in or near the social attraction site.

In 2012, habitat enhancement (i.e. placement of old tires filled with nesting material) and social attraction techniques (i.e. cormorant decoys and audio playback systems) were used for a second year on the USACE-constructed tern island in Sump 1B at Tule Lake National Wildlife Refuge and, for the first time, on the USACE-constructed tern island in Malheur National Wildlife Refuge to attract double-crested cormorants to nest at those two sites (Roby et al. 2013). We chose Tule Lake Sump 1B and Malheur Lake to continue feasibility studies after previous attempts to attract double-crested cormorants to nest using similar techniques on a floating platform had failed at both Fern Ridge Reservoir in the Willamette Valley (2007–2009) and at Dutchy Lake in the Summer Lake Wildlife Area (2010–2011). Human disturbance and a paucity of breeding age cormorants in these two areas were believed to have played a role in the lack of cormorant nesting activity at either of these sites. We thought that these factors might not pertain at the tern islands on Tule Lake Sump 1B or on Malheur Lake given that double-crested cormorants had previously nested near both new tern islands. There was an established arboreal colony of double-crested cormorants at Sodhouse Ranch, which is within sight of the artificial tern island in Malheur Lake. In addition, high water levels in 2011, the year before the artificial tern island at Malheur Lake was constructed, created a natural island on the north side of the lake at the Singhus Ranch property, whereupon a small double-crested cormorant nesting colony quickly formed within a larger colony of American white pelicans (*Pelecanus erythrorhynchos*), Caspian terns, and California and ring-billed gulls (*Larus californicus* and *L. delawarensis*).

4.3.10 Dispersal & Colony Connectivity

4.3.10.1 Color banding and band resighting/recovery – A total of 2,209 double-crested cormorants, including both adults and hatch-year young, were banded at the East Sand Island breeding colony during 1997–2014 (*Table 4.2*). Banding efforts were conducted in concert with telemetry studies to investigate foraging behavior in the Columbia River estuary and post-breeding dispersal from the estuary. Beginning in 2010, we banded double-crested cormorants to investigate breeding dispersal from and colony connectivity between the East Sand Island colony and other colonies that are part of the western population of the species. We banded each cormorant under federal permit with at least a metal leg band provided by the U.S. Geological Survey – Bird Banding Lab. During 1997–2001, cormorants were banded only with federal metal leg bands (n = 135), each with a unique numerical code. Starting in 2008, most banded cormorants (n = 1,985) were also marked with a field-readable Darvic plastic leg band engraved with a unique alphanumeric code on one leg and a federal numbered metal band on the other. The use of Darvic plastic leg bands for individually marking double-crested cormorants provided a more reliable protocol for field identification of individual cormorants at a distance. As part of banding studies, flightless cormorant chicks that were near fledging age were captured on-colony at the East Sand Island colony. Cormorant chicks were captured by herding flightless young into holding pens. During 1997–2001, we captured adult cormorants on-colony for banding using hand-held spotlights and aluminum fish-landing nets. Beginning in 2008, we captured adult cormorants on-colony for banding through removable panels on the sides of above-ground tunnels that bisected the colony area. Resightings of live banded cormorants or recoveries of leg-bands from dead birds were reported via the U.S. Geological Survey - Bird Banding Lab (BBL).

Table 4.2. Numbers of double-crested cormorants banded under federal permit at East Sand Island in the Columbia River estuary during 1997–2014. Starting in 2008, we used field-readable Darvic leg bands engraved with a unique alphanumeric code.

	USGS Metal Band Only		**Alphanumeric Band**		
Year	Adult	Hatch-Year	Adult	Hatch-Year	**Total**
1997	--	6	--	--	**6**
1998	13	4	--	--	**17**
1999	--	1	--	--	**1**
2000	57	1	--	--	**58**
2001	51	2	--	--	**53**
2002	--	--	--	--	**--**
2003	--	--	--	--	**--**
2004	--	--	--	--	**--**
2005	--	--	--	--	**--**
2006	--	--	--	--	**--**
2007	--	--	--	--	**--**
2008	28	--	--	8	**36**
2009	59	--	--	82	**141**
2010	1	--	125	13	**139**
2011	--	--	216	184	**400**
2012	1	--	274	438	**713**
2013	--	--	201	420	**621**
2014	--	--	24	--	**24**
Total	**210**	**14**	**840**	**1145**	**2209**
	Subtotal	**224**	**Subtotal**	**1985**	

4.3.10.2 Satellite telemetry studies – As part of our efforts to measure post-breeding dispersal and movements of double-crested cormorants nesting at East Sand Island, we captured and fitted adult cormorants with Platform Terminal Transmitter (PTT) tags during 2008–2009 (Courtot et al. 2012) and 2012–2013 (Peck-Richardson 2017). During both studies, nesting cormorants were captured by hand, at night, through removable panels built into the sides of an above-ground tunnel network within the cormorant colony. Once a nesting cormorant was captured, Courtot et al. (2012) attached either a battery-powered PTT (Kiwisat 202 [50 g]; Sirtrack Ltd., Havelock North, New Zealand) or a solar-powered PTT (GS55 [50 g] or GS25 [39 g]— High Water Pressure model; North Star Science and Technology, LLC, King George, Virginia) using a harness made of polytetrafluoroethylene (Teflon®) ribbon (described by Dunstan 1972, modified by King et al. 2000), or surgically implanted a coelomic PTT with an external antenna (Kiwisat implant [33 g and 46 g]; Sirtrack Ltd.; Hupp et al. 2006). Peck-Richardson (2017) attached either a Kiwisat 202 PTT (50 g; Sirtrack Ltd.) or a modified Kiwisat 303 PTT (50 g; Sirtrack Ltd.) using a Teflon® ribbon harness. In addition to attaching a PTT, Courtot et al. (2012) banded each captured cormorant with a USGS metal leg-band, while Peck-Richardson (2017) banded each captured cormorant with a USGS metal leg-band on one leg and a field-readable Darvic plastic leg band engraved with a unique alphanumeric code on the other. In both studies, PTTs and attachment materials weighed less than 3.2% of the tagged cormorant's body weight.

During the Courtot et al. (2012) and Peck-Richardson (2017) studies, positional data from satellite-tagged cormorants were recorded via the Argos satellite system (Woods Hole Group [formally CLS America, Inc.], Lanham, Maryland), and Argos assigned preliminary location classes (LCs; 3, 2, 1, 0, A, B, and Z, in descending order of accuracy), based on estimated accuracy of the location data. Location data were filtered using the Douglas Argos-Filter Algorithm version 7.03 (USGS, Alaska Science Center, https://alaska.usgs.gov/science/biology/spatial/douglas.html, accessed 12 April 2010 and 20 January 2013), utilizing SAS software version 9.3 (© 2011, SAS Institute, Cary, North Carolina). In both studies, all LC3 locations and locations within 2 km of the previous location were retained, consecutive locations requiring flying speeds > 70 km/hr were excluded, and the best single location per "on" period was then selected based on the LC and the most messages received during a satellite overpass. Our resulting roost location positions were estimated to have location errors of less than 1.5 km (LCs 3, 2, or 1; Argos 2007).

Courtot et al. (2012) and Peck-Richardson (2017) each described roosting sites of PTT-tagged cormorants relative to regional boundaries and active or historical cormorant colony sites. Courtot et al. (2012) adopted the regional boundaries of Carter et al. (1995), and the comprehensive review of double-crested cormorant nesting sites west of the Continental Divide by Adkins and Roby (2010), and used all of the known cormorant nesting sites documented in Adkins and Roby (2010) to categorize nesting sites as either active or inactive based on confirmed nesting during the 2008 or 2009 breeding seasons; nesting sites that were inactive during 2008 and 2009 were classified as historical nesting sites. Peck-Richardson (2017) adopted and modified the regional boundaries from Adkins and Roby (2010) and Courtot et al. (2012), while obtaining information on active/historical double-crested cormorant colonies from Adkins and Roby (2010) and the Pacific Flyway Council's Management Framework for double-crested cormorants (Pacific Flyway Council 2012). Detailed descriptions of capture, tagging, and statistical analysis methods can be found in Courtot et al. (2012) and Peck-Richardson (2017).

4.3.11 Lethal Methods to Reduce Colony Size

Phase I of the *Cormorant Management Plan* was intended for implementation during the 2015–2018 breeding seasons at East Sand Island (USACE 2015) but was only implemented during 2015-2017. Before the arrival of nesting cormorants in each year, the USACE constructed a 2.4-m high x 45-m long chain-link fence fitted with vinyl slats (i.e. dissuasion fencing) with an above-ground tunnel in parallel; the dissuasion fence and above-ground tunnel bisected the island near the eastern extent of the cormorant colony during the pre-management period (46.26°N, 123.98°W; *Figure 4.8*). This infrastructure was built to provide a visual barrier that helped conceal human activities on the island and to provide access to the cormorant colony while conducting egg-oiling forays and any colony monitoring efforts necessary to determine nest locations of double-crested cormorants and non-target species (i.e. Brandt's cormorants and glaucous-winged/western gulls). The USACE contracted with USDA-Wildlife Services to conduct lethal management (i.e. culling adult cormorants and oiling cormorant eggs on-colony), actions associated with Phase I of the *Cormorant Management Plan*. At East Sand Island, cormorant eggs present in nests selected for oiling were treated by USDA-Wildlife Services staff with food-grade corn oil to suppress embryo development and prevent eggs from hatching. Corn oil was applied using a backpack sprayer and access to the colony for the purpose of egg oiling was permitted during both daytime and nighttime; egg oiling was expected to occur every 2–3 weeks during the months of May through July (USACE 2015). During 2015–2017, USDA-Wildlife Services was contracted to oil up to ca. 46% of all double-crested cormorant nests at East Sand Island. During 2018–2019, USDA-Wildlife Services was permitted to take a maximum of 500 eggs from areas east of the dissuasion fence as a supplement to nonlethal dissuasion methods, such as human hazing and visual deterrents (e.g., eagle decoys and balloons), used to restrict nesting attempts by double-crested cormorants to the area west of the dissuasion fence.

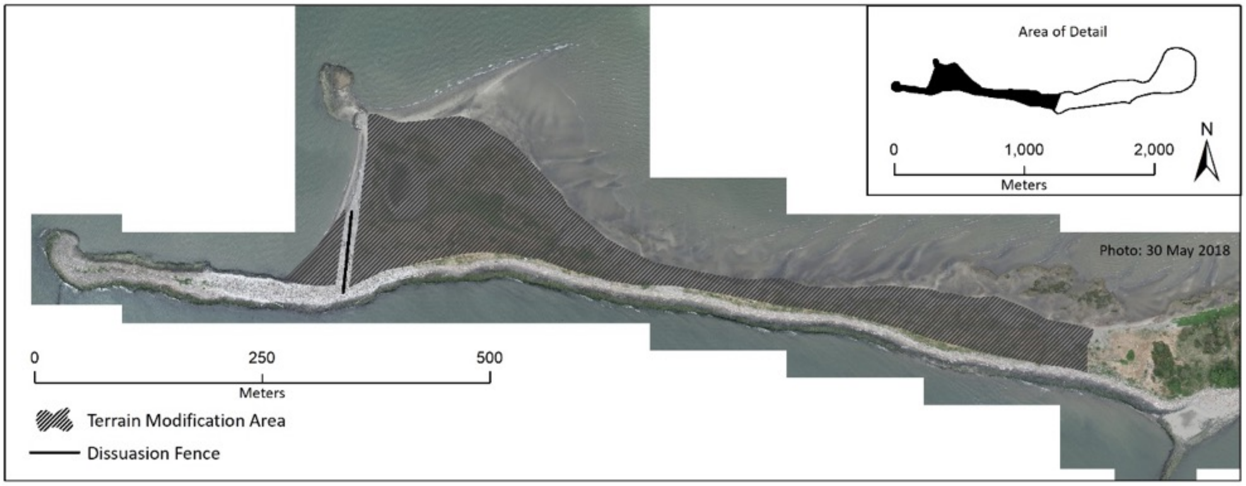

Figure 4.8. Upland terrain (shaded area) modified to restrict nesting habitat at the west end of East Sand Island as part of Phase II of the Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary during the winter of 2018–2019 overlaid on an aerial image taken during the 2018 breeding season. The solid black line indicates the location of the dissuasion fence; designated cormorant nesting habitat was restricted to the rock jetty west of the dissuasion fence.

USDA-Wildlife Services conducted boat-based lethal culling of adult double-crested cormorants over the water within 25 km of East Sand Island beginning in April and continuing throughout the breeding season during 2015 and 2016. In 2017, USDA-Wildlife Services began boat-based culling of adult double-crested cormorants in April but ceased operations after cormorants abandoned the nesting colony on East Sand Island in May and June. Boat-based culling of adult double-crested cormorants was conducted using shotguns ≤ 10-gauge in size and 0.22 caliber rifles (USACE 2015).

4.3.12 Reduction of Nesting Habitat to Limit Colony Size

Phase II of the *Cormorant Management Plan* was implemented during the 2018–2019 breeding seasons at East Sand Island (USACE 2015). Before the arrival of nesting cormorants in 2018, the USACE constructed a 2.4-m high x 50-m long chain-link fence fitted with vinyl slats (i.e. dissuasion fencing) that bisected the island about 75 m east of the base of the western jetty on East Sand Island (46.26°N, 123.99°W; *Figure 4.8* above), which marked the eastern boundary of the ca. 1.7-acre (ca. 0.69-ha) designated cormorant nesting area. During the winter of 2018–2019, the USACE implemented the terrain modification component of Phase II of the *Cormorant Management Plan*, wherein ca. 12 acres (ca. 4.9 ha) of historical upland cormorant nesting habitat on the western half of East Sand Island was converted into a tidally-influenced mudflats. Terrain modification was completed using heavy excavation equipment by a USACE contractor (LKE Corporation, Washougal, WA). In addition to terrain modification that lowered the elevation of the north side of the western half of the island, a ca. 90-m long dissuasion fence was constructed along a north-south running rock jetty prior to the 2019 breeding season (*Figure 4.8* above). Similar to the dissuasion fences constructed during Phase I of the management plan, the dissuasion fence built during Phase II was constructed to provide a visual barrier that concealed human activities east of the fence from cormorants nesting west of the fence. During 2018–2019, active hazing was conducted by USDA-Wildlife Services personnel using visual deterrents (e.g., eagle decoys and balloons) deployed to restrict nesting attempts by double-crested cormorants to the area west of the dissuasion fence.

4.4 Results

4.4.1 Colony Size, Nesting Habitat Use, & Nesting Density

4.4.1.1 Colony size – The number of breeding pairs of double-crested cormorants nesting at each colony site in the Columbia River estuary during 1997–2019 are presented in *Table 4.3*. The total number of double-crested cormorants nesting in the estuary gradually increased from 6,358 breeding pairs in 1997 to 13,983 breeding pairs in 2006. During 2007-2014, the total number of double-crested cormorants nesting in the Columbia River estuary remained relatively stable; the highest total number of double-crested cormorants nesting in the Columbia River estuary (15,484 breeding pairs) was recorded in 2013. Following implementation of the *Cormorant Management Plan*, the total number of breeding pairs of double-crested cormorants nesting in the Columbia River estuary declined; in 2019 only 4,323 breeding pairs of double-crested cormorants nested throughout the estuary (*Table 4.3*).

Table 4.3. Estimated seasonal peak number of breeding pairs of double-crested cormorants at colony sites in the Columbia River estuary during 1997–2019. The average annual number of breeding pairs is provided separately for the pre-management period (1997–2014) and the management period (2015–2019). Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data for 2019 provided by the U.S. Army Corps of Engineers.

	Lower Estuary			**Upper Estuary**						
Year	Channel Markers[a]	East Sand Island	Desdemona Sands	Astoria-Megler Bridge	Rice Island	Miller Sands Spit	Channel Markers[b]	Lewis & Clark Bridge	Total	% at East Sand Island
1997	--	5,023	130	0	1,141	0	64	--[f]	6,358	79.0%
1998	--	6,285	120	0	795	0	70	--[f]	7,270	86.5%
1999	--	6,561	--	0	0	0	--	--[f]	6,561	100.0%
2000	--	7,162	61	0	0	0	150	--[f]	7,373	97.1%
2001	--	8,120	0	0	150	12	75	--[f]	8,357	97.2%
2002	--	10,230	0	0	53	0	129	--[f]	10,412	98.3%
2003	--	10,646	0	0	211	0	183	--[f]	11,040	96.4%
2004	--	12,480	0	6	0	0	194	--[f]	12,680	98.4%
2005	--	12,287	0	14	0	0	208	--[f]	12,509	98.2%
2006	--	13,738	0	7	35	41	162	--[f]	13,983	98.2%
2007	--	13,771	0	11	0	90	160	--[f]	14,032	98.1%
2008	--	10,950	0	20	0	129	216	--[f]	11,315	96.8%
2009	--	12,087	0	24	0	0	235	--[f]	12,346	97.9%
2010	--	13,596	0	63	0	0	254	--[f]	13,913	97.7%
2011	--	13,045[c]	0	60	0	0	248	--[f]	13,353	97.7%
2012	8	12,301[c]	0	139	0	0	245	--[f]	12,693	96.9%
2013	7	14,916[c]	0	231	0	0	330	0	15,484	96.3%
2014	5	13,626	0	333	0	0	318	11	14,293	95.3%
2015	5	12,150[d]	0	425	0	0	319	122	13,021	93.3%
2016	18	9,772[d]	0	549	0	0	300	102	10,741	91.0%
2017	7	544[d]	0	834	0	0	204	147	1,736	31.3%
2018	41	3,672[e]	0	1,737	0	0	293	201	5,944	61.9%
2019	73	350[e]	0	3,542	0	0	219	139	4,323	8.1%
Pre-Mgmt Avg	**7**	**10,935**	**18**	**50**	**133**	**15**	**191**	**6**	**11,332**	**96.5%**
Mgmt Avg	**29**	**5,298**	**0**	**1,417**	**0**	**0**	**267**	**142**	**7,153**	**74.1%**

[a] *Channel markers located between the mouth of the Columbia River (Rkm 0) and the Astoria-Megler Bridge (Rkm 22).*

[b] *Channel markers located between the Astoria-Megler Bridge (Rkm 22) and Fitzpatrick Island (Rkm 51).*

[c] *Pilot study to test efficacy of non-lethal techniques for deterring nesting on part of the colony area.*

[d] *Phase I of the* Cormorant Management Plan*, which included culling of adults and oiling cormorant eggs in nests.*

[e] *Phase II of the* Cormorant Management Plan*, which included non-lethal management to limit the size of the cormorant colony; most cormorant nesting habitat on East Sand Island was converted to intertidal mudflat between the 2018 and 2019 nesting seasons.*

[f] *Very few if any nests present.*

Trends in the number of double-crested cormorants nesting throughout the Columbia River estuary during 1997–2019 were driven primarily by the size of the colony on East Sand Island (*Figure 4.1* above). In most years during the study period, greater than 95% of double-crested cormorants nesting in the Columbia River estuary nested at the colony on East Sand Island; the exceptions were the first two years of the study period (1997 & 1998), when a moderate-sized colony was present on Rice Island in the upper estuary (*Figure 4.1* above), and each of the five years during the management period (2015–2019; *Table 4.3* above). During the management period, the proportion of total breeding pairs in the estuary that nested at East Sand Island decreased from 93.3% in 2015 to 8.1% in 2019. The decline in the proportion of total breeding pairs in the estuary that nested on East Sand Island during 2015–2019, the management period, was due to both the decreasing number of breeding pairs at the East Sand Island colony (*Figure 4.9*) and the increasing size of the colony of double-crested cormorants on the Astoria-Megler Bridge (*Figure 4.10*).

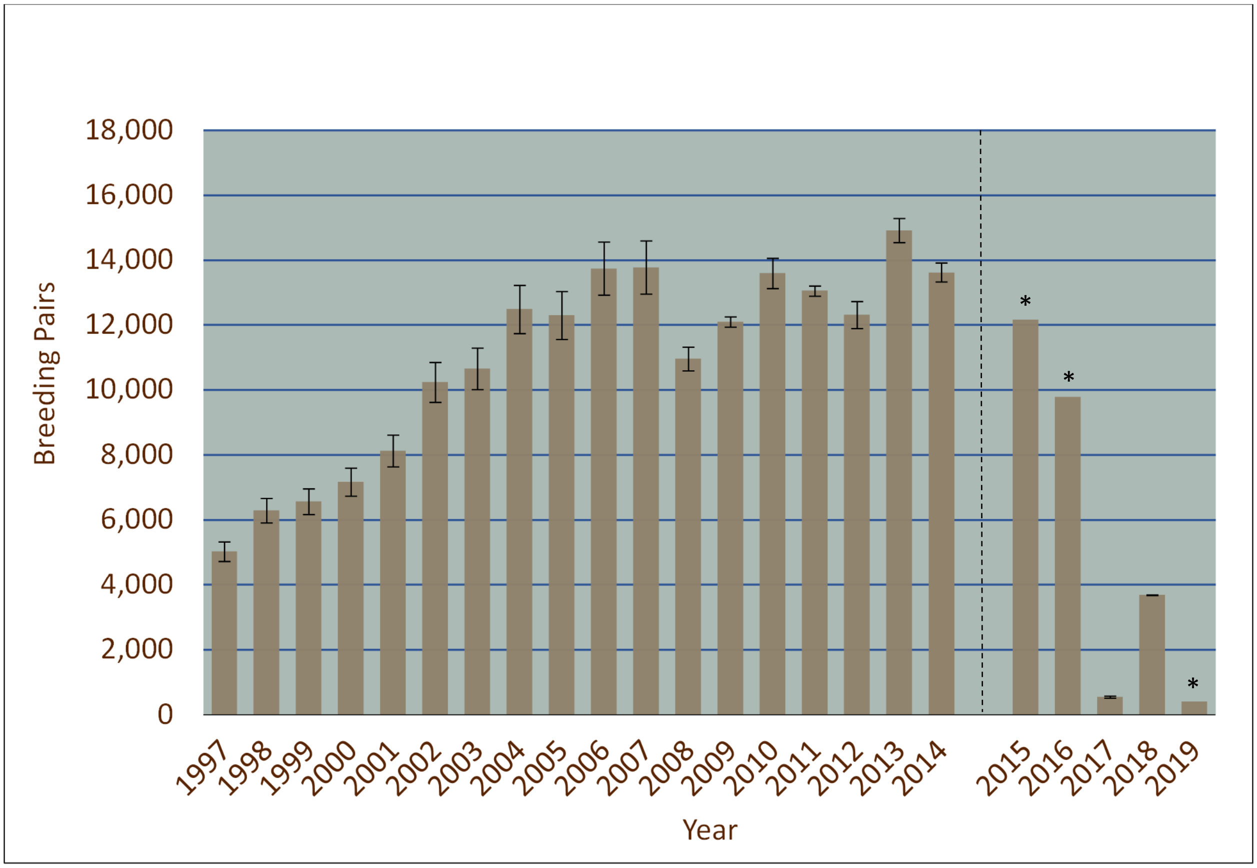

Figure 4.9. Size of the double-crested cormorant breeding colony (number of breeding pairs) at East Sand Island in the Columbia River estuary during the pre-management period (1997-2014) and the management period (2015–2019). Asterisks indicate that a 95% confidence interval for the estimate is not available. Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data for 2019 provided by the U.S. Army Corps of Engineers.

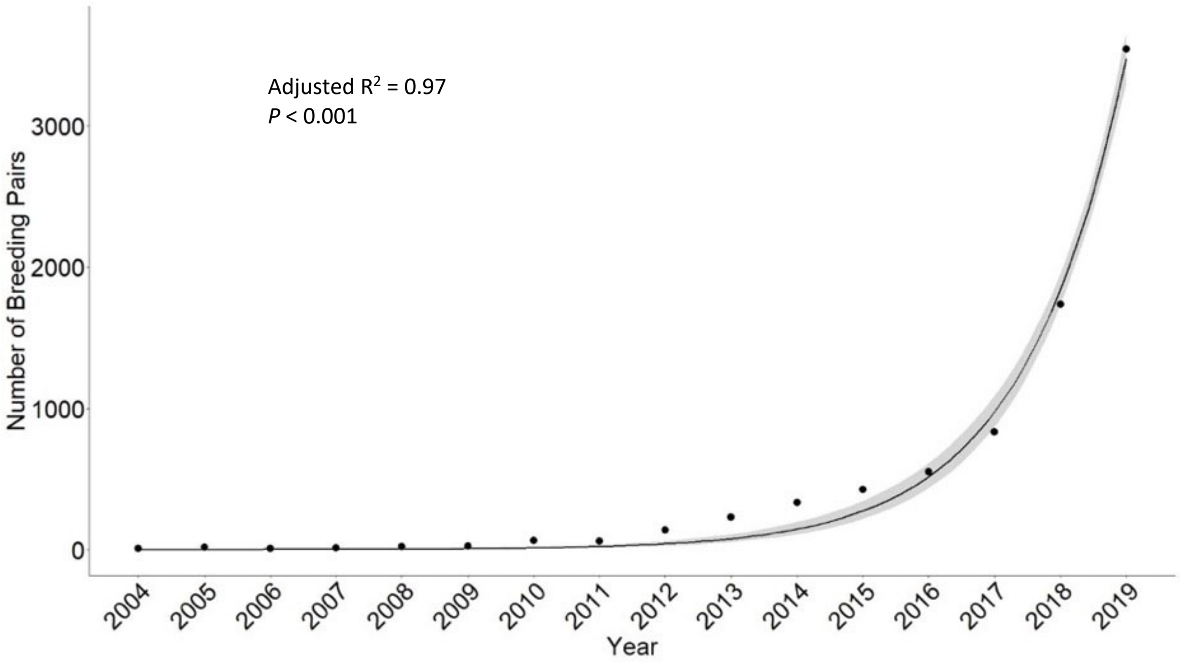

Figure 4.10. Size of the double-crested cormorant colony (number of breeding pairs) on the Astoria-Megler Bridge in the Columbia River estuary during 2004–2019. Double-crested cormorants first nested on the Astoria-Megler Bridge in 2004. Solid line represents the fitted log-linear trend in colony size (log number of breeding pairs, back-transformed). Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data for 2019 provided by the U.S. Army Corps of Engineers.

The size of the double-crested cormorant colony on East Sand Island increased rapidly from 91 breeding pairs in 1989, the first year when double-crested cormorant nests were counted on East Sand Island (Naughton et al. 2007), to 5,023 breeding pairs in 1997, the first year of the present study. This corresponds to an average annual growth rate (λ) in colony size of 1.65, well above the estimated maximum intrinsic rate of increase for the western population of double-crested cormorants of 1.16 (Lawonn, In prep.). The exceedingly high growth rate of the East Sand Island colony during 1989-1991 indicates that much of the increase was due to immigration from other colonies outside the estuary, presumably colonies in coastal Washington and British Columbia, where many colonies were in decline (Adkins et al. 2014). After 1991, however, when the rate of increase in colony size had slowed, intrinsic increase (i.e. recruitment of cormorants raised in the estuary) in the numbers of double-crested cormorants nesting in the Columbia River estuary may have contributed to growth in the size of the East Sand Island colony. Subsequently, the number of double-crested cormorants nesting on East Sand Island continued to increase from 5,023 breeding pairs in 1997 to 13,738 breeding pairs in 2006 (*Figures 4.9* above and *Figure 4.11*). During 1997–2006, however, the average growth rate of the East Sand Island colony (λ = 1.11) was much lower than during 1989-1997, suggesting that much of the growth in colony size during 1997-2006 was intrinsic. During the 8-year period of 2007–2014, the number of breeding pairs of double-crested cormorants nesting on East Sand Island was stable (λ = 1.00; *Figure 4.11*) and colony size averaged 13,040 breeding pairs. The maximum estimated size of the East Sand Island colony was 14,916 breeding pairs in 2013

(*Figure 4.9* above). During 2011-2014 the double-crested cormorant colony on East Sand Island experienced increasing disturbance rates from bald eagles, as well as human disturbance associated with pilot studies on East Sand Island to test the feasibility of non-lethal management techniques to limit cormorant colony size (see section *4.4.8.6 Dissuasion fencing & human hazing*). Nevertheless, the numbers of double-crested cormorants nesting at the East Sand Island colony remained stable (λ = 1.00). During the management period (2015–2019), when the East Sand Island colony of double-crested cormorants was limited by both lethal and non-lethal control techniques, the colony declined dramatically (λ = 0.49; $P = 0.06$; *Figure 4.11*) from 12,150 breeding pairs in 2015 to about 350 breeding pairs in 2019 (Lawonn, In prep.). There were no sustained nesting attempts by double-crested cormorants on East Sand Island in 2020 (M.J. Lawonn, ODFW, pers. comm.)

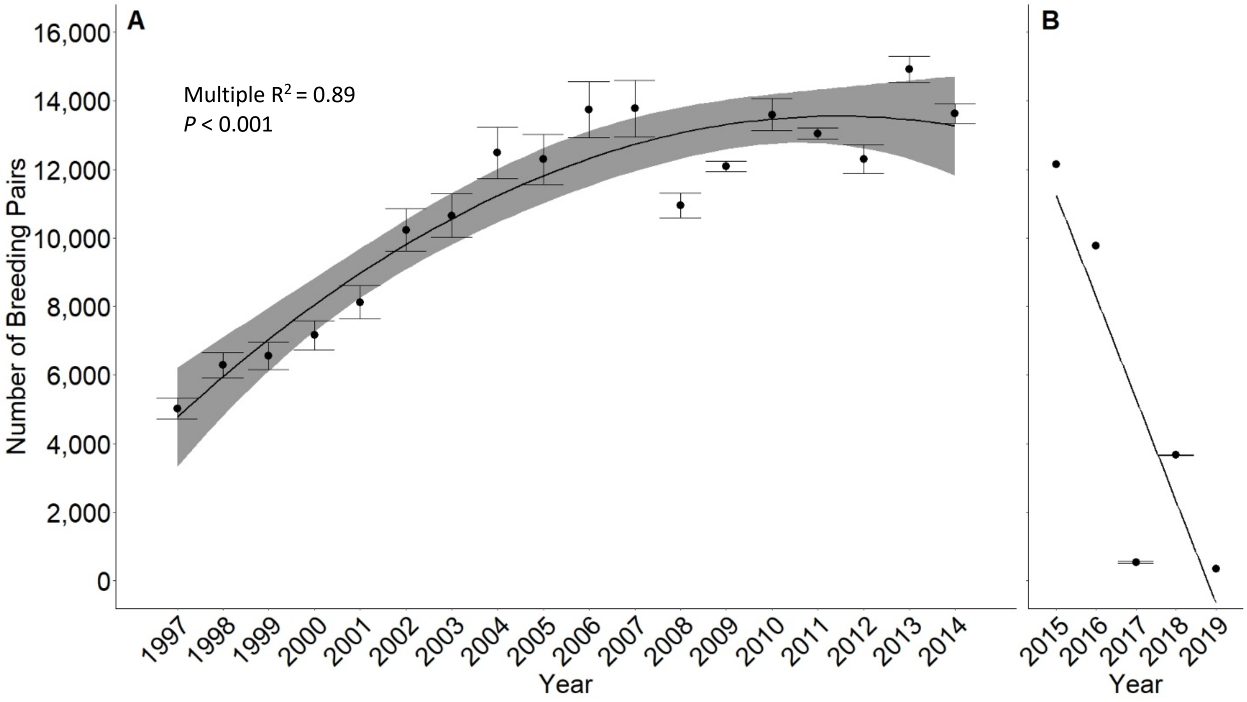

Figure 4.11. Trends in size (number of breeding pairs) of the double-crested cormorant colony on East Sand Island in the Columbia River estuary during the pre-management period (1997–2014; A) and during the management period (2015–2019; B). Fitted regression models are based upon point estimates for each year. Data for 2015 provided by David Smith and Associates, Inc., data for 2016 provided by Anchor QEA, and data for 2019 provided by the U.S. Army Corps of Engineers.

The seasonal peak in the number of double-crested cormorant nesting pairs on the East Sand Island colony occurred late in the incubation period during all pre-management years except for 2011, when peak colony size occurred early in the incubation period. This was likely due to exceptionally large numbers of bald eagles on or near the cormorant colony early in the 2011 nesting season (see section *4.4.3.2 Limiting factors for nesting success*). During the management period, however, the timing of the seasonal peak in number of cormorant nesting pairs varied widely; the colony peaked late in the chick-rearing period in 2015, immediately following a prolonged absence from the island in 2016, prior to egg-laying in 2017, and early in incubation in 2018 (see section *4.4.2.2 Colony attendance*).

The annual peaks in double-crested cormorant colony size on East Sand Island that are reported here were equivalent to the seasonal peak number of active nests counted in all years except 2017 and 2018. During the 2017 and 2018 breeding seasons, some early season estimates of the number of double-crested cormorants initiating nests on the East Sand Island colony exceeded those reported in *Table 4.3* (see above), but many of these early season nesting attempts soon failed, and the estimate of peak colony size was considerably lower in both years. In 2017, a maximum of 3,380 double-crested cormorant nests were counted in May, when cormorant nesting was initiated on the East Sand Island colony. But because the nests were only attended sporadically and then abandoned after just one week, we used the smaller peak count reported early in the incubation period (544 breeding pairs on 26 July; see section *4.4.2.1 Nesting chronology*). Similarly, in 2018 there was a dramatic one-week surge in the number of double-crested cormorant nests (5,999 nests counted on 30 May) when nesting was initiated on the East Sand Island colony, but we report the peak count in the middle of the chick-rearing period (3,672 breeding pairs on 25 July; Turecek et al. 2019). By selecting the smaller peak count in these two years, the estimate is more representative of the actual size of the breeding colony on East Sand Island during those two breeding seasons and we avoid double-counting breeding pairs that dispersed away from East Sand Island and re-nested at other sites in the Columbia River estuary.

Other double-crested cormorant colonies in the lower Columbia River estuary (i.e. below the Astoria-Megler Bridge) that were active during the study period were small compared to the colony on East Sand Island. The double-crested cormorant colony located at the Desdemona Sands pilings (46.21°N, 123.88°W), just downstream of the Astoria-Megler Bridge (*Figure 4.1* above), consisted of 130 breeding pairs in 1997 but declined rapidly thereafter; no nesting by double-crested cormorants has been reported at Desdemona Sands since 2000. The double-crested cormorants nesting on channel markers and navigational aids from Rkm 0 to Rkm 22 (downstream of the Astoria-Megler Bridge; *Figure 4.1* above) were first detected in 2012, when eight breeding pairs were counted; the maximum recorded number of breeding pairs at these channel markers was 73 breeding pairs in 2019.

Double-crested cormorants nested at several colony sites in the upper Columbia River estuary (i.e. above the Astoria-Megler Bridge) during 1997–2019. The maximum size of the double-crested cormorant colony on Rice Island during the study period was 1,141 breeding pairs in 1997, the first year of our study. The size of the Rice Island colony declined to 795 breeding pairs in 1998 and in 1999 and 2000 no double-crested cormorants nested on Rice Island (*Table 4.3* above). The managed relocation of the large Caspian tern colony from Rice Island to East Sand Island during 1999 and 2000 likely dissuaded double-crested cormorants from nesting on Rice Island in those two years. A small colony of double-crested cormorants reformed on Rice Island in 2001, but in 2004 and 2005 the colony was again abandoned. Following the 2006 nesting season, when nesting habitat enhancement and social attraction were used to induce double-crested cormorants to nest in small numbers on Rice Island, no cormorant nesting has occurred on Rice Island. Nesting dissuasion activities targeting Caspian terns prospecting for nest sites on Rice Island during 2010–2019 likely precluded nesting by double-crested cormorants on Rice Island. The double-crested cormorant colonies in the upper Columbia River estuary other than the colony on Rice Island were all small, each below 400 breeding pairs at maximum size. A small, incipient colony formed on Miller Sands Spit (46.25°N, 123.68°W; *Figure 4.1* above) in 2001, but the nesting attempts quickly failed, and the site was abandoned. In 2005 a small colony of double-crested cormorants formed again on the downstream end of Miller Sands Spit after deployment of social attraction and habitat enhancement (see section *4.3.9 Pilot Studies of Cormorant Social Attraction & Colony Relocation* above). This colony grew to a maximum size of 129 breeding pairs in 2008, the fifth and final year of deploying social attraction at the site. The colony site was abandoned later in the 2008 nesting season and no subsequent nesting by double-crested cormorants occurred on Miller Sands Spit (*Table 4.3* above). Double-crested cormorants nested on up to 12 different channel markers/navigational aids from river km 22 to 51 (upstream of the Astoria-Megler Bridge and downstream of Fitzpatrick Island; *Figure 4.3* above) during 1997–2019. There was a gradual increase in the total number of double-crested cormorants nesting on upper estuary channel markers, from 64 breeding pairs in 1997 to 330 breeding pairs in 2013. Since 2013, however, the number of double-crested cormorants nesting on channel markers in the upper estuary has remained relatively stable at between 204 and 319 breeding pairs ($\overline{x}$ = 276 breeding pairs). Finally, in 2014 double-crested cormorant nests were reported for the first time on the Lewis & Clark Bridge near Longview, Washington (46.10°N, 122.96°W; *Figure 4.1* above). The colony on the Lewis & Clark Bridge grew from 11 breeding pairs in 2014 to 201 breeding pairs in 2018; it seems likely that very few, if any, double-crested cormorant nests were present on the Lewis & Clark Bridge prior to 2014.

Currently (2020), the largest double-crested cormorant colony in the Columbia River estuary is located on the Astoria-Megler Bridge (46.21°N, 123.86°W), at the transition from the lower estuary to the upper estuary and about 15 km upstream of the double-crested cormorant colony on East Sand Island (*Figure 4.1* above). Double-crested cormorants were first detected nesting on the Astoria-Megler Bridge in 2004 and the colony remained below 25 breeding pairs until 2010 (*Figure 4.10* above). The growth rate of the Astoria-Megler Bridge colony during 2011–2014 (λ = 1.52), when pilot studies of nonlethal nest dissuasion for cormorants at East Sand Island were being conducted (see *Section 4.3.8*), was higher than during 2004–2010 (λ = 1.48), but only slightly so. During 2015-2019, the management period on East Sand Island, the growth rate of the Astoria-Megler Bridge colony increased dramatically (λ = 1.60). If the cormorant colony on the Bridge had continued to increase at the annual rate observed during 2011–2014 during the management period on East Sand Island, the extrapolated colony size estimate would have been 2,669 breeding pairs in 2019, about 25% less than the actual colony size in that year (3,542 breeding pairs). During each of these three periods, the growth rate of the Astoria-Megler Bridge colony of double-crested cormorants far exceeded the estimate of the maximum intrinsic rate of increase (i.e. number of births minus number of deaths with no immigration or emigration) for the western population of double-crested cormorants (λ = 1.16), suggesting that most of the growth in the double-crested cormorant colony on the Astoria-

Megler Bridge was likely due to immigration from East Sand Island (see *Section 4.9*; Lawonn, In prep.). There was a high count of 622 double-crested cormorant nests on the Astoria-Megler Bridge during the week of 6 June 2016, but this number was "unconfirmed" (Anchor QEA 2017). Instead, we used the peak count of 549 double-crested cormorant nests reported on 13 June 2016 (Anchor QEA 2017; *Table 4.3* above).

4.4.1.2 Nesting habitat use – In the Columbia River estuary, double-crested cormorants have nested in a variety of habitats and on various types of substrate, including rock jetties and revetment, wooden pilings, dredged material disposal islands, trees on islands, navigational aids (channel markers and towers), and support structures on road bridges (*Table 4.3* above and *Figure 4.1* above). At the western end of East Sand Island, double-crested cormorants mainly nested on the ground on the island's driftwood-strewn western jetty and on erosion control revetment along the southern shoreline, on vegetated uplands, and on researcher-provided automobile tires. In addition, double-crested cormorants nested on the ground adjacent to, or on top of, human-made structures deployed on the island to facilitate research and monitoring activities (i.e. above-ground access tunnels, blinds, and fences), and in the few trees that were present on the western half of East Sand Island. Although tree-nesting colonies of double-crested cormorants have formed in areas near the Columbia River estuary (e.g., Seaside, Oregon), the only known cormorant colony in the Columbia River estuary during 1997–2019 where some pairs nested in trees was the colony on East Sand Island.

When we began monitoring and mapping the distribution of cormorant nests at the western end of East Sand Island in 1997, nesting occurred along the top of the western-most rock jetty and extended eastward along the southern revetment of the island for several hundred meters; the cormorant colony occupied a total area of 1.26 acres (0.51 ha) on the island. As the cormorant colony on East Sand Island grew during 1997–2003, however, the nesting area shifted further eastward along the southern revetment and northward on to the sandy interior of the island (*Figure 4.12*). The area occupied by the double-crested cormorant colony on East Sand Island reached its maximum extent during the 2004 breeding season, when the colony occupied 4.22 acres (1.71 ha) of nesting habitat. By the 2010 breeding season, nesting double-crested cormorants had completely abandoned the western jetty and were distributed throughout the interior of the western half of East Sand Island and occupied an area of 2.62 acres (1.06 ha; *Figure 4.12*).

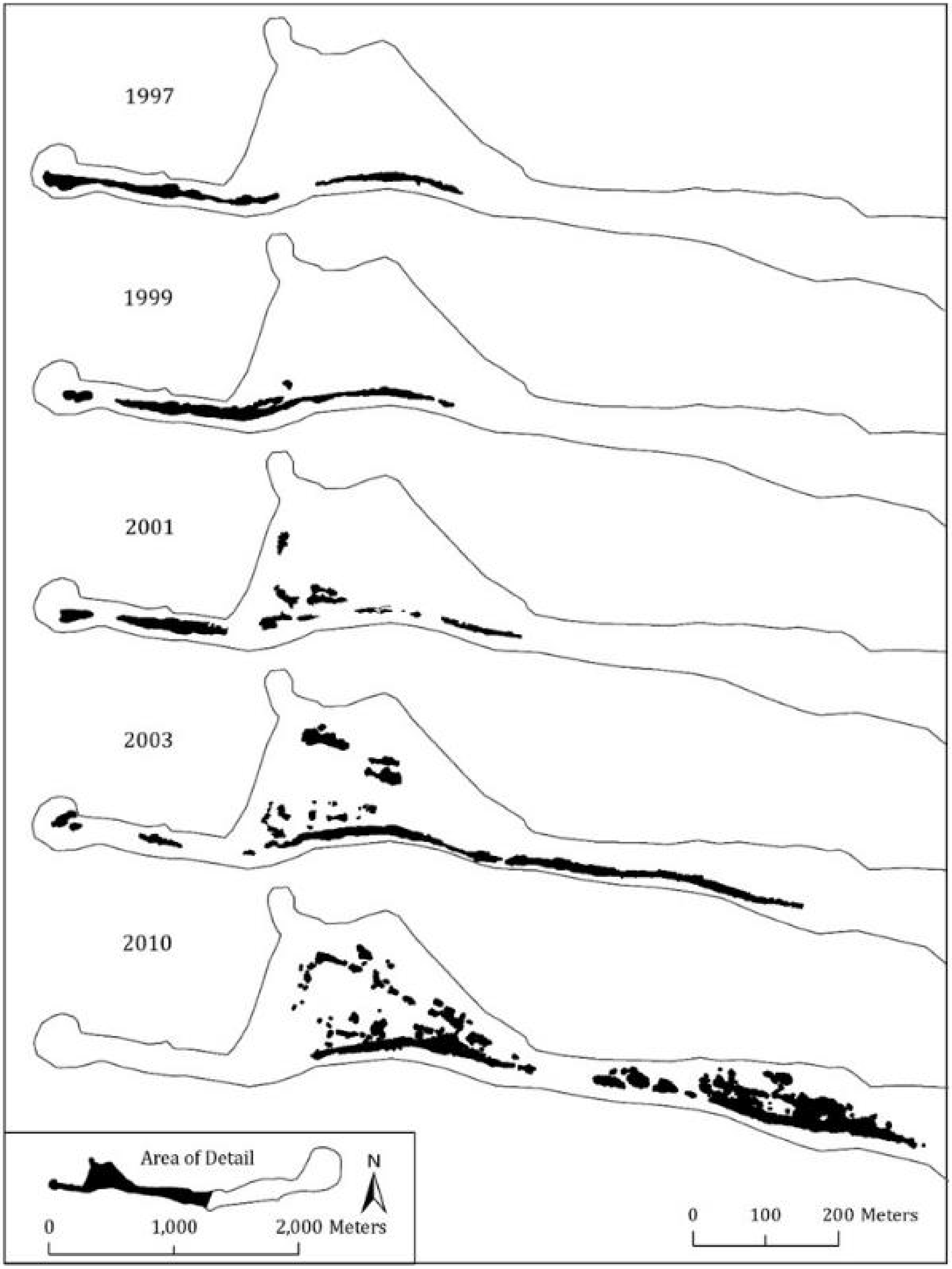

Figure 4.12. Distribution of active double-crested cormorant nests (shaded area) on the western half of East Sand Island in the Columbia River estuary in every other year from 1997 through 2003 and in 2010. There was an incremental shift in the area where cormorants nested on East Sand Island towards the north and east during this 14-year period.

During the 2011–2013 breeding seasons, the nesting distribution of cormorants on East Sand Island was intentionally constrained as part of pilot studies into the efficacy of non-lethal dissuasion techniques. During the 2011–2012 nesting seasons, the distribution of cormorants nesting on East Sand Island was pushed westward, and parts of the western jetty were reoccupied as nesting habitat (*Figure 4.4* above). During the 2013 breeding season, however, the cormorant colony was confined to a relatively small portion of the island between two dissuasion fences near the eastern extent of the nesting distribution used during the 2010 breeding season (*Figure 4.4* above). No effort was made to constrain the nesting distribution of cormorants during the 2014 breeding season, and nesting occurred in the upland areas to the east and west of the 4.0 acres (1.6 ha) of nesting habitat provided in 2013 (*Figure 4.4* above).

During 2015–2017, a dissuasion fence was deployed near the eastern extent of the cormorant nesting distribution to facilitate colony access during on-colony management activities associated with implementing Phase I of the *Cormorant Management Plan*. During 2015–2016, cormorants nested east of the dissuasion fence, west of the dissuasion fence (including adjacent to the blind and tunnel infrastructure), and throughout the interior of the western portion of the island (*Figure 4.13*). In 2016, however, fewer cormorants attempted to nest in the interior of the western portion of the island and those small satellite colonies that did form were restricted to the area near the southern shoreline (*Figure 4.13*). During the 2017 breeding season, cormorants first attempted to nest adjacent to the blind and tunnel infrastructure and dissuasion fence during June, but following nesting failures, reestablished nesting activities on a 0.27-acre (0.11-ha) area just east of the western jetty during July (*Figure 4.13*). In 2018, a dissuasion fence was erected ca. 75 m east of the base of the western jetty as part of Phase II of the *Cormorant Management Plan*, and nesting double-crested cormorants occupied 0.76 acres (0.31 ha) of nesting habitat, which was restricted almost exclusively to the jetty west of the dissuasion fence (*Figure 4.13*). During the winter of 2018–2019, the USACE modified the western half of East Sand Island to reduce the amount of available cormorant nesting habitat by converting former nesting habitat into an L-shaped jetty running east/west and with a northward running arm; a dissuasion fence and concrete barriers were installed along the northward running arm (*Figure 4.14*). During the 2019 breeding season, the few double-crested cormorant nests on East Sand Island were limited to areas adjacent to and to the west of the dissuasion fence (J. Macdonald, USACE, pers. comm.).

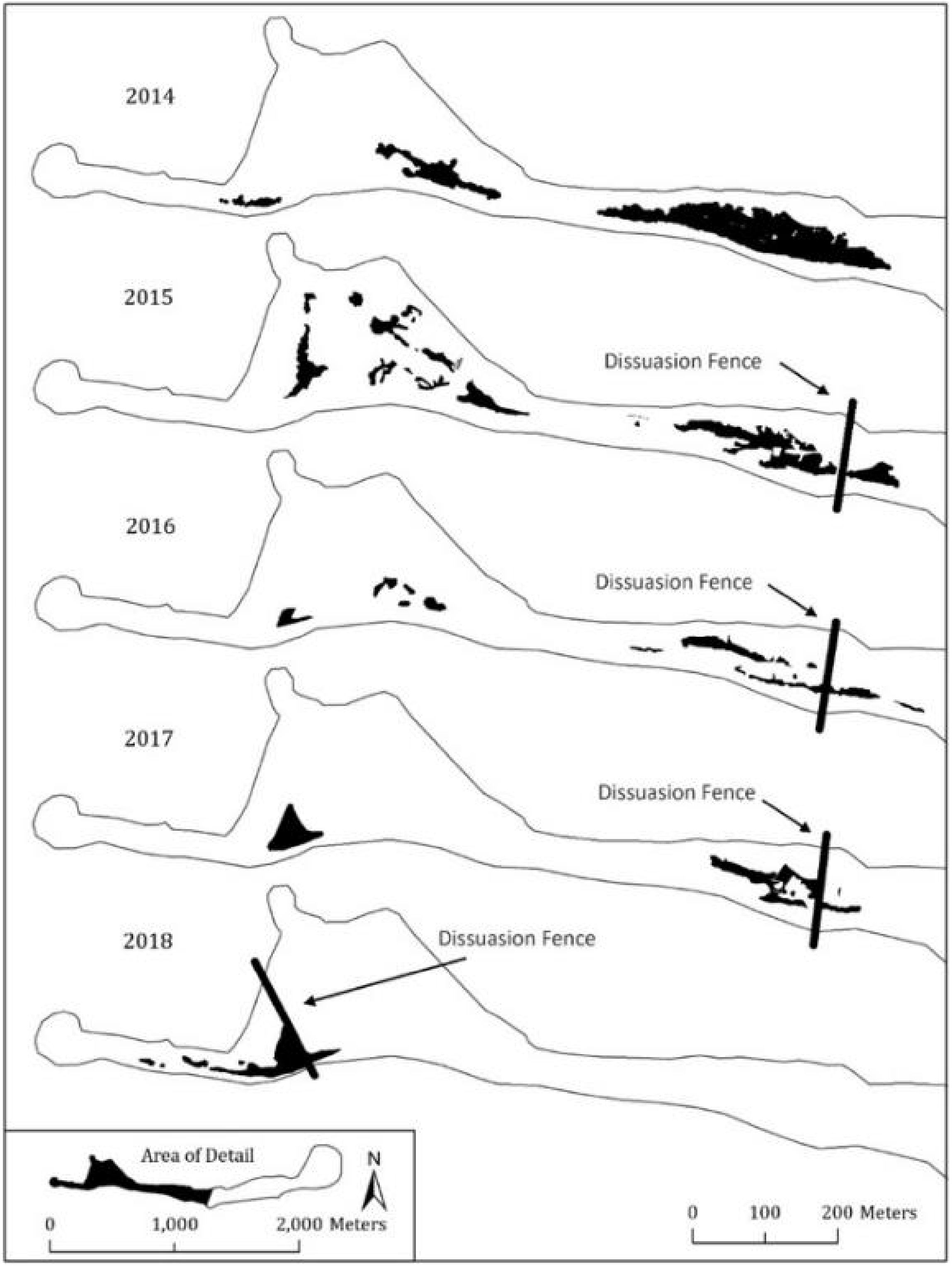

Figure 4.13. The distribution of active double-crested cormorant nests (shaded area) on the western half of East Sand Island in the Columbia River estuary during the 2014–2018 breeding seasons. Dissuasion fences were erected within the available nesting habitat during 2015–2018 to limit the size and distribution of double-crested cormorants nesting on East Sand Island as part of the Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary.

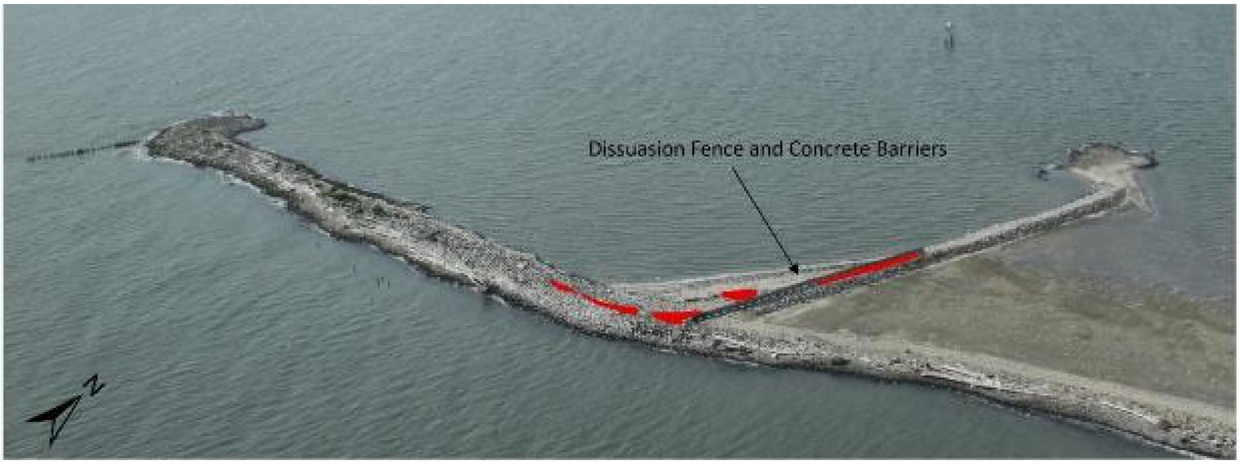

Figure 4.14. Oblique aerial image of the western end of East Sand Island in the Columbia River estuary during the 2019 breeding season (photo taken on 4 June 2019) showing the approximate distribution of double-crested cormorant nests (red shaded areas). The U.S. Army Corps of Engineers modified the island using heavy equipment during the winter of 2018–2019 to reduce the area of available nesting habitat for double-crested cormorants as part of the Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary. Designated cormorant nesting habitat was west of the dissuasion fence and concrete barriers.

4.4.1.3 Nesting density – Cormorant nesting density (nests/m^2 of occupied habitat) on the East Sand Island colony remained relatively constant during the 1997–2014 breeding seasons (*Figure 4.15*), despite the variability in colony size and nesting distribution at the western end of the island. The average of the estimated annual nesting densities for double-crested cormorants on the East Sand Island colony was 1.10 nests/m^2 (range = 0.73–1.28 nests/m^2) during the 1997–2014 breeding seasons. Following the initiation of management actions under the *Cormorant Management Plan*, however, average nesting density was estimated at 0.86 nests/m^2 (range = 0.5–1.2 nests/m^2) during the 2015–2018 breeding seasons (*Figure 4.15*). The lower average nesting density during the management period was mostly due to the very low estimated cormorant nesting density in 2017 (0.50 nests/m^2), the lowest ever recorded at the East Sand Island colony; average nesting densities in other years during the management period were within the range of nesting densities recorded during the pre-management period. The estimated average nesting density in 2015 of 1.01 nests/m^2 (David Smith and Associates 2016) was similar to the average nesting density recorded during the 1997–2001 breeding seasons. The estimated average nesting density in 2016 of 0.73 nests/m^2 (Anchor QEA 2017) was similar to the average nesting density during the 2004 breeding season. Finally, the estimated average nesting density in 2018 of 1.2 nests/m^2 (Turecek et al. 2019) was similar to the average nesting density during the 2005–2014 breeding seasons (*Figure 4.15*). An estimate of average cormorant nesting density on East Sand Island for the 2019 breeding season is not available.

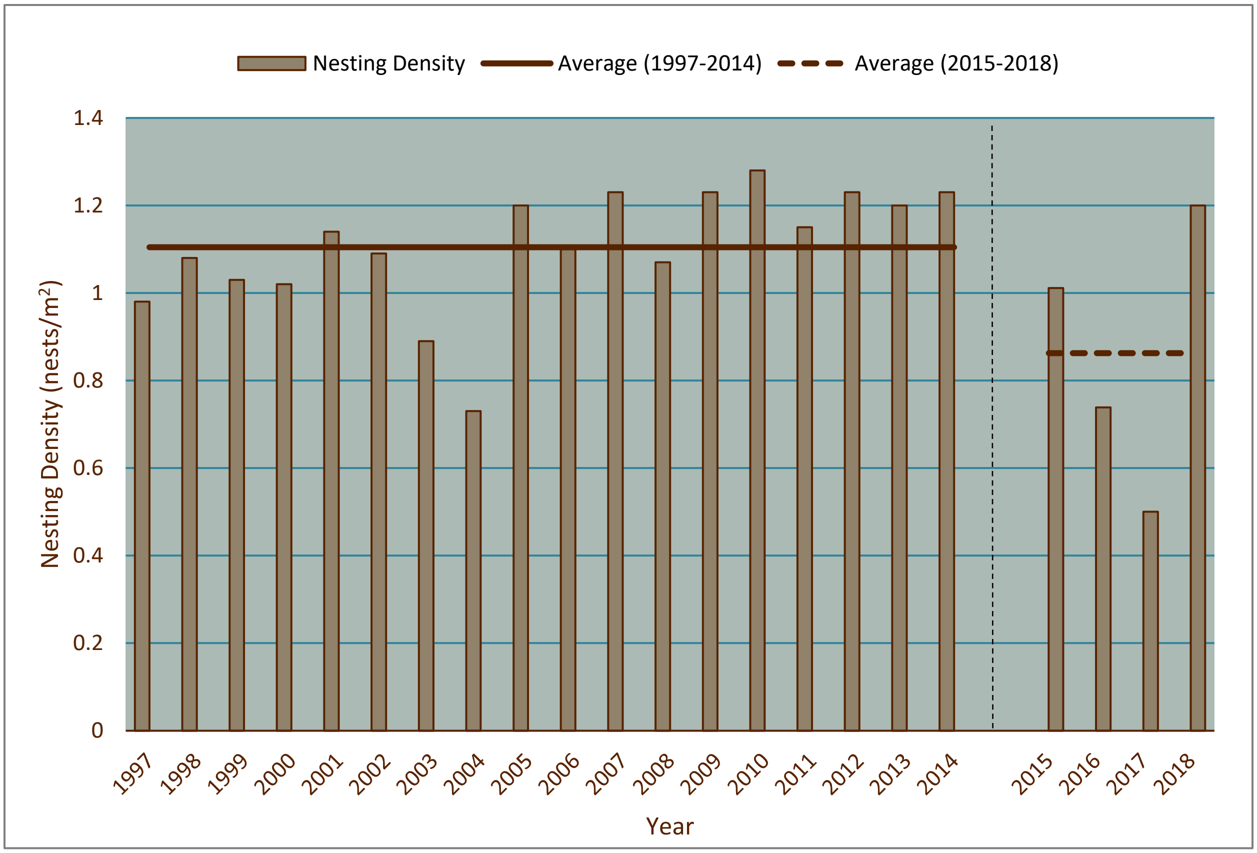

Figure 4.15. Double-crested cormorant nesting density (average number of nests/m^2) at East Sand Island in the Columbia River estuary during the pre-management period (1997–2014) and during the management period (2015–2018). Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data not available for 2019.

4.4.2 Nesting Chronology, Colony Attendance, & Nesting Success

4.4.2.1 Nesting chronology – The dates when double-crested cormorant eggs and chicks were first observed on the East Sand Island colony were recorded during 12 breeding seasons (2003–2014) prior to initiation of the *Cormorant Management Plan* and during three breeding seasons (2016–2018) following implementation of the *Plan*. Dates when double-crested cormorant chicks were first observed as fledglings (i.e. demonstrating sustained flight) at the nesting colony on East Sand Island were recorded during 10 breeding seasons (2005–2014) prior to the initiation of management and during two breeding seasons (2017–2018) following the implementation of management. The dates when the double-crested cormorant colony at East Sand Island attained peak size were estimated during 16 breeding seasons (2003–2018). No information on nesting chronology or colony attendance of double-crested cormorants was available for the 2019 breeding season at East Sand Island.

The median dates when the first double-crested cormorant egg, chick, and fledgling were observed on-colony, and the median date when colony size peaked at East Sand Island, were all significantly later in the breeding season during the management period (2015–2018) compared to the pre-management period (2003–2014; *Figure 4.16*). In addition to the delayed nesting chronology during the management period, nesting chronology also became more variable. Prior to the initiation of management, the first double-crested cormorant egg on East Sand Island was detected during a 2.5-week period from 19 April to 5 May. Following the implementation of management, however, the first double-crested cormorant egg was detected during a 6-week period from 3 May to 13 June. The median date when the first double-crested cormorant egg was observed on the East Sand Island colony was 23 April during the pre-management period (2003–2014), nearly one month earlier than the median first egg date of 20 May during the management period (2015–2018; *Figure 4.16*).

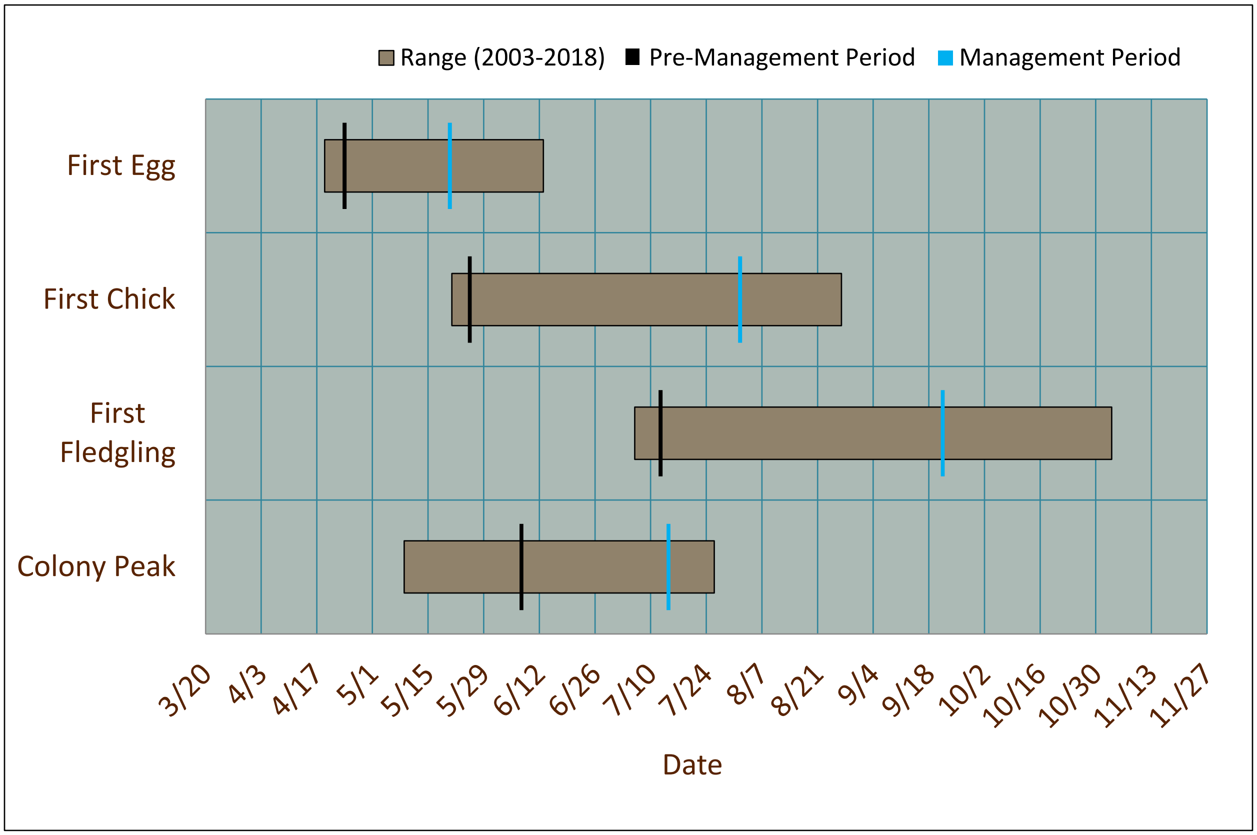

Figure 4.16. Nesting chronology for double-crested cormorants breeding on East Sand Island in the Columbia River estuary during 2003–2018. Median dates for each measure of nesting chronology are presented for both the pre-management period (2003–2014) and the management period (2015–2018). First egg date and first chick date are not available for 2015. First fledgling date is not available during 2003–2005 or during 2015–2016. Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data are not available for 2019.

During 2003–2014 we observed the first cormorant chicks hatching during the 2-week period from 21 May to 2 June; during the management period (2015–2018) the first cormorant chick was seen during the 9.5-week period from 22 June to 27 August. The median date when the first cormorant chick was observed on the East Sand Island colony during the management period (1 August) was nearly 10 weeks later than during the pre-management period (25 May: *Figure 4.16* above). During the pre-management period, the first fledgling cormorant chicks were observed during the 2-week period from 6 July to 19 July; during the management period, however, the first cormorant fledgling was reported during a 12.5-week period from 9 August to 3 November. The median date when the first cormorant fledgling was observed during the management period was 21 September, 10 weeks later than the median date when the first fledgling was observed during the pre-management period (14 July; *Figure 4.16* above). During the pre-management period, the estimated date when the double-crested cormorant colony on East Sand Island reached peak size was during a 33-day period from 9 May to 11 June (median date = 7 June). During the management period, however, the estimated peak size of the double-crested cormorant colony occurred during a 25-day period from 1 July to 26 July (median date = 14 July; *Figure 4.16* above), more than five weeks later than during the pre-management period.

We found no evidence for a continual, gradual delay in nesting chronology of double-crested cormorants at the East Sand Island colony during the pre-management period (2003–2014). Prior to implementation of the *Cormorant Management Plan*, we found no trend in the ordinal date of first chick hatch (Multiple $R^2 = 0.001$, $P = 0.98$; *Figure 4.17*). As with other measures of nesting chronology, however, the ordinal date of first chick hatch was delayed in years following implementation of the *Cormorant Management Plan* compared to the pre-management period ($P = 0.01$; *Figure 4.17*). During the management period, the median dates of first chick hatching and first chick fledging for double-crested cormorants nesting on East Sand Island were more delayed than the median date for the initiation of egg-laying compared to the pre-management period.

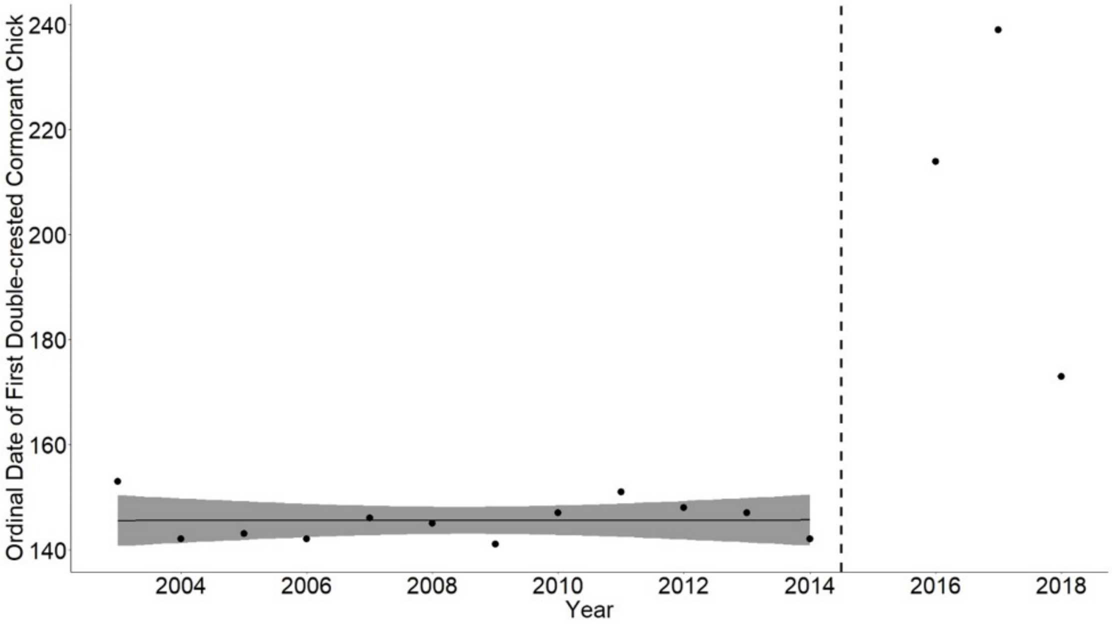

Figure 4.17. Date when the first double-crested cormorant chick hatched on the breeding colony at East Sand Island in the Columbia River estuary during 2003–2018. Trend line for the pre-management period (2003–2014) shown as a solid line and the 95% c.i. as gray shading. The Cormorant Management Plan was implemented beginning in 2015, represented by the vertical dashed line. The management period was 2016–2018; data are not available for 2015 or 2019. Data for 2016 were provided by Anchor QEA.

4.4.2.2 Colony attendance – Prior to implementation of the *Cormorant Management Plan*, the number of active cormorant nests peaked during the 2-week period ending 26 May, followed by a decline in the number of active cormorant nests through the 2-week period ending 7 July (*Figure 4.18*). Re-nesting attempts by cormorants that failed in their first nesting attempt may explain the slight increase in the average number of active nests counted during the 2-week period ending 21 July in the pre-management period (*Figure 4.18*). The average number of adult cormorants attending the East Sand Island colony declined rapidly by the middle of August during the pre-management period, as the nesting season ended. Cormorant colony attendance during the management period was different and reflected the delay in nesting chronology compared to the pre-management period (see section *4.4.2.1 Nesting chronology* above). Twice per month counts of cormorants on the East Sand Island colony during the 2018 breeding season, the only year during the management period when these data are available, indicated that the numbers of active cormorant nests and adult cormorants were reduced compared to the pre-management period (2008–2013; *Figure 4.18*). In 2018, the number of active cormorant nests during the 2-week period ending 26 May also exhibited a peak, but the colony did not reach its peak size that year until two months later during the 2-week period ending 21 July. There was likely a high rate of both nest failure and re-nesting attempts on the East Sand Island colony during May and June of 2018, attributed to frequent disturbances of the cormorant colony by bald eagles (Turecek et al. 2019). In 2018, however, the number of adult cormorants attending the colony had not yet started to decline when the collection of data on colony attendance ended after the 2-week period ending 18 August.

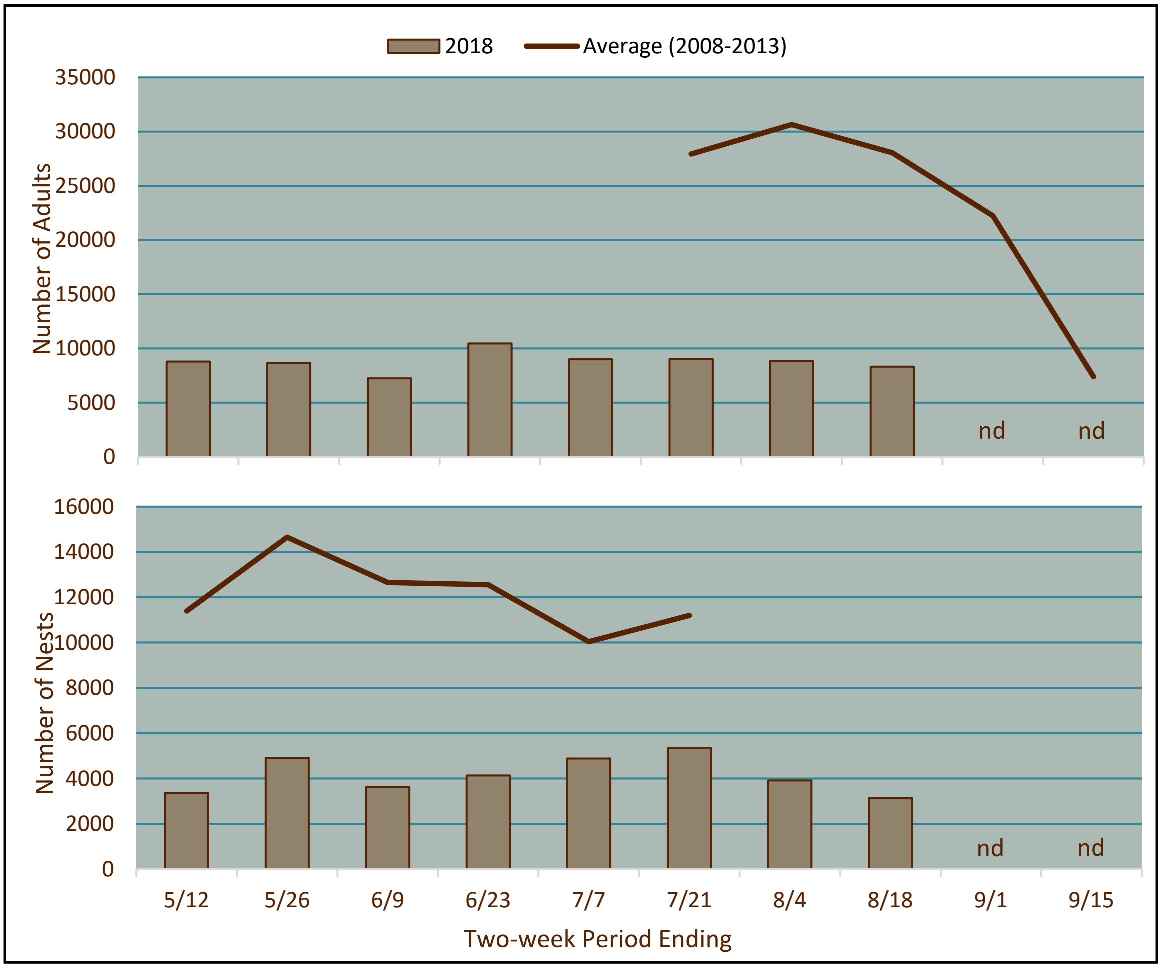

Figure 4.18. Bi-monthly averages of the number of adult cormorants on the East Sand Island colony in the Columbia River estuary during the 2018 breeding season (management period) compared to the pre-management period (2008–2013; above) and bi-monthly averages of the number of active cormorant nests on East Sand Island during 2018 compared to the pre-management period (2008–2013); no data were collected in 2014 or 2019, and few data are available for 2015, 2016, and 2017. No data are available on the number of adult cormorants attending the colony before 21 July and no data are available on the number of active cormorant nests on-colony after 21 July during the pre-management period. The counts of adult cormorants and cormorant nests include both double-crested cormorants and Brandt's cormorants. "nd" indicates that no data are available for 2018.

The extended delay in cormorant nesting chronology at the East Sand Island colony following implementation of the *Cormorant Management Plan* is further demonstrated by the numbers of active double-crested cormorant nests on-colony throughout each nesting season during the management period (*Figure 4.19*). In two of the four years during the management period, the cormorant colony was completely abandoned (no cormorants in attendance) for extended periods during the nesting season, including a 7-week period beginning 17 May in 2016 and a 5-week period beginning 21 May in 2017 (*Figure 4.19*). These were the first two instances of complete abandonment of the East Sand Island cormorant colony during the nesting season since monitoring began in 1997.

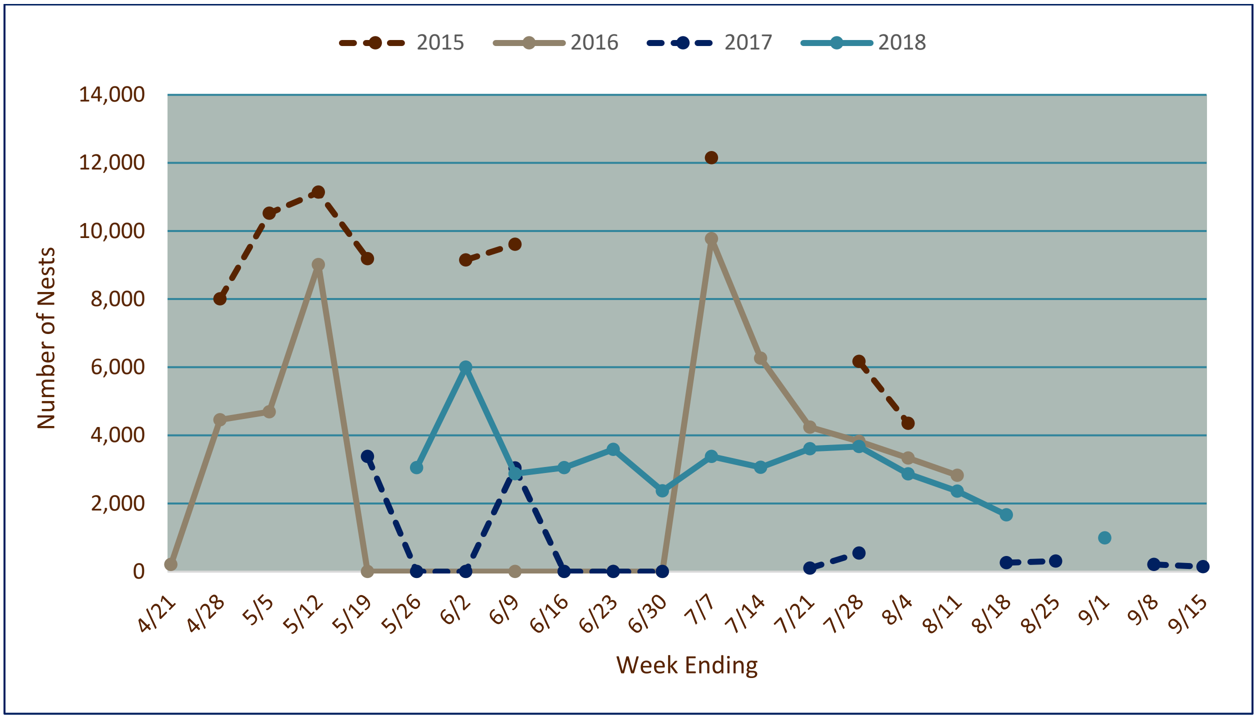

Figure 4.19. Weekly counts of the number of double-crested cormorant nests on the East Sand Island colony in the Columbia River estuary during the 2015 through 2018 breeding seasons (management period). Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data not available for 2019.

4.4.2.3 Nesting success – There was considerable inter-annual variation in average nesting success of double-crested cormorants at the East Sand Island colony during the pre-management period (1997–2013; *Figure 4.20*). Although there was an apparent trend of increasing nesting success for double-crested cormorants at the East Sand Island colony during the pre-management period, the trend was not significant (*P* = 0.11; *Figure 4.21*). Maximum annual average nesting success at the East Sand Island colony during the pre-management period was 2.8 young raised/breeding pair in 2009, which was 2.3 times greater than the minimum annual average nesting success of 1.2 young raised/breeding pair in 2000. The average annual nesting success at the East Sand Island colony during the pre-management period, 1.86 young raised/breeding pair, is considered typical for the species (Dorr et al. 2014).

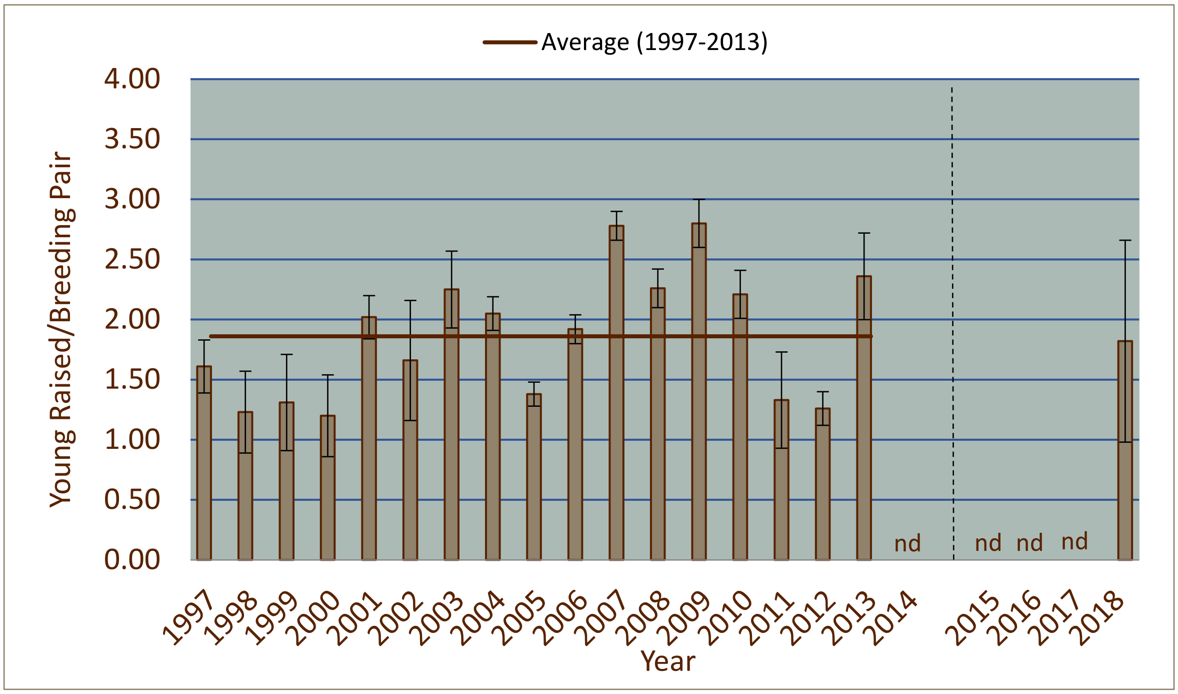

Figure 4.20. Nesting success (average number of young raised per breeding pair) of double-crested cormorants at the colony on East Sand Island in the Columbia River estuary during the pre-management period (1997–2014) and during the management period (2015–2018). "nd" indicates that no data are available.

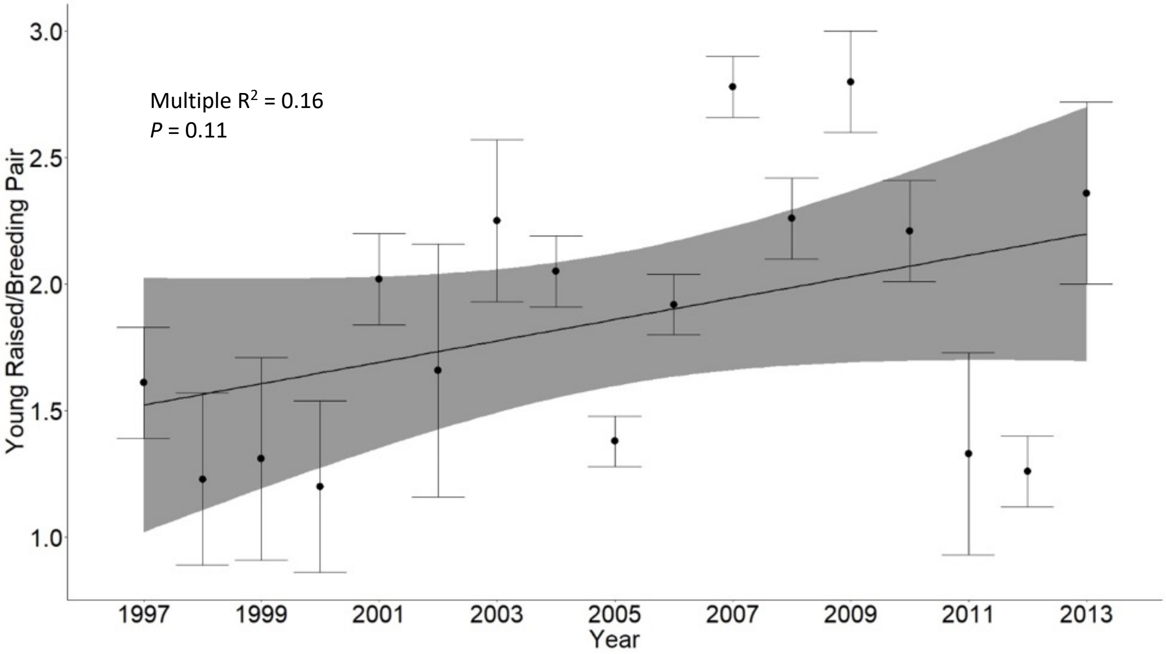

Figure 4.21. Trend in nesting success (average number of young raised/breeding pair) for double-crested cormorants nesting at the colony on East Sand Island in the Columbia River estuary during the pre-management period, 1997–2013. Best fit linear regression model (solid line) and 95% c.i. for the best fit model (shaded area) are based on the point estimates for each year.

During the management period (2015–2019), nesting success at the double-crested cormorant colony on East Sand Island was only estimated during the 2018 breeding season; average nesting success in 2018 was estimated as 1.8 young raised/breeding pair, similar to the average nesting success during the pre-management period. Although nesting success was not quantified in 2017 or 2019, it was considered to have been very low in both years, presumably well below the average nesting success during the pre-management period.

Data on nesting success for double-crested cormorants nesting at the Astoria–Megler Bridge colony are only available for one year (2018) and was estimated to be 2.7 young raised/breeding pair (Turecek et al. 2019). This is greater than the typical average nesting success for the species (1.2–2.4 young raised/breeding pair; Dorr et al. 2014) and is 1.5 times greater than the average nesting success of double-crested cormorants at the East Sand Island colony in 2018. If double-crested cormorant nesting success at the Astoria-Megler Bridge colony in 2018 was typical for that colony, then that would help explain why more and more breeding pairs of double-crested cormorants from the East Sand Island colony were attracted to nest at the Astoria-Megler Bridge colony during the management period, and why the size of the Astoria-Megler Bridge colony increased so rapidly during 2016–2019 (*Figure 4.10* above).

4.4.3 Factors Limiting Colony Size & Nesting Success

4.4.3.1 Limiting factors for colony size – The size of the double-crested cormorant colony on East Sand Island increased substantially during the pre-management period (*Figure 4.9* above). Colony size increased steadily from 5,023 breeding pairs in 1997 to 13,738 breeding pairs in 2006 and then remained stable from 2007 through 2014. During the pre-management period, the quadratic relationship between the size of the double-crested cormorant colony and year (*Figure 4.11* above) suggests that density-dependent factors increasingly limited colony size as the East Sand Island colony increased toward carrying capacity. If density-dependent factors were limiting colony size during 2006–2014, however, we would expect average nesting success as a function of year to decrease as colony size increased, and then stabilize at a lower-level once colony size had reached carrying capacity. Instead, nesting success at the East Sand Island colony trended upward, although not significantly so, as colony size increased (*Figure 4.21* above). However, nesting success is likely limited more by the availability of forage fish during the chick-rearing period (June-July), whereas colony size is likely limited by forage fish availability early in the nesting season (April-May). If forage fish availability early in the nesting season is not linked to forage fish availability late in the nesting season, nesting success could be a poor indicator of density-dependent limitations on colony size.

No correlation was found between the size of the double-crested cormorant colony on East Sand Island during the pre-management period and any single explanatory variable other than year. While it is likely that predation pressure and disturbance from bald eagles increased during the entire study period, no data on bald eagle numbers or disturbance rates are available for the cormorant colony on East Sand Island prior to 2008. Nevertheless, Isaacs and

Anthony (2011) reported a 738% increase in the number of nesting pairs of bald eagles in the lower Columbia River between 1978 and 2007. Also, the stabilization in size of the East Sand Island colony of double-crested cormorants by 2007 coincided with an increase in the proportion of salmonids, especially spring migrants, and a decrease in the proportion of clupeids in the diet of double-crested cormorants nesting on East Sand Island (see section *4.4.5 Diet Composition*). Both prey types are especially prevalent in the cormorant diet early in the nesting season. This apparent shift in the relative availability of spring migrant salmonids and clupeids early in the nesting season may help explain the stabilization of the size of the cormorant colony. At this same time (2006) there was an apparent change in the relationship between the proportion of salmonids in the cormorant diet and nesting success, whereby the prevalence of salmonids in the diet became negatively associated with nesting success (see section *4.4.3.2 Limiting factors for nesting success*). The available evidence suggests that a combination of top-down factors (disturbance and predation by bald eagles) and bottom-up factors (declining availability of marine forage fish early in the nesting season due to higher-than-normal river discharge and poor ocean conditions) were responsible for preventing any increase in the size of the double-crested cormorant colony on East Sand Island during 2006–2014.

During the management period, the double-crested cormorant colony on East Sand Island decreased from 12,150 breeding pairs in 2015 to 350 pairs in 2019 (Lawonn, In prep.). During the 2020 nesting season, there were no sustained nesting attempts by double-crested cormorants on East Sand Island (M.J. Lawonn, ODFW, pers. comm.). This dramatic decline in colony size coincided with the primarily lethal management techniques employed during Phase I of the *Cormorant Management Plan* (2015–2017) and the primarily non-lethal management techniques employed during Phase II of the *Plan* (2018–2019). While it is likely that the factors that limited colony size during the pre-management period (i.e. eagle disturbance and low availability of marine forage fish early in the nesting season) also influenced colony size during the management period, management actions implemented as part of the *Cormorant Management Plan* were the primary factors limiting colony size during 2015–2019 (see section *4.5.4 Management Action Effectiveness*). Concurrently, the large and rapidly growing colony of double-crested cormorants on the Astoria-Megler Bridge likely provided strong social attraction as a nearby alternative nesting and/or roosting site in the estuary for cormorants dissuaded from nesting on East Sand Island colony by the cumulative disturbance from management actions and eagle predation.

4.4.3.2 Limiting factors for nesting success – Inter-annual variation in nesting success at the double-crested cormorant colony on East Sand Island during the pre-management period was attributable in part to pressure from nest predators. Throughout the study period, bald eagles were frequently observed depredating both adult cormorants and the contents of cormorant nests at the East Sand Island colony. Associated with disturbances caused by bald eagle predation events, glaucous-winged/western gulls, which nest in large numbers on East Sand Island adjacent to the cormorant colony, frequently depredated unattended cormorant eggs and nestlings, a pattern that was also documented at the Caspian tern colony on East Sand Island (Collar et al. 2017, Bailey 2018). During 2008–2012 data were collected on the frequency and cause of large disturbance events at the cormorant colony on East Sand Island, and bald eagles were identified as the cause of 98% of disturbance events (*Figure 4.22*). The remaining sources of disturbance to the cormorant colony caused less than 2% of all colony disturbances, including disturbance due to research activities (1.1%), motorboats (0.4%), helicopters (0.2%), and other sources (0.2%). Unfortunately, no data were collected on the frequency and cause of disturbance events at the East Sand Island cormorant colony during the management period (2015-2019).

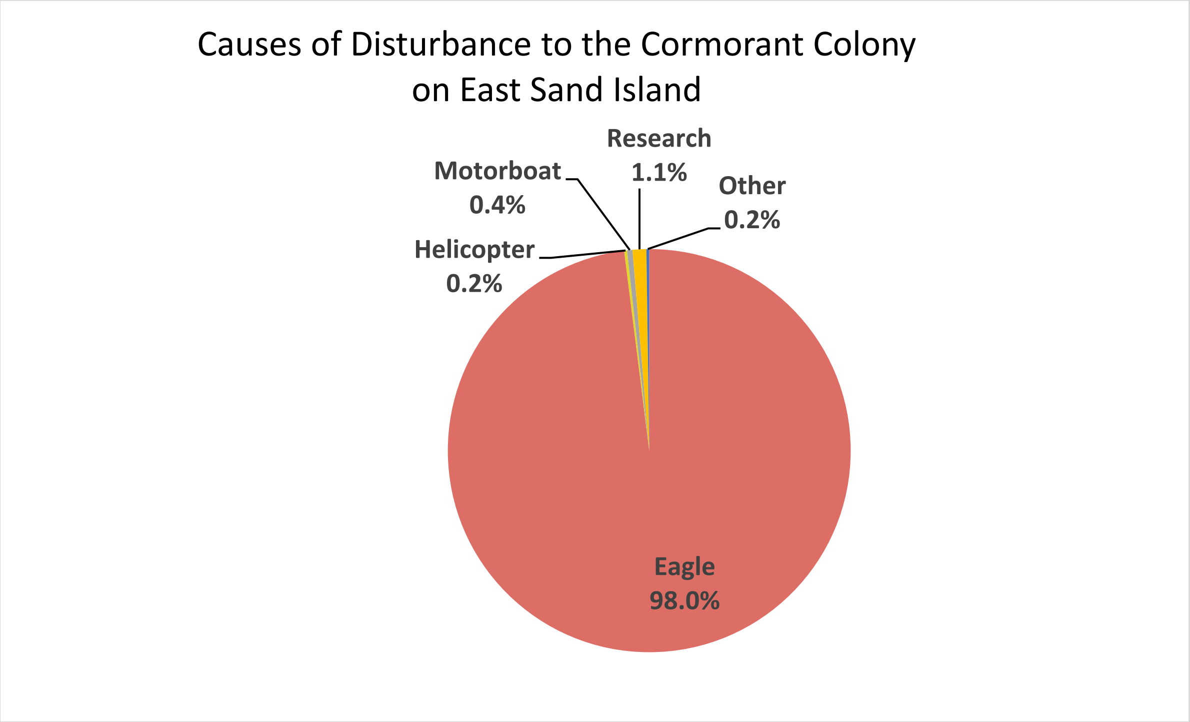

Figure 4.22. The relative proportion of all disturbance events to the double-crested cormorant colony on East Sand Island in the Columbia River estuary that were caused by different factors during the 2008–2012 nesting seasons.

During 2008–2012, there was variability both within and between years in the number of bald eagles present in the vicinity (within 300 m) of the East Sand Island cormorant colony (*Figure 4.23*). In each year, the seasonal peak in the average number of eagles present near the cormorant colony occurred early in the cormorant nesting period (late April), followed by a gradual decline in the average number of eagles until late in the cormorant chick-rearing period (late July). There were an estimated 2.3 times more bald eagles present near the cormorant colony at the beginning of the 2011 nesting season compared to the 2008–2010 nesting seasons ($P < 0.001$; 95% c.i. = 1.7 to 3.3 times more eagles). There were an estimated 1.6 times more bald eagles present near the cormorant colony at the beginning of the 2012 nesting season compared to the 2008–2010 nesting seasons ($P = 0.004$; 95% c.i. = 1.1 to 2.2 times more eagles). The rate of decline in the average number of eagles throughout the cormorant nesting season was not significantly different in 2012 than during 2008–2010 ($P = 0.54$); however, the number of bald eagles declined at a greater rate in 2011 than during 2008–2010 (μ = 1.1 times more eagles; 95% c.i. = 1.05 to 1.16 times more eagles, $P < 0.001$; *Figure 4.23*).

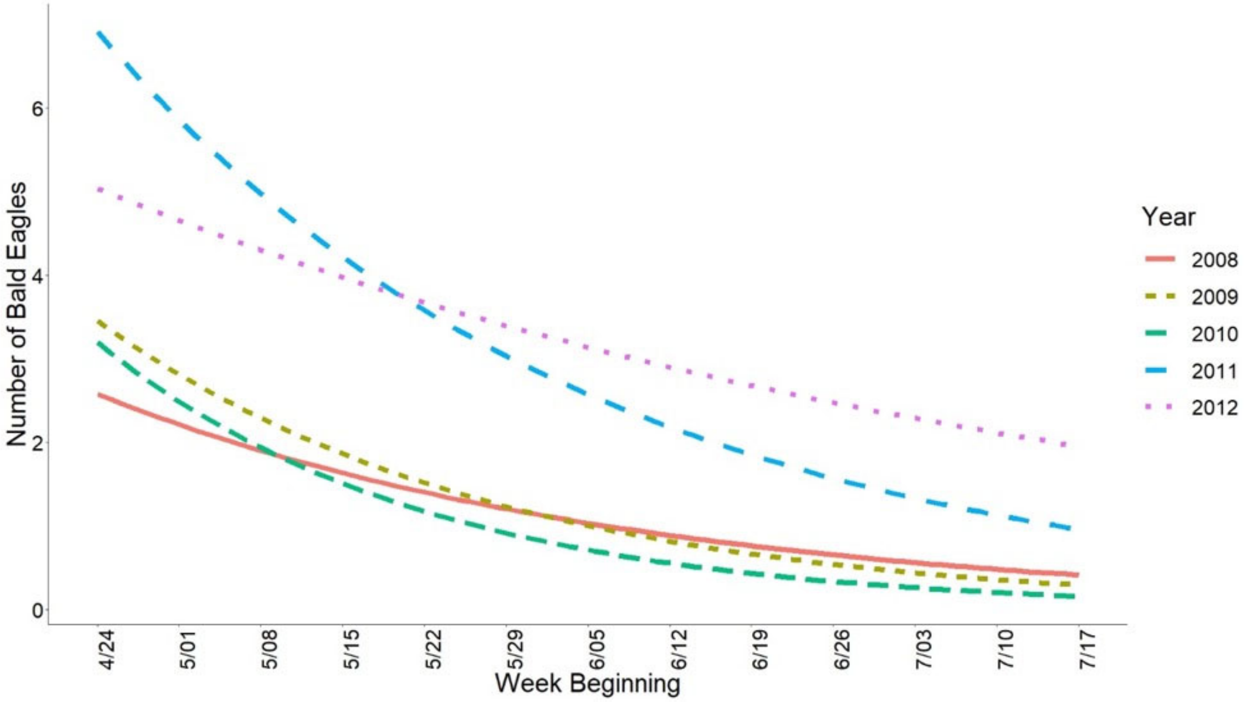

Figure 4.23. Seasonal trend in the average number of bald eagles within the vicinity (300 m) of the double-crested cormorant colony on East Sand Island during the breeding seasons of 2008–2012. Lines represent fitted models from multiple linear regression with a negative binomial distribution.

The average number of bald eagles counted in the vicinity of the cormorant colony at East Sand Island during 2008–2012 was significantly correlated with both the rate of Columbia River discharge in May of that year (1,000s of cubic feet/second [kcfs]; $P < 0.001$) and the year ($P < 0.001$; *Figure 4.24*). For every 100 kcfs increase in river discharge during May, there was on average 1.5 times more eagles present on or near the cormorant colony (95% c.i. = 1.3–1.7 times more eagles). For each increase of one year, there was on average 1.2 times more eagles on or near the cormorant colony (95% c.i. = 1.1–1.2 times more eagles). Thus, during the 2008–2012 study period bald eagles were increasingly prevalent in and near the cormorant colony on East Sand Island, especially when river discharge was relatively high. River discharge is negatively correlated with the prevalence of marine forage fish in the Columbia River estuary (Collar et al. 2017).

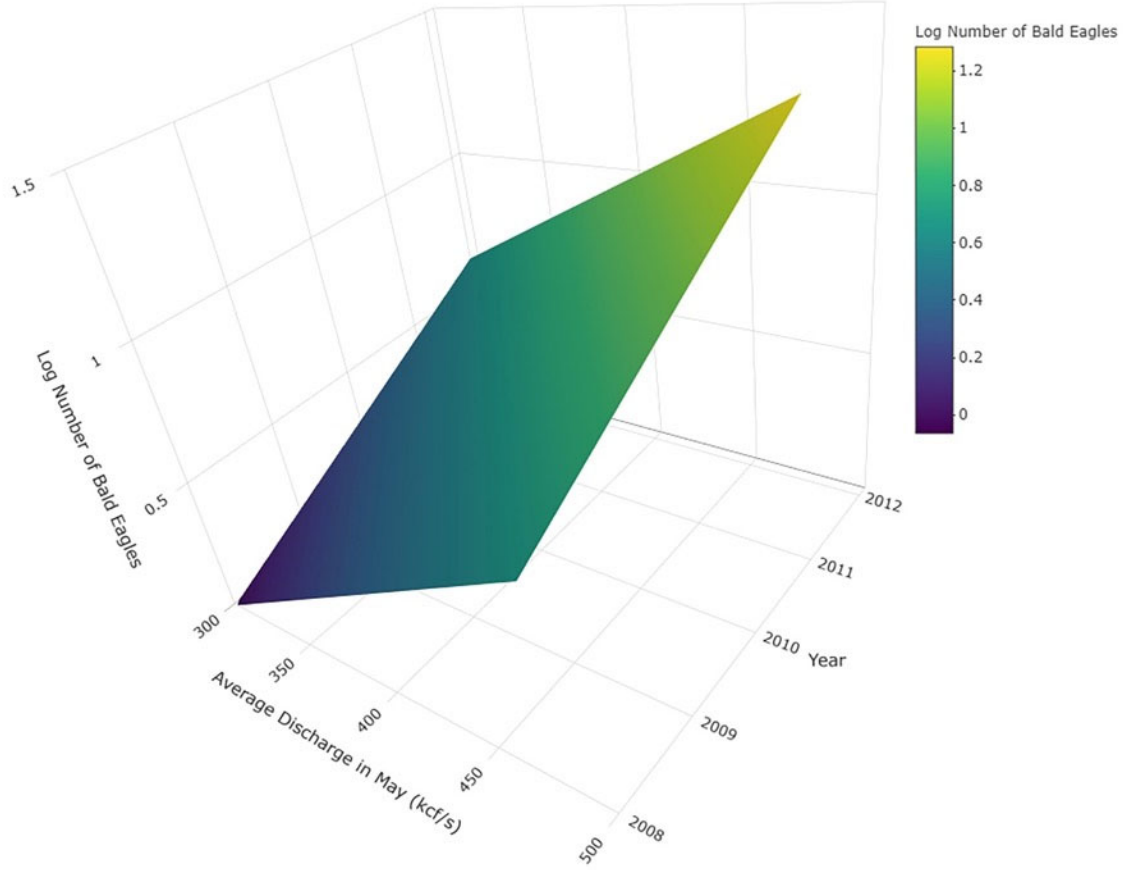

Figure 4.24. Number of bald eagles (log transformed) within the vicinity (300 m) of the double-crested cormorant colony on East Sand Island in the Columbia River estuary as a function of average Columbia River discharge (thousands of cubic feet per second; kcfs) in May and as a function of year (2008–2012). Discharge was measured at the USGS hydrological gage number 14246900 near Clatskanie, Oregon.

During 2008–2012, there was also a significant positive correlation between the number of disturbance events caused by bald eagles/hour and the average rate of Columbia River discharge in May (kcfs; Multiple $R^2 = 0.82$, $P = 0.03$; *Figure 4.25*). For every 100 kcfs increase in average river discharge during May, there were 0.4 more eagle disturbance events/hour (95% c.i. = 0.06–0.75 eagle disturbance events/hour). There was also a significant negative correlation between nesting success of double-crested cormorants and the eagle-caused disturbance rate of the cormorant colony at East Sand Island during 2008–2012 (Multiple $R^2 = 0.82$, $P = 0.03$; *Figure 4.26*). An increase in eagle disturbance rate of one disturbance event/hour was associated with an estimated 1.7 fewer young raised/breeding pair on average (95% c.i. = 0.2–3.2 fewer young raised/breeding pair).

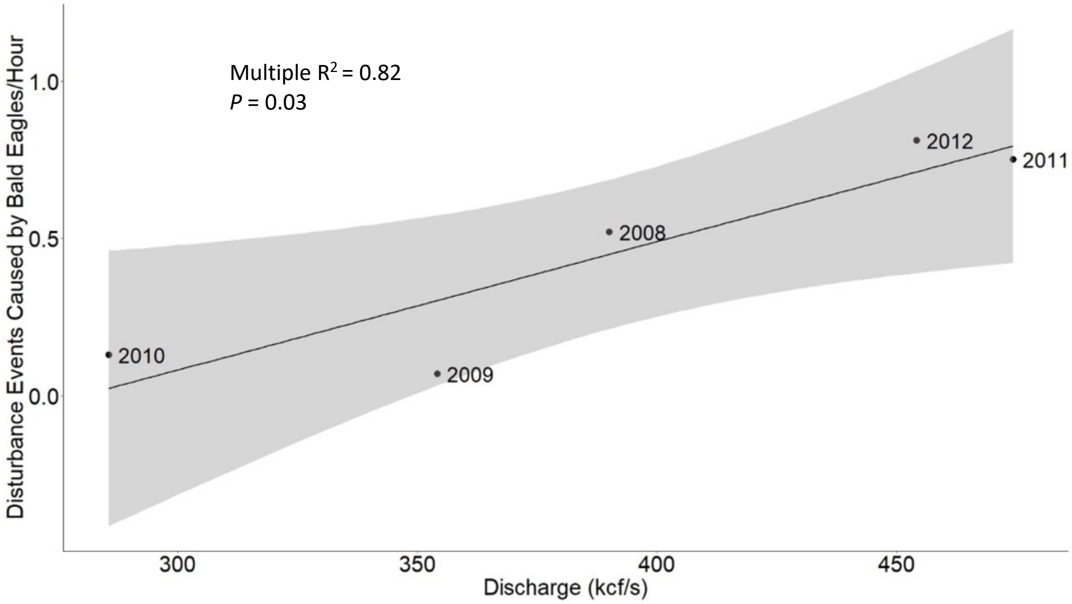

Figure 4.25. Bald eagle disturbance rate (disturbance events caused by bald eagles/hour) for double-crested cormorants nesting at the colony on East Sand Island in the Columbia River estuary as a function of average Columbia River discharge (thousands of cubic feet per second) in May during 2008–2012. Discharge was measured at the USGS hydrological gage number 14246900 near Clatskanie, Oregon.

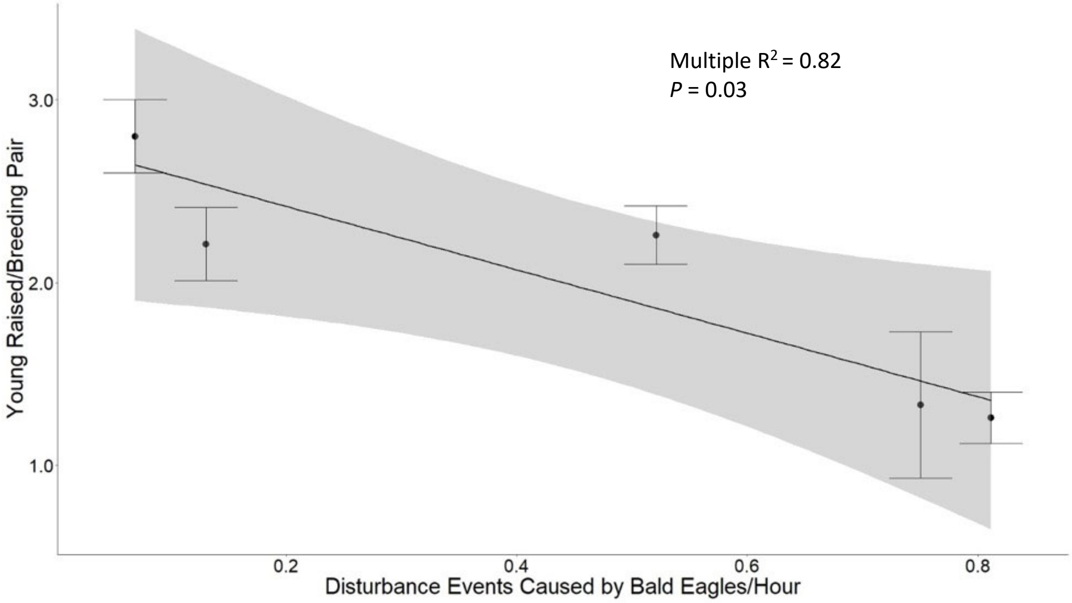

Figure 4.26. Nesting success (average number of young raised per breeding pair) as a function of bald eagle disturbance rate (disturbance events caused by bald eagles/hour) for double-crested cormorants nesting at the colony on East Sand Island in the Columbia River estuary during 2008–2012. The best fit linear regression model (solid line) and 95% c.i. (shaded area) are based on point estimates for each year.

Nesting success of double-crested cormorants at East Sand Island also appeared to be influenced by bottom-up factors, such as forage fish availability. During 1997–2005, the first half of the pre-management period, there was little evidence of a correlation between average cormorant nesting success (young raised/breeding pair) at East Sand Island and the proportion of juvenile salmonids in the diet (percent of total prey biomass; Adjusted $R^2 = 0.07$, $P = 0.29$; *Figure 4.27*). There was, however, a highly significant negative correlation between nesting success and the proportion of juvenile salmonids in the diet of double-crested cormorants nesting on East Sand Island during the second half of the pre-management period, 2006–2013 (Adjusted $R^2 = 0.80$, $P = 0.002$; *Figure 4.27*). A 10 percentage-point increase in the proportion of salmonids in the diet was associated with an estimated 1.1 fewer young raised/breeding pair (95% c.i. = -0.34 to -1.9 young raised/breeding pair). It is likely that the nesting success of double-crested cormorants was influenced by consumption of sub-yearling Chinook salmon smolts in particular given the relative importance of this species/run type to the salmonid portion of the diet (see section *4.4.6 Salmonid Smolt Consumption*). But the lower average nesting success in years when sub-yearling Chinook salmon were a higher proportion of the cormorant diet likely reflects the lower availability of marine forage fish in those years, rather than the relatively high availability of sub-yearling Chinook salmon.

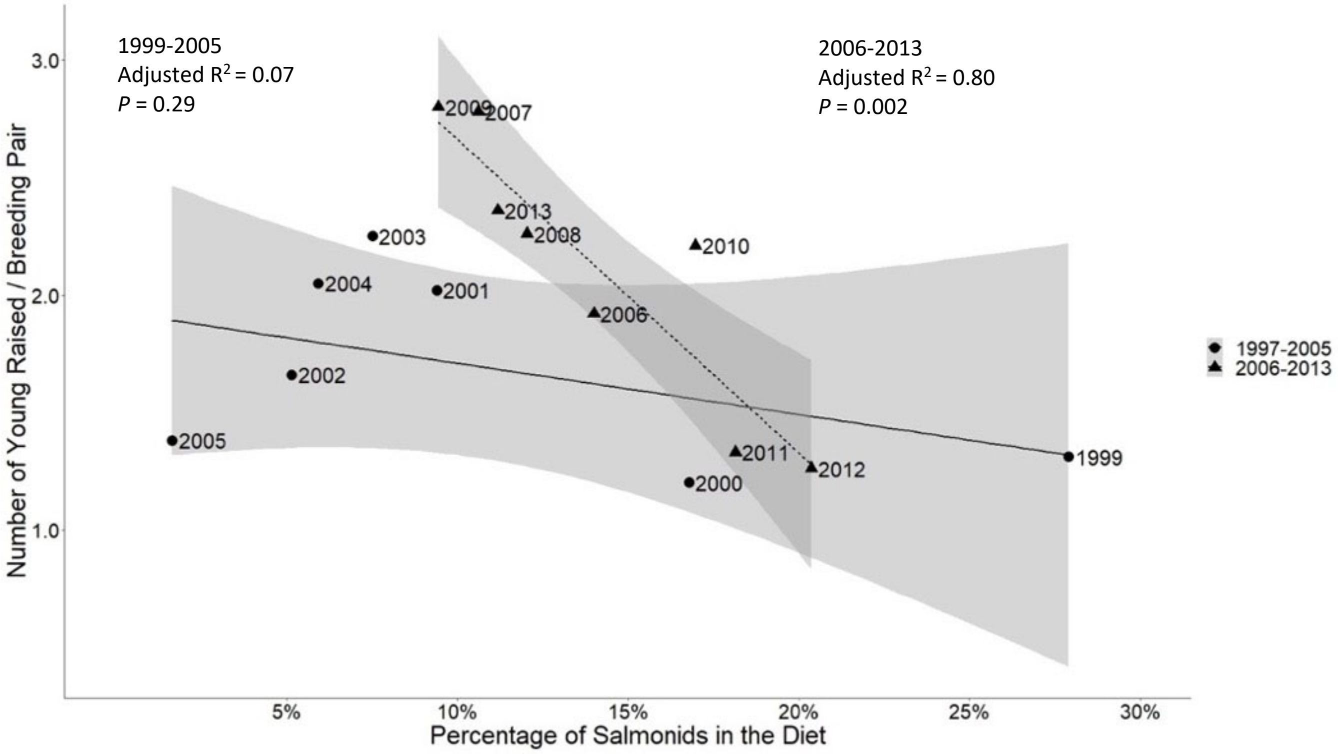

Figure 4.27. Nesting success (average number of young raised/breeding pair) as a function of average proportion of salmonids (percent of prey biomass) in the diet of double-crested cormorants nesting at the colony on East Sand Island in the Columbia River estuary during 1997–2005 and during 2006–2013.

The strong negative relationship between the proportion of salmonids in the diet and cormorant nesting success during 2006–2013 suggests that when non-salmonid forage fish were relatively less available fewer young cormorants were raised to fledging. Also, beginning in 2006 there was a notable shift in diet composition to a higher proportion of juvenile salmonids and a lower proportion of clupeids (herring, sardines, and shad) for double-crested cormorants nesting on East Sand Island. As the proportion of clupeids in the diet decreased it was somewhat compensated by an increase in the proportion of northern anchovy in the diet (see section *4.4.5 Diet Composition*). Clupeids were more prevalent in the cormorant diet early in the nesting season, whereas anchovies were more prevalent in the diet late in the nesting season (Lyons 2010), suggesting that the availability of anchovies was a larger factor for cormorant nesting success than the availability of clupeids.

The strong negative relationship between the proportion of juvenile salmonids in the diet and nesting success of double-crested cormorants on East Sand Island during the latter half of the pre-management period (2006–2013; *Figure 4.27* above) is more comprehensible when the role of Columbia River discharge is considered. During 1997–2005, the first half of the pre-management period, there was little evidence of a correlation between river discharge and nesting success of double-crested cormorants on East Sand Island; however, during 2006–2013 the correlation was negative and highly significant (P = 0.01; *Figure 4.28*). During 2006–2013, therefore, those years when river discharge in May was relatively high were also years when the proportion of salmonids in the cormorant diet was high. Because river discharge is negatively associated with the prevalence of marine forage fish in the Columbia River estuary (Weitkamp et al. 2012), the paucity of marine forage fish in the estuary during years when May discharge was high can potentially explain both the prevalence of juvenile salmonids in the diet (Lyons et al. 2014) and the relatively low nesting success by East Sand Island cormorants.

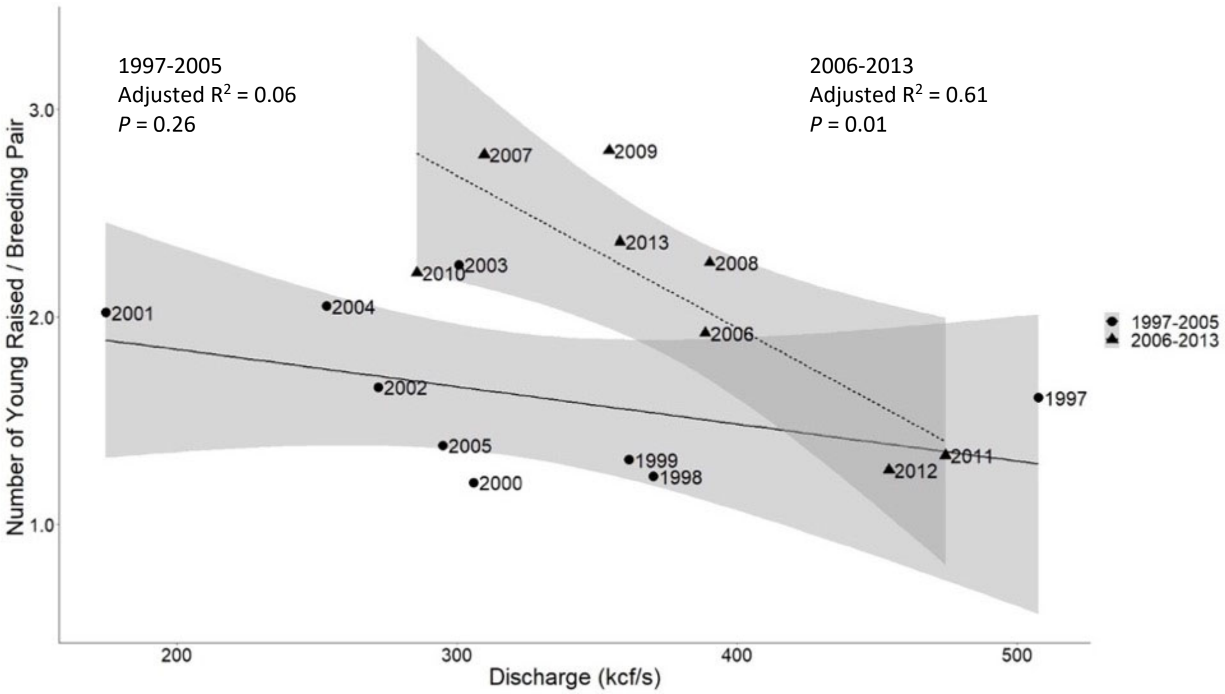

Figure 4.28. Nesting success (average number of young raised/breeding pair) for double-crested cormorants nesting at the colony on East Sand Island in the Columbia River estuary as a function of average Columbia River discharge (thousands of cubic feet per second; kcfs) in May during 1997–2005 and during 2006–2013. Discharge was measured at the USGS hydrological gage number 14246900 near Clatskanie, Oregon.

The nesting success of Caspian terns (average number of young raised/breeding pair) at the large colony on East Sand Island has also been shown to be negatively correlated with Columbia River discharge (Collar et al. 2017). Double-crested cormorant nesting success at East Sand Island was not, however, correlated with Caspian tern nesting success (Adjusted $R^2 = -0.07$, $P = 0.71$; *Figure 4.29*). The lack of any relationship between the nesting success of double-crested cormorants on East Sand Island and the nesting success of Caspian terns on the same island suggests there are major differences in the factors and/or the effects of the factors limiting nesting success for these two species of piscivorous colonial waterbirds. High Columbia River discharge likely reduced nesting success in cormorants and terns by both increasing the rate of disturbance caused by bald eagles (and subsequent nest depredation by gulls) and reducing the availability of marine forage fish in the estuary. The lack of a correlation in nesting success between cormorants and terns may be due in part to the shift in cormorant diet after 2005, but other factors are likely also responsible. One factor for the lack of a correlation is inter-specific differences in foraging mode as they influence availability of forage fish: double-crested cormorants are generalist predators that forage by pursuit-diving throughout the water column, whereas Caspian terns are more selective predators that forage by plunge-diving in the top meter of the water column. Also, cormorants transport food from foraging areas to the colony in their forestomach, whereas Caspian terns transport whole fish in their bills, which makes terns highly susceptible to kleptoparasitism by gulls, whereas cormorants are not.

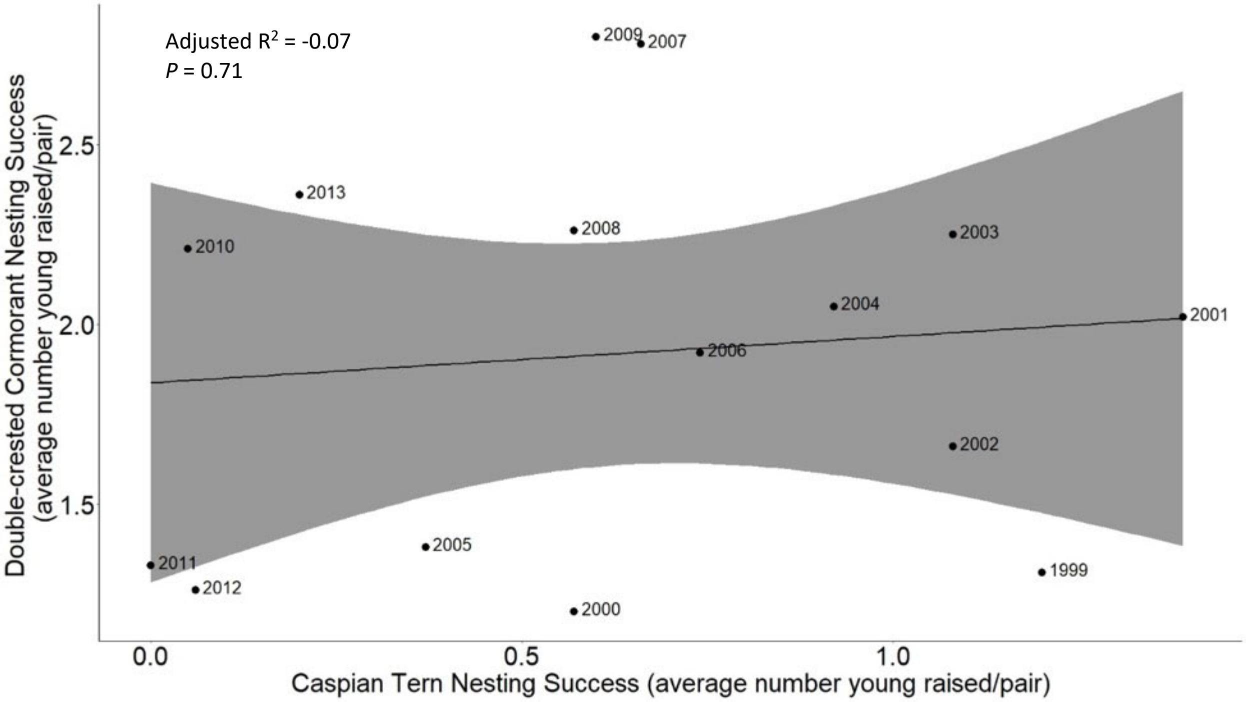

Figure 4.29. Annual nesting success (average number of young raised/breeding pair) of double-crested cormorants at East Sand Island in the Columbia River estuary vs. annual nesting success (average number of young raised/breeding pair) of Caspian terns at East Sand Island during 1999–2013.

During the management period (2015–2019), data on cormorant nesting success were collected in one year (2018) and no data were collected on cormorant diet composition or predator pressure on the cormorant colony at East Sand Island. Anecdotal observations of the cormorant colony in 2016, 2017, and 2019 suggest, however, that nesting success was very low compared to the pre-management period (Lawonn, *In prep.*), and even compared to 2018, the one year during the management period when nesting success was estimated. No cormorant young were apparently raised on East Sand Island during either the 2019 nesting season (Lawonn, In prep.) or the 2020 nesting season (M.J. Lawonn, ODFW, pers. comm.). The factors limiting cormorant nesting success on East Sand Island during the management period were likely the same as those that limited cormorant colony size pre-management: river discharge, eagle disturbance rates, and low availability of marine forage fish; however, human disturbance associated with management actions and effectiveness monitoring as part of the *Cormorant Management Plan* clearly had an additive impact on cormorant nesting success. In the absence of empirical data, however, it is not possible to determine the relative roles played by these four factors in limiting nesting success of double-crested cormorants on East Sand Island during the management period.

4.4.4 Foraging Behavior

Studies of the foraging behavior of double-crested cormorants nesting at East Sand Island using VHF telemetry in 2000–2001 (Anderson et al. 2004a) and GPS-telemetry in 2014 (Peck-Richardson et al. 2018) yielded complementary results. Detailed results from the two studies can be found in the respective peer-reviewed journal articles and are only briefly summarized here. Tracking of adult cormorants using VHF telemetry indicated differences between the sexes in foraging behavior, with nesting male cormorants commuting greater distances (mean = 15.98 ± 0.72 km) from East Sand Island to forage compared to nesting females (mean = 8.93 ± 0.55 km). Both studies found that the foraging range of some tagged breeding individuals extended at least 40 km from the breeding colony at East Sand Island.

Double-crested cormorants fitted with VHF radio-tags during the 2000–2001 tracking study were recorded foraging exclusively in the Columbia River estuary and predominantly in the freshwater zone and the estuarine mixing zone (*Figure 4.3* above for a map of the salinity zones in the Columbia River estuary); 37% of off-colony detections of tagged cormorants were in the freshwater zone, 35% of off-colony detections were in the estuarine zone, and 27% of off-colony detections were in the marine zone. As reflected in the mean commuting distance of VHF radio-tagged male and female cormorants, there were sexual differences in the spatial distribution of foraging among the three salinity zones in the Columbia River estuary; females foraged principally in the estuarine mixing zone (66% of telemetry locations) and males foraged mainly in the freshwater zone (52% of telemetry locations). Data from the VHF tracking study also indicated that the foraging locations of individuals were not randomly distributed in the estuary, suggesting that individual cormorants exhibited preferences for particular foraging areas.

During the 2014 tracking study (Peck-Richardson et al. 2018), double-crested cormorants fitted with GPS-TDlog data loggers were recorded foraging in some of the same areas of the estuary as in the Anderson et al. (2004a) study, but also over a much larger geographic area and a greater diversity of habitats. GPS-TDlog tagged cormorants were recorded foraging in nearshore marine habitats north and south of the mouth of the Columbia River, as well as marine waters at the mouth of the Columbia River, the estuarine mixing zone of the Columbia River estuary, and the freshwater and slough habitats of the upper Columbia River estuary. GPS-TDlog tagged cormorants were also recorded foraging in the estuarine mixing zone of Willapa Bay, approximately 10 km north of East Sand Island (*Figure 4.2* above), an area not used by VHF-tagged individuals during the Anderson et al. (2004a) study. These double-crested cormorants were tracked commuting directly between the colony on East Sand Island and Willapa Bay, using an overland route over the approximately 5-km wide strip of land separating the Columbia River estuary and Willapa Bay, which, without crossing over land, would otherwise require commuting greater than 50 km.

4.4.5 Diet Composition

The diet of double-crested cormorants nesting at the colony on East Sand Island during the 1999–2013 breeding seasons was composed of a diverse mix of marine, estuarine, and freshwater forage fishes, anadromous fishes, and even a few crustaceans. There was, however, a high degree of inter-annual variability in the relative proportions of prey types (% of total prey biomass) in the diet (*Table 4.4* and *Figure 4.30*). Despite the inter-annual variability in diet composition, the single prey type that comprised the largest proportion of the diet in most years was northern anchovies (*Engraulis mordax*), which, on average, accounted for 27.6% (range = 11.7–38.6%) of prey biomass. On average, juvenile anadromous salmonids were the second most prevalent prey type in the diet of double-crested cormorants nesting at East Sand Island, accounting for an average of 12.5% (range = 1.7–27.9%) of prey biomass (*Table 4.4* and *Figure 4.30*). On average, 7.7% (range = 4.2–14.9%) of prey biomass was flatfishes (Pleuronectidae), 7.5% (range = 3.9%–12.2%) of prey biomass was sculpins (Cottidae), 7.1% (range = 0.3–29.6%) of prey biomass was species in the family Clupeidae (i.e. Pacific herring [*Clupea pallasi*], Pacific sardine [*Sardinops sagax*], and American shad [*Alosa sapidissima*]), 6.4% (range = 0.1–15.9%) of prey biomass was sticklebacks (Gasterosteidae), 5.4% (range = 1.0–12.9%) of prey biomass was surfperches (Embiotocidae), and 3.5% (range 0.6–7.7%) of prey biomass was minnows (Cyprinidae; i.e. northern pikeminnow [*Ptychocheilus oregonensis*] and peamouth minnow [*Mylocheilus caurinus*]). On average, the remaining 9.2% (range = 3.4–8.5%) of identifiable prey biomass consisted of lampreys (*Lampetra* spp. and *Entosphenus* spp.), sand lance (Ammodytidae), smelts (Osmeridae), suckers (Catostomidae), cods (Gadidae), gunnels (Pholidae), snake pricklebacks (Ophichthidae), greenlings and lingcods (Hexagrammidae), mackerels (Scombridae), and invertebrate prey types such as native and non-native crayfish (Malacostraca). On average, 13.2% of prey biomass was not identifiable to family or species, although it could be determined to be non-salmonid in origin.

Table 4.4. Annual diet composition (percent of total prey biomass) of double-crested cormorants nesting at East Sand Island in the Columbia River estuary during 1999–2013, based on analysis of foregut contents from collected individuals. "Salmonids" = Oncorhynchus spp.; "Anchovy" = northern anchovy (Engraulis mordax); "Pleuronectids" = flatfishes; "Sculpins" = Cottidae; "Clupeids" = Pacific herring (Clupea pallasi), Pacific sardine (Sardinops sagax), and American shad (Alosa sapidissima); "Stickleback" = Gasterosteidae; "Surfperch" = Embiotocidae; "Cyprinids" = northern pikeminnow (Ptychocheilus oregonensis) and peamouth minnow (Mylocheilus caurinus).

Year	Salmonids	Anchovy	Pleuronectids	Sculpins	Clupeids	Stickleback	Surfperch	Cyprinids	Other
1999	27.9%	27.5%	7.7%	4.3%	3.2%	1.6%	6.5%	7.7%	7.0%
2000	16.8%	11.7%	14.9%	10.7%	9.2%	3.4%	5.2%	5.4%	10.1%
2001	9.4%	16.3%	10.1%	8.0%	15.9%	0.1%	4.5%	2.0%	15.7%
2002	5.1%	17.9%	7.8%	6.2%	29.6%	0.9%	4.8%	3.6%	9.6%
2003	7.5%	19.3%	8.0%	4.5%	6.4%	0.5%	6.7%	3.6%	18.5%
2004	5.9%	38.2%	10.3%	3.9%	13.1%	3.3%	4.2%	0.6%	5.8%
2005	1.7%	26.1%	6.3%	12.2%	9.3%	6.1%	12.9%	4.7%	3.4%
2006	14.0%	32.4%	4.3%	6.2%	3.3%	12.8%	8.5%	0.0%	5.0%
2007	10.6%	38.6%	9.5%	9.5%	1.7%	3.0%	4.6%	0.0%	7.8%
2008	12.0%	34.1%	4.3%	10.2%	2.5%	15.9%	4.1%	4.6%	7.8%
2009	9.4%	25.0%	9.5%	9.6%	3.6%	9.4%	6.0%	5.1%	7.7%
2010	17.0%	24.2%	4.8%	5.1%	3.2%	9.7%	6.2%	4.1%	11.4%
2011	18.1%	34.5%	4.2%	6.2%	3.4%	10.9%	1.0%	4.3%	8.3%
2012	20.4%	37.1%	5.1%	10.2%	1.1%	11.0%	3.9%	2.9%	4.0%
2013	11.2%	31.1%	9.5%	5.0%	0.3%	8.2%	1.5%	4.0%	16.0%
Average (1999–2013)	**12.5%**	**27.6%**	**7.7%**	**7.5%**	**7.1%**	**6.4%**	**5.4%**	**3.5%**	**10.4%**

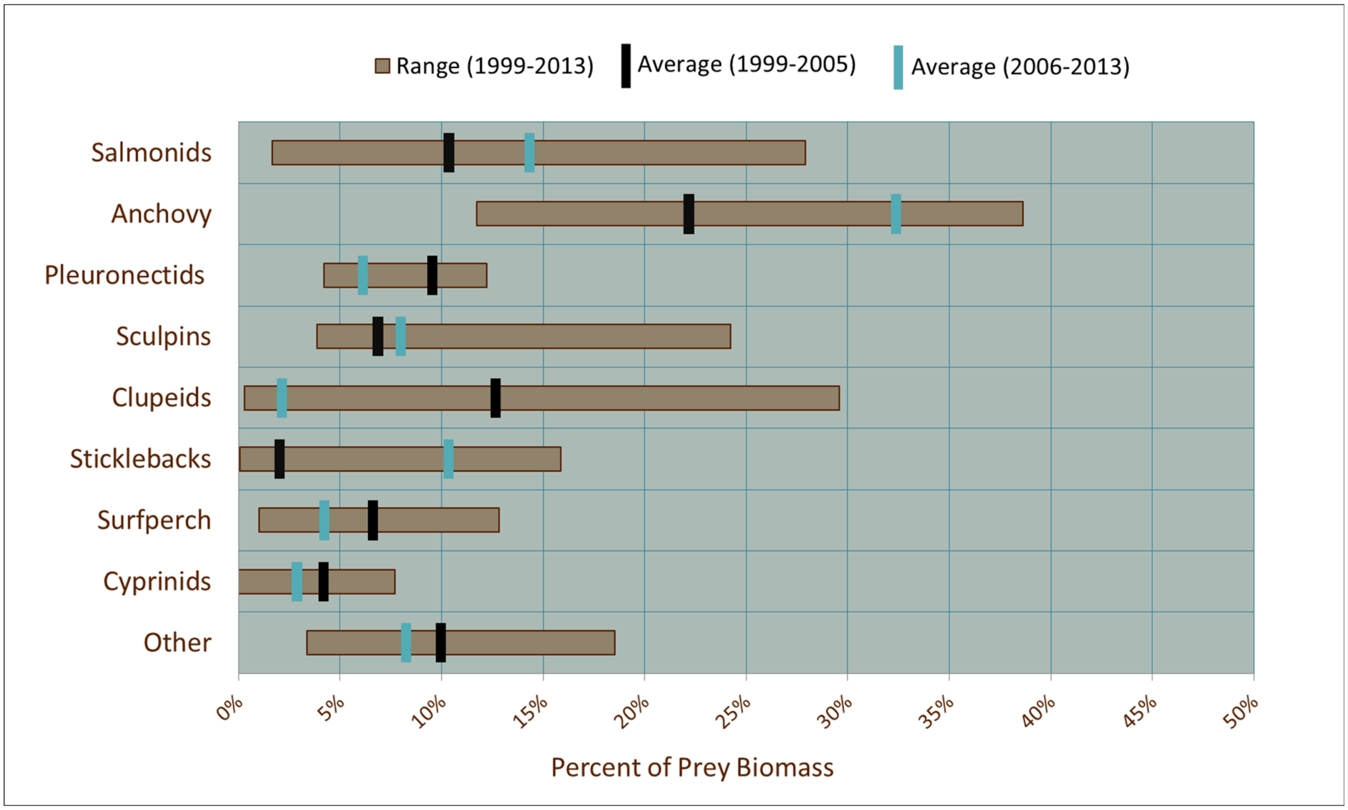

Figure 4.30. Percent biomass of major prey types in the diet of double-crested cormorants nesting on East Sand Island in the Columbia River estuary, based on the analysis of foregut contents from collected individuals. The range of average annual values for percent biomass during 1999–2013 is shown for each major prey type, as well as the mean percentage during the early part of the study period (1999-2005; black bar) and the later part of the study period (2006-2013; blue bar). "Salmonids" = Oncorhynchus spp.; "Anchovy" = northern anchovy (Engraulis mordax); "Pleuronectids" = flatfishes; "Sculpins" = Cottidae; "Clupeids" = Pacific herring (Clupea pallasi), Pacific sardine (Sardinops sagax), American shad (Alosa sapidissima); "Sticklebacks" = Gasterosteidae; "Surfperch" = Embiotocidae; "Cyprinids" = northern pikeminnow (Ptychocheilus oregonensis) and peamouth minnow (Mylocheilus caurinus).

During the study period (1999–2013) the diet of double-crested cormorants nesting at East Sand Island contained the highest proportion of salmonids (27.9% of identifiable prey biomass) in 1999 (*Table 4.4* above), the first year of study. The proportion of salmonids in the diet declined during the early 2000s, however, with the lowest proportion of salmonids in the diet observed in 2005 (1.7% of identifiable prey biomass; *Figure 4.31*). Overall, during the 2001–2005 breeding seasons, salmonids made up less than 10% of prey biomass in each year. Conversely, during the 2006–2013 breeding seasons salmonids generally accounted for greater than 10% of prey biomass each year; the exception was 2009 when salmonids comprised 9.4% of identifiable prey biomass (*Table 4.4* above and *Figure 4.31*). Concurrent with the relatively high prevalence of salmonids in the cormorant diet during 2006–2013, the prevalence of clupeids (i.e. Pacific herring, Pacific sardine, and American shad) in the diet was lower, less than 4% of identifiable prey biomass, similar to 1999 (*Table 4.4* above); clupeids were nearly absent from the cormorant diet (0.3% of prey biomass) during the 2013 breeding season, the last year of the diet study.

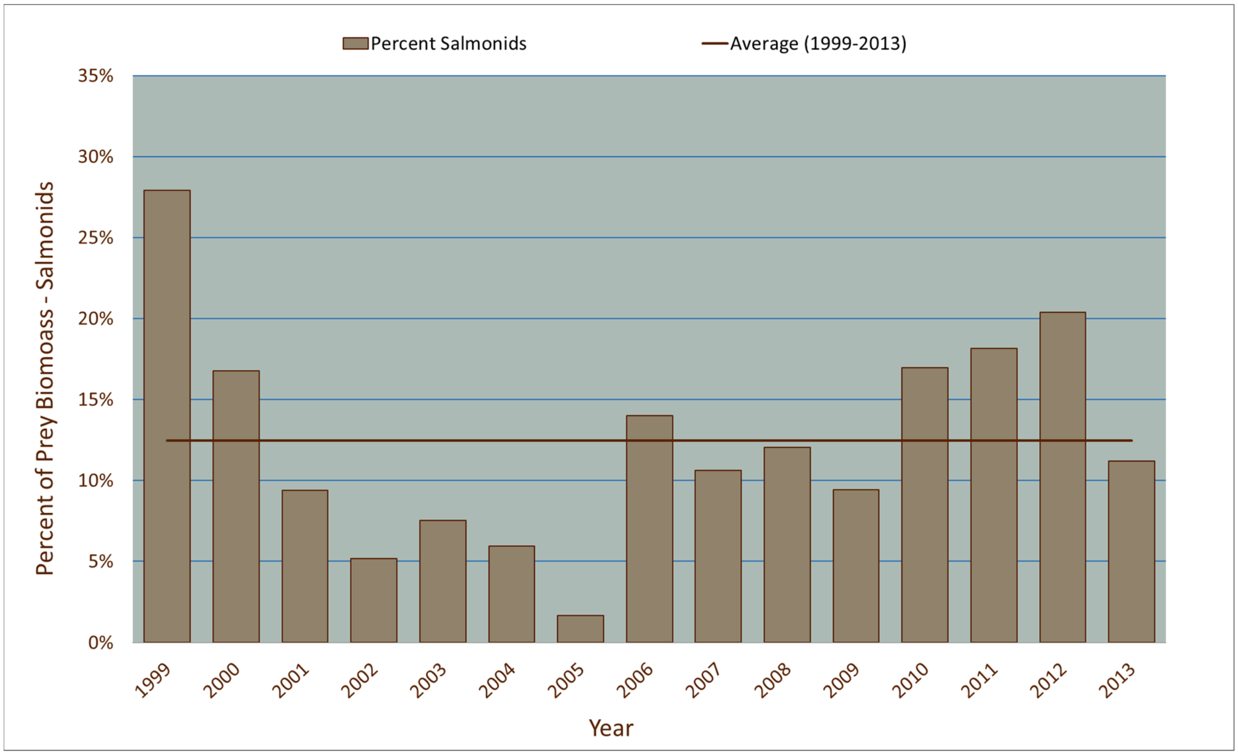

Figure 4.31. Average annual proportion of juvenile salmonids in the diet (percent of total prey biomass) of double-crested cormorants nesting at East Sand Island in the Columbia River estuary during 1999–2013. Solid line represents the average percent biomass of juvenile salmonids in diet samples from double-crested cormorants collected during 1999–2013.

Lyons et al. (2014) examined the relative impact of oceanographic and river conditions on the proportion of salmonids in the diet of double-crested cormorants nesting on East Sand Island using Principal Component Regression (PCR). Inter-annual variation in the proportion of salmonids in the diet during the 1999–2013 breeding seasons was best explained by three indices of ocean conditions as they influence availability of alternative prey (marine forage fish) in the Columbia River estuary: (1) date of spring transition to upwelling in the northern California Current system, (2) relative phase of the Pacific Decadal Oscillation (PDO), and (3) relative phase of the Multivariate ENSO Index (MEI). Lyons et al. (2014) also examined the relationship between estuary purse seine (EPS) catches and the diet composition of double-crested cormorants nesting at East Sand Island and showed that the annual proportion of salmonid biomass in the diet was significantly related to the annual proportion of salmonid biomass in EPS catches. Higher proportions of marine and freshwater/estuarine forage fishes in EPS catches were associated with lower proportions of salmonids in the diet of the East Sand Island cormorants (*Figure 4.32*). Finally, Lyons et al. (2014) found little support for the hypothesis that double-crested cormorants preferentially selected salmonids as prey based on the log odds ratios of the percent salmonids in the cormorant diet to that in the EPS catches (*Figure 4.33*). Based on this analysis, anchovy and flatfish appeared to be present in the cormorant diet at a level greater than expected while clupeids and smelts were present in the diet less than expected based on their abundance in EPS hauls. The EPS surveys were designed to sample out-migrating salmonids, however, and these results likely indicate issues related to the sampling approach (bias toward catching salmonids; see Weitkamp et al. 2012) rather than foraging preferences of double-crested cormorants in the Columbia River estuary on either northern anchovy (a primarily marine forage fish) or clupeids and smelt (both euryhaline estuarine residents).

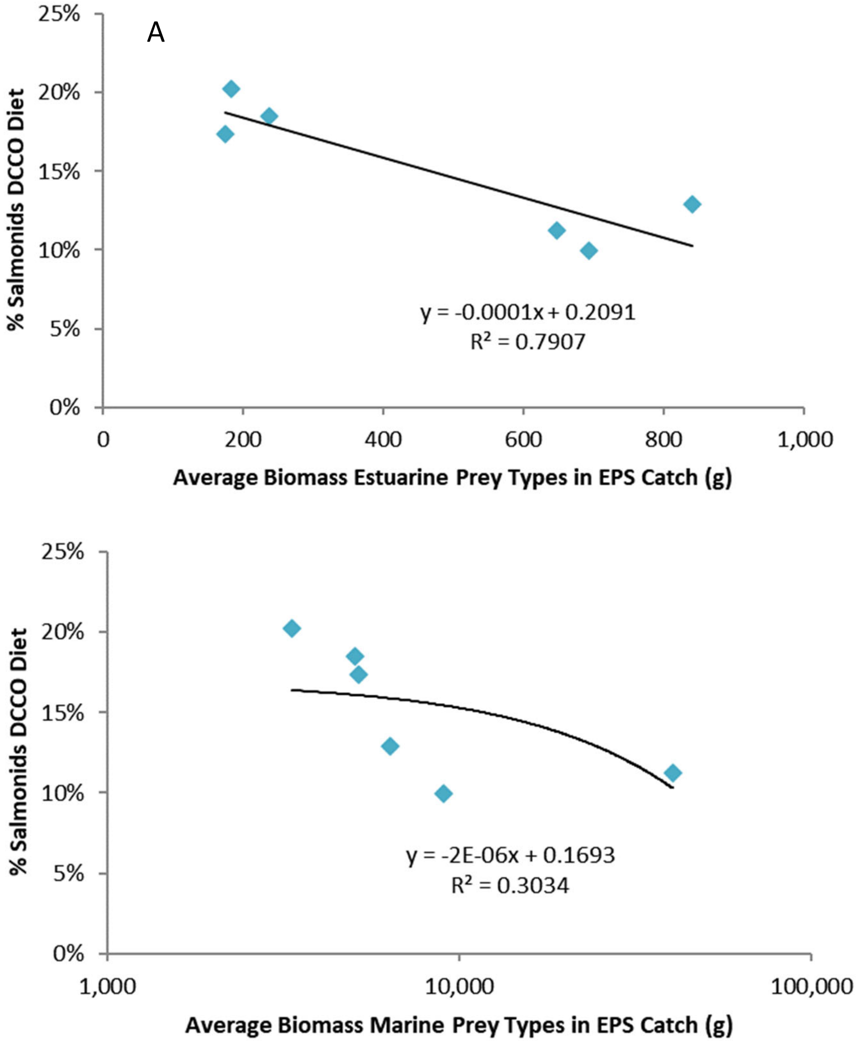

Figure 4.32. Relationships between the proportion of salmonids in the diet of double-crested cormorants (DCCO) nesting on East Sand Island and (A) the average total biomass of estuarine forage fishes in EPS catches and (B) average total biomass of marine forage fishes in estuary purse seine (EPS) catches. Each data point represents one year of study during 2007–2012. Greater absolute availability of alternative prey was associated with reduced cormorant reliance on juvenile salmonids. Figures from Lyons et al. (2014).

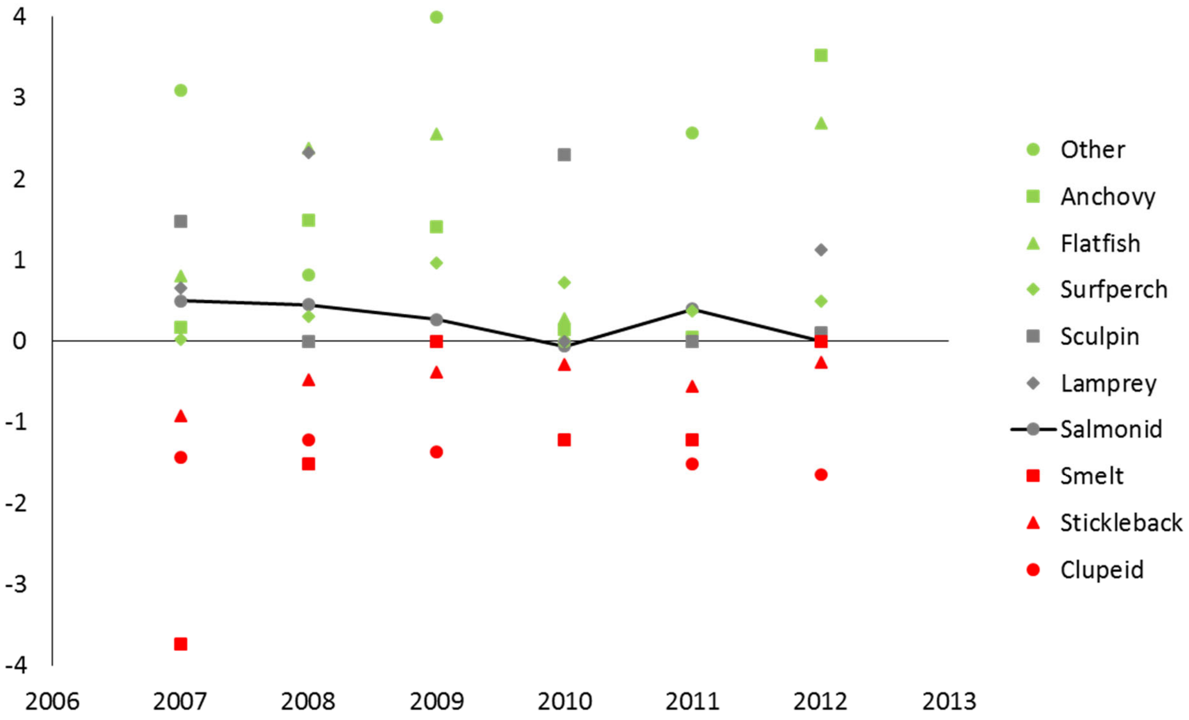

Figure 4.33. Log odds ratio of the percent of prey items by taxonomic group in the diet of double-crested cormorants nesting on East Sand Island in the Columbia River estuary to the percent of forage fish in estuary purse seine catches in the Columbia River estuary during April–May. Positive values indicate greater prevalence in the cormorant diet than in seine catches. Compared to values for anchovy, flatfish, and surfperch, selectivity by double-crested cormorants for salmonids (black circles and line) was minimal. Figure from Lyons et al. (2014).

4.4.6 Salmonid Smolt Consumption

Best estimates from bioenergetics modeling indicated that double-crested cormorants nesting at East Sand Island consumed on average about 10.8 million out-migrating juvenile salmonids per year during 1999–2013 (range = 2.9–20.9 million; *Table 4.5* and *Figure 4.34*). Uncertainty in the annual estimates of smolt consumption by cormorants was high, however, and 95% confidence intervals around the estimates were large (*Figure 4.34*) because the number of samples of cormorant stomach contents collected during each breeding season was small (less than 150 individuals per year) and some cormorant stomach contents samples contained no salmonids. Nevertheless, point estimates of the number of salmonids consumed per year by double-crested cormorants nesting on East Sand Island appeared to fit a quadratic model during 1999–2013 (Multiple $R^2 = 0.76$, $P = 0.0002$; *Figure 4.35*). The estimated number of juvenile salmonids consumed by double-crested cormorants nesting on East Sand Island declined during 1999–2005, despite strong growth in the size of the colony (number of breeding pairs; *Figure 4.9* above; see section *4.4.1.1 Colony size* above). Beginning in 2006, the best estimate of the number of juvenile salmonids consumed by double-crested cormorants nesting at East Sand Island increased, primarily because the proportion of salmonids in the diet increased (*Figure 4.31* above); during 2006–2013 the size of the East Sand Island colony was stable (*Figure 4.11* above). In addition, there was a period of above average discharge from the Columbia River during the 2011–2012 nesting seasons (*Figure 4.25* above), which was apparently associated with a shift in the diet composition of double-crested cormorants nesting at East Sand Island toward a higher prevalence of salmonid smolts (*Figure 4.31* above and *Figure 4.34*).

Table 4.5. Estimated annual consumption, in millions of fish (95% confidence interval), of five species/age-classes of juvenile salmonids by double-crested cormorants nesting on East Sand Island in the Columbia River estuary during 1998–2013. Smolt consumption estimates are based on the proportion of species/age classes of salmonids found in cormorant diet samples and bioenergetics modeling.

	Salmonid Species/Age-class					
Year	Sub-yearling Chinook Salmon	Yearling Chinook Salmon	Coho Salmon	Sockeye Salmon	Steelhead Trout	Yearly Total
1998	10.32 (5.70-19.79)	0.46 (0.11-1.25)	0.92 (0.35-1.99)	0.02 (0.00-0.20)	0.56 (0.22-1.22)	12.3 (6.37-24.45)
1999	8.32 (4.29-16.40)	0.86 (0.27-2.06)	1.65 (0.75-3.46)	0.03 (0.00-0.31)	1.03 (0.52-2.06)	11.9 (5.83-24.29)
2000	4.45 (2.19-9.38)	0.81 (0.20-2.06)	1.31 (0.52-2.87)	0.03 (0.00-0.31)	0.91 (0.38-2.20)	7.5 (3.29-16.81)
2001	5.00 (2.35-11.11)	0.44 (0.07-1.30)	0.81 (0.24-2.05)	0.02 (0.00-0.21)	0.51 (0.17-1.26)	6.8 (2.83-15.93)
2002	4.06 (1.61-8.88)	0.10 (0.01-0.34)	0.32 (0.06-0.82)	0.001 (0.00-0.02)	0.12 (0.02-0.35)	4.6 (1.70-10.42)
2003	1.37 (0.42-3.87)	0.65 (0.06-2.15)	0.95 (0.26-2.49)	0.03 (0.00-0.36)	0.65 (0.20-1.68)	3.7 (0.94-10.55)
2004	5.34 (1.94-11.76)	0.53 (0.10-1.35)	0.99 (0.37-2.40)	0.02 (0.00-0.27)	0.59 (0.20-1.42)	7.5 (2.57-17.21)
2005	2.18 (0.47-6.29)	0.12 (0.003-0.41)	0.35 (0.07-1.00)	0.005 (0.00-0.11)	0.20 (0.05-0.65)	2.9 (0.59-8.46)
2006	2.74 (0.94-6.30)	1.57 (0.52-4.06)	3.25 (1.55-6.97)	0.04 (0.00-0.60)	1.72 (0.78-3.72)	9.3 (3.79-21.65)
2007	4.99 (1.68-11.88)	1.05 (0.33-2.71)	2.51 (1.19-5.59)	0.03 (0.00-0.38)	1.31 (0.54-2.93)	9.9 (3.73-23.48)
2008	5.85 (2.56-10.72)	0.93 (0.29-1.81)	1.75 (0.94-2.81)	0.03 (0.00-0.32)	0.91 (0.40-1.56)	9.5 (4.20-17.23)
2009	8.71 (3.67-17.01)	0.67 (0.13-1.41)	1.43 (0.57-2.59)	0.03 (0.00-0.27)	0.78 (0.27-1.44)	11.6 (4.65-22.71)
2010	13.80 (6.77-24.19)	1.22 (0.24-1.63)	3.00 (1.64-4.56)	0.03 (0.00-0.39)	1.52 (0.74-2.36)	19.6 (9.48-33.91)
2011	15.71 (9.09-25.54)	0.86 (0.24-1.63)	2.85 (1.51-4.32)	0.15 (0.00-0.91)	1.28 (0.63-2.10)	20.9 (11.48-34.50)
2012	11.11 (5.93-17.43)	1.49 (0.54-2.67)	4.84 (2.96-7.17)	0.20 (0.00-1.08)	1.76 (1.01-2.71)	19.4 (10.44-31.06)
2013	11.92 (5.97-21.26)	0.98 (0.30-2.01)	2.78 (1.34-4.58)	0.26 (0.00-1.21)	1.12 (0.51-1.87)	17.1 (8.12-30.92)

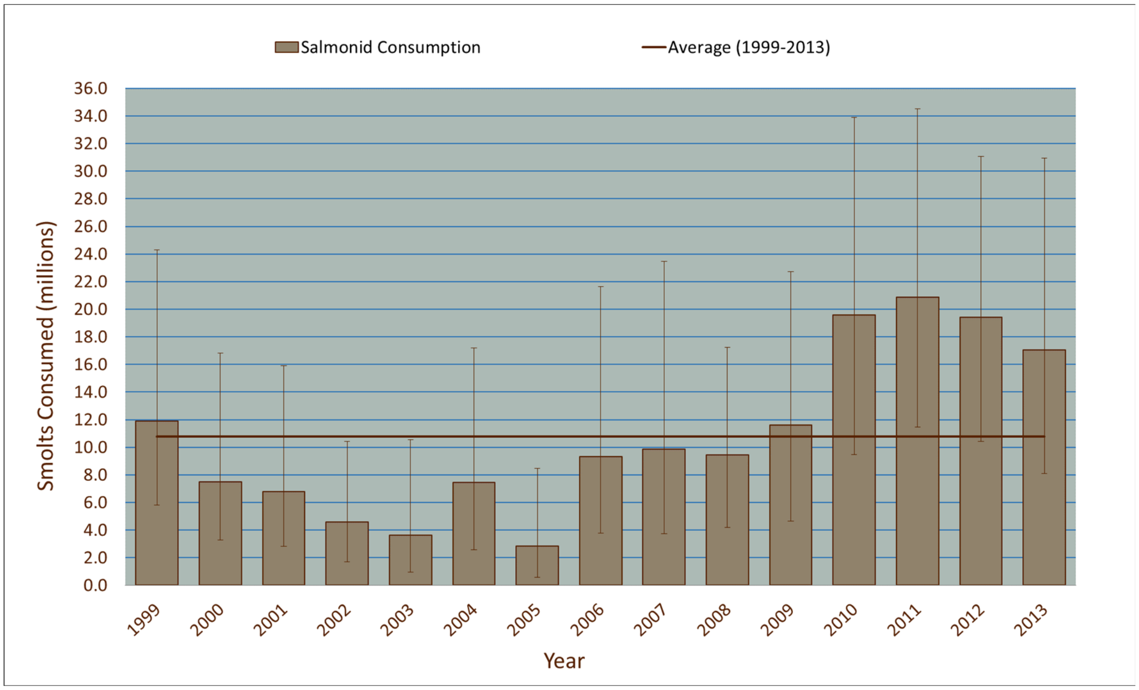

Figure 4.34. Estimated number (in millions) of juvenile salmonids consumed by double-crested cormorants nesting at East Sand Island in the Columbia River estuary during 1999–2013, based on bioenergetics modeling. Error bars represent the 95% confidence interval around each yearly estimate, and the solid line represents the average number of juvenile salmonids consumed per year during the study period, based on the best estimate for each year.

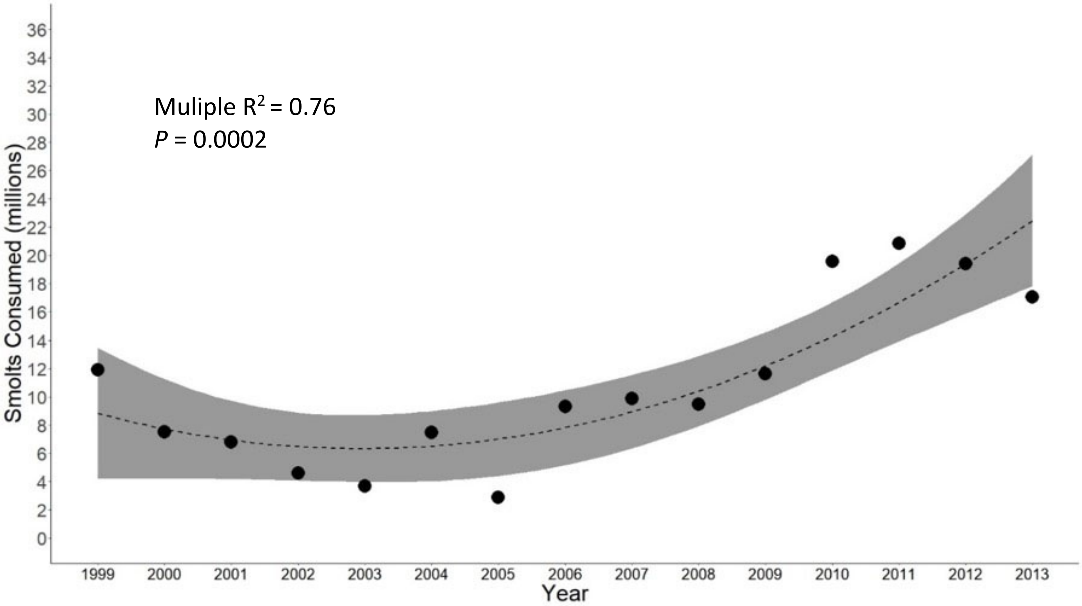

Figure 4.35. Estimated number (in millions) of juvenile salmonids consumed by double-crested cormorants nesting at East Sand Island in the Columbia River estuary during 1999–2013 based on bioenergetics calculations. Dashed line represents the best fit quadratic equation of the smolt consumption point estimates, and the shaded area represents the 95% confidence interval for that best fit quadratic equation.

During the 1998–2007 breeding seasons, Lyons (2010) found a positive correlation between average river discharge in May and the estimated number of spring migrant salmonid smolts consumed by double-crested cormorants nesting at East Sand Island. No such relationship existed, however, between May river discharge and cormorant consumption of sub-yearling Chinook salmon, likely because few sub-yearling Chinook are present in the estuary during May. Notably, the highest levels of consumption of sub-yearling Chinook salmon by cormorants typically occurred in June (Lyons et al. 2014). Lyons et al. (2014) used Principal Component Regression (PCR) analysis to show that colony size, river discharge, and the corresponding periodicity of the North Pacific Gyre Oscillation (NPGO) were the environmental factors that best explained the inter-annual variability in estimates of consumption of, and predation rates on, spring migrant smolts by double-crested cormorants nesting at East Sand Island during 1999–2013. Results of the PCR analysis indicated that colony size explained 12–17% of the inter-annual variability, river discharge explained 13%–15% of the inter-annual variability, and the NPGO explained an additional 6%–10% of the inter-annual variability in estimated numbers consumed and predation rates on spring migrant smolts (Lyons et al. 2014). A combination of relatively large cormorant colony size, relatively high river discharge, and relatively poor ocean conditions was responsible for the highest annual smolt consumption estimates by double-crested cormorants nesting at the East Sand Island colony during 2011 (*Table 4.5* above and *Figure 4.34* above).

Bioenergetics modeling was also used to estimate the numbers of smolts, separated by salmonid species/age-class, that were consumed by double-crested cormorants nesting on East Sand Island during 1999–2013 (*Table 4.5* above). Bioenergetics calculations indicated that double-crested cormorants did not consume salmonid species or age-classes with equal frequency (*Figure 4.36*), and that cormorants consumed a much greater number of sub-yearling Chinook salmon smolts than smolts of yearling Chinook salmon, coho salmon, sockeye salmon, or steelhead trout in most, but not all years (*Table 4.5* above). The bioenergetics model indicated, however, that during the 2006 breeding season, double-crested cormorants consumed more coho salmon smolts than those of any other salmonid species/age-class.

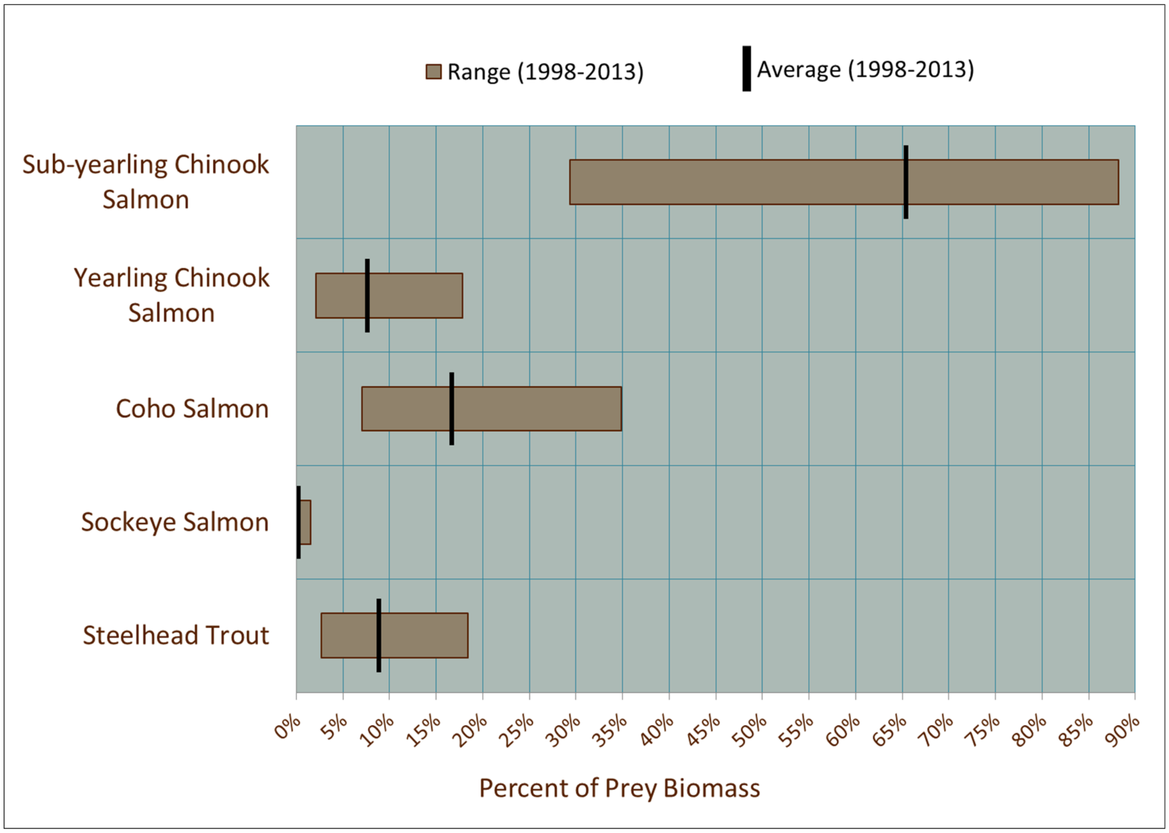

Figure 4.36. Percent contribution of five salmonid species/age-classes in the juvenile salmonid portion of the diet of double-crested cormorants nesting at East Sand Island in the Columbia River estuary, based on the analysis of foregut contents from collected individuals. The range of average annual values for percent biomass during 1998–2013 is shown for each salmonid species/age-class, as well as the average percentage (black bar) across the study period.

Based on bioenergetics calculations, sub-yearling Chinook salmon accounted for 66% on average (range = 29.4–88.2%) of the estimated total number of salmonid smolts consumed by double-crested cormorants nesting on East Sand Island during 1998–2013. Estimates derived from bioenergetics modeling indicated that the number of sub-yearling Chinook salmon consumed by cormorants nesting on East Sand Island ranged from a low of 1.4 million smolts (95% c.i. = 0.4–3.9 million smolts) in 2003 to a high of 15.7 million (95% c.i. = 9.1–25.5 million) in 2011 (*Table 4.5* above). Of the other four salmonid prey species/age-classes examined using bioenergetic models, coho salmon accounted for an average of 17% (range = 7.0–34.9%), steelhead trout accounted for an average of 9.1% (range = 2.7–18.4%), yearling Chinook salmon accounted for an average of 7.9% (range = 2.1–17.9%), and sockeye salmon accounted for an average of 0.44% (range = 0.02–1.15%) of the estimated total number of salmonid smolts consumed by double-crested cormorants nesting at East Sand Island (*Figure 4.36* above). The much higher prevalence of sub-yearling Chinook salmon in the diet of East Sand Island cormorants compared to other salmonid species/run-types is likely related to the run timing and residence time in the estuary for sub-yearling Chinook smolts, and the seasonal food energy requirements of nesting cormorants. Sub-yearling Chinook smolts enter the Columbia

River estuary later than spring migrants, such as yearling Chinook salmon, steelhead, and coho salmon, and reside in the estuary for an extended period (June-August) before ocean entry (Sebring et al. 2013; see also *Chapter 10*). The period of estuarine rearing for sub-yearling Chinook smolts coincides with the period of peak food energy requirements for nesting cormorants, when they are provisioning broods of large pre-fledging young at the colony (Lyons 2010).

4.4.7 Predation Rates on Juvenile Salmonids

Cormorant predation rates based on recoveries of juvenile salmonid (smolt) PIT tags were first measured at the double-crested cormorant colony on East Sand Island in 1999 (Ryan et al. 2001); these studies continued each year through 2018 (Evans et al. 2019a). During 1999–2018, over 340,000 smolt PIT tags (detections of individual fish) were recovered on the double-crested cormorant colony at East Sand Island, representing the second largest number of smolt PIT tags recovered on a single piscivorous waterbird colony in the Columbia River basin, second only to the Caspian tern colony on East Sand Island where over 530,000 PIT tags were recovered (Evans et al. 2019a; see also *Chapter 1*). Most fish tags recovered on the East Sand Island cormorant colony were from anadromous juvenile salmonids (> 99%), but small numbers of tags from other fishes (Pacific lamprey [*Lampetra tridentate*], white sturgeon [*Acipenser transmontanus*], and northern pikeminnow) were also recovered.

Because all anadromous juvenile salmonids from the Columbia River basin migrate through the estuary, all 13 ESA-listed ESUs/DPSs of salmonids from the basin are susceptible to predation by double-crested cormorants that nest and forage in the estuary (Lyons 2010, Evans et al. 2012; see also *Chapter 1*). The timing of the cormorant breeding season (April to September) also coincides with the peak smolt out-migration period in the estuary (April to August), resulting in near complete spatial and temporal over-lap between out-migrating juvenile salmonids and breeding double-crested cormorants (Evans et al. 2012; see also *Chapter 10*). Not all salmonid ESUs/DPSs, however, are PIT-tagged in sufficient numbers for estimation of cormorant predation rates based on PIT tag recoveries on breeding colonies in the Columbia River estuary (see *Appendix B*), so cormorant predation rates on all 13 ESA-listed populations of salmonids from the Columbia River basin were not available to this study.

Below we summarize ESU/DPS-specific annual predation rates and average annual predation rates by double-crested cormorants nesting at the East Sand Island colony during 2003–2018. Results are provided for each ESA-listed ESU/DPS where adequate data were available. To evaluate the efficacy of the *Cormorant Management Plan* to reduce predation rates on juvenile salmonids, predation rates were compared between the pre-management period (2003–2014) and the period when management actions aimed at reducing the size of the cormorant colony on East Sand Island were implemented (2015–2018). We focused the analysis on predation rates after 2002, the time period considered by the National Marine Fisheries Service as the “current” period for purposes of their Biological Opinion (NMFS 2008).

4.4.7.1 Upper Columbia River steelhead trout – Predation rate estimates on Upper Columbia River (UCR) steelhead by East Sand Island double-crested cormorants were highly variable, ranging annually from 0.7% (95% credible interval = 0.4–1.4%) to 11.4% (7.8–18.6%) of available PIT-tagged smolts during 2003–2018 (*Figure 4.37* and *Appendix B, Table B23*). Both wild and hatchery (from select hatcheries) steelhead are part of the ESA-listed UCR population (NMFS 2019) and comparisons of weekly and annual predation rates indicated that both rear-types were equally susceptible to predation by East Sand Island double-crested cormorants (Evans et al. 2019a; *Figure 4.38*; see also *Chapter 10*). During the pre-management period (2003-2014), the estimated average annual predation rate on UCR steelhead was 6.3% (5.5–7.2%; *Table 4.6*). During the management period (2015-2018), average annual predation rates were 5.8% (4.5–8.1%) during Phase I (2015-2017) and 0.7% (0.4–1.4%) during Phase II (2018; *Table 4.6*).

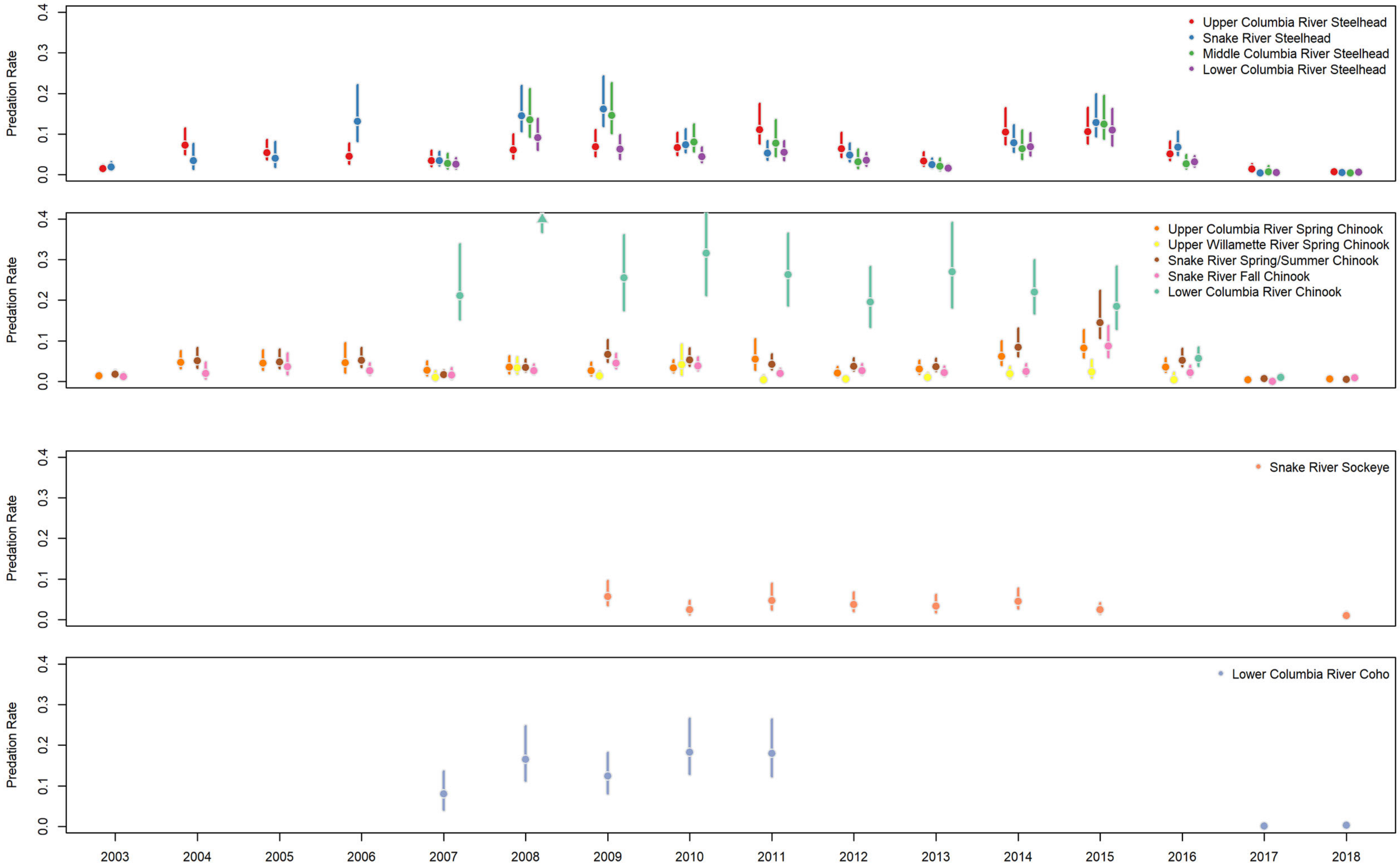

Figure 4.37. Annual predation rates (proportion of fish consumed) on ESA-listed juvenile salmonids by double-crested cormorants nesting on East Sand Island during 2003–2018. Error bars denote 95% credible intervals. Estimates during 2016–2018 represent minimum estimates of predation rates due to colony dispersal events that occurred during the peak smolt out-migration period (2016–2017) or when nesting was delayed (2018). Results are those of Evans et al. (2019a) or are newly reported for Lower Columbia River ESUs/DPSs (see also Appendix B, Table B23-B24).

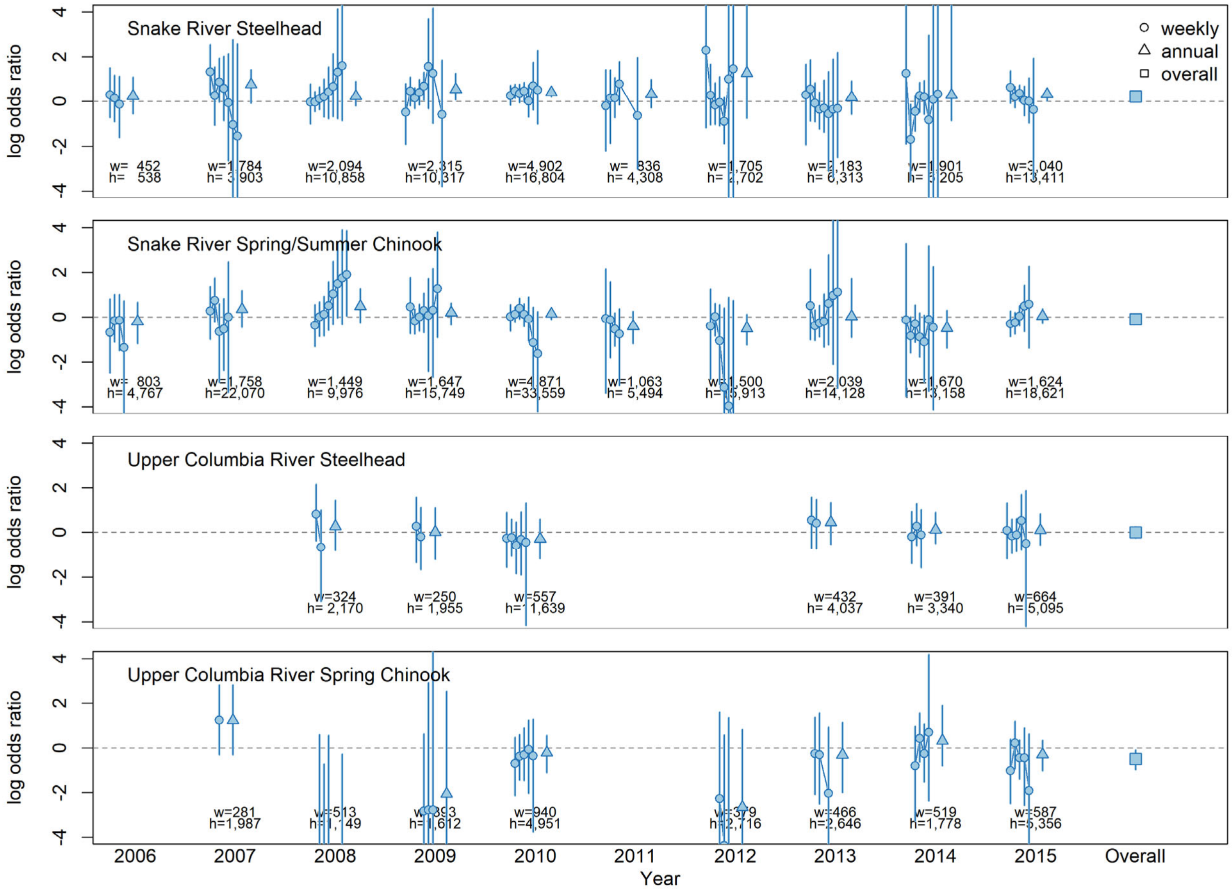

Figure 4.38. Relative susceptibility of hatchery and wild juvenile salmonids to predation by double-crested cormorants nesting on East Sand Island during 2006–2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for hatchery fish and values > 0 indicating greater predation odds for wild fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Only years when > 500 PIT-tagged fish of each rearing type were available are included. Only weeks when > 50 PIT-tagged fish of each rearing type were available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right. Figure from Evans et al. (2019a).

Table 4.6. Average annual predation rates on salmonid populations from the Columbia River basin (95% credible intervals) by double-crested cormorants nesting on East Sand Island prior to implementation of management actions (pre-management) and following implementation (Phase I and Phase II). ESA-listed salmonid populations (ESUs/DPSs) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. Asterisks denote statistically credible differences between pre-management and Phase I or Phase II of the management period. NA denotes that estimates were not available during that time period. See Appendix B, Tables B23-B24 for annual estimates.

Salmonid ESU/DPS	**Pre-management**	**Phase I** [1]	**Phase II** [1]
	2003–2014	2015–2017	2018
Upper Columbia River Steelhead	6.3% (5.5–7.2%)	5.8% (4.5–8.1%)	0.7% (0.4–1.4%)*
Upper Columbia River Sp Chinook	3.8% (3.2–4.6%)	4.1% (3.2–5.8%)	0.6% (0.3–1.2%)*
Snake River Steelhead	7.2% (6.3–8.5%)	6.8% (5.3–9.4%)	0.5% (0.3–0.9%)*
Snake River Sp/Su Chinook	4.6% (4.1–5.3%)	6.8% (5.3–9.4%)	0.5% (0.3–0.8%)*
Snake River Fall Chinook	2.7% (2.3–3.2%)	3.7% (2.6–5.4%)	0.9% (0.5–1.6%)*
Snake River Sockeye[2]	4.2% (3.3–5.3%)	2.4% (1.4–4.0%)	0.9% (0.4–1.9%)*
Middle Columbia River Steelhead[3]	7.5% (6.3–9.3%)	5.4% (4.0–7.0%)	0.4% (0.1–1.0%)*
Upper Willamette River Steelhead	NA	NA	NA
Upper Willamette River Sp Chinook[4]	1.8% (1.3–2.6%)	1.4% (0.6–2.9%)	NA
Lower Columbia River Steelhead[5]	5.4% (4.5–6.3%)	5.0% (3.7–6.9%)	0.6% (0.3–1.0%)*
Lower Columbia River Chinook[5]	27.5% (24.3–30.7%)	8.7% (6.2–12.1%)*	7.3% (4.8–11.6%)*
Lower Columbia River Coho[6]	15.0% (12.2–18.1%)	0.2% (0.0–0.7%)*	0.3% (0.1–0.8%)*
Lower Columbia River Chum	NA	NA	NA

[1] Minimum estimates due to dispersal events (2016–2017) or colony formation after the peak of smolt out-migration through the estuary (2018); atypical cormorant colony attendance influenced estimated predation rates to an unknown degree.
[2] Predation rate estimates were not available during 2003–2008 and 2016–2017.
[3] Predation rate estimates were not available during 2003–2006.
[4] Predation rate estimates were not available during 2003–2006 and in 2017.
[5] Predation rate estimates were not available during 2003–2006.
[6] Predation rate estimates were not available during 2003–2006 and 2012–2016.

4.4.7.2 Upper Columbia River spring Chinook salmon – Predation rate estimates on UCR spring Chinook by East Sand Island double-crested cormorants were lower than those of UCR steelhead in most, but not all years, ranging annually from 0.6% (0.3–1.2%) to 6.1% (3.9–10.1%) during 2003–2018 (*Figure 4.37* above and *Appendix B, Table B7*). Both wild and hatchery (from select hatcheries) spring Chinook salmon are part of the ESA-listed UCR population (NMFS 2019) and there was some evidence that hatchery fish were consumed at a slightly higher rate than wild fish (Evans et al. 2019a). Differences, however, were not consistently observed across all weeks within a year or across all years (*Figure 4.38* above), suggesting factors other than rear-type may explain differences in predation rates (see *Chapter 10*). During the pre-management period, the estimated average annual predation rate on UCR spring Chinook by East Sand Island cormorants was 3.8% (3.2–4.6%; *Table 4.6* above). During the management period, the average annual predation rate was 4.1% (3.2–5.8%) during Phase I and 0.6% (0.3–1.2%) during Phase II (*Table 4.6* above).

4.4.7.3 Snake River steelhead trout – Similar to UCR steelhead, annual predation rate estimates on Snake River (SR) steelhead by East Sand Island double-crested cormorants were highly variable, ranging annually from 0.4% (0.2–0.8%) to 16.6% (12.0–25.7%) during 2003–2018 (*Figure 4.37* above and *Appendix B, Table B7*). Both wild and hatchery (from select hatcheries) steelhead are part of the ESA-listed SR steelhead population (NMFS 2014) and analogous to UCR steelhead, there was no evidence of a difference in relative susceptibility based on a fish's rear-type, with wild SR steelhead just as likely to be consumed as their hatchery counterparts (*Figure 4.38* above). An investigation of East Sand Island cormorant predation rates between transported and in-river (naturally migrating) SR steelhead smolts also found no evidence of a difference in relative susceptibility, with both groups, on average, equally susceptible to cormorant predation (Evans et al. 2019a; *Figure 4.39*, see also *Appendix B, Table B30*). During the pre-management period, the estimated average annual predation rate on SR steelhead by East Sand Island cormorants was 7.2% (6.3–8.5%; *Table 4.6* above). During the management period, average annual predation rate was 6.8% (5.3–9.4%) during Phase I and 0.5% (0.3–0.9%) during Phase II (*Table 4.6* above).

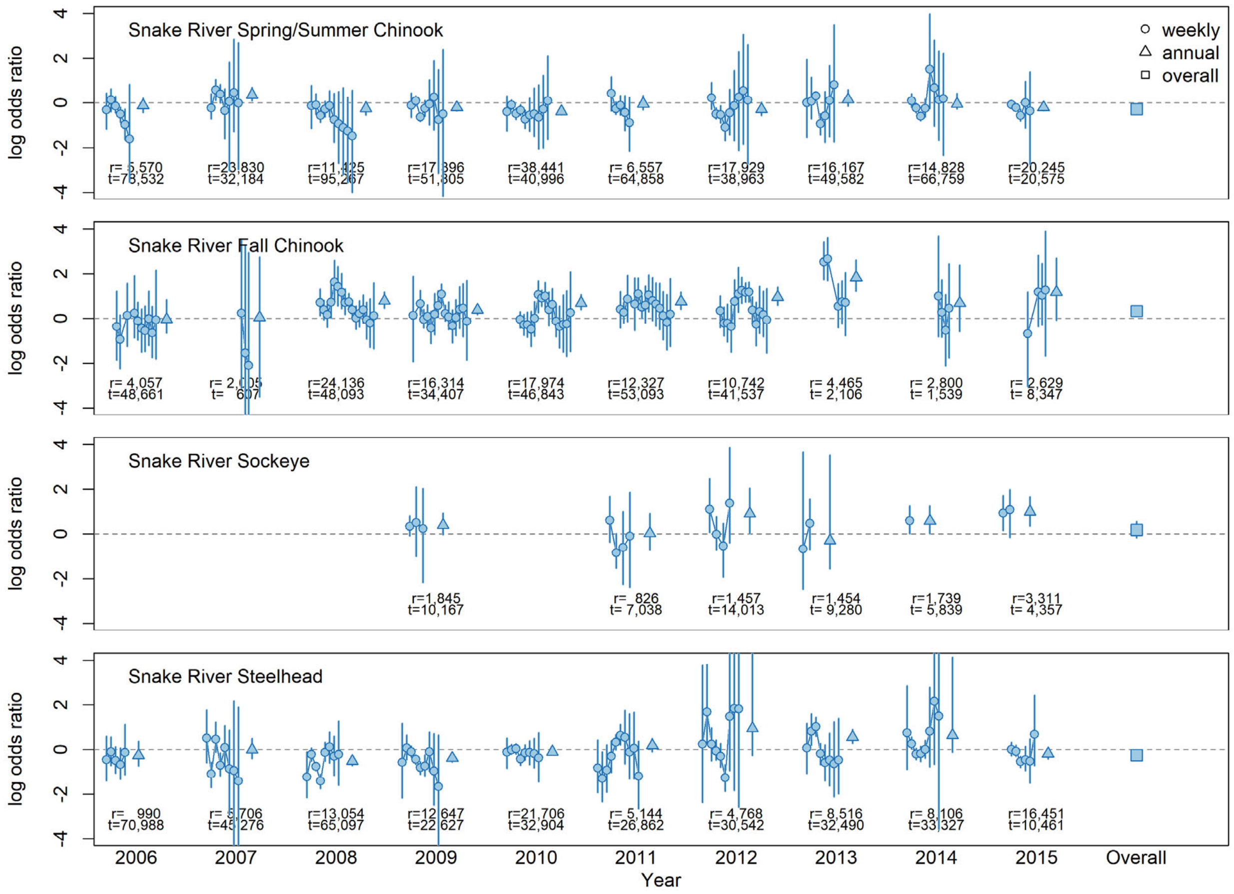

Figure 4.39. Relative susceptibility of in-river and transported juvenile salmonids to predation by double-crested cormorants nesting on East Sand Island during 2006–2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for in-river fish and values > 0 indicating greater predation odds for transported fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Only years when > 500 PIT-tagged fish of each migration history were available are included. Only weeks when > 50 PIT-tagged fish of each migration history were available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right. Figure from Evans et al. (2019a).

4.4.7.4 Snake River spring/summer Chinook salmon – Predation rate estimates on SR spring/summer Chinook by East Sand Island double-crested cormorants ranged annually from 0.5% (0.3–0.8%) to 8.5% (6.1–13.2%) during 2003–2018 (*Figure 4.37* above and *Appendix B, Table B7*). Estimated predation rates were similar to and not significantly different from those on UCR spring Chinook when comparisons were made on an annual basis (*Figure 4.37* above). Both wild and hatchery (from select stocks) SR spring/summer Chinook are part of the ESA-listed population (NMFS 2019) and there was no evidence of a difference in predation rate by rear-type, with wild fish just as likely to be consumed as hatchery fish (*Figure 4.38* above). There was also no evidence of a difference in the relative susceptibility of in-river versus transported SR spring/summer Chinook salmon to predation by East Sand Island double-crested cormorants, with in-river migrants just as likely to be consumed as their transported counterparts when data from all weeks and years were considered (*Figure 4.39* above; see also *Appendix B, Table B30*). During the pre-management period, the estimated average annual predation rate on SR spring/summer Chinook salmon by East Sand Island cormorants was 4.6% (3.2–4.6%; *Table 4.6* above). Following implementation of management actions, the average annual predation rate was 6.8% (5.3–9.4%) during Phase I and 0.5% (0.3–0.8%) during Phase II (*Table 4.6* above).

4.4.7.5 Snake River fall Chinook salmon – Predation rates by East Sand Island double-crested cormorants on SR fall Chinook were generally lower than those on UCR spring Chinook or SR spring/summer Chinook, ranging annually from 0.1% (< 0.1–0.2%) to 4.5% (3.2–7.1%) during 2003–2018 (*Figure 4.37* above and *Appendix B, Table B23*). Both wild and hatchery (from select hatcheries) SR fall Chinook are part of the ESA-listed population (NMFS 2019), but there were not enough tagged wild fall Chinook salmon available for relative comparisons of predation rates based on rear-type (Evans et al. 2019a). There was some evidence of a difference in the relative susceptibility of in-river versus transported SR fall Chinook to predation by East Sand Island cormorants, with transported fish disproportionately consumed in many, but not all, weekly and annual comparisons (*Figure 4.39* above; see also *Appendix B, Table B30*). Comparisons by out-migration history, however, were highly variable, with no consistent trend identified across all weekly and yearly comparisons (*Figure 4.39* above). During the pre-management period, the estimated average annual predation rate on SR fall Chinook salmon by East Sand Island cormorants was 2.7% (2.3–3.2%; *Table 4.6* above). During the management period, average annual predation rate was 3.7% (2.6–5.4%) during Phase I and 0.9% (0.5–1.6%) during Phase II (*Table 4.6* above).

4.4.7.6 Snake River sockeye salmon – There were inadequate numbers of PIT-tagged SR sockeye salmon for predation rate analyses in most, but not all years, with less than 500 tagged fish detected passing Bonneville Dam during 2003–2008 and again during 2016–2017 (*Appendix B, Table B23*). For those years with adequate sample sizes, predation rates ranged annually from 0.9% (0.5–1.9%) to 5.7% (3.5–9.8%) during 2009–2015 and 2018 (*Figure 4.37* above and *Appendix B, Table B23*). Estimates of predation rates by East Sand Island cormorants on SR sockeye salmon were similar to those on other salmon ESUs. Both wild and hatchery sockeye salmon are part of the SR population, but inadequate numbers of PIT-tagged wild fish prevented comparisons of relative susceptibility based on rear-type (Evans et al. 2019a). Small sample sizes of PIT-tagged sockeye salmon, particularly in-river fish, also prevented comparisons of susceptibility based on out-migration history (in-river vs. transported) in most, but not all, study years. Based on the limited data from those years with adequate sample sizes, there was no consistent trend in relative susceptibility, with in-river fish consumed at about the same rate as transported fish when data from all weeks and years were considered (*Figure 4.39* above; see also *Appendix B, Table B30*). Small sample sizes, however, resulted in imprecise comparisons (based on the width of 95% credible intervals; *Figure 4.39* above), so results should be interpreted cautiously (see also Evans et al. 2019a). During the pre-management period, the estimated average annual predation rate on SR sockeye salmon by East Sand Island cormorants was 4.2% (3.3–5.3%; *Table 4.6* above). During the management period, average annual predation rate was 2.4% (1.4–4.0%) during Phase I and 0.9% (0.4–1.9%) during Phase II (*Table 4.6* above).

4.4.7.7 Middle Columbia River steelhead trout – For those years with adequate samples sizes (2009–2018), predation rates by East Sand Island cormorants on MCR steelhead were highly variable, ranging annually from 0.4% (0.1–1.0%) to 14.9% (10.3–23.8%) during 2009–2018 (*Figure 4.37* above and *Appendix B, Table B23*). Estimates of cormorant predation rates on the MCR steelhead DPS were similar to, and not significantly different from, those on the UCR and SR steelhead DPSs (*Figure 4.37* above). Sample sizes of tagged MCR steelhead were generally smaller than those of UCR and SR steelhead, however, resulting in less precise annual estimates of predation rates (*Appendix B, Table B23*). Both wild and hatchery (from select stocks) steelhead are part of the ESA-listed MCR steelhead DPS (NMFS 2014), but inadequate numbers of PIT-tagged wild fish prevented comparisons between rear-types (Evans et al. 2019a). During the pre-management period, the estimated average annual predation rate on MCR steelhead by East Sand Island cormorants was 7.5% (6.3–9.3%; *Table 4.6* above). During the management period, average annual predation rate was 5.4% (4.0–7.0%) during Phase I and 0.4% (0.1–1.0%) during Phase II (*Table 4.6* above).

4.4.7.8 Upper Willamette River Chinook salmon – Predation rates on Upper Willamette River (UWR) Chinook by East Sand Island cormorants ranged annually from 0.4% (< 0.1–2.1%) to 4.2% (1.6–9.2%) during 2007–2014 (*Figure 4.37* above and *Appendix B, Table B23*). Annual predation rates and average annual predation rates by East Sand Island cormorants on UWR Chinook salmon were amongst the lowest of any ESU/DPS evaluated (*Table 4.6* above). Unlike UCR and SR spring/summer Chinook, the timing of out-migration for UWR Chinook varies considerably, with smolts out-migrating nearly year-round (NMFS 2019). Differences in run- timing and the relative availability of UWR Chinook in the estuary may explain the relatively low estimates of cormorant predation rates on this ESU (Evans et al. 2016; see also *Chapter 10*). Both wild and hatchery (from select hatcheries) UWR Chinook are part of the ESA-listed population (NMFS 2014), but small sample sizes of PIT-tagged wild UWR Chinook prevented comparison of predation rates based on rear-type. During the pre-management period, the estimated average annual predation rate on UWR Chinook salmon by East Sand Island cormorants was 1.8% (1.3–2.6%; *Table 4.6* above). During the management period, the average annual predation rate was 1.4% (0.6–2.9%) during Phase I; inadequate sample sizes of PIT-tagged UWR Chinook salmon prevented the estimation of predation rates during Phase II (*Table 4.6* above).

4.4.7.9 Lower Columbia River steelhead trout – Annual predation rate estimates on Lower Columbia River (LCR) steelhead by East Sand Island cormorants ranged from 0.6% (0.3–1.0%) to 9.0% (5.9–13.4%) during 2007–2014, years with adequate samples sizes of tagged steelhead and when standardized estimates of predation rates were available (*Figure 4.37* above and *Appendix B, Table B8*). Both wild and hatchery (from select hatcheries) LCR steelhead are part of the ESA-listed population (NMFS 2019) and, similar to results for other steelhead DPSs, wild and hatchery LCR steelhead were equally susceptible to predation by East Sand Island cormorants. For example, in most weekly and yearly comparisons, predation rates on wild LCR steelhead were not significantly different from those on hatchery LCR steelhead. Wild steelhead, however, generally out-migrated in April before most hatchery fish arrived in the estuary in May, resulting in only partial over-lap between rear-types and, consequently, relative comparisons of predation rates (see also *Chapter 10*). During the pre-management period, the estimated average annual predation rate on LCR steelhead by East Sand Island cormorants was 5.4% (4.5–6.3%; *Table 4.6* above). During the management period, average annual predation rate was 5.0% (3.7–6.9%) during Phase I and 0.6% (0.3–1.0%) during Phase II (*Table 4.6* above).

4.4.7.10 Lower Columbia River Chinook salmon – Predation rate estimates on LCR Chinook by East Sand Island cormorants were the highest and most variable of those ESUs/DPSs evaluated, ranging annually from 1.0% (0.5–1.7%) to 51.0% (38.3–59.0%) during 2007–2018, the years with adequate samples sizes of PIT-tagged smolts and years when standardized predation rate estimates were available (Figure 4.37 above and Appendix B, Table B24). Potential factors responsible for the high and variable predation rates on LCR Chinook are addressed in section *4.4.7.14 Per capita predation rates on juvenile salmonids* and discussed in section *4.5.4 Management Action Effectiveness*; see also Chapter 10. Both wild and hatchery (from select hatcheries) LCR Chinook are part of the ESA-listed population (NMFS 2014), but few wild LCR Chinook salmon were PIT-tagged annually, preventing robust comparisons of predation rates by rear-type. Lyons et al. (2014) found some evidence that hatchery LCR Chinook were consumed at higher rates by East Sand Island cormorants compared to their wild counterparts, but adequate samples sizes of wild fish were limited to a single year, with less than 500 wild fish tagged in that year. Conversely, several thousand (range = 11,734 to 33,031 fish annually) hatchery LCR Chinook were PIT-tagged annually. Numerically the vast majority of LCR Chinook smolts in the estuary were from hatcheries (Sebring et al. 2013, Lyons et al. 2014), so predation rate estimates summarized herein are based on the more abundant rear-type.

LCR Chinook salmon exhibit both sub-yearling and yearling life histories, with yearlings out-migrating in the spring and sub-yearlings in the late-spring and summer (NMFS 2019). Variations in these life histories correspond to different population groups or strata, groups that originate from distinct areas within the geographic range of the LCR Chinook salmon ESU (referred to as “Coastal,” “Cascade,” and “Gorge”; NMFS 2014). A large number and percentage (> 90%) of PIT-tagged hatchery LCR Chinook included in our analysis of predation rates were sub-yearling smolts that were released *en masse* in the Coastal and Cascade regions downstream of Bonneville Dam (see also Sebring et al. 2013). An investigation of predation rates based on LCR Chinook smolt release locations and age-classes (yearling vs. sub-yearling) indicated that sub-yearling smolts tagged and released downstream of Bonneville Dam were depredated at a significantly higher rate than sub-yearling and yearling LCR Chinook tagged and released upstream of Bonneville Dam, with statistically significant differences observed in most weeks and years during 2007–2014 (Sebring et al. 2013; this study). During the pre-management period, the estimated average annual predation rate on LCR Chinook salmon by East Sand Island cormorants was 27.5% (24.3–30.7%; *Table 4.6* above), the highest average annual predation rate documented for piscivorous waterbirds from any particular breeding colony in the Columbia River basin (*Table 4.6* above and *Appendix B, Tables B18-B30*). During the management period, the estimated average annual predation rate on LCR Chinook by East Sand Island cormorants was significantly less, but still higher than those observed for all other salmonid ESUs/DPSs, at 8.7% (6.2–12.1%) during Phase I and 7.3% (4.8–11.6%) during Phase II (*Table 4.6* above).

4.4.7.11 Lower Columbia River coho salmon – Adequate samples sizes of LCR coho were available for analyses of predation rates during 2007–2011 and 2017–2018 (*Appendix B, Table B8*). During those years, estimates of predation rate by East Sand Island cormorants ranged annually from 0.2% (< 0.1–0.7%) to 18.1% (13.0–26.1%) of available fish (*Figure 4.37* above and *Appendix B, Table B24*). Average annual predation rate was 15.0% (12.2–18.1%) during 2007–2011, among the highest observed for any ESU/DPS evaluated during the study period (*Table 4.6* above). Both wild and hatchery (from select hatcheries) LCR coho are part of the ESA-listed population (NMFS 2019), but analogous to LCR Chinook, only small numbers of wild LCR coho were PIT-tagged, preventing robust comparisons of predation rates by rear-type. Lyons et al. (2014) found some evidence that wild coho released into tributaries downstream of Bonneville Dam were less likely to be consumed by East Sand Island cormorants compared to their hatchery counterparts, but inadequate numbers of PIT-tagged wild fish were available in most years and there was little temporal over-lap in migration timing between rear-types (Lyons et al. 2014). Analogous to LCR Chinook, LCR coho from hatcheries are the most abundant rear-type present in the estuary, with very few wild smolts available as prey relative to hatchery fish (Lyons et al. 2014, NMFS 2014). During the pre-management period, the estimated average annual predation rate on LCR coho salmon by East Sand Island cormorants was 15.0% (12.2–18.1%; *Table 4.6* above). During the management period, the average annual predation rate was significantly less at 0.2% (< 0.1–0.7%) during Phase I and 0.3% (0.1–0.8%) during Phase II (*Table 4.6* above). Predation rate results during the management period, however, should be interpreted cautiously due to a lack of estimates during most management years and due to the sparse attendance by cormorants of the East Sand Island colony during the peak of the coho salmon out-migration period in May (see section *4.5.4 Management Action Effectiveness*).

4.4.7.12 Predation rates on other ESA-listed salmonid ESUs/DPSs – Two other ESA-listed ESUs/DPSs of Columbia Basin salmonids, Upper Willamette River steelhead and Columbia River chum salmon, are present in the estuary, but estimates of predation rates by East Sand Island cormorants on these two populations were not available due to a lack of tagging studies (PSMFC 2020). Only wild UWR steelhead are part of the ESA-listed population and smolts out-migrate as yearlings during the spring (NMFS 2014). Given the similarities in cormorant predation rates on SR, UCR, and MCR steelhead and similarities in out-migration timing among these populations and UWR steelhead, predation rates on UWR steelhead by East Sand Island cormorants may be similar to those of other steelhead DPSs. Chum salmon fry out-migrate as sub-yearlings shortly after emergence during late winter and early spring (NMFS 2014) and although they may reside and rear in the estuary before entering the open ocean (Groot and Margolis 1991), most fish are too small to PIT-tag (< 80 mm fork length). Based on genetic analyses of salmonid prey in cormorant stomach contents (see *section 4.3.4 Foraging Behavior* above), very few chum salmon have been confirmed in the diet of East Sand Island cormorants, indicating that cormorant predation rates on ESA-listed chum salmon in the estuary have likely been low.

4.4.7.13 Predation rates on juvenile salmonids by cormorants from other colonies – Standardized predation rate estimates were also available for double-crested cormorants nesting at the colony on Rice Island in the upper estuary during 1998, 2001, and 2006 (*Appendix A, Table B7*). Estimates were based on data originally collected by Collis et al. (2001), Ryan et al. (2001), and Collis et al. (2007), data that were retrospectively analyzed as part of this Synthesis Report (see *Chapter 5*). During the study period (1998–2014), the Rice Island cormorant colony was substantially smaller (range = 0 to 1,141 breeding pairs; *Table 4.3* above) compared with the East Sand Island cormorant colony (range = 5,023 to 14,916 pairs; *Table 4.3* above). Estimates of predation rates indicated that despite the relatively small size of the Rice Island cormorant colony, predation rates on juvenile salmonids were not insignificant. For example, in 1998, when 795 breeding pairs of double-crested cormorants nested on Rice Island, predation rates were as high as 2.4% (1.0–5.8%) and 5.3% (3.5–10.8%) on UCR spring Chinook salmon and SR steelhead, respectively (*Appendix B, Table B23*). In 2001 and 2006, when 150 breeding pairs and 35 breeding pairs, respectively, of double-crested cormorants nested on Rice Island, predation rates were less than 2% on all salmonid ESUs/DPSs evaluated, with the highest predation rate on SR steelhead at 1.0% (0.1–3.0%; *Appendix B, Table B23*). Estimates of predation rates were also available for a small cormorant colony (range = 41 to 90 pairs; *Table 4.3* above) on Miller Sands Spit in the upper estuary during 2006–2007. Estimates from the Miller Sand Spit cormorant colony were similar to those from the Rice Island cormorant colony in 2001 and 2006, with predation rates of less than 2% per ESU/DPS, and with the highest predation rates on steelhead DPSs (*Appendix B, Table B23*). Additional details regarding the impacts to smolt survival from predation by double-crested cormorants nesting on Rice Island and Miller Sands Spit in the upper estuary are provided in *Chapter 5*.

There were no standardized predation rate estimates available for double-crested cormorants nesting over water on bridges or navigational aids in the upper estuary (e.g., channel markers near Miller Sands Spit, the Astoria-Megler Bridge, and the Lewis & Clark Bridge in Longview, Washington). PIT tag detection and deposition probabilities for colonies of ground-nesting cormorants (i.e. East Sand Island, Rice Island, and Miller Sands Spit) likely differ considerably from those of cormorant colonies over water, where a large proportion of egested tags are presumably deposited in the water and not detected by researchers after the nesting season. As such, predation rate estimates for cormorant colonies situated over water are currently lacking but may be critical for evaluating predation rates on juvenile salmonids by all double-crested cormorants nesting in the Columbia River estuary.

4.4.7.14 Per capita predation rates on juvenile salmonids – Data on per capita (per breeding pair) predation rates have been used to help prioritize piscivorous waterbird colonies for potential management to reduce smolt mortality in the Columbia River basin (Roby et al. 2002, USFWS 2005, Evans et al. 2012, USACE 2015, Evans et al. 2019a). Estimates of per capita predation rates on ESA-listed salmonid ESUs/DPSs by double-crested cormorants nesting on East Sand Island are reported in *Table 4.7*. Results indicate that per capita cormorant predation rates on juvenile salmonids were similar among salmonid populations that originate entirely upstream of Bonneville Dam, those from UCR, SR, and MCR ESUs/DPSs. Per capita predation rates on LCR Chinook salmon, however, were significantly higher than those on all other

ESUs/DPSs evaluated (*Table 4.7*). This could be due to either the greater relative abundance of LCR Chinook salmon smolts compared to other salmonid ESUs/DPSs in the estuary (Lyons et al. 2014) and/or behavioral differences among salmonid ESUs/DPSs (i.e. run-timing and residence time in the estuary), differences that increased the susceptibility of LCR Chinook salmon smolts to cormorant predation relative to other ESUs/DPSs (Sebring et al. 2013; see also *Chapter 10*).

Table 4.7. Average annual per capita (per breeding pair) predation rates on salmonid populations (95% credible intervals) by double-crested cormorants nesting on East Sand Island during 2003–2014. ESA-listed salmonid populations (ESU/DPS) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated. NA denotes that estimates were not available. See Appendix B, Tables B23-B24 for annual estimates of predation.

Salmonid ESU/DPS	Per Capita Predation Rates [1]
Upper Columbia River Steelhead	0.0005% (0.0005–0.0006)
Upper Columbia River Sp Chinook salmon	0.0003% (0.0003–0.0004)
Snake River Steelhead	0.0006% (0.0005–0.0007)
Snake River Sp/Su Chinook salmon	0.0004% (0.0004–0.0005)
Snake River Fall Chinook salmon	0.0003% (0.0002–0.0003)
Snake River Sockeye salmon[2]	0.0003% (0.0002–0.0004)
Middle Columbia River Steelhead [3]	0.0007% (0.0006–0.0008)
Upper Willamette River Steelhead	NA
Upper Willamette River Sp Chinook salmon[3]	0.0001% (0.0001–0.0002)
Lower Columbia River Steelhead [3]	0.0007% (0.0006–0.0008)
Lower Columbia River Chinook salmon[3]	0.0029% (0.0026–0.0033)
Lower Columbia River Coho salmon[4]	0.0010% (0.0008–0.0012)
Lower Columbia River Chum salmon	NA

[1] *Estimates after 2015 were not considered due to atypical nesting behaviors that influenced per capita predation rates to an unknown degree.*

[2] *Predation rate estimates were not available during 2003–2008.*

[3] *Predation rate estimates were not available during 2003–2006.*

[4] *Predation rate estimates were not available during 2003–2006 or during 2012–2016.*

Based on estimates of average annual per capita (per breeding pair) predation rates, a double-crested cormorant colony of 5,939 breeding pairs on East Sand Island (the upper end of the range of colony sizes specified in the *Cormorant Management Plan*) would be expected to consume, on average, 3.0% (2.4–4.2%), 3.6% (3.0–4.3%), and 4.2% (3.6–4.8%) of UCR, SR, and MCR steelhead smolts, respectively. Estimates of per capita predation rates suggest that a cormorant colony of 5,939 pairs on East Sand Island would be expected to consume, on average, 5.9% (4.8–7.1%) and 17.2% (15.4–19.6%) of LCR coho salmon and LCR Chinook salmon, respectively, predation rates that are similar to or greater than predation rates by piscivorous waterbirds from any other colony on any other ESUs/DPSs of Columbia Basin salmonid. If average reductions in cormorant predation rates of this magnitude were achieved in the Columbia River estuary on a consistent basis, the resultant predation rates would represent an approximate 50% reduction in cormorant predation rates compared to levels during the base or pre-management period (NMFS 2008). As described above and in greater detail in *Chapter 10*, despite the stable size of the double-crested cormorant colony on East Sand Island during 2006–2014 (*Table 4.3* above), estimates of ESU/DPS-specific predation rates were highly variable, indicating that factors other than cormorant colony size were having a strong influence on the susceptibility of juvenile salmonids to cormorant predation in the Columbia River estuary (Lyons et al. 2014, Evans et al. 2016). As such, estimates of average annual per capita predation rates based on measures of colony size alone should be interpreted cautiously and used only as a gross predictor of potential future predation rates (see also *Appendix C*).

Comparisons of per capita predation rates between double-crested cormorants nesting on East Sand Island in the lower estuary and cormorants nesting on Rice Island or Miller Sands Spit in the upper estuary are provided in *Chapter 5*. In brief, our results indicate that per capita predation rates were significantly higher for double-crested cormorants nesting on islands in the upper estuary compared to those nesting on East Sand Island in the lower estuary during years when cormorants nested on islands in both the upper and lower sectors of the estuary. Results from this and other studies (Collis et al. 2002), provide strong evidence that double-crested cormorants nesting and foraging closer to the mouth of the Columbia River and within the marine zone of the estuary rely less on juvenile salmonids as a food source compared with cormorants nesting and foraging further upstream in the estuarine mixing zone or the freshwater zone of the estuary (*Figure 4.3* above).

4.4.8 Pilot Studies of Cormorant Nest Dissuasion Techniques

Pilot studies were conducted on East Sand Island during 2007-2013 to test the feasibility of several nonlethal methods to reduce the size of the large double-crested cormorant colony by dissuading cormorants from nesting in a portion of the colony area.

4.4.8.1 Artificial bald eagle perches – Bald eagles were attracted to the artificial perches deployed during the 2007 nesting season when the number of eagles using the area in and around East Sand Island was relatively high, especially during the early part of the cormorant breeding season (April). Cormorants temporarily abandoned the tidal pond satellite colony while eagles used the artificial perches nearby; however, bald eagles used the perches too infrequently to discourage cormorants from eventually nesting in the tidal pond satellite colony (Roby et al. 2008). Cormorants successfully nested and fledged chicks from this satellite colony, although nesting chronology was delayed compared to most other areas of the cormorant colony. Because of the ample number of natural perches available to eagles (e.g., other driftwood, pile dikes, etc.) on or near the East Sand Island cormorant colony, it was difficult to evaluate the efficacy of our artificial eagle perches at discouraging cormorant nesting. If other “natural” perches were not available, it is possible that eagles might have used our artificial perches more frequently, resulting in abandonment by cormorants attempting to nest nearby. Because of the excess of suitable cormorant nesting habitat available on the western half of East Sand Island, however, cormorants displaced from the vicinity of eagle perches with frequent use were able to simply relocate to other areas of the colony where eagle disturbances were less frequent.

4.4.8.2 Passive cormorant nest dissuasion – Double-crested cormorants nested in the area immediately north of the fence rows of landscape fabric in late April and gradually expanded into spaces between the fences early in the 2007 nesting season. By the end of April, a substantial number of double-crested cormorants had started nesting in the spaces between all the rows of landscape fabric fencing (Roby et al. 2008).

Cormorants prospecting for nest sites early in the 2007 nesting season initially avoided the area where a network of stakes, ropes, and flagging was erected in the interior portion of the colony. Once cormorants started occupying nest sites immediately adjacent to the network of dissuasion materials, prospecting cormorants began occupying spaces within the network. By the end of April, the density of nesting cormorants in the area covered with the network of stakes, ropes, and flagging was approximately the same as in other parts of the colony (Roby et al. 2008).

Landscape fabric fencing erected in the cormorant colony was susceptible to being blown down by high winds, while the network of stakes, ropes, and flagging lasted throughout the nesting season. The landscape fabric fencing erected on the rocky revetment did not discourage cormorants from nesting, although colonization of the area between and immediately outside the fencing by nesting cormorants appeared to be somewhat delayed compared to other areas of the colony. Cormorants also colonized the area where the network of stakes, ropes, and flagging had been erected in the interior of the colony and did so much faster than the areas between landscape fabric fences on the rocky revetment, presumably because the stakes, ropes, and flagging did not provide a visual barrier between cormorants nesting on the colony, which seems to discourage nesting in other colonial waterbirds (Roby et al. 2002). These results suggest that visual barriers (i.e. landscape fabric fencing) are more effective in discouraging cormorant nesting than physical barriers (i.e. networks of stakes, ropes, and flagging), although neither technique ultimately prevented cormorants from nesting (Roby et al. 2008). Both of these passive dissuasion techniques have proven highly effective in dissuading nesting by Caspian terns (see *Chapter 1*).

4.4.8.3 Covering nesting substrate with pond liner - The areas of rocky revetment on the cormorant colony that were covered with pond liner during the 2009 and 2010 nesting seasons remained free of nesting cormorants for the entirety of each nesting season (Roby et al. 2010, 2011a). The pond liner was compromised once, however, when a windstorm in late May 2010 lifted large sections of the pond liner, leaving several areas of rock revetment exposed. This did not compromise the efficacy of the pond liner in dissuading cormorants from nesting in 2010, however. Using data on double-crested cormorant nesting density in 2010, approximately 348 nests were excluded from the 2010 pond liner dissuasion area, as compared to 2009 when about 80 cormorant nests were excluded using pond liner. Although the pond liner material proved effective at deterring cormorants from nesting in areas of the breeding colony when covering nesting substrate formerly used by cormorants, the pond liner was expensive, heavy, and difficult to deploy and maintain. As such, scaling up the use of pond liner to prevent cormorants from nesting on multiple acres of the colony area was considered impractical.

4.4.8.4 Active nest dissuasion using a green laser – Technicians testing the green laser as an active cormorant nest dissuasion technique during the 2008 nesting season had low encounter rates with roosting double-crested cormorants when traveling to and from the colony. Most tests with the green laser in 2008 were conducted on cormorants that were roosting on the north beach at the west end of East Sand Island, where technicians applied the laser beam from just inside or near the mouth of the low fabric tunnel used to access the main observation tower. Seventeen tests with the LEM50 were completed in 2008, of which five were successful in flushing cormorants. All successful tests were conducted before sunrise or after sunset and at a distance of 55 m or less from roosting cormorants. Three unsuccessful tests were conducted under conditions that resulted in successful flushing of cormorants during additional attempts at closer range (distances of 40 to 80 m). The unsuccessful tests (n = 9) all elicited no behavioral reaction and were conducted during full daylight (09:20 to 18:46 PDT) under varying conditions of cloud cover and distance.

During the pre-nesting period in 2009, the green laser was effective at flushing cormorants from the targeted areas when used in low light conditions, primarily at dusk, but efficacy dropped considerably as light levels increased after sunrise and eventually failed completely to flush cormorants. While using the laser in the evenings, it had to be directed at the targeted area between 3–6 times for it to be successful in flushing cormorants. All 14 late evening (post-sunset) trials using the green laser were successful at keeping birds off the targeted area for greater than one hour. On at least two occasions the targeted area remained clear of birds until the next morning. Although partially successful in flushing cormorants, none of the 13 early morning (pre-sunrise) trials using the green laser were successful at keeping birds off the focal area for more than one hour. Although the laser was effective at disturbing cormorants in low light conditions, there was an average of 14 hours and 20 minutes of daylight each day when the birds were not disturbed by the laser. The experiment was terminated on 28 April, when an egg was observed in the targeted area, seven days after the first cormorant egg was observed on a non-targeted area of the colony.

4.4.8.5 Localized human hazing on-colony – In 2008, tests of human hazing to deter cormorant nesting in designated target areas of the colony began after many of the cormorants within the study area were defending nest territories and had well-established pair bonds. A total of six human hazing events, conducted over three days, were completed before cormorant eggs were observed in nests within the targeted area and hazing events ceased. We found that during the period right before egg-laying, short duration human hazing (< 5 min) successfully flushed cormorants from the target area, but that cormorants re-landed in the targeted area within 2 min after the disturbance was terminated. Human hazing was most effective at keeping cormorants off the targeted area of the colony when maintained for extended periods and repeated frequently; hazing events lasting longer than 10 min kept cormorants out of the target nesting area for > 10 min, and the length of time cormorants remained off the targeted area of the colony increased again when hazing was repeated immediately after cormorants re-landed. Once nest territories and pair bonds were established by nesting cormorants in a selected area, the duration and/or frequency of hazing would need to be much higher than was employed in this pilot study in order to preclude egg-laying (i.e. > 15 minutes/day and > 2 events/day). Initiating human hazing events earlier in the pre-laying period would be much more effective in discouraging cormorant nesting than the approach used in the pilot study. The use of a landscape fabric fence as a visual barrier was successful at limiting the portion of the colony affected by human hazing of the targeted area. Cormorants on the west side of the visual barrier were successfully screened from view of the technicians when they were outside the tunnel. Cormorants on the west side of the tunnel did, however, react at times to the alarm behaviors of cormorants on the east side of the visual barrier, which had an unobstructed view of the technicians. Within the targeted area, cormorants in view of technicians consistently flushed.

In 2009, human hazing proved to be an effective method of delaying, but not preventing, cormorants from nesting in the targeted area and caused little apparent disturbance to cormorants nesting in areas nearby (i.e. non-targeted areas; Roby et al. 2010). The length of time invested in dissuading cormorants from the targeted area seemed to be the limiting factor that eventually resulted in cormorants laying eggs in the targeted area on May 15. Although the efforts to disturb nesting cormorants in the targeted area were conducted daily, there was evidence of habituation by some targeted cormorants to the hazing. Also, access to the blind in the middle of the cormorant colony was limited by the tides (i.e. researchers could not enter or leave the blind during high tide without disturbing large numbers of nesting cormorants and roosting California brown pelicans). This constraint prevented technicians from disturbing nesting cormorants in the targeted area frequently enough to prevent the onset of egg-laying in mid-May. The disturbance to cormorants nesting in non-targeted areas from hazing of cormorants in the targeted area was minimal. Most disturbances to cormorants in non-targeted areas were recorded in the initial weeks of the trial, when cormorants were just beginning to prospect in the areas surrounding the blind. As nesting progressed and cormorants became more committed to their nest sites, the number of cormorants that flushed from non-targeted areas declined as a result of hazing the targeted area. Cormorants nesting within 10 m of the blind on the west side (non-targeted area) were observed to have eggs in early May, slightly earlier than the date when eggs were first laid in the targeted area (15 May).

4.4.8.6 Dissuasion fencing & human hazing – The human disturbance (hazing) experiments, in concert with a large visual barrier (privacy fence or “dissuasion” fence) and destruction of nest structures, was an effective method of preventing cormorants from nesting in the targeted dissuasion areas in 2011, 2012, and 2013 (*Figure 4.4* above; Roby et al. 2012, 2013, 2014). These methods also caused little apparent disturbance to cormorants nesting on the other side of the dissuasion fence from where human hazing was taking place. For example, double-crested cormorants established nests on the other side of the dissuasion fence and successfully raised young within 10 m of the dissuasion fence in all three years of the feasibility study. In 2011, double-crested cormorants were dissuaded from nesting in 15% of the area used by nesting double-crested cormorants in 2010. The dissuasion area was increased in 2012 to 62% of the area used by nesting cormorants in 2010. In 2011, approximately 8,400 double-crested cormorant nests were located in the 2012 dissuasion area, and the dissuasion area encompassed an area of approximately 6.5 acres (2.6 ha; Roby et al. 2013). In addition to the privacy fence, a camp, two observation blinds, and a tunnel system were constructed to provide researchers access to the area without disturbing nesting cormorants outside of the targeted dissuasion area. The camp concealed all routine non-hazing activity by technicians from cormorants within the dissuasion area, as well as those cormorants nesting on the non-dissuasion side of the dissuasion fence, and the blinds provided an elevated vantage point for observation of cormorants on either side of the dissuasion fence without disturbing nesting cormorants.

4.4.9 Pilot Studies of Cormorant Social Attraction & Colony Relocation

Double-crested cormorants were attracted to nest and successfully fledged young at all 11 test plots on East Sand Island where social attraction (decoys and audio playback of vocalizations) was deployed during 2004–2007, regardless of the method of nesting habitat enhancement or the year (Suzuki et al. 2015). The first observations of cormorants and the initiation of nesting behavior by cormorants on the plots were concurrent with the timing of cormorants that first settled on other parts of the colony (Collis et al. 2006, 2007) and were synchronous between plots within a year. The nesting density of cormorants on each test plot was similar to or greater than colony-wide nesting densities (*Figure 4.5* above). After habitat enhancement and social attraction materials were removed from the two plots following the 2004 nesting season, cormorants did not nest in those two areas during the subsequent three nesting seasons. The average nesting success of cormorants in test plots at East Sand Island was 2.3 (SE = 0.1, n = 43) and 1.9 (SE = 0.1, n = 64) young raised per breeding pair in 2004 and 2005, respectively. There was no significant difference in the average nesting success rate between test and control plots in 2004, while the average nesting success rate was greater in the two test plots compared to the control plots in 2005 ($t = 3.7$, $p = 0.0003$). These results indicate that nesting success of cormorants using the social attraction test plots was not lower than for cormorants nesting in the East Sand Island colony at large.

Prior to the 2006 nesting season a social attraction test plot was set up on Rice Island in the upper estuary, and prospecting cormorants and cormorants engaged in courtship display were first observed on the plot just one day after plot preparations were completed. A total of 30 pairs of double-crested cormorants nested within the test plot and five additional pairs nested immediately adjacent to the plot (*Figure 4.6* above). The best estimate of average nest success or cormorants that nested in or adjacent to the Rice Island plot in 2006 was 2.6 young raised per breeding pair, significantly higher than average nesting success at the East Sand Island colony in the same year (1.96 young raised per breeding pair). During the subsequent nesting season (2007), after the habitat enhancement and social attraction materials had been removed from the plot, double-crested cormorants did not attempt to re-nest on Rice Island.

The social attraction test plot on Miller Sands Spit in the upper estuary was first set up prior to the 2004 nesting season and redeployed in each subsequent year through 2008. Cormorants were observed congregated on the beach immediately adjacent to the social attraction test plot on Miller Sands Spit on several occasions during the 2004 nesting season, and also in the upland area near the plot on one occasion; however, there was no evidence that cormorants attempted to nest on the test plot in 2004. The first confirmation of cormorant nesting attempts in the test plot was recorded during the second year (2005), when cormorants were observed carrying nest material to the plot 24 days after completion of plot preparations (Collis et al. 2006). Subsequently, a total of 21 complete or partially built cormorant nests and six cormorant eggs in four different nests were confirmed both within and immediately adjacent to the test plot. All of these cormorant nests failed prior to eggs hatching, however, presumably due to egg predation by glaucous-winged/western gulls (*Larus glaucescens* x *L. occidentalis*), which nested in the vicinity of the cormorant social attraction plot (Suzuki et al. 2015). Double-crested cormorants successfully nested and fledged chicks at the social attraction test plot on Miller Sands Spit during the 2006 and 2007 nesting seasons (*Figure 4.7* above). Prospecting cormorants were first observed on the test plot 26 days and 28 days after completion of plot preparations in 2006 and 2007, respectively. A total of 41 breeding pairs nested in the plot or immediately adjacent to the plot in 2006, and an average of 2.2 young were raised per breeding pair. In 2007, we counted 90 active cormorant nests in and around the test plot, and an average of 1.7 young were raised per breeding pair. Finally, in 2008, the fifth and final year of deploying cormorant social attraction on the downstream end of Miller Sands Spit, approximately 129 breeding pairs were attracted to nest on the test plot; however, the colony was abandoned, and no young were successfully fledged due to disturbance from an unknown cause.

No double-crested cormorants were observed on or in the immediate vicinity of the social attraction plot on the rocky islet at the mouth of Trestle Bay during the 2005 nesting season. Cormorants were seen in the Trestle Bay area during all surveys of the site in 2005 (n = 16) except one; however, most cormorants observed in the area were in sub-adult plumage. Social attraction was not deployed at the Trestle Bay site after the 2005 nesting season.

Cormorants did not attempt to nest on the floating platform equipped with social attraction at Fern Ridge Wildlife Area near Eugene, Oregon during the 2007, 2008, or 2009 nesting seasons. Cormorants were not observed perching on the Fern Ridge floating platform during the nesting season in either 2007, 2008, or 2009. Although small numbers of double-crested cormorants were observed in Fisher Butte cell #2 during April, larger numbers of cormorants (approximately 100 individuals) were only observed in Kirk Pond at the north end of Fern Ridge Reservoir, and mainly in March and April, prior to the cormorant nesting season. Bald eagles were observed in the vicinity of the floating platform in Fisher Butte and may have served as a deterrent for prospecting adult cormorants. Although public access to the area was closed during the nesting season, occasionally people were observed walking through the area, which also might have disturbed prospecting cormorants near the floating platform.

Double-crested cormorants did not attempt to nest on the floating platform equipped with social attraction that was deployed on Dutchy Lake in Summer Lake Wildlife Area, Oregon, in either the 2010 or 2011 nesting seasons. Although double-crested cormorants were often observed roosting on the nearby floating tern island in Dutchy Lake, cormorants were not observed roosting on the cormorant nesting platform during either the 2010 or 2011 nesting seasons. Bald eagles and peregrine falcons (*Falco peregrinus*) were observed at Summer Lake Wildlife Area during both the 2010 and 2011 nesting seasons, but these predators were not observed near the cormorant nesting platform and likely were not a factor in the failure of cormorants to use the platform. Caspian terns did not nest on the floating tern island at Dutchy Lake in either 2010 or 2011, however, after successfully nesting there in 2009. Adverse weather and/or poor forage fish availability during the 2010 and 2011 nesting seasons may have contributed to the lack of nesting activity by both Caspian terns and double-crested cormorants at Dutchy Lake in those two years. Further, double-crested cormorants failed to nest anywhere else in Summer Lake Wildlife Area during either 2010 or 2011.

During the 2011 breeding season, double-crested cormorants did not nest on the Tule Lake Sump 1B tern island in Tule Lake NWR, where we had installed social attraction materials (Roby et al. 2012), even though cormorants were regularly seen loafing on a small natural island near the south shore of Tule Lake Sump 1B. Cormorants displayed interest in the social attraction materials, however, and were seen loafing in and flying over the social attraction site during the 2011 nesting season. Interestingly, gulls also did not nest on the Tule Lake Sump 1B tern island in 2011. Predation by great horned owls (*Bubo virginianus*) caused nearly a complete failure of the Caspian tern colony on the island in 2011 and may have deterred cormorants from nesting on the island. Human disturbance associated with technicians monitoring the Caspian tern colony on the island may have contributed to the reluctance of double-crested cormorants to nest at the social attraction site on Tule Lake Sump 1B tern island. During the 2012 breeding season, double-crested cormorants were regularly seen loafing at the south end of the tern island in Tule Lake Sump 1B, with up to 98 individuals counted on one occasion. Despite this, cormorants did not initiate nesting on or near the social attraction plot in 2012. It should be noted, however, that gulls again did not nest on the tern island in Tule Lake Sump 1B in 2012 and predation by raccoons (*Procyon lotor*) caused complete failure of the Caspian tern colony on the island in 2012. These factors, in addition to human activity on the island during monitoring of the Caspian tern colony, could have deterred double-crested cormorants from nesting at the site during the 2012 breeding season.

Double-crested cormorants were frequently observed loafing at the new Malheur Lake tern island during the 2012 breeding season, the first and only nesting season when cormorant social attraction was deployed at the island; up to 185 cormorants were counted on the island at one time. Cormorants were also regularly seen flying to and from the cormorant social attraction plot by technicians approaching the island by boat. One cormorant was observed carrying nesting material to the social attraction plot in late June, but no cormorant nesting on the Malheur Lake tern island was confirmed, either in the social attraction plot or elsewhere on the island during the 2012 nesting season (Roby et al. 2013).

4.4.10 Dispersal & Colony Connectivity

4.4.10.1 Dispersal & colony connectivity based on band resighting/recovery – Of the more than 2,200 double-crested cormorants that we banded at the East Sand Island colony during the 1997–2014 breeding seasons (*Table 4.2* above), a total of 286 band resighting/recovery reports for 231 banded individuals were received from the U.S. Geological Survey-Bird Banding Laboratory (BBL). These band resightings/recoveries were submitted to BBL by a number of avian research scientists, animal control technicians, and members of the public. From January 2009 to July 2020, we received reports of live double-crested cormorants that were banded at East Sand Island from locations as far south as Otay Mesa, California, near the U.S./Mexico border (32.6°N, 116.9°W; ca. 1,600 km from East Sand Island), and as far north as Hornby Island (49.5°N, 124.7°W; ca. 375 km from East Sand Island) in the northern Strait of Georgia, British Columbia, Canada; *Figure 4.40*). We also received a report of a dead banded cormorant from further north and in the interior of British Columbia (50.2°N, 124.2°W; ca. 550 km from East Sand Island). Little effort was made to resight banded cormorants during 2015–2018 surveys, but four double-crested cormorants banded at East Sand Island during 2012–2013 were resighted on the Astoria-Megler Bridge during surveys in June of 2019, of which three had been confirmed nesting previously at the East Sand Island colony (*Figure 4.41*). Of the 231 banded individuals reported to BBL as resightings/recoveries over an 11-year period from 2009 to 2020, 61 (26.4%) were cormorants shot and retrieved in the Columbia River estuary by USDA-Wildlife Services technicians during 2015–2017 as part of Phase I of the *Cormorant Management Plan* (USACE 2015). Because the bulk of the double-crested cormorant band resightings/recoveries were submitted by the public, except for those from the Columbia River estuary, we were not able to determine whether most of the resighted cormorants were associated with a particular nesting colony.

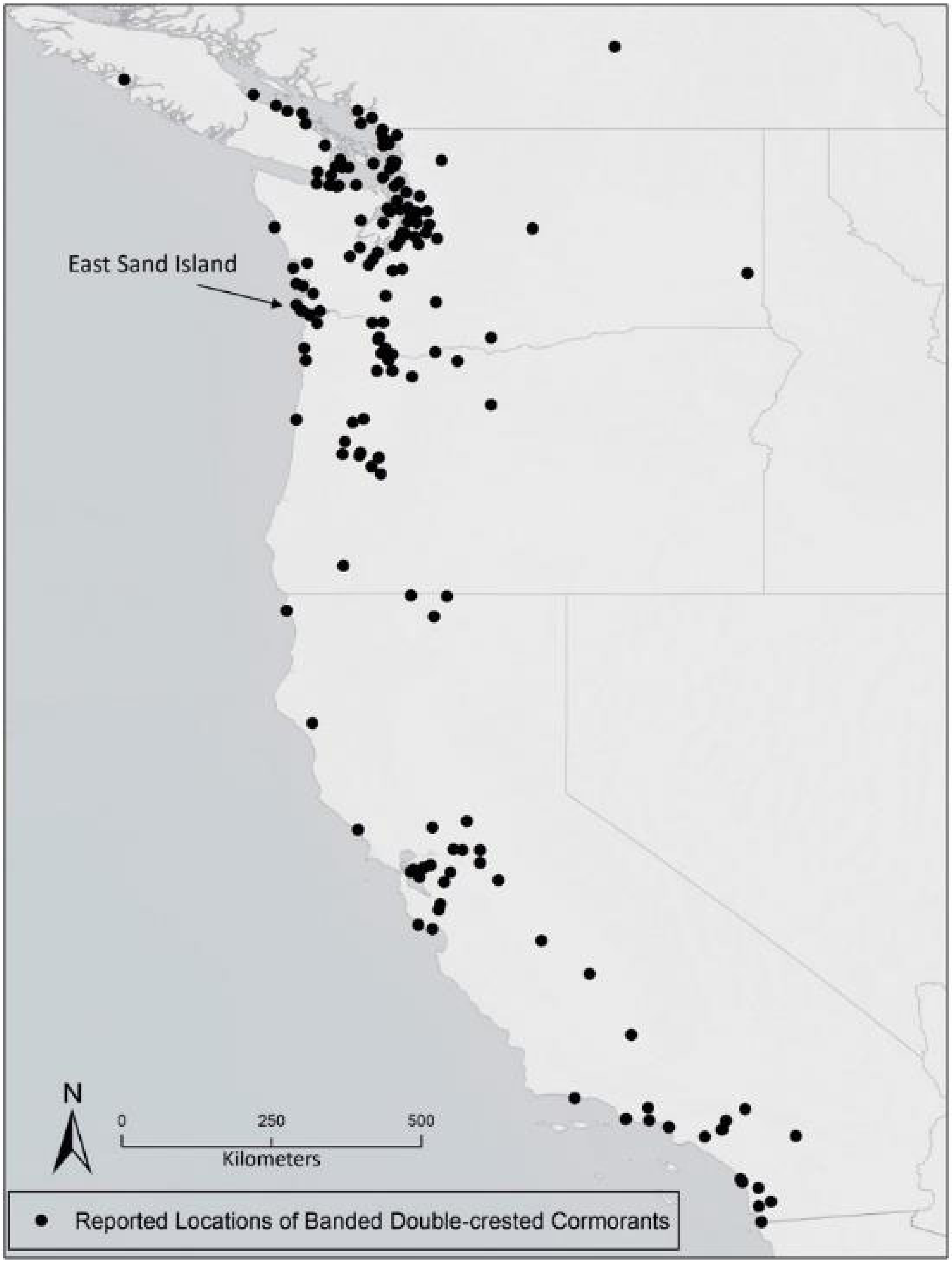

Figure 4.40. Resight/recovery locations during 2009–2020 of double-crested cormorants banded at East Sand Island in the Columbia River estuary during the 1997–2014 breeding seasons. Band resight/recovery records were reported to the U.S. Geological Survey-Bird Banding Laboratory (Patuxent Wildlife Research Center) by State and Federal employees, private research contractors, and members of the public.

Figure 4.41 Photographs of three of the four double-crested cormorants with field readable alphanumeric color bands resighted on the Astoria-Megler Bridge in the Columbia River estuary in June 2019. Each of these birds was banded at the East Sand Island cormorant colony during the 2012 or 2013 breeding seasons.

4.4.10.2 Dispersal & colony connectivity based on satellite telemetry – Tracking of double-crested cormorants fitted with platform terminal transmitter (PTT) tags during the 2008–2010 breeding seasons (n = 52) by Courtot et al. (2012) and during the 2012–2014 breeding seasons (n = 96) by Peck-Richardson (2017) was used to assess inter-colony movements of double-crested cormorants nesting at East Sand Island prior to the initiation of management actions under the *Cormorant Management Plan*. The distributional range of PTT-tagged cormorants was similar to the distributional range of banded cormorants reported to the BBL, as shown in *Figure 4.40* (see above). In the satellite telemetry study conducted during 2008–2010 by Courtot et al. (2012), post-breeding double-crested cormorants fitted with PTT tags dispersed to roost sites as far south as the mouth of the Colorado River, Baja California, Mexico (31.78°N, 114.68°W; ca. 1,800 km south of East Sand Island), and as far north as the mouth of the Powell River, British Columbia in the northern Strait of Georgia (49.98°N, 124.68°W; ca. 400 km north of East Sand Island); these sites are near the northern and southern extremes of the nesting range of the western population of double-crested cormorants (Adkins et al. 2014). In the satellite telemetry study conducted during 2012–2014 by Peck-Richardson (2017), PTT-tagged cormorants were tracked as far north as Vancouver, British Columbia (49.3°N, 123.0°W; ca. 350 km north of East Sand Island) and as far south as the Salton Sea, California (33.1°N, 115.5°W; ca. 1,700 km south of East Sand Island). In addition, the distributional range of double-crested cormorants dispersing from East Sand Island provided a detailed look into colony connectivity between the cormorant nesting colony on East Sand Island and nesting sites distributed throughout the Pacific Flyway (*Table 4.8* and *Table 4.9*; *Figure 4.42*; Courtot et al. 2012, Peck-Richardson 2017).

Table 4.8. Active (A) and historical (H) nesting colony sites outside of the Columbia River estuary region that were visited by double-crested cormorants that were satellite-tagged on East Sand Island during 2008 and 2009, including region, sub-region, count of birds, and months the site was visited. Region, sub-region, nesting site names, and status of colonies follow Carter et al. (1995) and Adkins and Roby (2010). Table modified from Courtot et al. (2012), with permission.

Region	Sub-region	Nest Site	Nesting Status	Count	Months Used
Coastal BC	Vancouver Area[a]	Sands Heads	H	1	Sep-Apr
		Second Narrows Bridge Power Tower	A	1	Jun and Nov
		Westshore Terminal	H	2	Sep-Dec
	Gulf Islands[a]	Mandarte Island	A	1	Jul
		Canoe Islet	H	1	Dec
Coastal WA	San Juan Islands[a]	Drayton Harbor	A	1	Jul
		Bare Island	H	1	Sep
		Bird Rocks	A	4	Jul-Oct
		Viti Rocks	H	5	Jul-Oct
	Olympic Peninsula	Seal Rock	H	1	Aug
		Petrel Island (Kohchaa)	A	1	Sep
		No Name 303	H	1	Aug
	Grays Harbor	Goose Island/Unnamed Sand Island	H	17	Jul-Oct
Coastal CA	Northern Coast	Arcata Bay Sand Islands	A	2	Oct
	Central Coast	Richmond-San Rafael Bridge	A	1	Jan
		San Francisco-Oakland Bay Bridge	A	3	Dec-Feb
		Donlon Island	H	1	Oct-Dec
Interior CA	Central Coast Range	Clear Lake (Lake Co.)	H	1	Nov-May
	Southern Interior	Buena Vista Lake, Kern Co.	H	1	Nov
		Mullet Is., Salton Sea (So.)	A	2	Oct-Mar

[a] *Sub-region in the Salish Sea region*

Table 4.9. Connectivity of satellite-tagged double-crested cormorants (n = 80) nesting at East Sand Island (ESI) with other regions and colony sites (indented and bolded values) away from ESI during the non-breeding season during 2012–2014 (September 16–April 15). Weighted distributions were adjusted to account for different satellite tag (PTT) sampling rates. The connectivity index is a measure of the relative magnitude of use of locations where cormorants were detected; the higher the value, the greater the apparent connectivity relative to the average within a colony group or region (zero indicates equal to the average connectivity). Connectivity indices are comparable among regions or among colonies, but are not comparable between regions and colonies. Table modified from Peck-Richardson (2017), with permission.

Region Colony (status)	**Individuals** Count (percentage)	**Weighted detections** Count (percentage)	**Connectivity Index**
Outer Vancouver Island	**4 (5)**	**656 (5)**	**-1.2**
Salish Sea	**25 (31)**	**4479 (33)**	**3.3**
Bird Rocks (A)	1 (1)	49 (<1)	-0.6
Mandarte Island (A)	1 (1)	14 (<1)	-0.8
Minor Island (A)	1 (1)	21 (<1)	-0.8
Snohomish River Mouth (A)	3 (4)	243 (2)	1.0
Outer Washington Coast	**47 (59)**	**1576 (12)**	**2.3**
Goose Island - Grays Harbor (H)	6 (8)	63 (<1)	0.2
Grays Harbor Channel Markers (A)	1 (1)	7 (<1)	-0.9
Unnamed Sand Island (A)	28 (35)	553 (4)	6.7
Columbia River Basin[a]	**19 (24)**	**2097 (15)**	**0.9**
Lewis and Clark Bridge - Longview (A)	6 (8)	48 (<1)	0.1
Smith and Bybee (H)	1 (1)	7 (<1)	-0.9
Troutdale Power Towers (A)	1 (1)	9 (<1)	-0.9
Columbia River Estuary[b]	**47 (59)**	**944 (7)**	**1.8**
Astoria-Megler Bridge (A)	39 (49)	631 (5)	8.8
Desdemona Sands Pilings (H)	3 (4)	27 (<1)	-0.5
Rice Island (H)	14 (18)	210 (2)	2.3
Upper Estuary Channel Markers (A)	4 (5)	9 (<1)	-0.4
Oregon Coast	**8 (10)**	**107 (1)**	**-1.4**
Table Rock (A)	1 (1)	2 (<1)	-0.9
Unnamed - Cape Lookout N. Rock (H)	1 (1)	7 (<1)	-0.9
Unnamed - Central Coast (A)	1 (1)	2 (<1)	-0.9
Unnamed - Southern Coast 1 (A)	1 (1)	7 (<1)	-0.9
Unnamed - Southern Coast 3 (A)	1 (1)	7 (<1)	-0.9
Northern California Coast	**14 (18)**	**657 (5)**	**-0.6**
Arcata Bay Sand Islands (A)	8 (10)	149 (1)	1.1
Castle Rock (A)	1 (1)	2 (<1)	-0.9
Hog Island (A)	2 (3)	90(1)	-0.2
Little River Rock (A)	1 (1)	1 (<1)	-0.9
Old Arcata Wharf (A)	1 (1)	30 (<1)	-0.7
Tolowa Rocks (H)	1 (1)	1 (<1)	-0.9
Northern California Interior	**1 (1)**	**21 (<1)**	**-1.9**

Table 4.9. (cont.)

Region Colony (status)	**Individuals** Count (percentage)	**Weighted detections** Count (percentage)	**Connectivity Index**
San Francisco Bay	**13 (16)**	**1449 (11)**	**0.0**
Cut off Slough - Bohannon (A)	2 (3)	117 (1)	0.0
Eucalyptus Island (A)	3 (4)	77 (<1)	-0.1
Greco Island Power Towers (A)	1 (1)	35 (<1)	-0.7
Lake Merritt (A)	2 (3)	14 (<1)	-0.7
Moffett B2 (A)	1 (1)	7 (<1)	-0.9
Moffett Power Towers (A)	1 (1)	7 (<1)	-0.9
Richmond - San Rafael Bridge (A)	5 (6)	116 (1)	0.4
San Fran. - Oakland Bay Bridge (A)	6 (8)	511 (4)	3.3
San Mateo Bridge & PG&E Towers (A)	2 (3)	23 (<1)	-0.6
Wheeler Island (A)	1 (1)	2 (<1)	-0.9
Central Valley	**14 (18)**	**658 (5)**	**-0.6**
Buena Vista Lake (H)	1 (1)	14 (<1)	-0.8
Gardner's Cove - San Joaquin River (A)	1 (1)	7 (<1)	-0.9
Kern County Water Agency (A)	1 (1)	85 (1)	-0.3
Milburn - San Juaquin River (A)	1 (1)	7 (<1)	-0.9
Southern California Bight	**4 (5)**	**235 (2)**	**-1.5**
Sepulveda Dam Rec. Area (A)	1 (1)	2 (<1)	-0.9
Salton Trough	**5 (6)**	**695 (5)**	**-1.1**
76th Avenue (H)	1 (1)	1 (<1)	-0.9
Mallard Rd. Gun Club (H)	3 (4)	58 (<1)	-0.2
Ramer Lake (A)	3 (4)	90 (1)	0.0
Salton Sea - North (A)	3 (4)	45 (<1)	-0.3

[a] *Not including the Columbia River estuary, below Puget Island*
[b] *Not including East Sand Island*

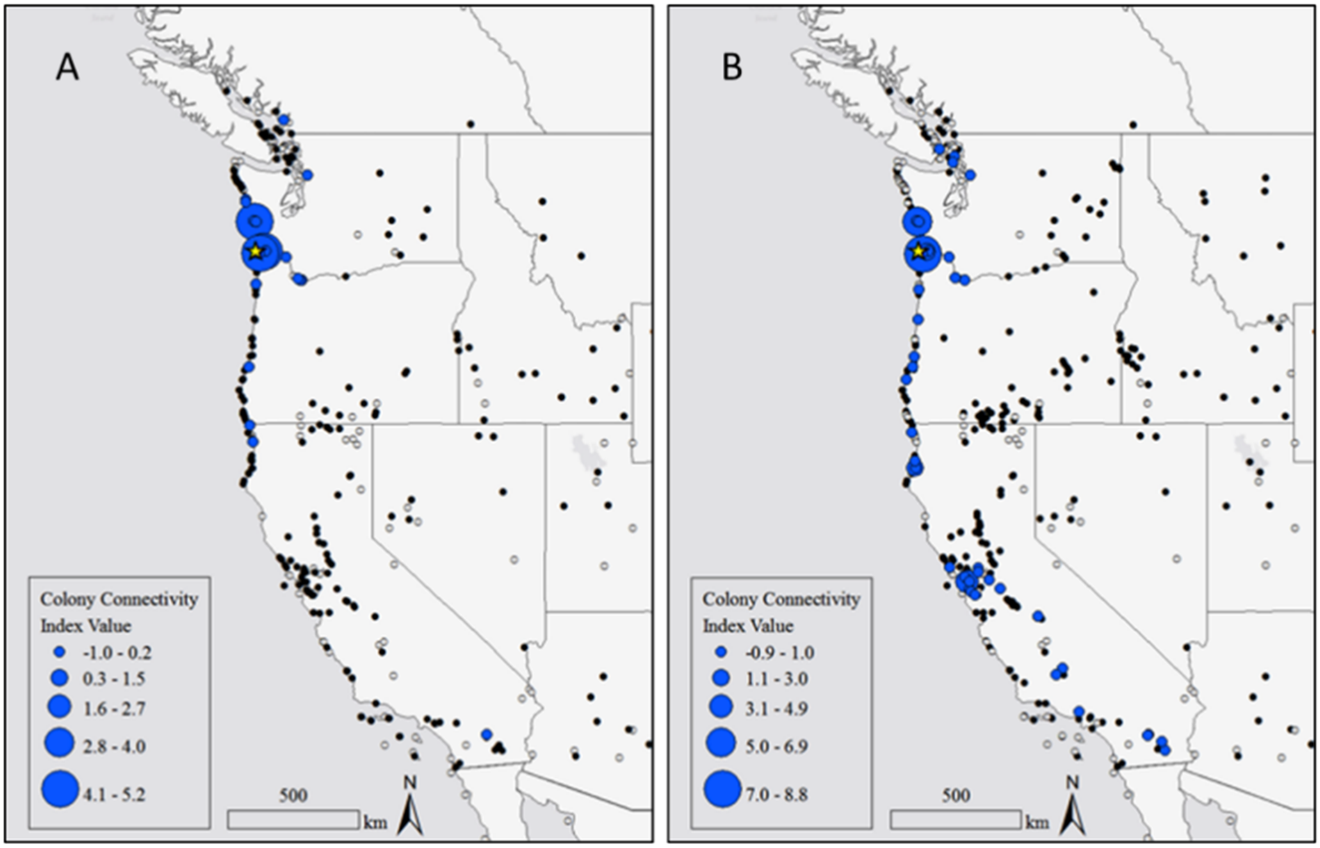

Figure 4.42. Relative regional connectivity for double-crested cormorants that were satellite-tagged on the East Sand Island colony in 2012 and 2013, based on roost locations during the breeding season (A; n = 84 tagged cormorants) and during the non-breeding season (B; n = 80 tagged cormorants). Distributions of active (●) and historical (○) colony locations for the western North America breeding population of double-crested cormorants that were not visited by satellite-tagged cormorants from East Sand Island during this period are also shown. Figures are modified from those in Peck-Richardson (2017), with permission.

During post-breeding dispersal, 75% of the 51 double-crested cormorants fitted with PTT tags at the East Sand Island colony during the 2008–2009 breeding seasons visited at least one known cormorant nesting colony site outside of the Columbia River estuary (Courtot et al. 2012); these colony sites were active cormorant breeding colonies contemporary to the study (n = 9) or historical breeding colony sites for cormorants (n = 11; *Table 4.8* above). During post-breeding dispersal, the PTT-tagged double-crested cormorants tracked during 2012–2014 (n = 80; Peck-Richardson 2017) visited about twice as many active or historical colony sites outside of the Columbia River estuary compared to those tagged during 2008–2009 (*Figure 4.42* and *Table 4.9* above). The historical nesting site in Grays Harbor, Washington (*Figure 4.42* above), located ca. 75 km north of East Sand Island, was found to be a particularly important roosting location for cormorants by both the Courtot et al. (2012) and the Peck-Richardson (2017) satellite telemetry studies (*Figure 4.42* above). PTT-tagged cormorants used the historical nesting site in Grays Harbor, in addition to roosting sites in nearby Willapa Bay, as staging locations away from East Sand Island and the Columbia River estuary prior to making longer distance post-breeding dispersal movements further to the north or south. Willapa Bay, about equidistant from Grays Harbor to the north and East Sand Island to the south, however, does not contain any known cormorant nesting locations.

Of the active and historical nesting colony sites used as post-breeding roosting sites by cormorants tagged during the 2008–2009 breeding seasons, sites were located in coastal British Columbia, coastal Washington, coastal California, and the interior of California (*Table 4.8* above; Courtot et al. 2012). PTT-tagged cormorants in 2008–2010 showed post-breeding connectivity between the nesting colony at East Sand Island and distant colony sites throughout much of the geographic range of the western population, but surprisingly little connectivity to comparatively nearby colony sites along the Oregon coast or east of the Cascade-Sierra Nevada mountain ranges. In contrast, the PTT-tagged double-crested cormorants tracked during 2012–2014 were recorded visiting colony sites along the Oregon coast, the southern California coast, and inland along the Columbia and Willamette rivers, in addition to colony sites in coastal British Columbia, coastal Washington, coastal California, and the interior of California (*Figure 4.42* above; Peck-Richardson 2017).

Neither satellite telemetry study documented the emigration by tagged cormorants from the East Sand Island colony to nest at colony sites outside of the Columbia River estuary or detected movement from East Sand Island to colony sites east of the Cascade-Sierra Nevada mountain ranges or to interior states. Nevertheless, both satellite telemetry studies documented that PTT-tagged cormorants roosting at known colony sites in the Columbia River estuary during the nesting season, in particular the Astoria-Megler Bridge colony (river km 23). During the 2012–2014 satellite telemetry study, Peck-Richardson (2017) documented comparatively greater connectivity between the cormorant colony at East Sand Island and the growing cormorant colony on the Astoria-Megler Bridge than was detected during the 2008–2010 study, as well as possible nesting attempts at the bridge by two of the 76 PTT-tagged cormorants (ca. 2.5% of the tagged individuals). In addition, the cormorant colony on the Astoria-Megler Bridge had the greatest connectivity with the East Sand Island colony of all colony sites visited by PTT-tagged cormorants during the 2012–2014 satellite telemetry study (*Table 4.9* above; Peck-Richardson 2017).

4.4.11 Results of Management Actions

4.4.11.1 Phase I – Lethal reduction of colony size – During Phase I of the *Cormorant Management Plan*, a total of 5,576 adult double-crested cormorant were culled in the Columbia River estuary and the eggs in 6,181 double-crested cormorant nests were oiled on East Sand Island (USACE, unpubl. data; *Table 4.10*). During 2015, the first year of management under the *Cormorant Management Plan*, 2,346 of the 3,489 adult double-crested cormorants that were permitted to be taken in that year were culled near East Sand Island under a depredation permit issued by the USFWS (*Table 4.10*). Those 2,346 individuals represented ca. 3% of the estimated number of breeding individuals in the western population of double-crested cormorants in 2015 (USFWS 2016). In addition, the eggs in 5,089 double-crested cormorant nests were oiled on the colony during the 2015 breeding season (*Table 4.10*), or approximately 42% of the estimated 12,150 active double-crested cormorant nests on the East Sand Island colony in that year (David Smith and Associates 2016). In 2016, USDA-Wildlife Services culled

2,982 adult double-crested cormorants in the Columbia River estuary out of a permitted take of 3,114 adults under the 2016 depredation permit and oiled the eggs in 1,092 double-crested cormorant nests on East Sand Island (*Table 4.10*). Lethal take in the Columbia River estuary during 2016 removed about 4% of the estimated number of breeding individuals in the western population of double-crested cormorants in that year (USFWS 2017) and egg-oiling destroyed the contents of about 11% of the estimated 9,772 active cormorant nests at the East Sand Island colony in 2016 (Anchor QEA 2017). Nearly all of the double-crested cormorant nests that were oiled in 2016 were oiled during a single night in May that was followed in a day or two by a large-scale colony abandonment event (Anchor QEA 2017). During the 2017 breeding season, after widespread nesting failures and colony abandonment events at the East Sand Island cormorant colony in 2016, 248 adult double-crested cormorants were culled out of the permitted take of 2,408 adults under the 2017 depredation permit. No eggs in cormorant nests were oiled in 2017 (*Table 4.10*). During Phase I of the *Cormorant Management Plan*, the estimated number of breeding individuals in the western population of double-crested cormorants declined from 73,437 birds in 2014 (95% c.i. = 67,124–79,751 individuals) to 43,075 birds in 2017 (95% c.i. = 36,132–50,016 individuals; USFWS 2020b).

Table 4.10. Maximum allowable lethal take authorized under permit and actual take of double-crested cormorant adults, double-crested cormorant nests oiled, and double-crested cormorant eggs destroyed as part of implementation of the Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary (USACE unpublished data: https://www.nwp.usace.army.mil/environment/cormorants; last accessed 11 November 2020).

	Maximum Permitted Take				**Actual Take (% of permitted take)**		
Year	Adults [a]	Nests [b]	Eggs [c]		Adults [a]	Nests [b]	Eggs [c]
2015	3,489	5,879	--		2,346 (67%)	5,089 (87%)	--
2016	3,114	5,247	--		2,982 (96%)	1,092 (21%)	--
2017	2,408	4,058	--		248 (10%)	0 (0%)	--
2018[d]	0	0	500		NA	NA	3 (0.6%)
Total	**9,011**	**15,184**	**500**		**5,576**	**6,181**	**3**

[a] *Adults shot while on colony or overwater near the East Sand Island colony using firearms.*
[b] *Nests with eggs oiled on the East Sand Island colony.*
[c] *Individual eggs destroyed outside the designated East Sand Island colony area.*
[d] *Beginning of Phase II of the* Cormorant Management Plan.

4.4.11.2 Phase II – Reduction of nesting habitat – Terrain modification efforts during the winter of 2018–2019 successfully converted ca. 12 acres (ca. 4.9 ha) of the former upland nesting habitat at the west end of East Sand Island into a tidally-influenced mudflat prior to the 2019 breeding season (*Figure 4.43*). In 2019, about 350 breeding pairs of double-crested cormorants nested on East Sand Island (see section *4.4.1.1 Colony size* above), and those nests were limited to the areas west of and adjacent to the installed dissuasion fence. During Phase II of the *Cormorant Management Plan*, the estimated number of breeding individuals in the western population of double-crested cormorants remained unchanged; estimated breeding population size at the end of Phase I (2017) was 43,075 individuals (95% c.i. = 36,132–50,016 individuals) and estimated breeding population size at the end of Phase II (2019) was 45,778 individuals (95% c.i. = 31,849–59,708 breeding individuals; USFWS 2020b). Concurrent with nesting habitat reductions on East Sand Island during Phase II of the *Cormorant Management Plan*, which resulted in a 90% reduction in the number of nesting double-crested cormorants at East Sand Island (3,672 breeding pairs in 2018; 350 breeding pairs in 2019), the colony of double-crested cormorants on the Astoria-Megler Bridge grew from an estimated 1,737 breeding pairs during the 2018 breeding season (the first year of Phase II) to an estimated 3,542 breeding pairs during the 2019 breeding season (see section *4.4.1.1 Colony size* above).

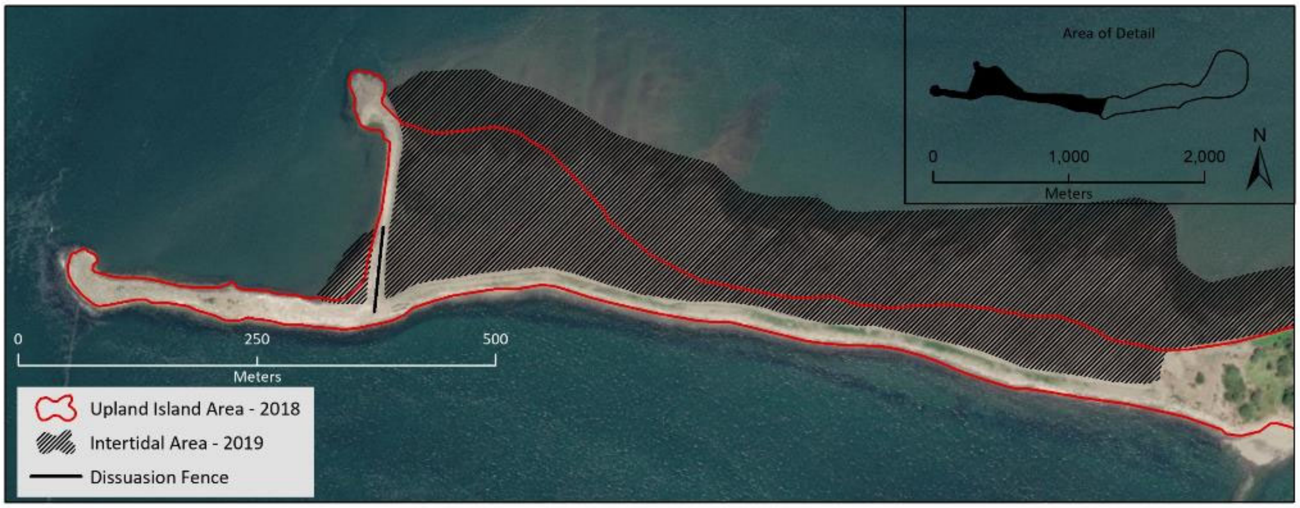

Figure 4.43. Approximate extent (shaded area) of the tidally influenced mudflats in 2019 following terrain modification at the west end of East Sand Island as part of Phase II of the Cormorant Management Plan. Solid black line indicates location of the dissuasion fence; solid red line indicates the outline of the west end of East Sand Island during the 2018 breeding season; cormorant nesting habitat was designated as west of the fence following terrain modification.

Phase I of the *Cormorant Management Plan* was initially intended to last for four years (2015–2018) and transition to Phase II in 2019 (USACE 2015). Management transitioned to Phase II in 2018, however, following repeated abandonment of the cormorant colony in 2016 and 2017, and a peak colony size of only 544 breeding pairs in 2017. The *Cormorant Management Plan* included this adaptive management provision should the size of the double-crested cormorant colony on East Sand Island drop below 5,380 breeding pairs. During the 2018 breeding season, the 3,672 breeding pairs of double-crested cormorants that nested on East Sand Island were restricted to the 1.7 acres (0.7 ha) of designated nesting habitat, and the colony occupied ca. 0.8 acres (0.3 ha) or about 46% of the designated nesting habitat (Turecek et al. 2019). Thus, the size of the cormorant colony in 2018 remained below the threshold of 5,380 breeding pairs for Phase I management (USACE 2015), and a depredation permit for culling adult cormorants and egg-oiling was not issued by the USFWS for 2018. Instead, the 2018 depredation permit allowed the take of up to 500 double-crested cormorant eggs in support of dissuading cormorants from nesting outside the 1.7-acre (0.7-ha) designated nesting area. Only three cormorant eggs were taken under the 2018 depredation permit (USACE, unpubl. data; *Table 4.10* above).

4.5 Discussion

4.5.1 Nesting, Foraging, & Trophic Ecology

The initial rapid increase in the size of the double-crested cormorant colony on East Sand Island, which went from 91 breeding pairs in 1989 to 2,026 breeding pairs in 1991 (Carter et al. 1995), was driven by immigration from other cormorant colonies throughout the range of the western population, but likely mostly from declining colonies in the Salish Sea region of Washington, USA, and British Columbia, Canada. Emigration from cormorant colonies in the Salish Sea was reportedly the result of increasing disturbance from bald eagles and humans (Chatwin et al. 2002, Adkins et al. 2014). During 1992-1997, the colony on East Sand Island continued to increase, but at a lower rate, and intrinsic growth (recruitment of adults that were raised in the Columbia River estuary) may have become increasingly important for continued growth in colony size. Immigrants from the Salish Sea region were likely attracted to the East Sand Island colony by (1) its large size, which may have provided a buffer from the impacts of disturbance and predation by bald eagles, (2) its location, which provided access to a diverse and abundant supply of marine, estuarine, freshwater, and anadromous forage fishes, and (3) the safety it afforded from terrestrial predators. During 1998–2006, the number of cormorants nesting at the colony on East Sand Island continued to increase and expand into new nesting habitats, but the relative contributions of intrinsic vs. extrinsic recruitment to continued colony growth are unknown.

From 2006 to 2014 the size of the colony remained stable at about 13,500 breeding pairs, while the numbers of bald eagles foraging at the cormorant colony and eagle disturbance rates on nesting cormorants early in the nesting season increased. Also, the prevalence of some marine forage fishes (e.g., clupeids) in the early season cormorant diet declined, especially in years when river discharge was high. The combination of increasing eagle disturbance and declines in availability of marine forage fish early in the nesting season was apparently responsible for curtailing the growth of the East Sand Island colony during this period. Further, during 2006–2013 the proportion of juvenile salmonids in the diet of double-crested cormorants nesting at the East Sand Island colony increased, with the estimated consumption of juvenile salmonids peaking at ca. 20 million smolts in 2011. Double-crested cormorants are generalist predators (Dorr et al. 2014) that tend to feed on fish prey types in relation to their availability. The proportion of juvenile salmonids in the diet of double-crested cormorants nesting on East Sand Island averaged 14% (% of prey biomass) during 2006–2013 and was negatively correlated with cormorant nesting success, indicating that double-crested cormorants nesting on East Sand

Island were neither specializing on juvenile salmonids as a prey source nor reliant on this prey type for successful nesting. Despite the large numbers of juvenile salmonids consumed by double-crested cormorants nesting on East Sand Island during 2010-2013 (average of ca. 19.3 million smolts per year), the large size and high nesting success of this colony was more dependent on the availability other prey types (e.g., anchovies, clupeids). In turn, the large size of the East Sand Island colony of double-crested cormorants (nearly 15,000 breeding pairs in 2013) was the attribute that likely provided the safety in numbers for nesting cormorants to continue to resist the increasing disturbance and attacks from a growing population of bald eagles in the Columbia River estuary.

Meanwhile, in 2004 a small number of double-crested cormorants began nesting on the Astoria-Megler Bridge, 15 km up-river from the East Sand Island colony. The Astoria-Megler Bridge colony grew from 7 breeding pairs in 2006 to 333 breeding pairs in 2014, prior to implementation of the *Cormorant Management Plan*. It appears the Astoria-Megler Bridge offered nest sites that were more protected from bald eagle depredation and the associated gull predation on cormorant eggs and nestlings compared to the colony on East Sand Island, where the only protection from eagles and gulls was the sheer size of the colony. Because the Astoria-Megler Bridge colony is located further up-river from the East Sand Island colony and in the transition zone between the marine portion of the estuary and the freshwater portion of the estuary, per capita predation rates on juvenile salmonids by cormorants nesting at the Bridge colony are likely higher than those of cormorants nesting at East Sand Island. Compared to East Sand Island, per capita predation rates by double-crested cormorants nesting on Rice Island in the freshwater zone of the Columbia River estuary are much higher (see *Chapter 5*), suggesting higher per capita predation rates for cormorants nesting on the Astoria-Megler Bridge as well.

While double-crested cormorants nesting on the Astoria-Megler Bridge likely experience lower availability of marine forage fish prey near their breeding colony than conspecifics nesting on East Sand Island, the much smaller size of the Astoria-Megler Bridge colony compared to the East Sand Island colony suggests that there was much less intra-specific competition for prey within proximity of the Bridge colony. The theory of Ashmole's halo (Birt et al. 1987) predicts that intra-specific competition for food among colonial-nesting seabirds increases as colony size increases and prey resources are depleted further and further from the colony. As of 2020, however, the double-crested cormorant colony on East Sand Island appears to have been abandoned and the colony on the Astoria-Megler Bridge has increased to over 5,000 breeding pairs (M.J. Lawonn, ODFW, pers. comm.), making it the largest double-crested cormorant colony in the Pacific Flyway. The relative security from eagle and gull depredations of the colony site on the Bridge perhaps more than compensates for the lower availability of marine forage fish.

4.5.2 Management Feasibility Studies

During 2007–2013, a number of feasibility studies were conducted to test and evaluate non-lethal approaches for reducing the size of the East Sand Island colony of double-crested cormorants, which had become the largest known breeding colony for the species. Most of these approaches were modeled on the non-lethal methods used to reduce the size of the Caspian tern colony on East Sand Island (USFWS 2005), which had also become the largest colony for the species in the world. The approach used for the Caspian tern colony had both “push” and “pull” components, the push consisting of reducing the availability of suitable nesting habitat (nest dissuasion) and the pull consisting of attracting breeding adults to alternative colony sites away from the Columbia River estuary.

4.5.2.1 Nonlethal nest dissuasion methods – Approaches for dissuading double-crested cormorants from nesting on East Sand Island belonged to three different categories: (1) providing perches for bald eagles on the cormorant colony, (2) modifying nesting habitat to render it unsuitable, and (3) actively hazing or disturbing cormorants in the initial stages of nesting.

Eagle perches: Erecting bald eagle perches to enhance eagle disturbance of the formerly large cormorant colony on East Sand Island may be a potential method for limiting the size and nesting distribution of the colony but was unlikely to be useful as a technique to reduce colony size to a certain target, such as 5,380–5,939 breeding pairs. The *Cormorant Management Plan*, however, was careful to stipulate that a colony of *less than* 5,380–5,939 breeding pairs was the management objective (USACE 2015). If complete elimination of the East Sand Island cormorant colony is an acceptable outcome for the *Cormorant Management Plan*, attracting more bald eagles to the cormorant colony by erecting perches or by providing carcasses as an attractant could be an effective method for enhancing predation and disturbance of nesting cormorants by bald eagles and thereby reducing the number of cormorants nesting on East Sand Island. The efficacy of this management approach is based on the supposition that (1) bald eagles have disrupted other cormorant colonies and caused their abandonment, (2) bald eagle predation and disturbance was a major factor limiting the expansion of the cormorant colony on East Sand Island during 2006-2014, and (3) the functional and numerical response of bald eagles preying on nesting double-crested cormorants hastened the time when eagle disturbance and depredation limited the growth of the cormorant colony (Roby et al. 2008). This management approach for the East Sand Island colony of double-crested cormorants likely has two significant down sides: (1) the negative effect on the more than 1,000 pairs of Brandt’s cormorants that formerly nested at the East Sand Island cormorant colony, and Brandt’s cormorants were not a target for management, and (2) disturbance by bald eagles has been identified as one factor in the repeated abandonment of the East Sand Island cormorant colony during implementation of the *Cormorant Management Plan* and played a role in the rapid increase in the size of the double-crested cormorant colony on the Astoria-Megler Bridge, where eagle disturbance and human disturbance appear to be quite limited. Although the complete elimination of the double-crested cormorant colony on East Sand Island has its advocates among some managers and stakeholders, it can be counter-productive for efforts to minimize smolt mortality due to cormorant predation in the estuary by dispersing cormorants to colony sites in the upper estuary where per capita predations rates on juvenile salmonids are greater.

Fencing, stakes, rope, & flagging: Our results suggest that passive nest dissuasion techniques such as landscape fabric fencing or networks of stakes, ropes, and flagging are much less effective in discouraging double-crested cormorants from nesting than they are for dissuading Caspian terns from nesting. Neither of the above passive nest dissuasion techniques ultimately prevented cormorants from nesting on the parts of the East Sand Island cormorant colony where they were deployed. Our pilot studies indicated that, in order to limit the size of a large cormorant colony, such as the former colony on East Sand Island, passive nest dissuasion methods need to be highly effective in deterring cormorants from nesting in certain areas, and capable of withstanding the extremes of weather at the colony site. Landscape fabric fencing and networks of ropes, stakes, and flagging failed to meet these criteria.

Pond liner: Use of a pond liner was highly effective in both 2009 and 2010. The high cost and difficulty of deploying pond liner as a passive nest dissuasion technique, however, should be considered before attempting to use the method to reduce the numbers of double-crested cormorants nesting at a large colony, such as the former colony on East Sand Island. Logistically, the most important consideration is the weight of pond liner material; one roll weighs over 100 kg (220 lbs.) and transport and installation over an area of several acres can be extremely challenging. Further, due to the harsh winter environment on East Sand Island, the pond liner would need to be removed from the island at the conclusion of each breeding season, adding to the already considerable transport challenges. We concluded that pond liner is an impractical method of passive nest dissuasion for limiting the size of a large ground-nesting colony of double-crested cormorants where there is an extensive area of suitable nesting substrate available.

Active nest dissuasion and hazing: Double-crested cormorants are much more wary of humans, and therefore more susceptible to human hazing and disturbance, than are terns or gulls. Prior to implementation of the *Cormorant Management Plan*, pilot studies to test the feasibility of various nest dissuasion were not permitted to cause egg loss or nest failure for cormorants nesting on East Sand Island, so all pilot studies focused on dissuading incipient cormorant nesting attempts before eggs were laid. During implementation of the *Cormorant Management Plan*, human disturbance to the cormorant colony was not restricted to the pre-laying period, and the cormorant colony was repeatedly abandoned, and breeding adults dispersed during Phase I of the Plan (2016 and 2017). Consequently, active nest dissuasion approaches that involve human disturbance (hazing) can be highly effective at causing colonies of double-crested cormorants to abandon and for breeding-age adults to disperse to other colonies (Carter et al. 1995, Duerr et al. 2009), especially colonies that are also susceptible to eagle disturbance and predation on nests by gulls. Therefore, management actions intended to reduce the size of, but not eliminate, breeding colonies of double-crested cormorants must address the challenge of hazing and disturbing some, but not all, breeding adults at the colony.

As described by the manufacturer, the green laser was most effective for hazing prospecting cormorants during low light conditions. All tests of the green laser that were successful in flushing double-crested cormorants were conducted early or late in the day under at least 40% cloud cover. Under the conditions tested, the laser appeared to be most effective at close range (< 60 m) relative to its potential range, which according to the manufacturer may exceed 2 km. Based on these results, any attempt to use green laser hazing to deter egg-laying on part of a cormorant colony should be: (1) initiated early in the breeding cycle, before pair bonds and nest territories have been established, (2) carried out at least daily during low light conditions (i.e. before 08:30 and after 21:00), and (3) employed for as long as necessary to clear the target area of any prospecting, pre-breeding cormorants. As such, using the green laser to dissuade thousands of pairs of double-crested cormorants from nesting on parts of a large, extensive colony, like the former colony on East Sand Island or the current colony on the Astoria-Megler Bridge, may be impractical. The green laser technique could, however, be used to dissuade double-crested cormorants from nesting at small, incipient colonies where the presence of nesting cormorants would cause conflicts.

Dissuasion fences (i.e. privacy fences), nest destruction, and targeted human hazing was demonstrated to be a feasible method for preventing double-crested cormorants from nesting on large, pre-defined areas of the East Sand Island cormorant colony without adversely impacting non-target cormorants and other colonial waterbird species outside the dissuasion area. Nest dissuasion experiments conducted during 2011–2013 were successful in progressively limiting the available nesting habitat on the western end of East Sand Island from the previous total of 16 acres (6.5 ha) available in 2010 down to 4 acres (1.6 ha) in 2013. Despite this large reduction, the available nesting habitat during the nest dissuasion experiments was not reduced to the point where nesting habitat became limiting for cormorants at the western end of East Sand Island. The growth rate of the East Sand Island cormorant colony during 2011–2014, when the nest dissuasion experiments were conducted, was stable ($\lambda = 1.00$), as was colony growth rate in the previous four years (2007-2010; *Figure 4.11* above). These trends in colony size suggest that, while the dissuasion experiments resulted in large changes in where cormorants nested on the island, it had little or no impact on the size of the East Sand Island cormorant colony. Most double-crested cormorants hazed from their nesting areas in 2013, the final year of study, were able to successfully relocate to nest in the non-dissuasion area between the two dissuasion fences, and the East Sand Island colony of double-crested cormorants was a record size in that year. If these methods were used to limit the size of the double-crested cormorant colony on East Sand Island to no more than 5,380–5,939 breeding pairs (USACE 2015), then available nesting habitat in the non-dissuasion area would need to be reduced to about 1.0 acres (0.4 ha: BRNW *unpubl. data*).

4.5.2.2 Social attraction & habitat enhancement methods – Results from the two experimental habitat modification and social attraction plots set up on Miller Sands Spit and on Rice Island in the Columbia River estuary, two islands at least 20 km from the large cormorant colony on East Sand Island, suggested that social attraction/habitat enhancement techniques can be effective for establishing new double-crested cormorant colonies outside the Columbia River estuary (Collis et al. 2007). A combination of placing at least 36 tires filled with fine woody debris on the ground on islands where mammalian nest predators are absent and placing several dozen cormorant decoys and at least two audio playback systems amongst the tires can be effective at attracting double-crested cormorants to establish new breeding colonies. These new colonies could support part of the population of cormorants that currently nests in the Columbia River estuary (e.g., the colony on the Astoria-Megler Bridge), thereby reducing cormorant predation rates on juvenile salmonids in the estuary. The efficacy of this approach, however, is likely to be limited by human disturbance and bald eagle disturbance to the extent that these factors cause cormorant nest failure at newly established alternative colony sites. Also, sites with a history of cormorant nesting attempts or, at least, cormorant roosting are more likely to support successful attempts at cormorant colony establishment using social attraction (Suzuki et al. 2015).

Habitat enhancement and social attraction techniques were demonstrated effective in establishing double-crested cormorant breeding colonies at sites in the Columbia River estuary where cormorant nesting attempts had previously occurred. But failed attempts to establish new double-crested cormorant colonies during the three-year social attraction feasibility study at Fern Ridge Wildlife Area, the two-year feasibility study at Dutchy Lake in Summer Lake Wildlife Area, the two-year feasibility study at Tule Lake Sump 1B in Tule Lake NWR, and the one-year feasibility study at Malheur Lake tern island in Malheur NWR suggest that these techniques may require longer periods of time to be successful at attracting cormorants to nest at sites where there is no prior nesting history. As such, the efficacy of habitat enhancement and social attraction techniques to establish new double-crested cormorant colonies outside the Columbia River basin remains uncertain.

Nevertheless, double-crested cormorants have nested successfully and appear to have established persistent breeding colonies on most of the islands constructed by the USACE as alternative nesting habitat for Caspian terns outside the Columbia River estuary. Double-crested cormorants have established persistent breeding colonies on artificial tern islands built on Sheepy Lake in Lower Klamath NWR, California; Malheur Lake in Malheur NWR, Oregon; East Link Impoundment in Summer Lake Wildlife Area, Oregon; Crump Lake in the Warner Valley, Oregon; and Tule Lake Sump 1B in Tule Lake NWR, California, but only after Caspian terns, California gulls, and ring-billed gulls had established breeding colonies on each island. Of the seven islands built as alternative colony sites for Caspian terns in interior Oregon and northeastern California, only the 1-acre (0.4-ha) tern island at Fern Ridge Reservoir and the 0.5-acre (0.2-ha) tern island in Gold Dike Impoundment, Summer Lake Wildlife Area have not supported new double-crested cormorant colonies, and these two islands were not colonized by other colonial waterbirds either.

Of the five USACE-constructed tern islands where new cormorant colonies formed, two had conspecific (i.e. double-crested cormorant) social attraction deployed during one or two nesting seasons immediately after island completion. The presence of other species of nesting colonial waterbirds on the USACE-constructed tern islands was apparently more attractive to prospecting adult cormorants than decoys and audio playback of conspecifics. Also, double-crested cormorants first nested on these five USACE-constructed tern islands 4–8 years after the islands were completed and nesting terns and/or gulls had colonized them. Sheepy Lake tern island has supported a cormorant colony as large as 240 breeding pairs (2018), Malheur Lake tern island has supported a cormorant colony as large as 95 breeding pairs (2016), East Link tern island has supported a cormorant colony as large as 43 breeding pairs (2018), Tule Lake Sump 1B tern island has supported a cormorant colony as large as 41 breeding pairs (2018), and Crump Lake tern island has supported a cormorant colony as large as 22 breeding pairs (2016). These results suggest that establishing colonies of ground-nesting double-crested cormorants at sites without a history of cormorant nesting can best be accomplished by first establishing breeding colonies of colony associates, such as terns, gulls, or pelicans, which then signal to prospecting cormorants that the site is safe for nesting. Even in those cases where new colony sites were immediately colonized by other species of colonial waterbirds, it took at least four years for double-crested cormorants to finally initiate nesting. In the case of the five USACE-constructed tern islands that were ultimately colonized by nesting double-crested cormorants, however, there were no efforts to displace nesting cormorants from other nearby colony sites. For example, double-crested cormorants have quickly established large new colonies on novel sites in Coos Bay, Oregon, after other nearby colonies were abandoned (M.J. Lawonn, ODFW, pers. comm.).

4.5.3 Dispersal & Colony Connectivity

Satellite-tracking studies of adult double-crested cormorants tagged with PTTs at the East Sand Island colony (Courtot et al. 2012, Peck-Richardson 2017) demonstrated connectivity with other double-crested cormorant colonies from southern British Columbia to southern California. After the breeding season on East Sand Island, PTT-tagged adults dispersed over much of the breeding range of the western population of double-crested cormorants. This suggests that the East Sand Island colony has connectivity with most other cormorant colonies throughout the Pacific Flyway. While this connectivity may simply reflect the suitability of other colony sites as over-winter roost sites, it also suggests that natal dispersal from other colonies throughout the breeding range of the Pacific Flyway population to the East Sand Island colony may have occurred, especially during the 1990s when the numbers of cormorants nesting in the Columbia River estuary were increasing rapidly. For example, some double-crested cormorants banded as nestlings at Mullet Island in the Salton Sea, California, have been resighted on the East Sand Island colony (Y. Suzuki, OSU, unpubl. data). The fact that most of the resightings/recoveries of double-crested cormorants banded on East Sand Island were in the Salish Sea region of Washington and British Columbia (Clark et al. 2006, see section *4.4.10.1 Dispersal & colony connectivity based on recoveries of banded birds* above), plus the high connectivity between the East Sand Island colony and current and former cormorant colony sites in the Salish Sea as demonstrated by satellite-tagged cormorants (Courtot et al. 2012, Peck-Richardson 2017), suggests that much of the immigration to the East Sand Island colony during the early years (1989–1997) was from colonies in the Salish Sea region. This is further supported by declines in size and abandonment of a number of double-crested cormorant colonies in the Salish Sea region during the period when the colony on East Sand Island was growing rapidly (Moul and Gebauer 2002, Adkins et al. 2014).

The capture and satellite-tagging of pre-nesting double-crested cormorants on the East Sand Island colony was intended, in part, to reveal where cormorants would disperse to if not allowed to nest on East Sand Island (Peck-Richardson 2017). The general pattern of short dispersal trips (< 2 weeks) and high return rates to the East Sand Island colony following satellite-tagging suggests high site fidelity to the East Sand Island colony site. High colony site fidelity may have been a result of the prolonged history of successful nesting on East Sand Island (many individual cormorants having nested on East Sand Island their entire lives), conspecific attraction and facilitation from the very large colony on East Sand Island, availability of unoccupied suitable nesting habitat for double-crested cormorants on East Sand Island, and the lack of suitable nesting opportunities at alternative colony sites both within and outside the Columbia River estuary.

To induce double-crested cormorants to permanently emigrate to alternative colony sites outside the Columbia River estuary, it would be necessary to both restrict suitable nesting habitat on East Sand Island to 1.0 acres (0.4 ha) or less, plus prevent immigration to colony sites elsewhere in the Columbia River estuary with demonstrated high connectivity to the East Sand Island colony (i.e. the Astoria-Megler Bridge and Rice Island). Cormorants nesting at these alternative colony sites further up-river in the Columbia River estuary have greater per capita predation rates on juvenile salmonids than cormorants nesting at East Sand Island.

The high colony site fidelity demonstrated by double-crested cormorants that were tagged on the East Sand Island colony makes the repeated partial and total colony abandonments during implementation of the *Cormorant Management Plan* difficult to explain without invoking the disturbances associated with implementation of the Plan. Despite the absence of data on the frequency and magnitude of disturbances to the cormorant colony on East Sand Island during the management period, we conclude that the cumulative impact of human disturbance and colony size reduction associated with implementation of the Plan, combined with eagle disturbances associated with a growing bald eagle population in the Columbia River estuary, repeatedly crossed a threshold for cormorant tolerance at East Sand Island. Once that threshold had been crossed, abandonment of nest sites by breeding cormorants occurred *en masse*, affecting large portions or the entire colony, and leaving nest contents unprotected and vulnerable to predation. These episodes of partial or complete colony abandonment were associated with large numbers of cormorants dispersing, at least temporarily, to other colony sites in the Columbia River estuary.

4.5.4 Management Action Effectiveness

The *Cormorant Management Plan* succeeded in reducing the size of the cormorant breeding colony on East Sand Island to the target of no more than 5,380–5,939 breeding pairs (USACE 2015). It also initiated some unintended changes to the nesting and foraging ecology of cormorants in the Columbia River estuary, however, which likely compromised efforts to reduce predation impacts on ESA-listed salmonids in the Columbia River estuary, the primary goal of the Plan. In 2019, only about 350 pairs of double-crested cormorants attempted to nest on East Sand Island, and in 2020 there were no sustained nesting attempts by double-crested cormorants on East Sand Island (Lawonn, In prep.). The apparent near total abandonment by breeding cormorants of the East Sand Island colony site in 2020 followed unprecedented partial and total colony abandonments each year during the 2016-2019 nesting seasons (*Figure 4.19* above; USACE *unpubl. data*), concurrent with implementation of the *Cormorant Management Plan*.

The unexpected and unprecedented episodes of partial and total abandonment of the double-crested cormorant colony on East Sand Island during the 2016 and 2017 nesting seasons resulted in a peak colony size of 544 breeding pairs in 2017, well below the minimum colony size needed to continue lethal take under the depredation permit issued to the USACE by the USFWS. As such, Phase I of the *Cormorant Management Plan* was terminated after the 2017 nesting season and management transitioned to Phase II of the Plan in 2018. The move to Phase II was a year earlier than expected but was consistent with adaptive management provisions written into the Plan. Whereas the Plan had called for the culling of up to 10,912 adult cormorant and the oiling of eggs in up to 15,184 nests during 2015–2018, Phase I ended with a total of 5,576 adult double-crested cormorants culled and the eggs in 6,181 cormorant nests oiled. Phase II of the *Cormorant Management Plan* shifted to primarily nonlethal management actions to prevent the double-crested cormorant colony on East Sand Island from exceeding the threshold of 5,380–5,939 breeding pairs, as stipulated in the *Plan* (USACE 2015). During the first year of Phase II in 2018, the size of the double-crested cormorant colony on East Sand Island rebounded to a peak colony size of 3,672 breeding pairs. During the second year of Phase II in 2019, following major habitat modification to the west end of East Sand Island, the peak size of the double-crested cormorant colony declined to ca. 350 breeding pairs.

The large decline in size of the East Sand Island colony cannot be directly explained by the lethal removal implemented during Phase I of the *Cormorant Management Plan*, during which a total of 5,576 adult double-crested cormorants were reportedly culled and the eggs in 6,181 cormorant nests were oiled (USACE, unpubl. data). Given an average colony size of about 13,500 breeding pairs during the five years preceding implementation of the *Cormorant Management Plan*, the culling of 5,576 adults would be expected to reduce the size of the East Sand Island colony to between 10,000 and 11,000 breeding pairs (assuming otherwise constant survival and reproductive rates). Consequently, the near total abandonment of the East Sand Island cormorant colony in 2020 has led to roughly 20,000 adult double-crested cormorant prospecting for new nest sites.

The factors responsible for the repeated and unprecedented partial and total abandonments of the East Sand Island colony during the management period are not fully understood, but the primary factor appears to be the cumulative disturbance and stress to cormorants attempting to nest at the East Sand Island colony from management activities (Ellison and Cleary 1978, Strickland et al. 2011), plus carry-over effects of those disturbances from one breeding season to the next on reproductive investment and nest site fidelity (Blackmer et al. 2004, Harrison et al. 2011, Salas et al. 2020). Intense disturbance of nesting cormorants by bald eagles was also clearly a factor in episodes of colony abandonment, but intense eagle disturbance was prevalent late in the pre-management period as well, especially in 2011, but never caused the mass abandonment behavior at the East Sand Island colony that was repeatedly recorded during the 2016-2017 nesting seasons. During 18 breeding seasons of research, monitoring, and evaluation of the cormorant colony on East Sand Island, our team experienced many challenges to working in the colony without disturbing nesting cormorants and went to extraordinary lengths to minimize that disturbance (e.g., building blinds and above-ground tunnels, accessing tunnels only during low tide). Nevertheless, it proved impossible to completely avoid researcher disturbance (*Figure 4.22* above). Despite research disturbance to the cormorant colony during 1997-2014, and occasionally other large disturbance events caused by unauthorized landings by recreational boaters or helicopter overflights, our field crews never witnessed the type of mass abandonment of part or all the cormorant colony as was repeatedly documented during the management period.

During the management period the USACE and its contractors were engaged in efforts to reduce the size of the cormorant colony and were not obligated to take similar steps to avoid all disturbances to nesting cormorants that might lead to nest failure (David Evans and Associates 2016; Anchor QEA 2017; Turecek et al. 2018; Turecek et al. 2019; Lawonn, in prep.; USACE, unpubl. data). As such, it is fair to assume that their activities may have continued to contribute in some degree to the cumulative disturbance experienced by cormorants during both Phase I and Phase II management actions. While the exact threshold of tolerance by double-crested cormorants nesting on East Sand Island to disturbances during the nesting season and alterations in nesting habitat would have been difficult to predict, that threshold was apparently exceeded multiple times during the management period.

As a result of the major decline in size of the double-crested cormorant colony on East Sand Island, *predation rates on juvenile salmonids by cormorants nesting on East Sand Island* were lower for most, but not all, salmonid ESUs/DPSs following implementation of Phase I management actions under the *Cormorant Management Plan* (see section *4.4.7 Predation Rates on Juvenile Salmonids* above). Predation rates by cormorants nesting on East Sand Island trended lower following the initiation of Phase I management for eight of the 11 ESUs/DPSs of salmonids evaluated, with statistically significant declines in predation rates observed for two salmonid populations: LCR Chinook salmon and LCR coho salmon. Predation rates were not significantly lower for the other three ESUs/DPSs evaluated, however, and for some ESUs/DPSs (e.g., Snake River spring/summer Chinook salmon) predation rates were higher during Phase I management compared to pre-management, despite a substantial decline in the average numbers of double-crested cormorants nesting on East Sand Island during Phase I. The high levels of intra-annual variability in cormorant predation rates on juvenile salmonids that were observed pre-management may, at least in part, explain why predation rates on some salmonid ESUs/DPSs remained similar following implementation of Phase I management actions. In 2015, the first year of implementation of Phase I of the *Cormorant Management Plan*, lethal take removed a relatively small number of adult double-crested cormorants (n = 2,346), or less than 10% of the numbers nesting on East Sand Island pre-management, and these cormorants were culled between 22 May and 31 August 2015 (USACE 2015), after most PIT-tagged smolts had out-migrated through the estuary that year (Evans et al. 2016). In 2015, predation rates by double-crested cormorants nesting on East Sand Island were among the highest ever recorded for that colony (*Appendix B, Table B23*). As such, inclusion of estimates of cormorant predation rates from 2015 in the management period average increased the average annual estimate of predation rates during Phase I, but reductions in cormorant predation rates on juvenile salmonids due to reductions in size of the East Sand Island colony did not occur until 2016 (Evans et al. 2018).

Estimates of average annual predation rates by East Sand Island cormorants were significantly lower for all salmonid ESUs/DPSs evaluated following implementation of Phase II management actions on East Sand Island beginning in 2018. The predation rates by East Sand Island cormorants on salmonids in 2018 were lower than those recorded for the double-crested cormorant colony on East Sand Island during any year of the pre-management period. At 3,672 breeding pairs, the size of the East Sand Island double-crested cormorant colony was also much smaller than in any year during the pre-management period (1997–2014). Predation rates by East Sand Island cormorants in 2018 were less than 1.0% of available smolts for all ESUs/DPSs evaluated, with the notable exception of LCR Chinook salmon, where the predation rate was estimated at 7.3%. Predation rates during Phase II were approximately 70% to 90% lower, depending on the salmonid ESU/DPS, than those observed during 2003–2014, prior to implementation of the *Cormorant Management Plan*. Although these reductions in predation rates were coincident with large reductions in cormorant colony size at East Sand Island, they were not commensurate with the magnitude of those reductions. The greater than expected reduction in predation rates by double-crested cormorants nesting on East Sand Island in 2018 is likely due to the delayed onset of cormorant nesting at East Sand Island compared to the pre-management period (see *Section 4.4.1.2*) and the high inter-annual variability in predation rates on salmonid smolts by East Sand Island cormorants observed during the pre-management period (see section *4.4.5 Diet Composition* above and section *4.4.6 Salmonid Smolt Consumption* above).

Estimates of predation rates by double-crested cormorants nesting on East Sand Island during the management period were influenced by several factors, some management-related and others independent of management, and comparisons of smolt losses to predation by East Sand Island cormorants between the pre-management period and the management period should be interpreted cautiously. For example, in 2016 and 2017 cormorants dispersed from and temporarily abandoned the East Sand Island colony for extended periods during April and May, which coincided with the peak smolt out-migration period in the estuary for several salmonid ESUs/DPSs (Evans et al. 2019a). Rather than dispersing to locations outside the estuary, large numbers of cormorants (ca. 20,000 individuals) remained in the estuary where they continued to forage on juvenile salmonids and other forage fishes but were no longer depositing smolt PIT tags on the East Sand Island colony (Evans et al. 2019a). As such, measured predation rates by double-crested cormorants from the East Sand Island colony in 2016 and 2017 represent minimum estimates. In 2018, the first year of implementation of Phase II of the *Cormorant Management Plan*, cormorants did not consistently attend the East Sand Island colony until late May, but some cormorants attempted to nest in late April and early May

(Turecek et al. 2019). Thus, cormorants that attempted to nest on East Sand Island early in the 2018 nesting season, but whose nesting attempts failed, were nevertheless depositing some smolt PIT tags, but these adult cormorants may not have been included in the peak 2018 colony count of 3,672 breeding pairs.

At the time reasonable and prudent alternative (RPA) 46 was written as part of the National Marine Fisheries Service's Biological Opinion (NMFS 2008), predation rate estimates for double-crested cormorants nesting on East Sand Island were minimum estimates and there were no estimates of predation rates on salmonid ESUs/DPSs from the Lower Columbia River. The more accurate, standardized estimates of predation rates presented herein indicate that East Sand Island cormorants disproportionately consumed LCR Chinook salmon smolts relative to smolts from other ESA-listed salmonid ESUs/DPSs (see section *4.4.7 Predation Rates on Juvenile Salmonids* above). Sebring et al. (2013) also reported higher rates of predation by East Sand Island cormorants on LCR Chinook salmon relative to other salmon ESUs and hypothesized that lower migration rates and prolonged estuarine residence times increased the susceptibility of LCR Chinook salmon to cormorant predation, particularly Chinook salmon smolts originating from tributaries downstream of Bonneville Dam (see also *Chapter 10*). Diet composition and smolt consumption estimates also indicate that sub-yearling Chinook salmon, which are predominately from the Lower Columbia River ESU, are the most intensely depredated population of juvenile salmonids by double-crested cormorants nesting on East Sand Island (see section *4.4.6 Salmonid Smolt Consumption* above). Lyons et al. (2014) estimated that the LCR Chinook salmon and LCR coho salmon ESUs would receive the greatest benefits from management actions aimed at reducing predation by double-crested cormorants nesting at the East Sand Island colony.

Prior to 2016, most double-crested cormorants nesting in the Columbia River estuary nested on East Sand Island (*Table 4.1* above; Lyons 2010, Turecek et al. 2018). As described above, however, during 2016–2017 cormorants dispersed from the East Sand Island colony site for extended periods during both the cormorant nesting season and the smolt out-migration period. In 2018, the cormorant colony did not form on East Sand Island until late May (Turecek et al. 2019). During 2016–2018, the peak number of nesting pairs increased dramatically at nearby colonies in the Columbia River estuary, including those on the channel markers between Rkm 0 and Rkm 22 and the colony on the Astoria-Megler Bridge (*Table 4.1* above and *Figure 4.1* above). By 2019, the double-crested cormorant colony on the Astoria-Megler Bridge, at 3,542 breeding pairs, was the largest breeding colony for the species in the Columbia River estuary and within the breeding range of the western population (USFWS 2020b). The most recent surveys of the Astoria-Megler Bridge colony of double-crested cormorants, conducted in June 2020, indicated that the colony had grown to over 5,000 breeding pairs, and there remains enough suitable nesting habitat on the Bridge for the colony to more than double in size (Lawonn, In prep.)

Previous research suggests that the impact of piscivorous colonial waterbirds on survival of juvenile salmonids in the Columbia River basin varies greatly depending on where the colony is located and, therefore, where breeding birds forage. For example, Collis et al. (2002) and this study (see *Chapter 5*) found large differences in the proportion of juvenile salmonids consumed by double-crested cormorants nesting on Rice Island in the upper Columbia River estuary compared with those nesting on East Sand Island in the lower estuary. Although the two islands are only 26 km apart, the proportion of juvenile salmonids in the diet of double-crested cormorants nesting on or near Rice Island was two to three times that of double-crested cormorants nesting on East Sand Island. Further, per capita consumption of PIT-tagged smolts by cormorants was about 5-18 times higher for birds nesting on islands in the upper estuary compared with those nesting on East Sand Island (see *Chapter 5*). Based on these results, efforts to prevent a large and sustained double-crested cormorant colony from persisting elsewhere in the estuary, such as on the Astoria-Megler Bridge or further upriver in the upper estuary, will likely be paramount to achieving the smolt survival goals set by the National Marine Fisheries Service as part of RPA 46.

While the *Cormorant Management Plan* was effective at nearly eliminating the double-crested cormorant colony on East Sand Island, the immigration of large numbers of double-crested cormorants, most of which had likely nested previously on East Sand Island, to the large and increasing colony on the Astoria-Megler Bridge has compromised the management goal of reducing cormorant predation on ESA-listed juvenile salmonids throughout the estuary. The weight of evidence suggests that estuary-wide cormorant predation rates have declined little, if at all, for most ESA-listed ESUs/DPSs of salmonids and, as such, the Plan was likely not effective at meeting its objectives for conservation of ESA-listed salmonids.

4.5.5 Adaptive Management Considerations

In our professional opinion, there are several important considerations that can potentially impact future action effectiveness and the necessity for adaptive management in order to fully achieve the objectives of the *Cormorant Management Plan* (USACE 2015). These uncertainties include (1) impacts on smolt survival of the large and growing double-crested cormorant colony (> 5,000 breeding pairs) that has formed on the Astoria-Megler Bridge since implementation of management actions to reduce the size of the cormorant colony on East Sand Island; (2) the continued suitability of East Sand Island as nesting habitat for double-crested cormorants at a reduced colony size; (3) continuing efforts to prevent double-crested cormorants from recolonizing Rice Island or other dredged material disposal islands in the upper Columbia River estuary where cormorants have nested historically; (4) the long-term status and conservation of the western population of double-crested cormorants; and (5) ensuring that predation rates by double-crested cormorants on ESA-listed juvenile salmonids in the Columbia River estuary, particularly on Lower Columbia River Chinook salmon smolts, do not exceed acceptable levels.

4.5.5.1 Manage cormorant colony on the Astoria-Megler Bridge

– At the time of writing (late 2020) the double-crested cormorant breeding colony on the Astoria-Megler Bridge was growing exponentially, had exceeded 5,000 breeding pairs, and was the largest breeding colony for the species in the western United States. We view the large and increasing size of this colony as problematic because there is a strong likelihood that per capita predation rates on juvenile salmonids by double-crested cormorants nesting on the Bridge (Rkm 23) are substantially higher than those by double-crested cormorants nesting on East Sand Island (Rkm 8). Although predation rates on juvenile salmonids by double-crested cormorants nesting on the Bridge have not been measured, per capita predation rates by double-crested cormorants nesting at the former colony on Rice Island (Rkm 34) were significantly greater (roughly 6 times greater) than those of double-crested cormorants nesting at the former colony on East Sand Island (see *Chapter 5*). Fortunately, smolt PIT tags can be recovered from a sub-sample of all cormorants nesting on the Astoria-Megler Bridge by sampling the nesting areas on the bridge pillar 'cribs' at the Astoria end of the Bridge. As part of a pilot study, smolt PIT tags (n = 1,048) were recovered by researchers on the northern crib of the Astoria-Megler Bridge in October of 2020. PIT tag detection probabilities in these areas also need to be measured, however, before estimates of smolt predation rates for the entire Astoria-Megler Bridge colony can be calculated through extrapolation. Field work to estimate predation rates on juvenile salmonids by double-crested cormorants nesting on the Bridge is currently planned for 2021 (G. Smith, Bonneville Power Administration, pers. comm.).

In addition to increased per capita predation rates on salmonids, the large colony of double-crested cormorants on the Astoria-Megler Bridge is also problematic because of damage to the Bridge from accumulated cormorant guano on bridge support structures, causing accelerated corrosion. The weight of evidence suggests that the vast majority of double-crested cormorants that have colonized the Astoria-Megler Bridge in recent years are emigrants from the former East Sand Island colony, where disturbance rates from management actions and bald eagles have been high since 2015. The resighting on the Astoria-Megler Bridge of double-crested cormorants that were banded on East Sand Island and formerly nested there (*Figure 4.12* above and see section *4.4.1.1 Colony size* above), the high connectivity of the East Sand Island cormorant colony with the colony on the Astoria-Megler Bridge (*Table 4.9* above and section *4.5.3 Dispersal & Colony Connectivity* above), and the exponential growth of the Astoria-Megler Bridge colony (*Figure 4.10* above) concurrent with the abandonment and dispersal of cormorants from the East Sand Island colony (*Figure 4.11* above) all suggest that the vast majority of the more than 5,000 breeding pairs of double-crested cormorant nesting on the Astoria-Megler Bridge formerly nested on East Sand Island, where the colony has been almost completely abandoned. It appears likely that the cormorant colony on the Astoria-Megler Bridge will continue to grow, perhaps to the size of the former colony on East Sand Island, and with colony growth would come increasing predation impacts on ESA-listed populations of Columbia Basin salmonids and damage to the Bridge. As of late 2020, there was no federal plan in place to manage the double-crested cormorant colony on the Astoria-Megler Bridge, although a State of Oregon plan to start dissuading cormorants from nesting on the Bridge in 2022 was under development (Lawonn, In prep.).

4.5.5.2 Re-establish & stabilize East Sand Island cormorant colony – The cormorant colony on East Sand Island was mostly or entirely abandoned during the 2020 nesting season, while this document was in preparation. Although about 350 breeding pairs of double-crested cormorants attempted to nest on East Sand Island in 2019, no nesting attempts were apparently successful in raising young, and no sustained nesting activity by double-crested cormorants was reported on East Sand Island in 2020 (M.J. Lawonn, ODFW, pers. comm.). The objective of the *Cormorant Management Plan* was to reduce the size of the East Sand Island colony of double-crested cormorants to no more than 5,380–5,939 breeding pairs and to ensure colony size remained less than 5,380–5,939 breeding pairs in the future. Whereas this objective has been met in recent years, the goal of substantial improvements in survival of out-migrating salmonid smolts has likely not been achieved due to the mass dispersal of double-crested cormorants from East Sand Island to other nesting sites in the upper Columbia River estuary (i.e. the Astoria-Megler Bridge). East Sand Island is the colony site in the Columbia River estuary where the per capita impact of nesting double-crested cormorants on the survival of juvenile salmonids from the Columbia Basin is lowest. The diet of double-crested cormorants nesting on East Sand Island averages 12.5% salmonids (% of total prey biomass), whereas in the upper estuary the diet of nesting cormorants can exceed 50% salmonids (see *Chapter 5*).

Re-establishing a colony of double-crested cormorants on East Sand Island would provide an alternative colony site for cormorants currently nesting on the Astoria-Megler Bridge. Given the strong social attraction of the large cormorant colony currently occupying the Bridge, plus the apparent low rates of disturbance from eagles, gulls, and humans at the Bridge colony, it is unlikely that the East Sand Island cormorant colony can be restored without a concurrent effort to dissuade cormorants from nesting on the Bridge. Devising methods to dissuade cormorants from nesting on the Bridge will pose significant challenges, however, and will likely require pilot studies to test the feasibility of new approaches. But the development of practical and effective methods to deter cormorant nesting on the Astoria-Megler Bridge will provide substantial benefits to both salmonid smolt survival and bridge maintenance over the long term.

While effective cormorant nest dissuasion at the Astoria-Megler Bridge would provide the “push” required to re-establish the East Sand Island colony, re-locating the Bridge colony back to East Sand Island will likely also require a “pull”. The nesting habitat enhancement and social attraction techniques described *above* have been demonstrated effective at restoring abandoned cormorant colonies in the Columbia River estuary even in the absence of a “push” (see section *4.4.9 Pilot Studies of Cormorant Social Attraction & Colony Relocation* above). It appears that the current site of designated cormorant nesting habitat on East Sand Island (the jetty at the western end of the island) may be too susceptible to bald eagle disturbance, especially at much-reduced colony sizes, for cormorants to successfully re-colonize the island. Instead, we suggest that a designated cormorant colony area may need to be provided further east on the island, closer to the center of occupancy for the colony during 2015 and 2016. In our opinion, restoring a colony of double-crested cormorants on East Sand Island that is no more than 5,380–5,939 breeding pairs will likely require (1) selecting a more secure designated colony site on East Sand Island, (2) providing nesting habitat enhancement and social attraction at the new designated site, (3) closely monitoring the designated colony site during the nesting season to identify and manage limiting factors for colony establishment, (4) dissuading cormorants from nesting on the Astoria-Megler Bridge, and (5) convincing skeptical managers and stakeholders that re-establishing a breeding colony of no more than 5,380–5,939 breeding pairs of double-crested cormorants on East Sand Island is an effective management action for reducing estuary-wide predation rates on ESA-listed salmonid smolts.

4.5.5.3 Prevent formation of cormorant colonies in the upper Columbia River estuary – Double-crested cormorants foraging in the Columbia River estuary are apparently preying on juvenile salmonids in proportion to their availability relative to other forage fishes (Lyons et al. 2014), and the availability of marine and estuarine forage fishes is much greater in the lower estuary near East Sand Island compared to the upper estuary near Rice Island. Per capita predation rates on juvenile salmonids by double-crested cormorants nesting on dredged material disposal islands in the upper estuary (i.e. Rice Island and Miller Sands Spit) are approximately 6 times greater than per capita predation rates by cormorants nesting on East Sand Island (see *Chapter 5*). Formation of even a moderate-sized cormorant colony (ca. 1,000 breeding pairs) on Rice Island or other upper estuary dredged material disposal islands could negate most of the benefits achieved by reducing the East Sand Island colony from about 13,500 breeding pairs down to no more than 5,380–5,939 breeding pairs, the objective of the *Cormorant Management Plan*.

At the time of writing and to our knowledge, there is no plan in place to prevent the formation of new breeding colonies of double-crested cormorants on dredged material disposal islands in the upper Columbia River estuary (i.e. Rice Island, Miller Sands Spit, Pillar Rock Sands) over the long-term. Regular surveys of these potential cormorant colony sites early in each breeding season could detect incipient cormorant colonies and, once detected, hazers could be deployed to dissuade nesting cormorants early in the nesting season, when hazing is a highly effective deterrent and before cormorants develop philopatry to the site.

4.5.5.4 Monitor predation rates on juvenile salmonids by cormorants – Robust estimates of cormorant predation rates based on recoveries of smolt PIT tags require adequate sample sizes of tagged fish (Evans et al. 2012, Evans et al. 2014, Evans et al. 2019b). Enough PIT-tagged smolts were available for most, but not all, ESA-listed salmonid populations for inclusion in this study. Notable exceptions include Upper Willamette River steelhead and Columbia River chum salmon, where no estimates of cormorant predation rates were available because no or very few smolts were tagged. Sample sizes of tagged smolts were also limited for several other ESUs/DPSs of Columbia Basin salmonids, including Snake River sockeye salmon, Upper Willamette River Chinook salmon, and Lower Columbia River coho salmon. Adequate sample sizes of tagged wild or naturally produced smolts were also lacking for several salmonid populations and years, tagging that is necessary to achieve a more representative sample of smolts from each ESU/DPS.

Estimates of double-crested cormorant predation rates on Lower Columbia River ESUs/DPSs were based on fish that originated from tributaries located both upstream and downstream of Bonneville Dam (see *Appendix B*). Analogous to estimates of cormorant predation on Upper Columbia River, Snake River, and Middle Columbia River ESUs/DPSs of salmonids, estimates of predation rates on Lower Columbia River ESUs/DPSs account for fish survival to Bonneville Dam and also account for within-season differences in run-timing and the relative availability of tagged fish each week (see *Appendix B*). The predation rate estimates do not, however, account for mortality following release into tributaries to the mainstem Columbia River downstream of

Bonneville Dam, where fish from other ESUs/DPSs are also available as prey. If large numbers of PIT-tagged Lower Columbia River smolts released into tributaries downstream of Bonneville Dam died prior to reaching the Columbia River estuary, predation rate estimates would be biased low and to an unknown degree.

Although the double-crested cormorant breeding colony on the Astoria-Megler Bridge is the largest for the species in the Columbia River basin, or anywhere else in the Pacific Flyway, there are no estimates available for predation rates on juvenile salmonids by cormorants nesting at this colony. Most cormorants nesting on the Astoria-Megler Bridge are nesting over water, so recovery of smolt PIT tags consumed by cormorants nesting on the Bridge is problematic (see section *4.5.5.1 Manage cormorant colony on the Astoria-Megler Bridge* above). Also, while smolt PIT tag detection and deposition probabilities at ground-nesting cormorant colonies in the Columbia River estuary (e.g., the East Sand Island and Rice Island colonies) have been measured, these probabilities are likely very different from those at cormorant colonies over water, like the Astoria-Megler Bridge colony. Predation rate models used in studies summarized herein were based on detection and deposition probabilities measured at ground-nesting cormorant colonies, colonies where tags can readily be detected by researchers after the breeding season. Several cormorant colonies in the Columbia River estuary, including what is currently the largest colony in western North America on the Astoria-Megler Bridge, are not ground-nesting, so robust estimates of cormorant predation rates for cormorants nesting in these colonies are not currently available. Estimates of predation impacts by cormorants nesting at the Astoria-Megler Bridge colony are essential for evaluating the cumulative effects of avian predation on the survival of ESA-listed salmonid ESUs/DPSs in the Columbia River estuary (see *Appendix C*). In the absence of estimates of predation rates by double-crested cormorants nesting on the Astoria-Megler Bridge it is impossible to determine whether the *Cormorant Management Plan* has been effective at reducing losses of ESA-listed juvenile salmonids to cormorant predation in the Columbia River estuary.

4.5.5.5 Monitor conservation status of the western population of cormorants – The estimated size of the western population of double-crested cormorants was 22,890 breeding pairs in 2019 (95% c.i. = 15,925–29,855 pairs; USFWS 2020b), the most recent year when the Flyway Council's monitoring strategy (Pacific Flyway Council 2013) was implemented. This represents a ca. 43% decline (95% c.i. = 10–66%) in the estimated number of breeding pairs in the western population since 2014, just prior to implementation of the *Cormorant Management Plan* (USFWS 2016). Such a large decline in the size of the breeding population in western North America suggests that the near elimination of the East Sand Island colony, where over 40% of the western population formerly nested (Adkins et al. 2014), contributed substantially to this decline. A large decline in the number of breeding pairs in the western population following implementation of the *Cormorant Management Plan* was, however, predicted by the USFWS in the Final Environmental Impact Statement (FEIS) for the *Plan* (USACE 2015). In the FEIS, the USFWS anticipated that by the fifth year of implementing the *Cormorant Management Plan* (2019) the western population of double-crested cormorants would have declined to about 19,182 breeding pairs (USACE 2015; Table 5-4), considerably less than the estimate from the monitoring strategy in 2019 (22,890 breeding pairs; USFWS 2020b). Nevertheless, because many of the breeding adults displaced from the East Sand Island colony during 2017-2019 were likely non-breeders that had not yet recruited as breeders to a new colony when the 2019 monitoring strategy was implemented, the decline in numbers of breeding pairs in the western population during 2014-2019 was not likely a direct reflection of a decline in the number of breeding-age adults in the population.

The USFWS has taken the position that the western population of double-crested cormorants is resilient, and that even if the population was reduced below 20,000 breeding pairs during implementation of the *Cormorant Management Plan,* the population would stabilize and then slowly recover given that the western population has increased from a population size much less than 20,000 breeding pairs in the past (USACE 2015). We conclude that the future trajectory of the western population is highly uncertain, however, based on the apparent complete abandonment of the formerly productive colony on East Sand Island, our status assessment of the western population of double-crested cormorants (Adkins and Roby 2010, Adkins et al. 2014), the recent decline or abandonment of several large, inland colonies in the Pacific Flyway, and a recent new rule from the USFWS allowing the lethal take of up to 4,539 double-crested cormorants from the western population per year (USFWS 2020a), with implementation scheduled to start in 2021.

Growth of the East Sand Island colony during the 1990s and early 2000s was the sole reason that the western population increased during that period; total numbers of breeding double-crested cormorants elsewhere in the Pacific Flyway were stable (Adkins et al. 2014). Not only was the East Sand Island colony by far the largest breeding colony for double-crested cormorants in the Pacific Flyway, it supported consistently high nesting success in a region where nesting failure due to disturbance by bald eagles and humans was increasingly frequent (Carter et al. 1995, Chatwin et al. 2002, Hipfner et al. 2012, Adkins et al. 2014; *Figure 4.24* above). Further, nesting success at the East Sand Island colony may have compensated for the loss in 2013 of the second largest colony of double-crested cormorants in the Pacific Flyway (Mullet Island in Salton Sea, California, due to drought) and declines at other large inland colonies (e.g., Upper Klamath Lake, Oregon, and Potholes Reservoir, Washington; USFWS 2020b).

The single recent development that suggests that the trajectory of the western population of double-crested cormorants could stabilize and then slowly increase, as predicted by the USFWS in the FEIS (USACE 2015), is the exponential increase in colony size on the Astoria-Megler Bridge, upstream of East Sand Island. As of 2020, the cormorant nesting colony on the Astoria-Megler Bridge had supplanted the East Sand Island colony as the largest double-crested cormorant breeding colony in the Pacific Flyway at over 5,000 breeding pairs. Although based on limited nesting success data, it appears that the Astoria-Megler Bridge colony has been a highly productive colony. If left unmanaged, the double-crested cormorant colony on the Bridge is expected to expand in the future and it is estimated that there is enough space to support a colony of as many as 10,000 breeding pairs (M.J. Lawonn, ODFW, pers. comm.) The growth of this highly productive colony, however, is largely an unintended and undesirable consequence of the near total abandonment of the East Sand Island colony. Given the apparent impact of the Astoria-Megler Bridge cormorant colony on survival of ESA-listed salmonid smolts in the Columbia River estuary (see *Chapter 5*) and the impact of guano from thousands of nesting cormorants on the structural integrity of the bridge, it seems very unlikely that the cormorant colony on the bridge will be allowed to persist. Plans are currently being developed to prevent cormorants from nesting on the Astoria-Megler Bridge starting in the 2022 nesting season. Therefore, the future trajectory of the western population of double-crested cormorants largely depends on whether management actions to reduce or eliminate the Bridge colony are coupled with management efforts to restore a colony of no more than 5,380–5,939 breeding pairs on East Sand Island.

Lastly, we believe that the recent new rule for management of double-crested cormorants at the federal level (USFWS 2020a) further enhances the uncertainty over the future population trajectory of the western population. The rule has established a new system for issuing special permits to States and Tribes for lethal take of double-crested cormorants and applies not just to regions of the U.S. formerly included under the Public Resource Depredation Order for double-crested cormorants (50 CFR 21.48; USFWS 2003), but throughout the contiguous 48 states (USFWS 2020a). The rule sets a maximum allowable take of 4,539 double-crested cormorants per year within the Pacific Flyway (USFWS 2020a), or about 10% of the ca. 45,780 breeding individuals enumerated during the most recent count of the western population based on the Pacific Flyway Monitoring Strategy (Pacific Flyway Council 2013, USFWS 2020b). While it is not clear what level of annual take will occur in the Pacific Flyway under the new rule, if the take level approaches the maximum allowable take the stabilization and recovery of the western population in the aftermath of the *Cormorant Management Plan* is unlikely, especially once the Astoria-Megler Bridge colony is dispersed and if the colony on East Sand Island is not restored.

For all the above reasons, we believe that it is important to sustain the U.S. Fish and Wildlife Service's Double-crested Cormorant Monitoring Strategy for the Pacific Flyway (Pacific Flyway Council 2013) in order to ensure that the western population does not decline to a level where it becomes of conservation concern. The monitoring strategy, which is based on a dual frame sampling approach, is currently conducted every three years and has as its objective to detect a 5% change/year in the size of the western population of double-crested cormorants with 80% power and 10% Type I error rate (Pacific Flyway Council 2013). Because of the transitory nature of many double-crested cormorant breeding colonies and the high inter-annual variability in colony size among extant colonies, the Cormorant Monitoring Strategy has struggled to achieve its monitoring objectives (USFWS 2020b). We believe that the dual frame sampling approach for monitoring the western population of double-crested cormorants can achieve its objectives for accuracy and precision if the strategy is modified and updated after each implementation and if surveys are conducted throughout the Flyway to identify new and incipient colonies as they form. Periodically, perhaps every 10-12 years, a Flyway-wide census of double-crested cormorant breeding colonies may be required to ensure that the dual frame sampling approach is achieving its objectives.

4.6 Conclusions

1. The double-crested cormorant breeding colony on East Sand Island in the Columbia River estuary was a small colony of 91 breeding pairs in 1989 but rapidly increased to a colony of over 13,000 breeding pairs by 2006 and became the largest known breeding colony for the species in North America.

2. The rapid increase in size of the East Sand Island colony of double-crested cormorants between 1989 and 1991 apparently resulted from immigration from outside the Columbia River estuary, mostly from colonies in coastal Washington and British Columbia. The colony continued to grow during 1992–2006, but the relative contribution of immigration and intrinsic recruitment to colony growth is unknown. The large size of the colony provided safety in numbers from bald eagle predation, the location of East Sand Island provided access to an abundant supply of marine and estuarine forage fish, and the island's insularity provided security from terrestrial predators and human disturbance.

3. Juvenile salmonids, including smolts from 11 of the 13 ESUs/DPSs of Columbia Basin salmonids listed under the U.S. Endangered Species Act, averaged 12.5% of the diet of double-crested cormorants nesting on East Sand Island. Estimated annual losses of juvenile salmonids to cormorant predation in the Columbia River estuary frequently exceeded 10 million smolts and approached 20 million smolts (mostly Lower Columbia River Chinook salmon) in some years prior to management due to the large size and high energy requirements of the East Sand Island colony. These smolt losses likely represented 5% to 15% of all smolts that survived to the Columbia River estuary.

4. Predation rates on juvenile salmonids (proportion of available smolts consumed) by double-crested cormorants nesting at the East Sand Island colony generally exceeded 5% for steelhead populations, 15% for Lower Columbia River coho salmon, and were as high as 30% for Lower Columbia River Chinook salmon; predation rates on smolts from other ESUs/DPSs of Columbia Basin salmonids were generally less than 5%.

5. By 2013 the East Sand Island colony of double-crested cormorants consisted of nearly 15,000 breeding pairs, which accounted for over 40% of the total number of breeding adults in the western population of the species. Colony size was not limited by available nesting habitat on East Sand Island and double-crested cormorant nesting success averaged more than two young raised per breeding pair. Instead, colony size was apparently limited by the synergistic effects of high rates of bald eagle disturbance and low availability of marine forage fish early in the nesting season.

6. The objective of the *Cormorant Management Plan* to limit predation on smolts by reducing the size of the double-crested cormorant colony on East Sand Island to no more than 5,380–5,939 breeding pairs during Phase I of the Plan was achieved earlier than expected because the cormorant colony was repeatedly abandoned beginning in May of 2016, apparently due to the synergistic effects of management-related human disturbance and eagle disturbance. During the 2020 nesting season, the second nesting season following completion of the habitat modification prescribed in Phase II of the Plan, there was no sustained nesting activity at the cormorant colony on East Sand Island.

7. Reductions in the size of the cormorant colony on East Sand Island due to management resulted in unknown changes in predation rates on ESA-listed salmonids in the Columbia River estuary because many of the double-crested cormorants that formerly nested on East Sand Island immigrated to the colony on the Astoria-Megler Bridge, where predation rates on juvenile salmonids have not been measured, but where per capita predation rates on smolts are likely greater than for cormorants nesting on East Sand Island.

8. The legacy of the *Cormorant Management Plan* remains controversial. Some managers and stakeholders point to the apparent elimination of the cormorant colony on East Sand Island, accomplished without driving the number of breeding pairs of double-crested cormorants in the western population below levels predicted in the Plan, and the resultant reduction in predation on juvenile salmonids in the estuary as indicative of management success. Conversely, other managers and stakeholders remain concerned that cormorant predation rates on juvenile salmonids in the estuary have not declined because of the exponential increase in the size of the double-crested cormorant colony on the Astoria-Megler Bridge. Resolution of this controversy will require continued research, monitoring, and evaluation of outcomes from the *Cormorant Management Plan*, as well as adaptive management to ensure that outcomes are consistent with the fisheries conservation goals of the *Plan*.

9. For managers to be confident that the *Cormorant Management Plan* has achieved its goal of reducing cormorant predation on juvenile salmonids in the Columbia River estuary to “base period” levels, predation by cormorants nesting at the large and growing colony on the Astoria-Megler Bridge, which is currently unmanaged, must be drastically reduced or eliminated. Such management will be most effective if efforts to dissuade cormorant nesting at the Astoria-Megler Bridge are paired with efforts to attract cormorants to suitable colony sites where predation impacts will be lower, most notably the former colony site on East Sand Island. Because the Columbia River estuary is a highly attractive location for cormorant breeding, it is possible, or even likely, that annual management to dissuade cormorants from nesting at undesired sites will be

necessary, especially if cormorants are unable to successfully recolonize East Sand Island. This will require additional collaboration and support from Federal, State, and Tribal natural resource management agencies.

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CHAPTER 5: Relative Impacts of Double-crested Cormorants and Caspian Terns on Survival of Juvenile Salmonids in the Columbia River Estuary: A Retrospective Analysis

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5.1 Summary

To address concerns about avian predation on juvenile salmonids (*Oncorhynchus* spp.) in the Columbia River estuary (CRE), management plans were implemented to reduce impacts on smolt survival by reducing the number of double-crested cormorants (*Phalacrocorax auritus*) and Caspian terns (*Hydroprogne caspia*) nesting at specific colonies and relocating colonies to alternative sites within the estuary. Multiple colony sites for these two species exist in the CRE, both active and historical, where birds from managed colonies might relocate to nest, and where impacts on survival of juvenile salmonids may vary. We evaluated the relative susceptibility of passive integrated transponder (PIT)-tagged juvenile salmonids (smolts) to predation by cormorants and terns nesting at spatially explicit locations in the marine and freshwater zones of the CRE. Colony sites were located on East Sand Island (river km [Rkm] 8), in the marine zone, and on Rice Island (Rkm 34) and Miller Sands Spit (Rkm 38) in the freshwater zone of the CRE. A historical dataset of colony size (number of breeding pairs) and smolt PIT tag recoveries on cormorant and tern colonies was used to evaluate the per capita (per breeding pair) impacts of these two species nesting at various locations in the CRE on survival of juvenile salmonids during 1999–2007. Results indicated that per capita predation probabilities on tagged smolts were significantly higher for birds nesting in the freshwater zone relative to those nesting in the marine zone of the CRE. Spatial differences in relative susceptibility to predation were consistently observed across all salmonid species (i.e. Chinook salmon [*O. tshawytscha*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*], and steelhead trout [*O. mykiss*]) and years evaluated. Estimates of average per capita predation probabilities were 5.1 (95% credible interval = 2.9–7.9) to 17.9 (9.0–28.2) times higher, depending on the salmonid species, for cormorants nesting in the freshwater zone compared to those nesting in the marine zone. Estimates of average per capita predation probabilities were 2.3 (1.6–3.2) to 2.9 (2.0–4.1) times higher, depending on the salmonid species, for terns nesting in the freshwater zone compared to those nesting in the marine zone, such as on East Sand Island. Results from this study indicate that differences in the diet composition of piscivorous waterbirds nesting at different locations in the CRE were considerable and that impacts of predation on survival of salmonid smolts are significantly higher for cormorants and terns nesting in the freshwater zone compared to the marine zone of the CRE. These results have important implications for the management of cormorants and terns in the CRE to reduce predation rates on salmonid smolts listed under the U.S. Endangered Species Act.

5.2 Introduction

The Columbia River estuary (CRE) has supported exceptionally large breeding colonies of double-crested cormorants (*Phalacrocorax auritus*) and Caspian terns (*Hydroprogne caspia*), with colony sizes exceeding 10,000 breeding pairs in some years (Lyons 2010, Adkins et al. 2014). Predation impacts on juvenile salmonids (*Oncorhynchus* spp.) by double-crested cormorants (hereafter “cormorants”) and Caspian terns (hereafter “terns”) nesting in the estuary have been substantial, with millions of smolts consumed by terns and cormorants annually (Roby et al. 2003, Lyons 2010). Concerns over the impacts of avian predation on survival of juvenile salmonids listed under the U.S. Endangered Species Act (ESA) have prompted the development and implementation of management plans to reduce the size of large cormorant and tern colonies in the CRE (USFWS 2005, USFWS 2006, USACE 2015). These plans are only successful if other active and historical colonies within the CRE do not increase in size following implementation of management, thereby partially or entirely offsetting the reduction in impacts to smolts from managed colonies.

Previous research indicates that predation rates on juvenile salmonids by cormorants and terns nesting in the CRE can vary greatly based on where these two species nest and forage within the CRE. Collis et al. (2002) observed that juvenile salmonids (i.e. Chinook salmon [*O. tshawytscha*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*], and steelhead trout [*O. mykiss*]) accounted for an average of 46% of the diet (percent of prey biomass) of cormorants nesting on Rice Island in the CRE during 1997–1998. Rice Island is located in the freshwater zone of CRE at river km (Rkm) 34, as defined by distance to the Pacific Ocean. By comparison, juvenile salmonids accounted for just 15% of the diet (percent of prey biomass) for cormorants nesting on East Sand Island (Rkm 8) in the marine zone of the CRE (Collis et al. 2002) during the same two years. These site-specific differences in cormorant diet composition suggested that the location of colonies of piscivorous waterbirds in the CRE could have a strong influence on diet composition.

Based on the results from this study, resource managers decided to attempt to relocate a colony of about 9,000 breeding pairs of Caspian terns from Rice Island to East Sand Island based on the hypothesis that terns nesting on East Sand Island, like cormorants, would be less reliant on juvenile salmonids as prey. Management efforts to relocate the Rice Island tern colony to East Sand Island during 1999–2001 were successful (Roby et al. 2002) and, as hypothesized, the proportion of salmonids in the diet of terns nesting on East Sand Island was significantly lower than that observed on Rice Island. Juvenile salmonids accounted for an average of 77% of the diet (percent of prey items) of terns nesting on Rice Island during 1997–1998, whereas juvenile salmonids accounted for 42% of the diet (percent of prey items) of terns nesting on East Sand Island in the marine zone of the estuary during 1999–2001 (Roby et al. 2002).

Following management to relocate the tern colony from Rice Island to East Sand Island, colony size for both terns and cormorants on East Sand Island continued to grow; over 10,000 breeding pairs of terns and nearly 15,000 breeding pairs of cormorants nested on East Sand Island during years of maximum colony size (Lyons 2010; Adkins et al. 2014). In response to increases in cormorant and tern colony sizes and continued concern over predation rates on juvenile salmonids by birds from these two colonies, resource managers implemented additional management actions to reduce the numbers of cormorants and terns nesting on East Sand Island. Management to reduce the size of the tern colony on East Sand Island by about 60% was implemented starting in 2008 (USFWS 2005, 2006) and management to reduce the size of the cormorant colony on East Sand Island by at least 60% was implemented starting in 2015 (USACE 2015).

During 2001–2014, nearly all nesting by cormorants and terns in the CRE occurred on East Sand Island. Over the past three decades, however, both species have nested at other locations in the CRE. For example, in addition to Rice Island, cormorants have nested on Miller Sands Spit (Rkm 38), on a number of navigational aids (Rkm 5–225; Carter et al. 1995, AQEA 2017), and, more recently, on the Astoria-Megler Bridge (Rkm 21; see *Chapter 4*). Although suitable nesting habitat is less abundant in the CRE for Caspian terns compared to double-crested cormorants, they too have nested or attempted to nest on both Rice Island and Miller Sands Spit (Shuford and Craig 2002, Harper and Collis 2018). Given the long lifespan and philopatry to historical colony sites demonstrated by both species (Courtot et al. 2012, Peck-Richardson 2017, Suzuki et al. 2018), the potential for managed birds displaced from East Sand Island to relocate to other colonies (extant or historical) in the CRE, where predation rates on juvenile salmonids may be as great or greater, is an ongoing concern for resource managers (Collis et al. 2002, Roby et al. 2002, Lyons 2010).

Passive integrated transponder (PIT) tags inserted into out-migrating juvenile salmonids and subsequently recovered on piscivorous waterbird colonies have been one of the primary tools used to assess avian predation rates in the CRE (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Sebring et al. 2013, Evans et al. 2019). PIT tags recovered from colonies provide specific information on individual fish (e.g., species, population, and release location) and have been used extensively to estimate predation rates (percentage of available fish consumed by birds). Methods used to estimate predation rates have improved incrementally over the years and now include corrections for PIT tags consumed by nesting birds that were not deposited on breeding colonies (deposition probability), and for those tags deposited on-colony but not detected by researchers following the breeding season (detection probability; Hostetter et al. 2015). For instance, although PIT tag recovery in the CRE has occurred annually since 1996, estimates generated prior to 2014 were reported as minimum estimates of predation because they were not corrected for both deposition probabilities and detection probabilities (Hostetter et al. 2015).

To better understand the relative impacts on smolt survival of predation by cormorants and terns nesting at different locations within the CRE, we re-analyzed the historical PIT tag dataset using modern methods, data that were collected in years when cormorant and tern colonies were concurrently nesting at sites located within both the marine zone (East Sand Island) and the freshwater zone (Rice Island, Miller Sands Island) of the CRE. We estimated annual per capita (per breeding pair) consumption of PIT-tagged salmonids for cormorants and terns nesting in the freshwater zone of the CRE and compared results to cormorants and terns nesting in the marine zone of the CRE. These results may assist in evaluating the efficacy of ongoing management to reduce avian predation in the CRE to acceptable levels and may help to develop adaptive management actions to maximize the benefits to survival of ESA-listed juvenile salmonids.

5.3 Methods

5.3.1 Study Area

Predation rates on PIT-tagged juvenile salmonids were evaluated for cormorants and terns nesting on three different islands located in the CRE: (1) East Sand Island, (2) Rice Island, and (3) Miller Sands Spit (*Figure 5.1*). Both terns and cormorants nested on East Sand Island and Rice Island, while only cormorants nested on Miller Sands Spit during the study period. Rice Island and Miller Sands Spit are within the freshwater zone of the CRE (upstream of Rkm 29), while East Sand Island is within the marine zone of the CRE (Rkm 0–12; Simenstad et al. 1990).

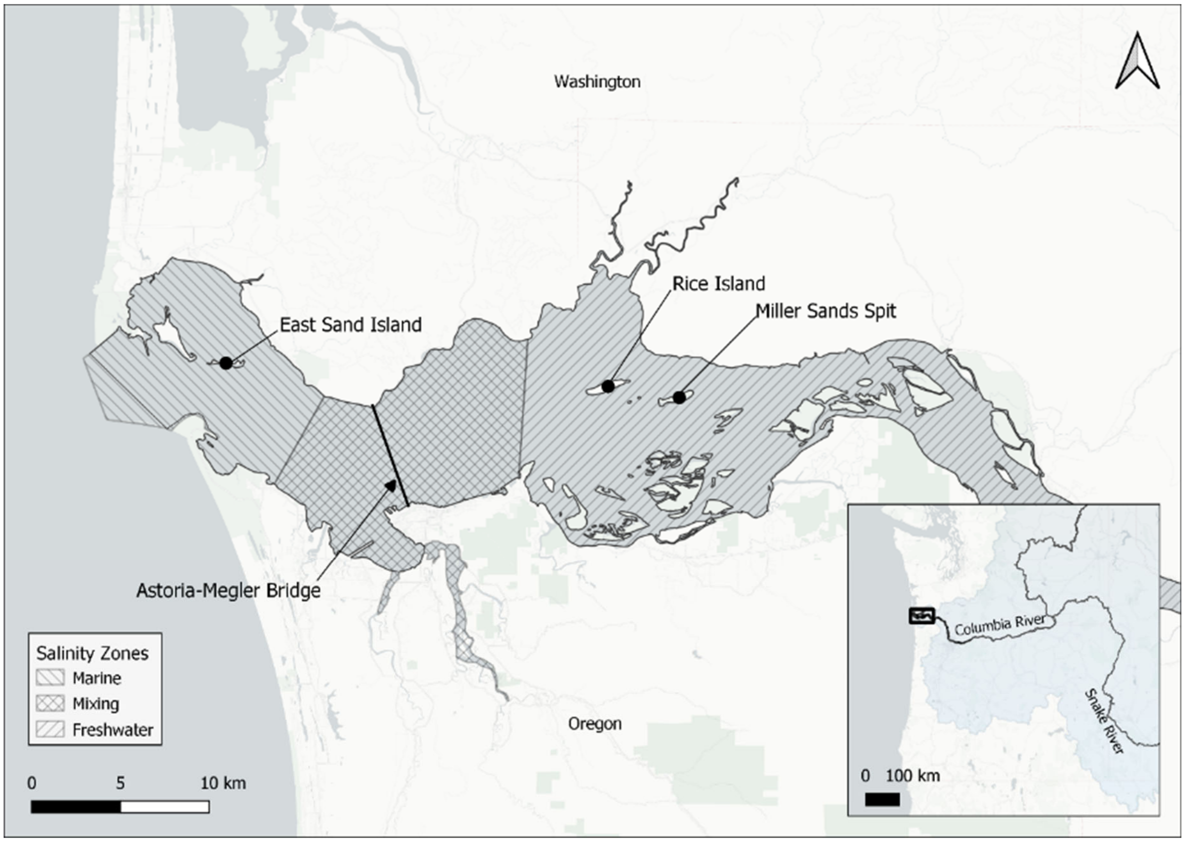

Figure 5.1. Locations of double-crested cormorant and Caspian tern colonies referred to in this study and salinity zones within the Columbia River estuary (derived from Simenstad et al. [1990], as modified by Anderson et al. [2004]).

5.3.2 Colony Size

Estimates of concurrent colony size (i.e. number of breeding pairs) for cormorants and terns when they nested in both the marine and freshwater zones of the CRE were available in 2001, 2006, and 2007 for cormorants and in 1999–2000 for terns. The numbers of cormorant and tern breeding pairs were estimated via land-, boat-, or aerial-based surveys, depending on colony size. The size of large colonies (those greater than 1,000 breeding pairs) was estimated using photographs taken during aerial-based surveys late in the incubation period, when maximum colony attendance of breeding birds was assumed (Bullock and Gomersal 1981, Gaston and Smith 1984). Colony size estimates from photography were either direct counts of all incubating birds (or active nests) or direct counts of all adults on-colony corrected using simultaneous ground counts of incubating and non-incubating birds in plots located within the colony. An average of two direct counts from aerial photography was used to estimate colony size (see Collis et al. 2002 for further details of aerial photography census methods). Smaller colonies were estimated using ground- or boat-based counts of attended nests during late incubation.

5.3.3 Smolt Availability

To maximize the sample size of PIT-tagged smolts available as prey to cormorants and terns nesting in the CRE that was used in analyses of predation rates (see *below*), fish were grouped by salmonid species (i.e. Chinook salmon, sockeye salmon, coho salmon, steelhead trout) and migration year. To ensure that fish were available to cormorants and terns nesting at all colonies included in the study, only fish released upstream of Rkm 34 were considered available as prey to birds nesting at all the islands included in this study. Data regarding each PIT-tagged fish were retrieved from the PIT Tag Information System (PTAGIS), a regional database maintained by the Pacific States Marine Fisheries Commission (PSMFC 2019).

5.3.4 PIT Tag Recovery & Analysis

Two techniques were used to recover (i.e. electronically detect) PIT tags *in situ* on cormorant and tern colonies in the CRE following each breeding season (September–October): (1) a vehicle-operated, flat-plate detection system and (2) multiple hand-held, pole-mounted detection systems. Beginning in 1998, researchers with the National Marine Fisheries Service deployed a flat-plate PIT tag antenna system that was towed by a four-wheel-drive vehicle (Ledgerwood et al. 1997, Ryan et al. 2001). This system was used to detect PIT tags on bare sand habitat used by terns nesting at East Sand Island and Rice Island during 1999–2000, with at least three passes (complete sweeps) of each colony area conducted in each year (Ryan et al. 2001, Ryan et al. 2003). For cormorant colonies on Miller Sands Spit, Rice Island, and East Sand Island, hand-held, pole-mounted detection systems were used to recover tags from stick nests and the surrounding colony area (Ryan et al. 2001, Glabek et al. 2003, Evans et al. 2012), with at least one complete pass of each area of the colony conducted in each year (i.e. 2001, 2006, and 2007).

Not all PIT tags consumed by piscivorous colonial waterbirds are deposited by birds on their nesting sites, nor are all deposited tags recovered by researchers following the breeding season (Evans et al. 2012, Osterback et al. 2013, Hostetter et al. 2015). Instead, a portion of consumed tags are either damaged following consumption or are deposited at off-colony locations. Deposition probabilities (i.e. the probability that a tag consumed by a nesting bird is deposited on its breeding colony) were estimated using the methods and results of Hostetter et al. (2015). In brief, PIT-tagged fish were fed to cormorants and terns nesting on East Sand Island and the proportion subsequently recovered on the respective colony was used to estimate deposition probabilities (see Hostetter et al. 2015 for details). The median deposition probabilities derived from these studies were 0.51 (95% credible interval = 0.34–0.70) for cormorants and 0.71 (0.51–0.89) for terns. Deposition probability priors informed by these experiments were used to infer deposition probabilities for data collected in all years of this study, whereby we assumed predator-specific deposition probability, δ_h, for a colony h to be

$$\delta_h \sim Beta(a_h, b_h).$$

Not all PIT tags deposited on breeding colonies are subsequently found by researchers after the nesting season (Evans et al. 2012, Frechette et al. 2012, Hostetter et al. 2015, Payton et al. 2019). PIT tags can be transported off-colony during storm events, damaged during the breeding season, or simply missed (not detected) by researchers during colony scans. Detection probabilities (i.e. the probability that a tag deposited on a colony is detected by researchers after the nesting season) often vary within and between breeding seasons (Hostetter et al. 2015, Payton et al. 2019, Evans et al. 2019). When possible, direct measures of detection probabilities were determined by sowing PIT tags with known codes onto the colonies included in this study prior to and immediately following the breeding season. During each PIT tag sowing period, control tags were haphazardly sown throughout the area occupied by breeding birds (Evans et al. 2012). Detections (i.e. recoveries) of control tags following the breeding season were then used to model the probability of detecting tags deposited throughout the breeding season via logistic regression (see *below*). Explicitly, we let $N_{h,w}$ be the number of intentionally sown tags on colony h in week w, $X_{h,w}$ be the proportion of $N_{h,w}$ subsequently recovered following the breeding/migration season, and assuming

$$X_{h,w} \sim \text{Binomial}\left(N_{h,w}, \psi_h(w)\right) \text{ where } \psi_h(w) = \text{logit}^{-1}(\alpha_h + \beta_h).$$

Detection probabilities based on empirical data were available for some, but not all, study years (*Table 5.1*). To address this, we used the methods of Payton et al. (2019) to incorporate information from other migration years to inform estimates of $\psi_h(w)$ for years when empirical data were lacking. We assumed the parameters defining the logistic relationship outlined above vary among years according to a multivariate-normal distribution. That is, we let

$$\begin{bmatrix} \alpha_h^y \\ \beta_h^y \end{bmatrix} \sim \text{Normal}\left(\begin{bmatrix} \mu_{\alpha_h} \\ \mu_{\beta_h} \end{bmatrix}, \mathbf{\Sigma}_\text{h}\right).$$

where $\mathbf{\Sigma}_{\mathrm{h}}$ is the covariance matrix among years. This sharing of information among years allows for unbiased estimates while recognizing the inherent uncertainty involved in the prior distributions. The colonies and years which necessitated this informed prior approach are listed in *Table 5.1* along with the years and colonies used to inform them.

Table 5.1. Colonies and years where direct measures of detection probability were insufficient, and the colony (years) used to inform prior detection probability distributions.

Colony	Years	Informed by Colony (Years)
Double-crested cormorants		
Rice Island	2001, 2006, 2007	Miller Sands Spit (2006–2007)
Caspian terns		
Rice Island	1999 - 2000	East Sand Island (2000–2004)
East Sand Island	1999 - 2000	East Sand Island (2000–2004)

PIT tag recoveries were corrected for both deposition and detection probabilities (collectively referred to as “recovery probabilities”) to estimate per capita (per breeding pairs) consumption rates for each predator species (i.e. cormorants, terns), each salmonid species (i.e. Chinook salmon, coho salmon, sockeye salmon, steelhead trout), each colony (i.e. East Sand Island, Rice Island, Miller Sands Spit), and each year. Deposition probabilities were assumed to be constant across years (Hostetter et al. 2015). Detection probabilities were averaged across all weeks during which smolts were assumed to be available (April–August). Thus, consumption for colony h, C_h, was calculated as the number of tags recovered, R_h, corrected for deposition, δ_h, and average detection, $\bar{\psi}_h$, such that

$$C_h = \frac{R_h}{\overline{\psi_h} * \delta_h}.$$

This estimate of consumption was then divided by the estimated colony size (i.e. number of breeding pairs) at colony h to produce per capita consumption estimates. Per capita consumption estimates for each colony were assumed to be mutually independent, allowing for assessments of differences in per capita estimates to be inferred directly from calculated differences in random draws from the respective posterior distributions. Because colony monitoring effort varied by site and year, variation in intra-annual colony size (i.e. colony attendance) could not be fully assessed. Anomalous attendance patterns (e.g., colony abandonment, delayed nesting) could bias relative measures of per capita consumption rates based on peak colony size. Except for the delayed nesting by cormorants on Miller Sands Spit in 2006 (see *Results*), however, there was no evidence to suggest significant differences in nesting chronology and colony attendance patterns occurred during this study, so we assumed that nesting chronology was similar for colonies in both the freshwater and marine zones of the CRE.

Relative differences in per capita consumption of PIT-tagged smolts were determined by dividing random draws from the posterior distribution of per capita consumption estimates for a freshwater zone colony (i.e. Rice Island or Miller Sands Spit) by draws from the corresponding posterior distribution of consumption estimates from the marine zone colony (i.e. East Sand Island) in the same year. Ratios between the two draws that were greater than 1.0 were considered statistically significant.

5.4 Results

5.4.1 Colony Size

Most cormorants in the CRE nested on East Sand Island during the study period (i.e. 2001, 2006, and 2007), with annual colony size estimates ranging from 8,120 to 13,738 breeding pairs (*Table 5.2*). During this same study period, comparatively small numbers of cormorants nested on Rice Island and Miller Sands Spit, with annual colony size estimates ranging from 35 to 150 breeding pairs (*Table 5.1* above). All terns in the CRE nested on either Rice Island or East Sand Island during the study period (i.e. 1999 and 2000); during 1999 8,096 breeding pairs of terns nested on Rice Island and 1,400 breeding pairs nested on East Sand Island, whereas during 2000 8,513 breeding pairs nested on East Sand Island and 588 breeding pairs nested on Rice Island (*Table 5.2*). The change in the nesting distribution of terns in the CRE during 1999–2000 was due to active management to relocate the tern colony from Rice Island to East Sand Island (see *above*).

Table 5.2. Estimated number of breeding pairs for double-crested cormorants and Caspian terns nesting at islands located in the Columbia River estuary. "NC" denotes no colony established in that year.

Year	East Sand Island	Rice Island	Miller Sands Spit
Double-crested cormorants			
2001	8,120	150	12[1]
2006	13,738	35	41
2007	13,771	NC	90
Caspian terns			
1999	1,400	8,096	NC
2000	8,513	588	NC

[1] *No PIT tag recovery data.*

5.4.2 PIT Tag Recovery

More than 21,000 smolt PIT tags were recovered on cormorant colonies in the CRE during the study period (*Table 5.3*). In all three study years, most PIT tags were recovered on the East Sand Island cormorant colony due to the much larger size of that colony compared to the colonies on Rice Island and Miller Sands Spit (*Table 5.2* above). For all cormorant colonies and all years, PIT tags from Chinook salmon smolts were the most numerous, followed by steelhead, coho salmon, and sockeye salmon. This result suggests that cormorants nesting in the CRE consumed PIT-tagged juvenile salmonids in proportion to their relative availability, because Chinook salmon represent more than 60% of all PIT-tagged smolts released into the Columbia River basin annually (PTAGIS 2019).

More than 66,000 smolt PIT tags were recovered on tern colonies in the CRE during the study period (*Table 5.3*). In 1999, the majority (91%) of smolt PIT tags were recovered on Rice Island (*Table 5.3*). Conversely, more than 82% of smolt PIT tags were recovered on East Sand Island during 2000 due to efforts to relocate the colony from Rice Island to East Sand Island. In contrast to cormorants, terns disproportionately consumed steelhead smolts, with similar numbers of tags recovered on-colony from Chinook salmon and steelhead (*Table 5.3*), even though many more Chinook salmon were PIT-tagged relative to steelhead during both study years (PTAGIS 2019).

Table 5.3. Number of smolt PIT tags recovered by salmonid species on double-crested cormorant and Caspian tern colonies in the Columbia River estuary. Only PIT-tagged smolts released upstream of river km (Rkm) 34 (Rice Island) that were recovered during the same year are included. "NC" denotes no colony established or no PIT tag recovery effort in that year.

Year	Species	East Sand Island	Rice Island	Miller Sands Spit
Double-crested cormorants				
2001	Chinook	3,336	244	NC
	Sockeye	7	0	
	Coho	14	2	
	Steelhead	260	31	
2006	Chinook	7,828	120	64
	Sockeye	56	1	2
	Coho	277	5	5
	Steelhead	4,083	79	43
2007	Chinook	3,187	NC	221
	Sockeye	36		6
	Coho	116		8
	Steelhead	1,112		127
Caspian terns				
1999	Chinook	1,801	12,796	NC
	Sockeye	5	53	
	Coho	147	1,374	
	Steelhead	1,328	19,259	
2000	Chinook	11,789	2,862	NC
	Sockeye	46	10	
	Coho	2,093	384	
	Steelhead	10,118	2,136	

PIT tag recovery probabilities for double-crested cormorant colonies were relatively low, ranging from 0.28 (0.14–0.42) to 0.36 (0.32–0.40; *Table 5.4*). For Caspian terns, recovery probabilities were consistently higher, ranging from 0.64 (0.43–0.85) to 0.69 (0.50–0.84). Higher PIT tag recovery probabilities observed on tern colonies relative to cormorant colonies were due to higher deposition and detection probabilities associated with tern colonies (see *above*).

Table 5.4. Average annual recovery probabilities (95% credible intervals) of smolt PIT tags on double-crested cormorant and Caspian tern colonies in the Columbia River estuary. "NC" denotes no colony established or no PIT tag recovery in that year. Detection and deposition probabilities were those previously reported by Evans et al. (2012), Sebring et al. (2013), Hostetter et al. (2015), Evans et al. (2016), and Evans et al. (2019). Recovery probability is shown as the deposition probability multiplied by the annual weighted average of detection probability.

Year	East Sand Island	Rice Island	Miller Sands Spit
Double-crested cormorants			
2001	0.34 (0.22–0.48) [1]	0.32 (0.18–0.47) [1]	NC
2006	0.31 (0.19–0.44)	0.32 (0.17–0.46) [1]	0.36 (0.32–0.40) [1]
2007	0.28 (0.14–0.42)	NC	0.32 (0.26–0.37)
Caspian terns			
1999	0.65 (0.42–0.84) [1]	0.64 (0.43–0.85) [1]	NC
2000	0.69 (0.50–0.84)	0.65 (0.45–0.85) [1]	NC

[1] *Variation in detection probability partially inferred from other years (see also Payton et al. 2019).*

5.4.3 Per Capita Smolt Consumption

For all study years and all salmonid species evaluated, per capita consumption of PIT-tagged smolts was greater for cormorants nesting at colonies in the freshwater zone of the CRE (i.e. Rice Island and Miller Sands Spit) than for those nesting at the colony in the marine zone (i.e. East Sand Island; *Figure 5.2*, *Table 5.5*). Per capita consumption of PIT-tagged smolts varied by year and colony location, with median estimates of consumption ranging from less than 0.1 to 1.9 PIT-tagged smolts consumed per breeding pair for cormorants nesting on East Sand Island and from less than 0.1 to 10.7 for those nesting on Rice Island and Miller Sand Spit. For all three estuary cormorant colonies, annual estimates of per capita PIT tag consumption probabilities were greatest for Chinook salmon (*Figure 5.2*, *Table 5.5*). Recoveries of PIT tags from coho and sockeye salmon were sparse, particularly at the smaller Rice Island and Miller Sands Spit colonies. Despite small sample sizes, however, estimates of per capita consumption rates at colonies in the freshwater zone were significantly greater than those observed on East Sand Island in the marine zone of the CRE (*Figure 5.2*).

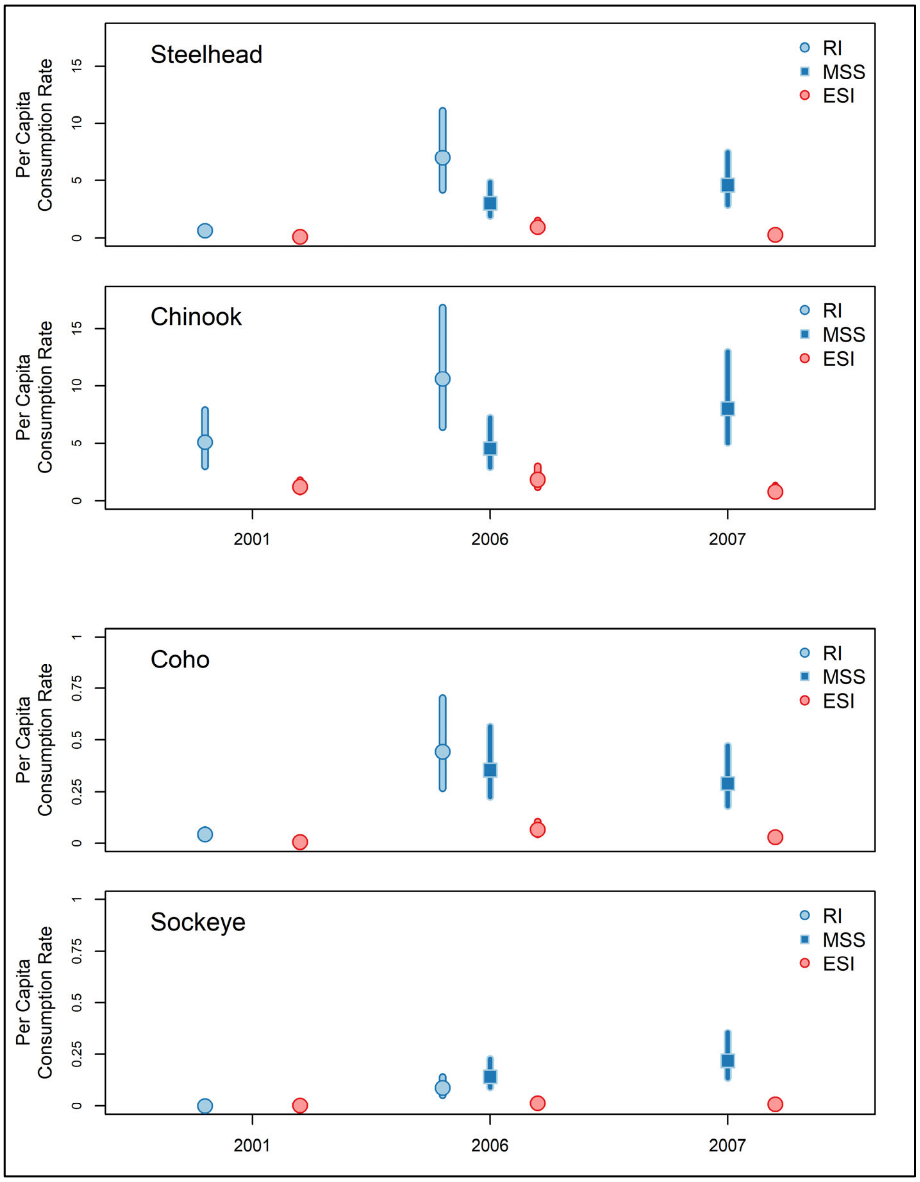

Figure 5.2. Estimated per capita (per breeding pair) consumption rate (95% credible intervals) by salmonid species for double-crested cormorants nesting at colonies on East Sand Island (ESI), Rice Island (RI), and Miller Sands Spit (MSS). See Table 5.5 for the data used in these figures.

Table 5.5. Estimated per capita (per breeding pair) consumption rates of PIT-tagged juvenile salmonids (95% credible intervals) for double-crested cormorants and Caspian terns nesting at colonies in the Columbia River estuary during years when PIT tags were recovered from colonies in both the marine zone (East Sand Island) and freshwater zone (Rice Island, Miller Sands Spit)of the estuary. "NC" denotes no colony was established or no PIT tag recovery effort occurred in that year.

Year	Species	East Sand Island	Rice Island	Miller Sands Spit
Double-crested cormorants				
2001	Chinook	1.21 (0.77–1.75)	5.09 (3.19–8.00)	NC
	Sockeye	<0.01	<0.01	
	Coho	0.01 (<0.01–0.01)	0.04 (0.03–0.07)	
	Steelhead	0.09 (0.06–0.14)	0.65 (0.41–1.02)	
2006	Chinook	1.85 (1.24–2.88)	10.68 (6.91–17.90)	4.60 (2.97–7.37)
	Sockeye	0.01 (<0.01–0.02)	0.09 (0.06–0.15)	0.14 (0.09–0.23)
	Coho	0.07 (0.04–0.10)	0.45 (0.29–0.75)	0.36 (0.23–0.58)
	Steelhead	0.96 (0.65–1.50)	7.03 (4.55–11.79)	3.09 (1.99–4.96)
2007	Chinook	0.83 (0.50–1.35)	NC	7.97 (4.93–13.40)
	Sockeye	0.01 (0.01–0.02)		0.22 (0.13–0.36)
	Coho	0.03 (0.02–0.05)		0.29 (0.18–0.49)
	Steelhead	0.29 (0.18–0.47)		4.58 (2.83–7.70)
Caspian terns				
1999	Chinook	1.98 (1.51–3.03)	2.46 (1.83–3.59)	NC
	Sockeye	0.01 (<0.01–0.01)	0.01 (<0.01–0.01)	
	Coho	0.16 (0.12–0.25)	0.26 (0.20–0.39)	
	Steelhead	1.46 (1.12–2.23)	3.70 (2.76–5.41)	
2000	Chinook	2.01 (1.58–2.64)	7.48 (5.53–10.62)	NC
	Sockeye	0.01 (<0.01–0.01)	0.03 (0.02–0.04)	
	Coho	0.36 (0.28–0.47)	1.00 (0.74–1.42)	
	Steelhead	1.73 (1.36–2.27)	5.58 (4.13–7.92)	

Average annual per capita consumption rates of PIT-tagged smolts was 5.1 (2.9–7.9) to 17.9 (9.0–28.2) times greater for cormorants nesting at colonies in the freshwater zone of the CRE relative to those nesting on East Sand Island, depending on the salmonid species evaluated (*Table 5.6*). Estimates of annual per capita consumption rates were at least 4.2 (2.1–7.4) times greater for cormorants nesting on Rice Island in 2001 and 2006, and at least 2.5 (1.3–4.5) times greater for those nesting on Miller Sand Spit in 2006 and 2007 and were significantly greater in all salmonid species-specific comparisons (*Table 5.6*). It should be noted that estimates of per capita consumption rates for cormorants nesting on Miller Sand Spit in 2006 were generally the lowest of all estimates available for cormorants nesting in the freshwater zone of the estuary (*Table 5.6*). These low estimates in 2006 may have been due to a delay in colony formation during that year (Suzuki et al. 2015), when cormorants were not nesting and depositing tags on Miller Sands Spit until May. On average, however, relative per capita consumption estimates were similar between the two colonies in the freshwater zone, with no apparent difference in relative susceptibility of PIT-tagged smolts to cormorants nesting on Rice Island versus Miller Sands Spit (*Table 5.6*).

Table 5.6. Relative susceptibility of PIT-tagged juvenile salmonids to predation by double-crested cormorants and Caspian terns nesting in the freshwater zone versus the marine zone of the Columbia River estuary. Comparisons represent the proportion of fish consumed per breeding pair for birds nesting on Rice Island or Miller Sands Spit in the freshwater zone relative to birds nesting on East Sand Island in the marine zone of the estuary (95% credible intervals). Uncertainty ranges greater than 1.0 were considered statistically significant differences in relative susceptibility. "NC" denotes no colony established in that year; "NA" denotes PIT tag recoveries were insufficient to estimate relative susceptibility.

		Per Capita Consumption Rate (95% CI)	
Year	Species	Rice Island	Miller Sands Spit
Double-crested cormorants			
2001	Chinook	4.2 (2.1–7.4)	NC
	Sockeye	NA	
	Coho	8.0 (4.0–14.2)	
	Steelhead	6.8 (3.3–11.6)	
2006	Chinook	5.9 (2.7–10.1)	2.5 (1.3–4.5)
	Sockeye	6.7 (3.1–11.8)	10.9 (5.9–20.1)
	Coho	6.9 (3.6–12.8)	5.5 (2.5–10.3)
	Steelhead	7.3 (3.6–13.1)	3.23 (1.5–5.5)
2007	Chinook	NC	9.9 (3.6–18.6)
	Sockeye		24.1 (9.3–43.5)
	Coho		9.7 (4.4–17.3)
	Steelhead		16.3 (7.6–30.2)
Average	**Chinook**	**5.1 (2.9–7.9)**	**6.3 (2.9–10.5)**
	Sockeye	**NA**	**17.9 (9.0–28.2)**
	Coho	**7.7 (4.6–11.8)**	**7.7 (4.6–12.4)**
	Steelhead	**7.3 (4.7–11.3)**	**9.9 (4.9–16.7)**
Caspian terns			
1999	Chinook	1.2 (0.7–2.0)	NC
	Sockeye	1.8 (1.1–2.7)	
	Coho	1.6 (0.9–2.5)	
	Steelhead	2.5 (1.4–4.1)	
2000	Chinook	3.7 (2.3–5.7)	NC
	Sockeye	2.6 (1.9–3.7)	
	Coho	2.8 (1.8–4.3)	
	Steelhead	3.2 (2.0–4.9)	
Average	**Chinook**	**2.5 (1.7–3.5)**	**NC**
	Sockeye	**2.6 (1.7–3.7)**	
	Coho	**2.3 (1.5–3.0)**	
	Steelhead	**2.9 (2.1–4.1)**	

During both 1999 and 2000, terns nesting on Rice Island in the freshwater zone of the estuary consumed more PIT-tagged salmonid smolts per breeding pair than those nesting on East Sand Island in the marine zone (*Figure 5.3*, *Table 5.5* above). Per capita consumption of PIT-tagged smolts varied by year and colony location with median estimates ranging from less than 0.1 to 2.0 PIT-tagged smolts per breeding pair for terns nesting on East Sand Island and from less than 0.1 to 7.5 PIT-tagged smolts per breeding pair for terns nesting on Rice Island (*Table 5.5* above). Average annual per capita consumption rates of PIT-tagged smolts were 2.3 (1.6–3.2) to 2.9 (2.0–4.1) times greater for terns nesting on Rice Island relative to those nesting on East Sand Island, depending on the salmonid species evaluated. Estimates of annual per capita consumption rates for Rice Island terns were significantly greater than those for East Sand Island terns in six of eight possible salmonid species comparisons (*Table 5.6* above). Although the point estimates for consumption rates were greater for Rice Island terns for all salmonid species in all years, results were not significantly different for Chinook salmon and coho salmon in 1999. Estimates of PIT-tagged smolt consumption rates were 1.2 (0.7–2.0) to 2.5 (1.4–4.1) times greater for terns nesting on Rice Island compared to East Sand Island in 1999, the first year of colony relocation efforts. Once most Caspian terns nesting in the CRE had relocated to East Sand Island in 2000, however, estimates of PIT-tagged smolt consumption rates for terns nesting at the colony on Rice Island were 2.6 (1.9–3.7) to 3.7 (2.3–5.7) times greater than those for terns nesting on East Sand Island.

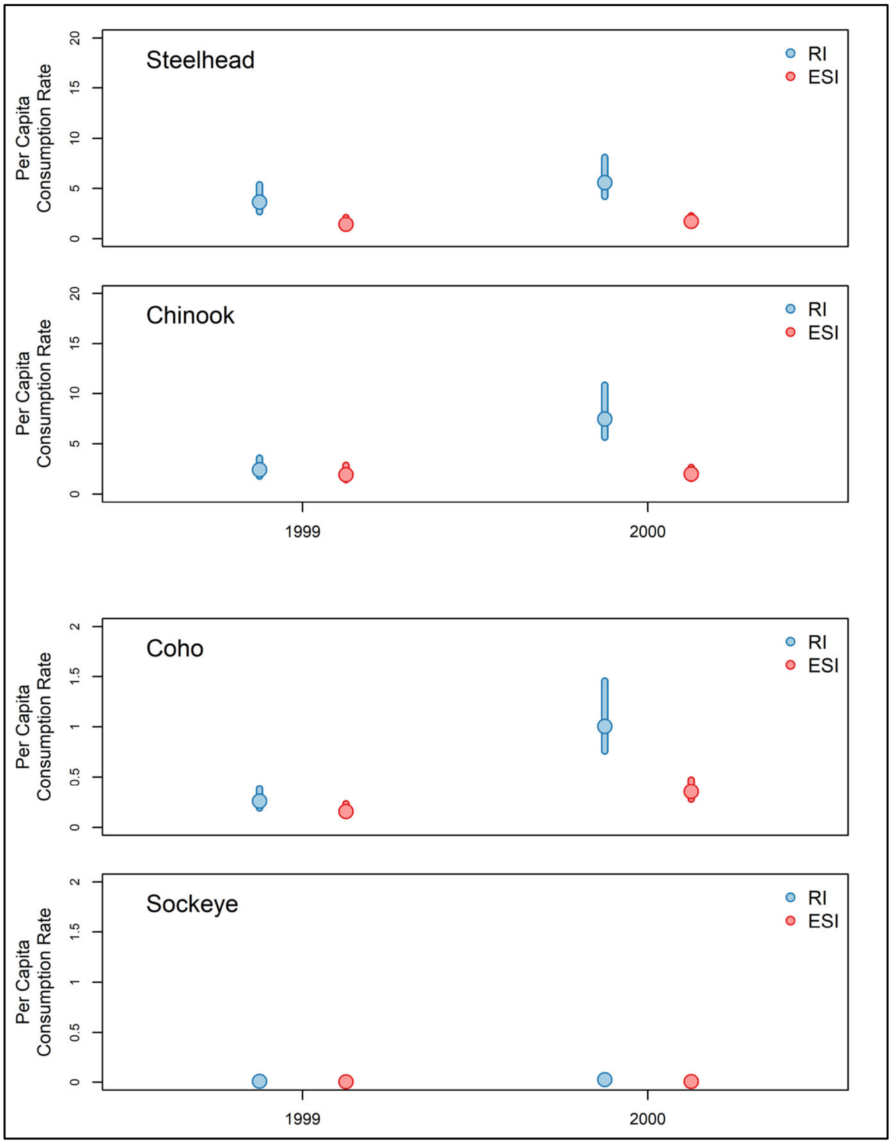

Figure 5.3. Estimated per capita (per breeding pair) consumption rates (95% credible intervals) by salmonid species for Caspian terns nesting at colonies on East Sand Island (ESI) and on Rice Island (RI). See Table 5.5 above for the data used in these figures.

5.5 Discussion

Results from this study indicate that per capita predation rates on juvenile salmonids by cormorants and terns nesting in the CRE were related to where the birds nested, with consistently, and often significantly, higher per capita smolt consumption rates for cormorants and terns nesting at colonies in the freshwater zone of the CRE as compared to colonies in the marine zone of the CRE. This finding is consistent with the results and conclusions drawn from other related studies. For example, studies investigating the diet composition and foraging behavior of piscivorous colonial waterbirds nesting concurrently on both Rice Island and East Sand Island found that juvenile salmonids were significantly more prevalent in the diet of cormorants, terns, and gulls nesting at colonies in the freshwater zone versus the marine zone of the CRE (Collis et al. 2002, Roby et al. 2002, Lyons et al. 2005). These differences in diet composition were most dramatic for cormorants, where the proportion of salmonids in the diet of cormorants nesting at colonies on or near Rice Island were approximately three times greater than for cormorants nesting at the colony on East Sand Island during 1997–1998 (Collis et al. 2002). Using a bioenergetics modelling approach, Lyons (2010) estimated significantly higher annual per capita consumption rates of juvenile salmonids by cormorants nesting on Rice Island (2,900 smolts/breeding pair) relative to those nesting on East Sand Island (2,000 smolts/breeding pair) in 1998 . These studies also found that marine and estuarine forage fish (e.g., Pacific herring [*Clupea pallasii*], northern anchovy [*Engraulis mordax*], and surf smelt [*Hypomesus pretiosus*]) were significantly more prevalent in the diet of cormorants and terns nesting in the marine zone versus the freshwater zone of the estuary (Collis et al. 2002, Roby et al. 2002, Lyons et al. 2005, Lyons 2010). Finally, a radio telemetry study that compared foraging behaviors of terns nesting on Rice Island and East Sand Island during the same year indicated that terns nesting on Rice Island foraged primarily near the colony in the freshwater zone of the CRE, while terns nesting on East Sand Island foraged predominately in the marine and mixing zones of the estuary, especially during the early chick-rearing period (28 May–18 June; Lyons et al. 2005), which corresponds to the period of peak smolt abundance in the estuary (FPC 2019). Taken together, these results suggest colony location significantly influenced foraging behavior for both cormorants and terns nesting in the CRE, consistent with predictions based on the central-place foraging theory (Orians and Person 1979).

For all estuary cormorant colonies, per capita PIT tag consumption estimates were greatest for Chinook salmon relative to coho salmon, sockeye salmon, and steelhead. This finding is consistent with Chinook salmon being the most abundant species of salmonid smolt in the CRE, and therefore the most heavily depredated salmonid species by generalist predators such as double-crested cormorants (Sebring et al. 2013, Dorr et al. 2014). This is particularly true for sub-yearling Chinook salmon originating from tributaries below Bonneville Dam, smolts that are more susceptible to cormorant predation in the estuary due, in part, to their immediate availability (i.e. hatchery release sites are within commuting distance of cormorant colonies in the estuary) and increased exposure due to the prolonged residency of sub-yearling Chinook salmon in the estuary compared to other salmonid run-types (i.e. yearling) and species (Bottom et al. 2011, Roegner et al. 2012, Sebring et al. 2013, McNatt et al. 2016). The PIT tag recovery data used in this study were insufficient to determine relative differences in per capita consumption of smaller salmonid cohorts beyond the species level (e.g., a specific population, stock, or rearing type) due to the small size of cormorant colonies in the freshwater zone of the CRE during the study period. Although considerable foraging in the estuarine-mixing and freshwater zones of the CRE has been documented by cormorants nesting on East Sand Island (Anderson et al. 2004, Lyons 2007, Peck-Richardson 2017), Lyons (2010) determined via bioenergetics modelling that, proportionately, sub-yearling Chinook salmon were significantly more prevalent in the diet of cormorants nesting on East Sand Island compared to those nesting on Rice Island.

Other studies have shown that terns nesting in the CRE do not consume juvenile salmonids in proportion to their relative availability, with terns depredating steelhead trout smolts at higher rates than other salmonid species (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012). The greater susceptibility of juvenile steelhead to tern predation relative to other salmonid species is likely due to the greater size and nutritional value of steelhead smolts (Lyons 2010, Hostetter et al. 2012), plus the greater surface-orientation of steelhead during out-migration compared to other species of salmonid (Beeman and Maul 2004), factors that increase their risk of predation by plunge-diving predators such as Caspian terns (Cuthbert and Wires 1999). The primary impetus to manage terns in the CRE has been to reduce tern predation on steelhead trout (USFWS 2005, 2006). Roby et al. (2003) and Lyons (2010) found that salmonids were less prevalent in the diet of terns nesting on East Sand Island in the marine zone relative to Rice Island in the freshwater zone of the CRE. Results from this study corroborate those findings and provide additional information on the per capita rates of tern predation on steelhead trout and other salmonid species in the CRE.

Although results of this and other studies clearly indicate that cormorants and terns consume fewer salmonid smolts per capita when nesting in the marine zone (i.e. East Sand Island) compared to the freshwater zone of the CRE, some results should be interpreted cautiously. First, few historical data points were available for analysis during years when each bird species nested at different colony sites in the same year. For example, tern colonies have only been active at more than one location in the CRE during four years since long-term PIT tag recovery efforts on piscivorous waterbird colonies were initiated, and only in two years were both tag recovery and colony count data available (e.g., in 2017 PIT tags were recovered on Rice Island but colony size could not be determined due to nest dissuasion efforts conducted at that site during that year [Harper and Collis 2018]). While cormorants have nested at multiple locations in CRE for well over a decade, suitable concurrent PIT tag recovery efforts have only occurred on three occasions. For example, there was a sizable cormorant colony (795 breeding pairs), and PIT tags were recovered on Rice Island in 1998, but no data were collected from the East Sand Island cormorant colony in 1998. It should be noted, however, that the per capita consumption estimates for Rice Island cormorants in 1998 were comparable to estimates for Rice Island cormorants in 2001 and 2006 (A. Evans, unpublished data), alleviating potential concern over the effects of smaller colony size on the per capita consumption rates presented herein. Since 2007, most cormorant nesting in the CRE that has occurred away from the East

Sand Island colony has been at over-water sites where PIT tag recovery is severely limited (i.e. bridges, pilings, navigational aids) or where colony size estimates were uncertain (i.e. Rice Island in 2002). Because opportunities for PIT tag recovery at multiple colonies in the same year have been limited and due to other limitations (see below), our understanding of colony-specific predation rates by cormorants and terns on juvenile salmonids in the CRE is also limited.

Results from this and other studies indicate that colony location and colony size are important factors influencing losses of juvenile salmonids to predation by cormorants and terns in the CRE (Collis et al. 2002, Roby et al. 2003, Evans et al. 2012, Evans et al. 2016b). Other factors, however, have also been linked to seasonal and interannual variation in predation by cormorants and terns in the CRE. For instance, Lyons et al. (2014) found that environmental factors that regulate the availability of alternative prey, such as river discharge, sea surface temperatures, and the timing and strength of upwelling, explained a large proportion of variation in the diet composition of cormorants nesting on East Sand Island during 1999–2015. Additionally, variation in predation rates by cormorants and terns nesting in the CRE can be influenced by smolt run-timing, abundance, survival, and by the nesting chronology and nesting success of at particular colonies of piscivorous waterbirds (Evans et al. 2016). Given the complex suite of factors that influence smolt susceptibility to cormorant and tern predation in the CRE, we caution against using these results to extrapolate smolt losses to piscivorous waterbird colonies where empirical data are lacking, especially without taking into consideration how these factors might influence smolt susceptibility to avian predation in the CRE.

Beginning in May 2016, the East Sand Island cormorant colony has been abandoned several times during the breeding season, which has coincided with the peak smolt out-migration period (April–June; Evans et al. 2019). Although the factors responsible for these colony abandonment events are not fully understood, they have coincided with the onset of cormorant management at the East Sand Island colony and frequent bald eagle (*Haliaeetus leucocephalus*) disturbances of the colony (Turecek et al. 2018, Turecek et al. 2019). Rather than dispersing to alternative colony locations outside the CRE, large numbers of cormorants (more than 15,000; Turecek et al. 2019) have remained in the estuary throughout the smolt out-migration period. Nesting and roosting cormorants have been observed at numerous locations within the freshwater zone and the estuarine-mixing zone of the CRE, including the Astoria-Megler Bridge, Rice Island, Miller Sands Spit, and several nearby channel markers. In particular, the size of the double-crested cormorant breeding colony on the Astoria-Megler Bridge has increased exponentially since colony abandonment events began on East Sand Island (Turecek et al. 2019; see *Chapter 4*). PIT tag recovery at the Astoria-Megler Bridge colony has not been conducted and no diet composition data for cormorants nesting at this colony currently exist; therefore, predation rates on juvenile salmonids by cormorants nesting in the estuarine-mixing zone have not been quantified and remain a critical uncertainty. Given that the Astoria-Megler Bridge colony of double-crested cormorants has grown to more than 5,000 breeding pairs as of the 2020 nesting season (M.J. Lawonn, ODFW, unpubl. data) and may as much as double in size in the future (M.J. Lawonn, ODFW, pers. comm.), continued research at the Astoria-Megler Bridge cormorant colony is clearly warranted and may be paramount to evaluating the efficacy of the *Cormorant Management Plan* for lowering predation rates on juvenile salmonids in the CRE by reducing the size of the East Sand Island cormorant colony. Similarly, major reductions to tern nesting habitat at East Sand Island since 2008 have resulted in smaller tern colony size on East Sand Island in recent years but have coincided with an increase in recolonization attempts at historical colony sites in the freshwater zone of the CRE, particularly on Rice Island where in some years several thousand terns have attempted but failed to nest (Harper and Collis 2018). As such, until the impacts on smolt survival from the growing number of cormorants and terns utilizing the freshwater and estuarine-mixing zones of the CRE can be quantified, resource manager cannot reasonably assess the efficacy of ongoing management actions to reduce predation rates by cormorants and terns nesting in the CRE. Management that results in cormorants and terns dispersing from East Sand Island to alternative colony sites in the CRE could, depending on colony size and location, have the unintended effect of increasing impacts of avian predation on juvenile salmonids (see *Chapters 1* and *4* for a more detailed description of management considerations and recommendations stemming from this and other studies).

Collectively, results from this study confirm and support previously published studies on diet composition that indicate juvenile salmonids are more susceptible to predation by cormorants and terns that nest at colony sites located upstream of East Sand Island within the freshwater zone of the CRE. Relative differences in smolt susceptibility to predation varied by salmonid species, predator, and year, but estimates of consumption rates of PIT-tagged smolts were consistently higher at all nesting colonies in the freshwater zone of the estuary compared to colonies in the marine zone. Although avian predation is just one of many mortality factors for salmonids during the smolt life-stage, current impacts on smolt survival from avian predation in the CRE may be limiting recovery of some ESA-listed salmonid populations (Evans et al. 2019). Recent management initiatives have resulted in reduced smolt mortality from cormorants and terns nesting at colonies on East Sand Island, but these benefits are likely being offset to an unknown degree by the growing number of cormorants and terns that are nesting, roosting, and prospecting for nest sites within the freshwater and estuarine-mixing zones of the CRE.

5.6 Acknowledgments

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CHAPTER 6: Predation on Juvenile Salmonids by Colonial Waterbirds Nesting at Unmanaged Colonies in the Columbia River Basin

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6.1 Summary

The primary objectives of this chapter are to (1) summarize the best available information on several piscivorous waterbird species nesting at unmanaged colonies in the Columbia River basin (CRB) and (2) identify those colonies whose impacts on survival of juvenile salmonids (*Oncorhynchus* spp.) from populations listed under the U.S. Endangered Species Act (ESA) may be significant. Predation on juvenile salmonids (smolts) by Caspian terns (*Hydroprogne caspia*) from colonies in the Columbia River estuary and the Columbia Plateau region and predation by double-crested cormorants (*Phalacrocorax auritus*) from colonies in the Columbia River estuary have been well documented since the late 1990s. The magnitude of predation in each case led to the implementation of regional management plans aimed at improving smolt survival during out-migration to the Pacific Ocean. This is a synthesis of the available data for unmanaged colonies of piscivorous waterbirds in the CRB, namely colonies of double-crested cormorants, California gulls (*Larus californicus*), ring-billed gulls (*L. delawarensis*), and American white pelicans (*Pelecanus erythrorhynchos*) in the Columbia Plateau region (CPR) and colonies of Brandt's cormorants (*P. penicillatus)*, glaucous-winged/western gulls (*L. glaucescens* X *L. occidentalis*), ring-billed gulls, and American white pelicans in the Columbia River estuary (CRE). In most cases, data on unmanaged colonies of piscivorous waterbirds were limited to estimates of colony size and smolt predation rates for terns, cormorants, and pelicans or smolt consumption rates for gulls (see *below*); also summarized in this chapter are limited data that were collected on nesting chronology, nesting success, factors that limit colony size and nesting success, and diet composition.

Surveys were conducted throughout the CRB during 2003–2018 to assess the regional distribution and size of colonies of piscivorous waterbirds. Breeding colonies of piscivorous waterbirds were detected at more than 20 different locations, from East Sand Island near the mouth of the Columbia River to islands on several inland waterbodies in the CPR region of eastern Washington. Colony locations ranged from sites with a long-term history of nesting by certain species to ephemeral sites used during a single breeding season. Gulls (i.e. glaucous-winged/western gulls, California gulls, ring-billed gulls) were the most abundant and well-distributed taxon of piscivorous colonial waterbirds in the CRB, with nesting colonies documented at 15 different locations. In some years the estimated number of gulls breeding in the CRB was more than 75,000 individuals, mainly California and ring-billed gulls nesting at colonies in the CPR. The number of gulls breeding in the CPR declined during this study and the decline was most evident at colonies located on islands in the Columbia River. Nearly twice as many gulls were counted at colonies on the Columbia River during the late 1990s compared to during 2013–2018, when aggregate counts averaged about 29,000 individuals. Double-crested cormorants were much less numerous in the CPR than gulls, but cormorant colonies were detected at 11 different locations during 2003–2018. Comprehensive regional surveys conducted during 2005-2013 indicated that the number of nesting double-crested cormorants in the CPR is stable or declining, with annual estimates ranging from 1,150 to about 1,575 breeding pairs during this period. After 2013, the regional breeding population of double-crested cormorants appeared to be in decline, due largely to the dramatic decline and apparent abandonment of the Potholes Reservoir area as a nesting site for double-crested cormorants. Numbers of American white pelicans breeding in the CRB increased substantially during 2003–2018, chiefly at the colony on Badger Island in the CPR, where colony size grew from about 1,000 individuals in 2005 to about 3,300 individuals in 2018. Numbers of American white pelicans breeding in the Columbia River estuary (CRE) were much smaller than in the CPR, but also increased during the study period; a small colony was first confirmed at Miller Sands Spit in 2010 and subsequently grew to several hundred breeding individuals. Brandt's cormorants, a marine species, also nested in the CRE, mainly on East Sand Island, where the colony grew from fewer than 50 breeding pairs in 2006 to more than 2,100 breeding pairs in 2018, making it one of the largest Brandt's cormorant colonies recorded along the coasts of Washington and Oregon. Brandt's cormorants apparently abandoned the breeding colony on East Sand Island in 2020, however, following the abandonment of the colony by double-crested cormorants. The numbers of glaucous-winged/western gulls nesting on East Sand Island, the largest gull colony in the Columbia River estuary, declined from more than 7,000 individuals in 1997 to about 4,000 individuals in 2015. Finally, a ring-billed gull colony first became established on East Sand Island in 2004, when a few hundred individuals colonized the northeastern tip of the island. This colony subsequently grew to over 2,200 individuals by 2009 and subsequently fluctuated between 1,417 and 2,676 individuals during 2010–2015.

Other data on the nesting ecology of glaucous-winged/western gulls, California gulls, ring-billed gulls, American white pelicans, and Brandt's cormorants in the CRB are generally sparse. Factors observed to limit colony size and nesting success of piscivorous waterbirds nesting at unmanaged colonies in the CRB included nesting habitat loss (both natural and anthropogenic), predation, food availability, and human disturbance. Monitoring for most species and colonies during this study was often too limited to measure nesting success or detect specific causes of poor nesting success or colony abandonment. Similarly, other metrics like diet composition and the number of smolts consumed, which have been used to estimate the magnitude of predation by Caspian terns and double-crested cormorants on juvenile salmonids at managed colony sites, were either limited (i.e. double-crested cormorants nesting on Foundation Island, certain gull colonies) or not measured (American white pelicans, Brandt's cormorants). The average proportion of juvenile salmonids (percent of prey biomass) in the diet of double-crested cormorants nesting on Foundation Island was about 22% when measured during 2005–2010. Diet composition data for CRB gull colonies was last collected during 1997–1998, when the gull diet at the now abandoned colonies on Three Mile Canyon Island and Island 18 generally consisted of few fish. Conversely, juvenile salmonids were detected in the diet of California gulls nesting at Little Memaloose Island near The Dalles Dam (ca. 15%; colony now abandoned) and at Miller Rocks near John Day Dam (ca. 3%), and in the diet of glaucous-winged/western gulls nesting at Rice Island (ca. 11%) and East Sand Island (ca. 4%) in the CRE. Other ancillary data sources were included (where available) in this synthesis to provide a more complete retrospective analysis of avian predation/consumption rates, and to highlight data gaps where additional research may be warranted. Collectively, per capita predation/consumption rates and information on the foraging ecology of gulls, American white pelicans, Brandt's cormorants, and double-crested cormorants in the CPR indicated that colony size alone was not generally a reliable predicter of impacts on smolt survival from avian predation/consumption. Instead, the proximity of colonies to hydroelectric dams on the Columbia River, availability of alternative prey, and species-specific foraging traits were key factors influencing predation/consumption rates on salmonid smolts.

Over the last two decades, recoveries of smolt passive integrated transponder (PIT) tags at unmanaged piscivorous waterbird colonies in the CRB have been used to quantify avian predation/consumption rates (see *below*; percentage of the available tagged fish that were depredated/consumed). Despite such a long time series, many of the initial estimates of predation/consumption rates were reported as minimums due to a lack of information on what proportion of ingested PIT tags were subsequently deposited by birds on their breeding colonies. Consequently, the full impact on survival of juvenile salmonids from predation/consumption by piscivorous waterbirds from unmanaged colonies in the CRB remained unknown. As part of this study, smolt PIT tag data from unmanaged piscivorous waterbird colonies were retrospectively analyzed to generate standardized, accurate estimates of predation/consumption rates of salmonid smolts from ESA-listed populations (hereafter referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]); these predation/consumption rates incorporate both detection probabilities and deposition probabilities. Because California gulls, ring-billed gulls, and glaucous-winged/western gulls are known to consume dead, moribund, and injured smolts and are known to kleptoparasitize (steal) smolts that have been depredated by other piscivorous waterbirds, we refer to their impacts on juvenile salmonids as “consumption rates.” That is not the case for the other piscivorous colonial waterbirds investigated here (i.e. Caspian terns, double-crested cormorants, Brandt’s cormorants, and American white pelicans); consequently, impacts on smolts by these bird species are referred to as “predation rates”.

Results from analyses of smolt PIT tags recovered from piscivorous waterbird colonies indicated that smolt predation/consumption rates by double-crested cormorants, Brandt’s cormorants, California gulls, ring-billed gulls, and American white pelicans nesting at unmanaged colonies varied considerably by predator species, salmonid ESU/DPS, colony location, and year. For example, smolt predation/consumption rates by double-crested cormorants and gulls from several colonies in the CPR, particularly those located more than 30 km from the Columbia River, were consistently less than 1% of available smolts, per salmonid ESU/DPS. Estimates of predation rates by Brandt’s cormorants nesting in the CRE and American white pelicans nesting in the CPR were also consistently less than 1% of available smolts, per ESU/DPS. Estimates of consumption rates by gulls nesting on islands in the Columbia River, however, were significantly greater than those by gulls nesting at off-river sites. For example, average annual consumption rates by California gulls nesting at Miller Rocks were 8.2% (95% credible interval = 6.9 –9.3%), 7.2% (6.5–8.1%), and 6.2% (4.8–7.7%) of available Upper Columbia River steelhead, Snake River steelhead, and Snake River sockeye salmon, respectively, during 2007–2019. Estimates of smolt consumption rates by gulls nesting at other colonies on the Columbia River (i.e. Blalock Islands, Crescent Island, Badger Island, and Island 20) were generally less than those of gulls nesting at Miller Rocks, but were also appreciable in some years, with consumption rates consistently greater than 2% per salmonid ESU/DPS, but generally less than 5% of available smolts, depending on the colony, ESU/DPS, and year. Of those double-crested cormorant colonies in the CPR that were evaluated, smolt predation rates were highest for cormorants nesting on Foundation Island, where average annual predation rates were 4.0% (3.4 –4.7%) and 3.4% (2.4 –4.5%) on Snake River steelhead and Snake River sockeye salmon, respectively, during 2007-2014. Collectively, the results indicated that consumption rates by gulls nesting at certain colonies on the Columbia River and predation rates by double-crested cormorants nesting at Foundation Island were similar to or greater than those of Caspian terns nesting at managed colonies (i.e. Crescent Island and Goose island) in the CPR.

Several data gaps and uncertainties associated with estimating predation/consumption rates on juvenile salmonids by piscivorous waterbirds nesting at unmanaged colonies were identified as part of this study. For example, estimates of smolt predation/consumption rates are currently lacking or represent minimum estimates for several extant colonies of piscivorous waterbirds in the CRB, including glaucous-winged/western gulls nesting in the CRE, American white pelicans nesting in the CRE and the CPR, and for several of the largest gull colonies in the CPR. An unknown proportion of smolts consumed by gulls at several colonies in the CPR were kleptoparasitized (stolen) from Caspian terns (i.e. killed by terns but consumed by gulls) and gulls may be disproportionately consuming dead, moribund, injured, or temporally disoriented smolts in the tailraces of dams. Data to evaluate to what degree gull consumption of juvenile salmonids constitutes predation, and to what extent the mortality associated with gull predation is additive versus compensatory, would be crucial for understanding the potential benefits of managing gull colonies in the CRB to increase smolt survival. Finally, study results have demonstrated that the population dynamics, diet composition, and predation/consumption rates of juvenile salmonids by piscivorous colonial waterbirds nesting in the CRB can change rapidly, especially when new breeding colonies become established and/or extant colonies increase dramatically in size. Some baseline monitoring is therefore necessary to detect changes in the number, distribution, and size of piscivorous waterbird colonies, changes that could potentially alter the impact of avian predators on smolt survival in the CRB.

6.2 Introduction

6.2.1 Research on Piscivorous Colonial Waterbirds in the Columbia River Basin

The Columbia River basin (CRB) supports large breeding colonies of several species of piscivorous waterbirds, specifically Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), Brandt's cormorants (*P. penicillatus)*, glaucous-winged/western gulls (*Larus glaucescens* X *L. occidentalis*), California gulls (*L. californicus*), ring-billed gulls (*L. delawarensis*), and American white pelicans (*Pelecanus erythrorhynchos*). Predation by colonial waterbirds on juvenile salmonids (i.e. Chinook salmon [*Oncorhynchus tshawytscha*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*]*,* and steelhead trout [*O. mykiss*]) during out-migration to the Pacific Ocean is an on-going concern for fisheries managers in the CRB, especially for those stocks that are listed under the U.S. Endangered Species Act (ESA; NOAA 2019). Predation impacts on juvenile salmonids by piscivorous waterbirds nesting in the CRB can be substantial, and in some cases account for a large fraction of total mortality for out-migrating smolts (Roby et al. 2003, Lyons 2010, Evans et al. 2019a; see also *Chapter 7*). To mitigate the impact of predation by Caspian terns and double-crested cormorants nesting in the CRB, resource managers have developed and implemented three management strategies to reduce the size of or eliminate selected breeding colonies (USFWS 2005, USFWS 2006, USACE 2014, USACE 2015). The breeding and foraging ecology of Caspian terns in the CRB and double-crested cormorants in the Columbia River estuary (CRE) has been well-documented and is the subject of numerous studies since 1997 (Collis et al. 2002, Roby et al. 2003, Antolos et al. 2004, Lyons 2010, Hostetter et al. 2012). Lacking, however, is comparable information that would contextualize the impact of predation/consumption on ESA-listed salmonids by piscivorous waterbirds nesting at unmanaged colonies (i.e. California gulls, ring-billed gulls, double-crested cormorants, and American white pelicans nesting in the Columbia Plateau region [CPR]; glaucous-winged/western gulls, ring-billed gulls, American white pelicans, and Brandt's cormorants nesting in the CRE) relative to that of managed species and colonies of avian predators in the CRB (Collis et al. 2002, Lyons et al. 2011b, Evans et al. 2012, Adkins et al. 2014a, Evans et al. 2016a; Evans et al. 2019).

Over the last two decades, studies of avian predation/consumption based on recoveries of passive integrated transponder (PIT) tags from juvenile salmonids on bird colonies have been conducted at numerous unmanaged piscivorous waterbird colonies in the CRB (Ryan et al. 2001, Collis et al. 2001, Evans et al. 2012, Lyons et al. 2013, Evans et al. 2016a, Payton et al. 2019, Evans et al. 2019a, Collis et al. 2020). Like studies conducted at managed colonies of Caspian terns and double-crested cormorants, studies of smolt predation/consumption rates at unmanaged colonies of piscivorous waterbirds were initially conducted to determine which ESA-listed salmonid populations (hereafter referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]) were the most susceptible to avian predation/consumption during out-migration. Estimates of predation rates from these studies date back to 1999 (Ryan et al. 2001). Despite such a long time series, however, many of the initial estimates of predation rates were reported as minimum estimates of smolt losses (e.g., Antolos et al. 2005, Lyons et al. 2011, Evans et al. 2012, USACE 2014) due to a lack of data on PIT tag deposition probabilities (probability a consumed tag is deposited by a bird on its nesting colony; see also Hostetter et al. 2015). During the development of the *Inland Avian Predation Management Plan* (*IAPMP*), for example, accurate estimates of smolt consumption rates by gulls nesting at colonies in CPR were not available because information on what proportion of consumed tags were deposited by gulls at off-colony loafing and roosting sites were unknown at the time (Evans et al. 2012, USACE 2014). Also, it was unknown, and remains unknown, what proportion of smolts consumed by gulls were depredated as opposed to scavenged or kleptoparasitized (stolen) from other piscivorous waterbirds. As such, the potential impact on survival of salmonid smolts from these other unmanaged predator species and colonies has not been fully considered to date. As part of this chapter, recently obtained data on PIT tag detection probabilities and deposition probabilities (See *Appendix A.2, Predation Rate Methods*) were used to retrospectively analyze and summarize smolt PIT tag recoveries at selected gull, cormorant, and pelican colonies in the CRB during 2007–2019.

The primary focus of this chapter is to (1) summarize the best available data for unmanaged colonies of piscivorous waterbirds in the CRB, (2) identify those colonies with an appreciable potential impact on survival of smolts from ESA-listed salmonid ESUs/DPSs, and (3) identify critical uncertainties that resource managers may wish to consider when developing future management strategies to address avian predation on juvenile salmonids in the CRB.

6.2.2 Double-crested Cormorant Nesting in the Columbia Plateau Region

The double-crested cormorant is a colonial waterbird found along coasts and at interior waterbodies throughout much of North America (Dorr et al. 2014). Five subspecies have been described, with two found along the Pacific Coast of North America, one in the Pacific Flyway from northwestern Mexico to southern British Columbia and inland to the Continental Divide (*P. a. albociliatus*) and the other in Alaska (*P. a. cincinatus*; Wires et al. 2001); however, Mercer et al. (2013) concluded that the Pacific Flyway population was not sufficiently genetically distinct from cormorant populations east of the Continental Divide to warrant separate subspecies status. Double-crested cormorants utilize a diverse array of nesting habitats, including islands where they can nest on the ground, isolated cliffs, in trees and shrubs, and on artificial structures like bridges and transmission line towers (Dorr et al. 2014). Double-crested cormorants are mostly piscivorous, foraging on fish in open water, where they dive from the surface to pursue their prey underwater (Coleman et al. 2005). They are opportunistic predators, with more than 250 fish species reported in their diet (Dorr et al. 2014).

Double-crested cormorants were first documented to nest in the CPR during the early 1930s (Hanson 1968), but likely nested at scattered locations in the CPR before the 1930s. Double-crested cormorants were considered common in the CPR as late as 1953; however, island nesting habitat in the region was lost and the numbers of nesting cormorants declined because of impoundment of the Columbia River behind McNary Dam beginning in 1954 (Hanson 1968). During this period, range-wide declines in numbers of double-crested cormorants occurred, attributable at least in part to widespread use of dichlorodiphenyltrichloroethane (DDT) and other persistent insecticides (Wires and Cuthbert 2006). Double-crested cormorants were first documented to nest at Potholes Reservoir during the late 1970s and the colony in North Potholes Reserve numbered ca. 650 breeding pairs by 1997 (Finger and Tabor 1997; *Figure 6.1*). There has also been a cormorant nesting colony on Foundation Island in the Columbia River near Wallula, Washington since at least 1998, when ca. 100 breeding pairs were present (Collis et al. 2002). More recently, as part of assessing the status of the Pacific Flyway population of double-crested cormorants, Adkins et al. (2014b) found that during 2003—2009 roughly 4% of the Pacific Flyway population nested in the CPR. While cormorant nesting was documented at seven sites in the CPR during this period, most nesting occurred at four sites in eastern Washington: Potholes Reservoir, Foundation Island, Sprague Lake, and the mouth of the Okanogan River (Adkins et al. 2014b). Since 2010, double-crested cormorant colonies have also become established on power transmission towers near Troutdale, Oregon (2013), in the Locke

Island Complex on the Columbia River (2014), Lenore Lake (2014), and along Crab Creek near Wilson Creek, Washington (2018; *Figure 6.1*).

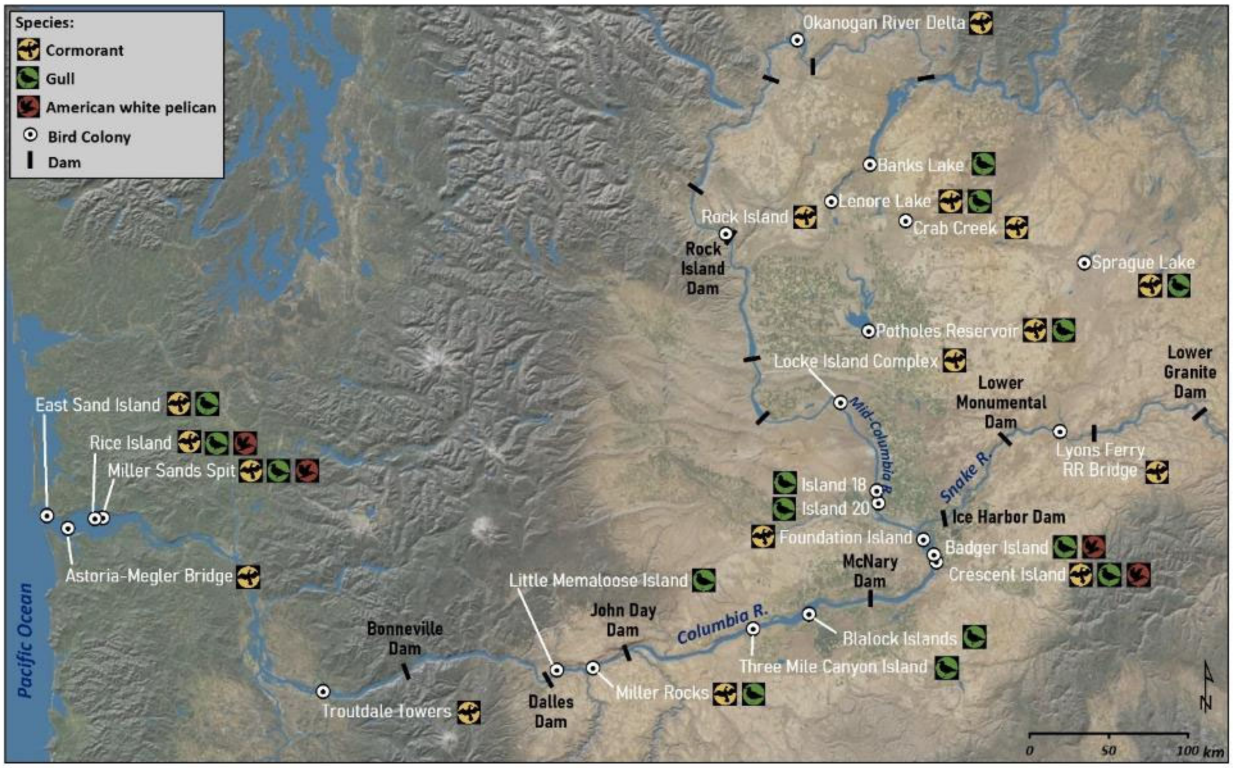

Figure 6.1. Map of the Columbia River basin showing locations of breeding colonies of double-crested cormorants, American white pelicans, and various gull species during 1997–2018.

6.2.3 Gull Nesting in the Columbia River Basin

Four species of gulls are known to nest in the CRB: California gulls, ring-billed gulls, glaucous-winged gulls, western gulls, and their hybrids (i.e. glaucous-winged/western gulls). California gulls are endemic to western North America, breeding at interior locations from central California to Colorado and northward on both sides of the Continental Divide into portions of Manitoba, British Columbia, and Northwest Territories (Winkler 1996). California gulls typically nest on bare or sparsely vegetated islands on rivers, lakes, and reservoirs and commonly nest associated with ring-billed gulls (Vermeer 1970). They are opportunistic foragers and their diet composition is dependent on the locally-available food resources. California gulls consume a wide variety of food resources, including small mammals, fish, birds, eggs, invertebrates, agricultural products, and garbage, and frequently scavenge or kleptoparasitize dead prey items as well as capture live prey (Winkler 1996). California gulls actively pursue fish either by grasping fish at the surface or by shallow plunge dives (Behle 1958, Merrell 1959). Ring-billed gulls are primarily an inland-nesting gull species and are common throughout North American. The western population breeds from northeastern California to parts of Alberta, Saskatchewan, and Manitoba (Pollet et al. 2012). Ring-billed gulls nest on the ground on low islands with sparse vegetation, typically on freshwater lakes, reservoirs, and large rivers. Ring-billed gulls are opportunistic foragers with feeding behaviors and diets similar to California gulls (Pollet et al. 2012). In the western U.S., Conover et al. (1983) found that ring-billed gull colonies were often found within 36 km of towns with populations > 1,000 and were likely to be located near agricultural areas with irrigated crops.

California and ring-billed gulls, which often nest together in mixed species colonies, were first recorded breeding in the CPR during the 1930s (Kitchin 1930, Decker and Bowles 1932). By the 1950s breeding was documented throughout the region with colonies on Moses Lake, Sprague Lake, and along the Columbia River from Boardman, Oregon to Pasco, Washington (Hudson and Yocom 1954, Johnsgard 1954, Broadbooks 1961, Hanson 1963). The apparent expansion of breeding populations of gulls in the CPR continued through the 1970s and has been linked to the rapid development of irrigation-based agriculture and new islands created by dam impoundments (Conover et al. 1979). Conover et al. (1979) observed mixed colonies of ring-billed and California gulls at nine different sites in the CPR during the late 1970s, totaling 17,000 ring-billed gulls and 9,000 California gulls. Colonies were observed on the Columbia River at Little Memaloose Island near the Dalles Dam, Miller Rocks near the mouth of the Deschutes River, Three Mile Canyon Island near Boardman, Oregon, and multiple islands near Richland, Washington. Gull colonies were also detected at Banks Lake, Sprague Lake, and Potholes Reservoir (*Figure 6.1* above). More than two decades later, Collis et al. (2002) found mixed gull colonies at many of the same locations in the CPR, with the notable addition of a colony on Crescent Island, an artificial island near Wallula, Washington that was colonized by gulls shortly after its creation in 1985 (Ackerman 1994).

Gulls belonging to the glaucous-winged gull and western gull hybrid complex breed along the Pacific Coast of North America from southwest Alaska to Baja California, Mexico (Bell 1992). Both species and their hybrids nest at colonies from the Strait of Juan de Fuca, Washington to Coos Bay, Oregon (Hayward and Verbeek 2020). For the purposes of this study, both species and their hybrids are grouped as glaucous-winged/western gulls. Nesting mainly occurs on islands with bare ground or sparse herbaceous cover, but has also occurred on piers, roof-tops, and other man-made structures near water (Pierotti and Annett 1995). Glaucous-winged/western gulls are generalist foragers that consume marine invertebrates, fish, eggs and chicks of other seabirds, carrion, and human refuse. They capture fish by surface-dipping, via shallow plunge dives, and in areas where prey are concentrated by other predators (Pierotti and Annett 1995, Pierotti 1998). Glaucous-winged/western gulls have nested in the CRE on East Sand Island since at least the late 1980s (Bell 1997). More recently, glaucous-winged/western gulls have been documented nesting at three locations in the CRE: East Sand Island, Rice Island, and Miller Sands Spit (*Figure 6.1* above; Collis et al. 2002).

6.2.4 American White Pelican Nesting in the Columbia River Basin

American white pelicans are indigenous to the western and southern portions North America. For the western population, breeding occurs in northeastern California, the Great Basin, and the Pacific Northwest on isolated islands, typically in shallow freshwater lakes, reservoirs, and rivers (Knopf and Evans 2004). Nesting substrate is diverse, ranging from bare ground to sparsely vegetated ground with woody cover (e.g., sagebrush and willows); shallow nest structures are built on the ground with gravel, soil, and/or vegetation. American white pelicans are mainly piscivorous, capturing prey in shallow water less than three meters deep by dipping their bill into the water and scooping prey items (Knopf and Evans 2004). Foraging behavior ranges from individual efforts to cooperative efforts by large flocks (Anderson 1991, Mcmahon and Evans 1992). Relative to other piscivorous colonial waterbirds in the CRB, American white pelicans are the largest species and can consume larger prey items, including adults of salmonids, carp, and suckers (BRNW 2015, Teuscher et al. 2015, Evans et al. 2016b). In the CRB, American white pelicans were thought to have historically nested at Moses Lake and possibly other inland waterbodies such as Sprague Lake (Dawson and Bowles 1909). The first documented breeding record for American white pelicans in the CPR was at Moses Lake in 1926 (Brown 1926). While pelicans were observed in the region in the interim, their breeding status was unclear until 1994, when they were recorded nesting on Crescent Island in the Columbia River (*Figure 6.1* above; Ackerman 1994). In 1997, the pelican breeding colony shifted from Crescent Island to nearby Badger Island, which remains the only known American white pelican colony in the CPR (Ackerman 1997, Adkins et al. 2014a). In the CRE, American white pelicans were first documented nesting on Miller Sands Spit in 2010. There is no historical evidence of American white pelicans having previously nested in the CRE.

6.2.5 Brandt’s Cormorant Nesting in the Columbia River Estuary

Brandt’s cormorants are endemic to the California Current System along the Pacific Coast of North America, breeding from northwestern Mexico to Vancouver Island, British Columbia (Wallace and Wallace 1998). They nest on the ground, typically on rocky islands and isolated cliffs along the coast. Brandt’s cormorants are mainly piscivorous and capture prey during pursuit dives from the surface. Foraging habitat consists of nearshore coastal waters, bays, and occasionally estuaries (Wallace and Wallace 1998). Relative to other piscivorous waterbirds found in the CRB, Brandt’s cormorants are potentially capable of exploiting prey resources at greater depths (> 100 m; Ainley et al. 1984). Brandt’s cormorants were first detected nesting in the CRE in 1997, when a small colony was found nesting on a pile dike off the west end of East Sand Island (Couch and Lance 2004), adjacent to a much larger colony of double-crested cormorants. In 2006 the Brandt’s cormorant colony moved to the upland portion of East Sand Island where they began nesting amongst the large double-crested cormorant colony after winter storms destroyed much of the pile dike where they formerly nested. This rare estuarine colony of Brandt’s cormorants subsequently grew to several thousand pairs, one of the largest colonies for the species in Oregon and Washington.

6.3 Methods

6.3.1 Nesting Distribution

Beginning in 1997, aerial, boat, and ground surveys were initiated to determine the breeding distribution of piscivorous colonial waterbirds (i.e. Caspian terns, double-crested cormorants, several species of gulls, and American white pelicans) in the CRB (Collis et al. 2002). During 2004–2018 standardized reconnaissance flights were conducted beginning in late April, early in the incubation period, to identify active breeding colonies. These surveys were conducted over several days from a manned fixed-wing aircraft that covered the Columbia River from the mouth near Astoria, Oregon to Chief Joseph Dam near Bridgeport, Washington, and along the Snake River from the confluence with the Columbia River to the mouth of the Clearwater River. In addition, waterbodies off the mainstem rivers and within foraging range (~ 100 km) of the Federal Columbia River Power System (FCRPS; *Figure 6.2*) were surveyed. Flights were often specifically designed to locate Caspian tern colonies (see *Chapter* 2), but observations of other colonial waterbirds were also recorded. At each locale where colonial waterbirds were observed, data were collected on the site (i.e. GPS coordinates, island name, description, habitat type), the number and species of colonial waterbirds present, and waterbird behavior (i.e. nesting, foraging, or loafing). When colonial waterbirds were observed in habitat and on substrate that could be suitable for nesting, oblique aerial photography was taken using a digital SLR camera with an image-stabilizing, zoom lens. When in-flight observations or post-flight inspection of digital images suggested the presence of a potential breeding colony, ground- or boat-based surveys were conducted to assess breeding status and other colony metrics (see *below*). Although it is possible that small colonies (i.e. < 10 breeding pairs) may have been missed during these surveys, all colonies of consequence were identified within the study area due to the extent of coverage by aerial surveys and communication with other researchers and agency personnel familiar with colonial waterbird nesting in the region (see *Appendix A* for a detailed description of these methods).

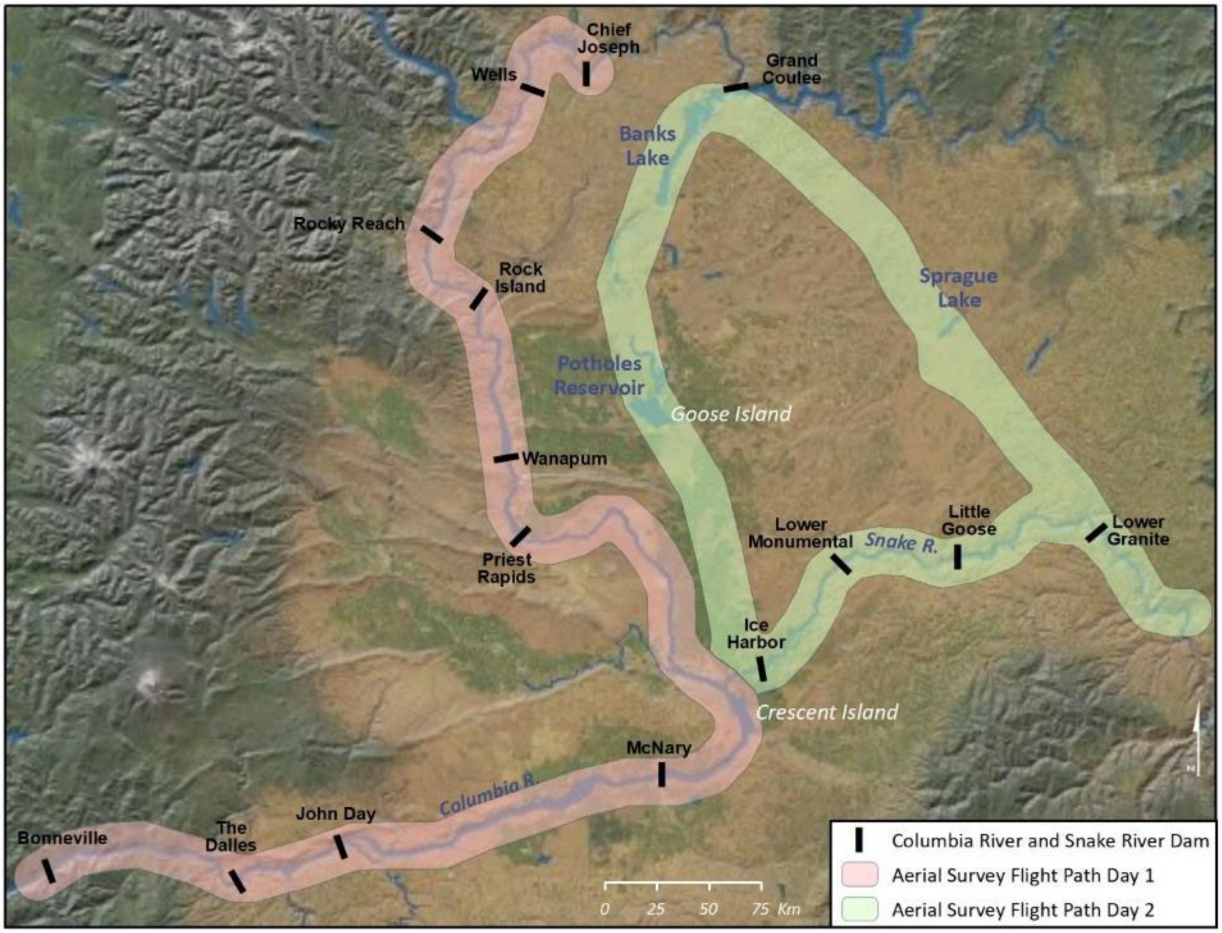

Figure 6.2. Flight path followed during aerial reconnaissance surveys conducted to locate piscivorous waterbird colonies in the Columbia Plateau region.

6.3.2 Nesting Chronology

6.3.2.1 Double-crested cormorants – During the 2006–2012 breeding seasons, field crews visited Foundation Island on the Columbia River (river km [Rkm] 518; *Figure 6.1* above) each week during April–July and monitored this arboreal cormorant colony from an observation blind positioned in the river approximately 25 meters from the colony to prevent researcher disturbance. Nesting chronology for double-crested cormorants was described as the date when the first egg was seen in a nest (i.e. first egg), when the first hatchling chick was seen (i.e. first chick), and when the first fledgling was observed (i.e. first fledging). All other double-crested cormorant colonies in the region were observed less frequently and from more distant vantage points (from boats or on land), which limited the ability to collect data on breeding chronology.

6.3.2.2 Gulls – Nesting chronology data for California, ring-billed, and glaucous-winged/western gulls nesting in the CRB were collected opportunistically during irregular visits to active colonies. In some years (e.g., at Goose Island, Potholes Reservoir during implementation of the

IAPMP (see *Chapter 2*) frequent visitation allowed for more detailed colony observations. When possible, nesting chronology for gulls was described as the date when the first egg was seen in a nest (i.e. first egg), when the first hatchling chick was seen (i.e. first chick), and when the first fledgling was observed (i.e. first fledging).

6.3.2.3 American white pelicans – No standardized data on nesting chronology for this species were collected.

6.3.2.4 Brandt's cormorants – No standardized data on nesting chronology for this species were collected.

6.3.3 Colony Size & Nesting Success

6.3.3.1 Double-crested cormorants – In the CPR, double-crested cormorant colonies were observed from blinds or other nearby vantage points and during survey flights to avoid disturbing nesting birds. Colony size (number of breeding pairs) was estimated using aerial-, ground-, or boat-based counts of the peak number of active nests during the late incubation and early chick-rearing periods. Any nest structure attended by a cormorant adult and/or chick was considered an active nest and indicative of a breeding pair. Estimates of colony size at arboreal colonies should be considered minimums, as vegetation partially obscured some nests over the course of the breeding season. Methods to estimate colony size for double-crested cormorants followed those of Adkins et al. (2014a) and are described in *Appendix A.1*. In-season colony attendance was monitored for double-crested cormorants nesting on Foundation Island in select years (see *above*). When possible, these observations were used to estimate nesting success based on the number of chicks raised to 28 days post-hatching in monitored nests (up to 50 nests, annually). No measure of variance for estimates of colony size and nesting success at double-crested cormorant colonies in the CPR are available.

6.3.3.2 Gulls – Vertical or oblique aerial photography of gull colonies taken during late incubation were counted to estimate the total number of individual gulls on-colony as an index to colony size. Nearly all the gulls present on each colony were visible in the aerial imagery because few nests were obscured by vegetation, but these counts should nevertheless be considered minimums. Furthermore, due to a lack of ground counts to determine the ratio of incubating gulls to loafing gulls at the time of the aerial photography, counts of adults on-colony are an index to the number of breeding pairs using the colony. Methods to estimate colony size for gulls followed those of Adkins et al. (2014a) and are described in *Appendix A.1*. Although gull nesting success was not measured as part of this study, attempts were made to confirm whether fledglings were produced at each gull colony using ground and boat-based surveys at the onset of the fledging period. At mixed-species gull colonies (i.e. California and ring-billed gulls), species-specific counts were generally not conducted as species delineations within colonies were not always possible.

6.3.3.3 American white pelicans – Aerial photography taken during the late incubation period was used to estimate the number of breeding pairs using American white pelican colonies in the CRE (i.e. Miller Sands Spit and Rice Island), and to count the number of individual pelicans present at the large colony on Badger Island in the CPR (*Figure 6.1* above). Although most white pelicans present on these colonies were visible in the aerial imagery, a few nesting individuals were likely obscured by overhanging trees or shrubs, especially at the Badger Island colony where pelicans nest amongst dense sagebrush vegetation. Counts of American white pelican breeding pairs at colonies in the CRE should be considered minimum estimates, whereas counts of adult pelicans at Badger Island should be considered an index to the number of breeding pairs present. Although pelican nesting success was not measured as part of this study, attempts were made to confirm whether fledglings were produced at each pelican colony using ground and boat-based surveys at the onset of the fledging period.

6.3.3.4 Brandt's cormorants – Methods to estimate the number of breeding pairs of Brandt's cormorants on East Sand Island were the same as those for double-crested cormorants (see *Chapter 4*). In brief, three independent counts of the numbers of attended nests were derived from high-resolution vertical aerial imagery taken during peak incubation. Because Brandt's cormorants nest within the much larger double-crested cormorant colony on East Sand Island, species-specific areas of the colony were delineated based on in-season observations from on-colony blinds and by ground-truthing the cormorant colony after the breeding season to identify those cormorant nests that had been used by Brandt's cormorants as opposed to double-crested cormorants, based on the distinctly different nests built by the two cormorant species (BRNW 2010).

6.3.4 Limiting Factors for Colony Size & Nesting Success

Factors that may limit colony size or nesting success of piscivorous waterbirds at unmanaged colonies were recorded opportunistically when observed, including predation, availability of suitable nesting habitat, variable water levels, food availability, disease, displacement by other colonial waterbirds, and human disturbance. These data provide qualitative comparisons of factors that may limit colony size or nesting success of piscivorous waterbirds nesting at different breeding colonies within the CRB.

6.3.5 Dispersal & Colony Connectivity

6.3.5.1 Double-crested cormorants – No systematic data collection or studies of dispersal and colony connectivity for double-crested cormorants nesting in the CPR were conducted. The limited information on cormorant dispersal and colony connectivity in the CPR comes from three data sources: post-breeding surveys of the lower Snake River during 2008–2010, resightings of banded individuals or band recovery in the CPR, and satellite telemetry data from nesting cormorants tagged on East Sand Island (see *Chapter 4*).

During 2008–2010 an assessment of the post-breeding dispersal and overwintering use of the lower Snake River (LSR) portion of the FCRPS by double-crested cormorants and other piscivorous waterbirds was conducted by Cramer et al. (2011). In brief, during October–February of the winters of 2008-2009 and 2009-2010 monthly boat-based surveys of the LSR from the confluence with the Columbia River to Lewiston, Idaho were conducted (Rkm 0–224). Abundance and distribution of cormorants were estimated for each reach during each month of boat-based surveys and summed across all reaches to estimate total abundance for the LSR in each month. Densities for each reach were calculated by dividing the number of individuals observed in a reach by the length of the reach (Rkm). Any double-crested cormorants encountered within 0.25 Rkm of LSR dams were spatially assigned to that dam. Each dam was then assigned a fixed length of 0.5 Rkm to calculate the density of double-crested cormorants observed in the forebay and tailrace of each dam and compared to cormorant densities in each reach away from the dams (see Cramer et al. 2011 for a detailed description of survey and analytical methods).

No double-crested cormorants were banded or satellite-tagged in the CPR as part of this study. A total of 2,209 adult and juvenile double-crested cormorants were banded and/or tagged at East Sand Island in the CRE, and 1,985 of these marked individuals were banded with field-readable leg bands (see *Chapter 4*). Subsequent re-sighting effort and tracking in the CPR of cormorants banded/tagged in the CRE was opportunistic.

6.3.5.2 Gulls – No systematic data collection or studies of dispersal and colony connectivity for gull species in the CRB were conducted as part of this study. In some years, apparent connectivity between gull colonies was detected when a large increase in size of a gull colony occurred coincident with the failure or major decline of a nearby colony.

6.3.5.3 American white pelicans – No systematic data collection or studies of dispersal and colony connectivity for American white pelicans in the CRB were conducted as part of this study. Small numbers of American white pelican chicks were marked with field-readable leg bands in the CRE during the 2011–2012 breeding seasons, but subsequent re-sighting efforts have been limited. Post-breeding dispersal data are limited to observations during LSR surveys in 2008–2009 and 2009-2010 (see *above*) and resightings of banded birds at over-wintering sites in California.

6.3.5.4 Brandt's cormorants – No systematic data collection or studies of dispersal and colony connectivity for Brandt's cormorants nesting in the CRE were conducted as part of this study.

6.3.6 Foraging Behavior

Species-specific studies to better understand the foraging behavior of piscivorous waterbirds nesting at unmanaged colonies in the CRB are generally lacking. Information on the foraging behavior associated with predation/consumption on juvenile salmonid by double-crested cormorants, California and ring-billed gulls, and American white pelicans nesting in the CPR is limited to just a few studies (Evans et al. 2016a, Roby et al. 2016, Roby et al. 2017). These studies used last known detections of acoustic-tagged and PIT-tagged smolts (i.e. double-tagged) using the Juvenile Salmon Acoustic Telemetry System (JSATS, McMichael et al. 2010) and subsequent recoveries of PIT tags on bird colonies to evaluate species- and colony-specific foraging locations. Fish were tagged as part of smolt survival studies in CRB, with acoustic telemetry arrays deployed throughout the middle Columbia River, lower Snake River, and lower Columbia River. This network of telemetry arrays resulted in multiple spatial scales in which to evaluate predation/consumption by piscivorous waterbirds from colonies within foraging distance of out-migrating, tagged fish (see Evans et al. 2016a for a detailed description of these methods).

During the 2011 breeding season (April–July), 14 boat-based surveys to determine the numbers and distributions of double-crested cormorants were conducted, Caspian terns, and American white pelicans that were foraging, loafing, or roosting in a portion of the middle Columbia River above McNary Dam. Surveys were conducted from the mouth of the Walla Walla River (Rkm 509) upstream to Ice Harbor Dam (Rkm 538, Snake River) and to the upstream end of the Richland Islands (Rkm 545). The GPS coordinates of birds, along with information on the types of structures used as loafing or roosting sites were recorded during each survey. Similar data were not recorded for California and ring-billed gulls (see Roby et al. 2012 for a detailed description of these methods).

6.3.7 Diet Composition & Smolt Consumption

6.3.7.1 Double-crested cormorants – During 2005–2010, small numbers of adult double-crested cormorants nesting on Foundation Island were lethally sampled as they were commuting back to the colony after foraging trips to determine the diet composition of cormorants nesting at this colony. Diet samples from cormorants were also collected nondestructively by collecting regurgitations from chicks during the latter part of nesting season (see Collis et al. 2002 for a detailed description of methods). Because of small sample sizes of collected foregut contents and uneven distribution of collected samples across the breeding season within any sample year, samples were pooled across years. A total of 140 adult cormorants across all years were collected during seven different 2-week periods of the nesting season. The taxonomic composition of double-crested cormorant diets was expressed as the percentage of identifiable prey biomass and calculated for five 2-week periods during the nesting season. The diet composition of cormorants over the entire 10-week nesting season was based on the average of these 2-week percentages for samples collected during 2005–2010. All diet samples were analyzed using the same methods described in *Chapter 4* (see also Lyons et al. 2011 for a detailed description of methods specific to the Foundation Island cormorant colony).

The number of prey items consumed by double-crested cormorants nesting on Foundation Island during the 2005–2010 breeding seasons were estimated using a bioenergetics modeling approach after Roby et al. (2003), as modified by Lyons (2010). In brief, the model used several input parameters (i.e. number of consumers present [adults and chicks], energy requirements of adults and chicks, predator diet composition, and energy content of the various prey types consumed) to estimate the total energy requirements of the Foundation Island colony. Estimates of the numbers of prey consumed to meet energy demands of the cormorant colony were then derived from measurements of the proportion (% biomass) of salmonids and other prey types in the diet (see *above*), the mean mass of individual prey types, and the mean energy density of each prey type. Due to the relatively small size of the Foundation Island cormorant colony, it was not possible to collect enough samples in one year to accurately characterize diet for that year. Consequently, during 2005–2010 diet sampling was conducted during different periods of the cormorant breeding season to obtain a single, composite representation of the diet of cormorants nesting at the Foundation Island colony, with more samples collected during the out-migration period for juvenile salmonids from late April to early June. The length of the breeding season was partitioned into seven 2-week intervals, in each of which the number of smolts consumed was calculated. Inter-annual variation in the estimated numbers of juvenile salmonids and lamprey consumed by cormorants was a function of between-year differences in colony size, seasonal colony attendance, and fledging success. As mentioned above, because of constraints on diet sampling, diet composition was assumed to be the same across all years. Two techniques for converting salmonid biomass consumed by cormorants nesting on Foundation Island into numbers of juvenile salmonids consumed were compared. The first approach used genetic identification of salmonid soft tissue recovered from the foreguts of collected adult cormorants (*Appendix A*). The second approach presumed that cormorants took salmonids in proportion to their availability and used estimates of smolt passage at McNary Dam to estimate availability. The relative proportion of each salmonid species (i.e. Chinook salmon, sockeye salmon, steelhead) during each 2-week period of calculation was used, along with average smolt mass, to partition the salmonid biomass consumed by cormorants into numbers of each species consumed. Given the uncertainties associated with either approach (small sample sizes for genetics analysis or assumption of the number of smolts consumed in proportion to availability), results are reported as estimated total number of salmonid smolts consumed, without further breakdown to species (see *Appendix A* and Lyons et al. 2011 for a detailed description of these methods).

During the winters of 2008–2009 and 2009-2010, diet composition of double-crested cormorants over-wintering on the lower Snake River (LSR) was determined by lethally collecting cormorants (n = 130) between Lower Monumental Dam and Lower Granite Dam, where cormorant densities were observed to be comparatively high during monthly surveys (i.e. at dams, Lyons Ferry Bridge, Central Ferry Bridge). Taxonomic composition of each cormorant foregut sample was determined as the percent of identifiable prey biomass and followed processing methods identical to those described *above* and in *Chapter 4.* Prey composition was calculated for each month, with monthly collections averaged across all study years. Due to the opportunistic nature of diet collections and the uneven distribution of samples across years, diet composition data were pooled across all years. All diet samples collected from cormorant foreguts were also scanned for PIT tags from salmonid smolts. When present, detailed information from PIT tags were used to verify visual identification of prey species and to identify any ESA-listed salmonids in the cormorant diet. Cormorant diet data and abundance estimates were used to estimate the overall mass of salmonids consumed by cormorants over-wintering on the LSR (see Cramer et al. 2011 for a detailed description of these methods).

Estimates of the number of prey consumed by double-crested cormorants over-wintering on the LSR were performed as described above for cormorants nesting on Foundation Island but were modified based on several simplifying assumptions. First, daily energy requirements for over-wintering double-crested cormorants in the LSR have not been measured. Thus, calculations based on two different plausible scenarios were performed: (1) daily energy expenditure (DEE) was similar to that estimated for double-crested cormorants post-breeding (August–October) in the Columbia River estuary (Lyons 2010) and (2) DEE was similar to that measured in great cormorants (*Phalacrocorax carbo*) wintering in Germany (Keller and Visser 1999). Second, it was assumed that the average number of cormorants seen across the season during surveys (n = 256 individuals; see section *6.4.5 Dispersal & Colony Connectivity*) was a reasonable approximation of the number of cormorants that was present to prey upon salmonids across the winter. Third, a 5-month (154-day) period of possible predation (October–February) was assumed. All other model parameters and the bioenergetics model structure followed those used to estimate the number of smolts consumed by double-crested cormorants at breeding colonies (Lyons 2010). Due to small sample sizes and a lack of information on the life stage (juvenile, adult) of salmonids in the diet of cormorants over-wintering in the LSR, conversions of the total salmonid biomass consumed could not be translated into an estimate of the total numbers of salmonids consumed.

6.3.7.2 Gulls – During the 1997–1998 breeding seasons, limited data on diet composition of California and ring-billed gulls nesting at some colonies in the CPR were collected (Collis et al. 2002). Methods were like those for double-crested cormorants nesting at the Foundation Island colony, whereby foregut samples of lethally sampled adults were collected during the nesting season. Diet samples from gulls were also collected nondestructively by collecting regurgitations from chicks during the latter part of nesting season (see Collis et al. 2002 for a detailed description of methods).

6.3.7.3 American white pelicans – No systematic data collection or studies of diet composition for American white pelicans nesting in the CRB were conducted as part of this study.

6.3.7.4 Brandt’s cormorants – No systematic data collection or studies of diet composition for Brandt’s cormorants in the CRB were conducted as part of this study.

6.3.8 Predation/Consumption Rates on Juvenile Salmonids

A detailed description of methods used to estimate predation/consumption rates by double-crested cormorants, California gulls, ring-billed gulls, American white pelicans, and Brandt’s cormorants at unmanaged colonies in CRB are presented in *Appendix A.2*. Because California gulls, ring-billed gulls, and glaucous-winged/western gulls are known to consume dead, moribund, and injured smolts and that they kleptoparasitize (steal) smolts that have been depredated by other piscivorous waterbirds, we refer to their impacts on juvenile salmonids as "consumption rates". That is not the case for the other piscivorous colonial waterbirds (i.e. Caspian terns, double-crested cormorants, Brandt's cormorants, and American white pelicans) investigated here, as a result smolt impacts by these bird species are expressed as "predation rates".

Standardized predation/consumption rate estimates were derived by dividing the number of smolt PIT tags found on a given colony by the number of PIT-tagged smolts available, as indicated by detection at upstream dams. These raw estimates of predation/consumption rates were then adjusted for the proportion of PIT tags predated/consumed by avian predators that were subsequently deposited on their nesting colony (referred to as "deposition probability") and the proportion of PIT tags intentionally sown on the colony by researchers during the breeding season that were subsequently detected by researchers following the nesting season (referred to as "detection probability"). For gull and cormorant colonies, these methods account for tag loss associated with on-colony PIT tag detection and deposition probabilities, resulting in accurate estimates of predation/consumption (Hostetter et al. 2015). Accurate estimates of predation rates by American white pelicans nesting on Badger Island in McNary Reservoir were not available, however, because there were no empirical data on PIT tag deposition probabilities. As such, estimates of predation rates by American white pelicans reported here are minimums because, while corrected for PIT tag detection probabilities, they are not corrected for PIT tag deposition probabilities.

6.4 Results

6.4.1 Nesting Distribution

6.4.1.1 Double-crested cormorants – During 2003–2018, there were 13 different colony sites used by nesting double-crested cormorants in the CRB above the Columbia River estuary. Colony locations ranged from historical sites where cormorant nesting has occurred annually since the 1970s, to ephemeral sites used during a single nesting season. Colony locations were widely distributed along the mainstem Columbia and Snake rivers and across inland lakes and reservoirs in the CPR within potential foraging distance of the FCRPS (*Figure 6.1* above). On the Columbia River, nesting was documented at eight colony sites upstream of the Columbia River estuary, from Lewis and Clark Bridge at Longview, Washington (Rkm 106) and the power transmission towers near Troutdale, Oregon (Rkm 186) to the confluence with the Okanogan River (Rkm 859); cormorant nesting was confirmed at one colony site on the lower Snake River, at the Lyons Ferry Railroad Bridge. In addition, there were cormorant colonies at four inland waterbodies in the CPR: Potholes Reservoir, Sprague Lake, Lenore Lake, and Crab Creek. Of the 13 cormorant colony sites above the Columbia River estuary, six were arboreal colonies, four were on anthropogenic structures, and three were on the ground on islands. At arboreal colonies, cormorants were generally associated with nesting great blue herons (*Ardea* *herodias*), black-crowned night-herons (*Nycticorax nycticorax*), and/or great egrets (*A. alba*). Colonies on the ground were associated with ground-nesting gull colonies.

6.4.1.2 Gulls – During 2004–2018, there were 15 different colony sites in the CRB where California, ring-billed, and/or glaucous-winged/western gulls were confirmed to have nested, three of which were in the Columbia River estuary (*Figure 6.1* above). At some colony sites, nesting occurred on multiple islands in close proximity; thus, the actual number of distinct breeding sites is greater (see section *6.4.3 Colony Size & Nesting Success*). Colony locations ranged from historical sites with evidence of nesting dating back to at least the 1950s to those thought to have been colonized by gulls for the first time during this study. Eleven separate colonies were confirmed on the Columbia River from East Sand Island (Rkm 8) to Island 18 (Rkm 549) near Richland, Washington, and at four inland waterbodies within potential foraging distance of the FCRPS: Potholes Reservoir, Banks Lake, Sprague Lake, and Lenore Lake. Most gull colonies in the CPR were mixed-species colonies with both California and ring-billed gulls, while gull colonies in the CRE consisted of either glaucous-winged/western gulls or ring-billed gulls. During parts of the study period ring-billed gulls nested on either East Sand Island, Miller Sands Spit, or Rice Island in the CRE.

6.4.1.3 American white pelicans – During 2004–2018, by far the largest colony of American white pelicans in the CRB was on Badger Island (Rkm 512) in McNary Reservoir on the Columbia River (*Figure 6.1* above). Pelicans attempted to nest on both Badger Island and nearby Crescent islands (Rkm 510) in 2010, but all nesting attempts on Crescent Island failed and since then all white pelican nesting has occurred on Badger Island. Starting in 2010 American white pelicans also attempted to nest on two dredged material disposal islands in the CRE: Miller Sands Spit (Rkm 38) and Rice Island (Rkm 34). These four colony sites are the only known breeding sites for American white pelicans in the CRB.

6.4.1.4 Brandt's cormorants – During 2000–2018, Brandt's cormorants are known to have nested at three different sites in the CRE: a pile dike attached to the west end of East Sand Island, the upland portion of the western end of East Sand Island, and, beginning in 2018, on the Astoria-Megler Bridge (Rkm 21; *Figure 6.1* above).

6.4.2 Nesting Chronology

6.4.2.1 Double-crested cormorants – Nesting chronology data for double-crested cormorants nesting in the CPR were limited to data collected during the 2006–2012 breeding seasons at the Foundation Island colony, where nesting cormorants were observed beginning in March and the first eggs were seen by late-March (*Figure 6.3*). The first visible chicks in nests were seen usually during the third week in April, and the first fledglings were observed in early June. Colony attendance (i.e. the number of adult cormorants observed on-colony) typically peaked in late April, with few cormorants remaining on the colony by late July (*Figure 6.4*). Date ranges for the observation of the first egg and the first nestling are likely to be biased late to an unknown extent because the contents of most nests at the Foundation Island colony were difficult to determine for this arboreal colony from ground-based observation blinds. There were no comparable nesting chronology and colony attendance data for other double-crested cormorant colonies in the CPR because those sites were not visited as frequently as Foundation Island or access was limited and prevented such observations. Anecdotally, nesting chronology at other cormorant colonies in the CPR appeared to be similar to that at Foundation Island, with cormorants first observed at colonies in March, and the first fledglings observed in early June.

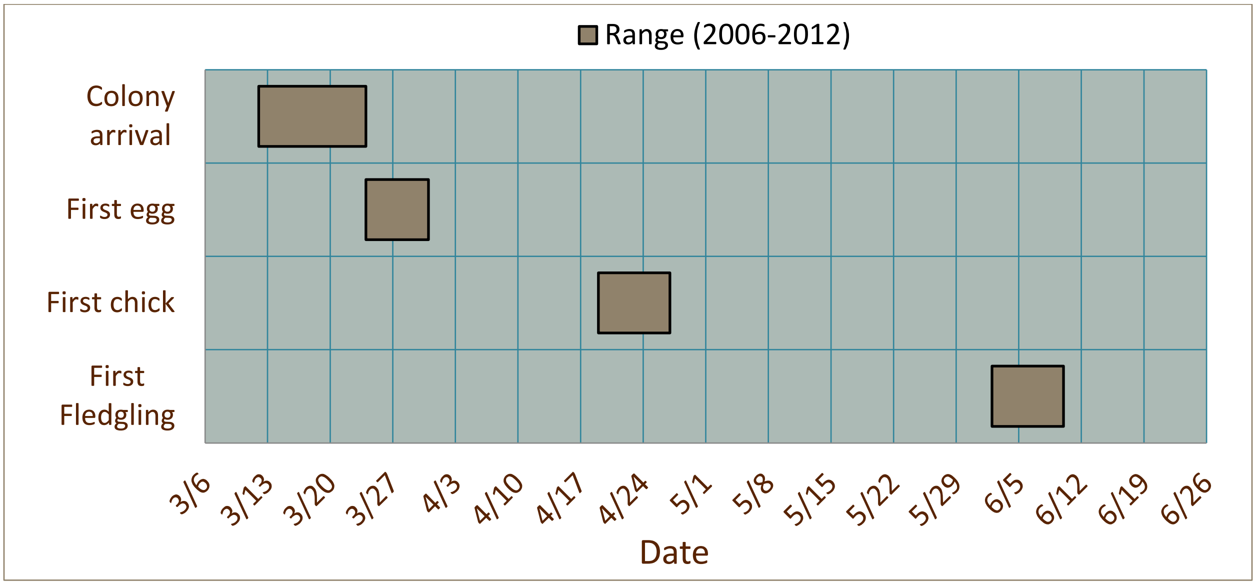

Figure 6.3. General nesting chronology for double-crested cormorants at the Foundation Island colony during 2006–2012.

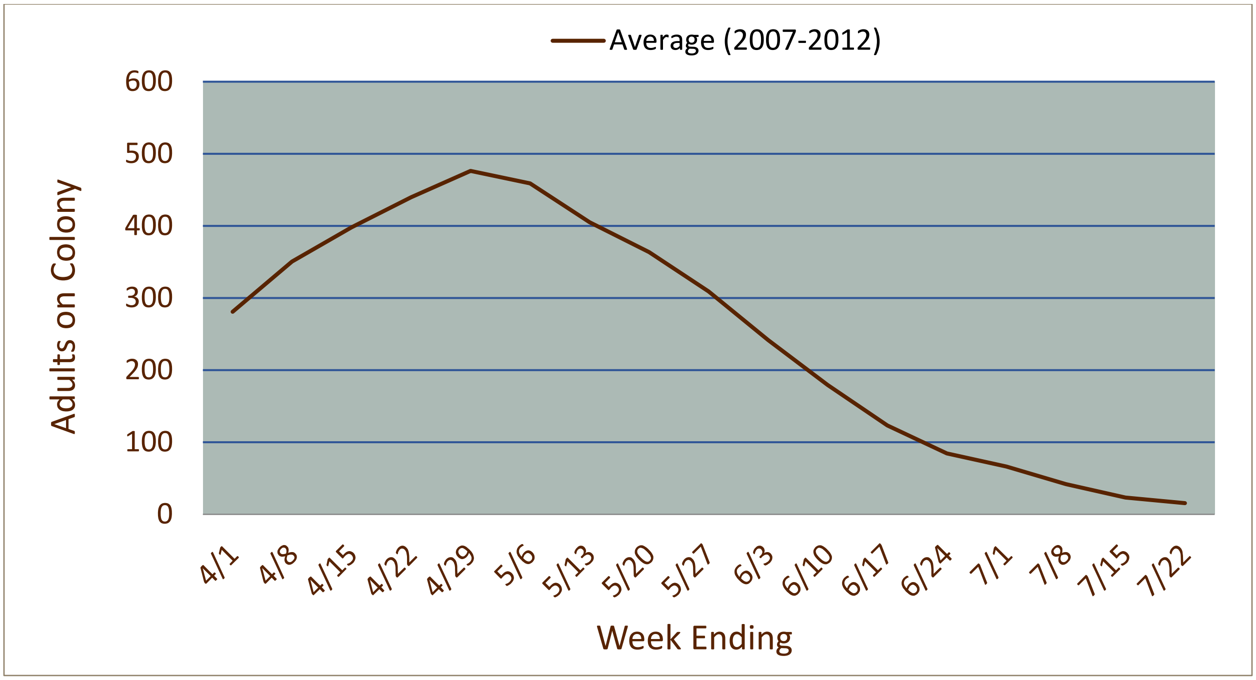

Figure 6.4. Average weekly number of adult double-crested cormorants counted on the Foundation Island breeding colony during 2007–2012.

6.4.2.2 Gulls – Data on nesting chronology and colony attendance for California and ring-billed gulls in the CPR are generally lacking. Standardized data were collected regularly at the mixed California and ring-billed gull colony on Goose Island, Potholes Reservoir during implementation of the *IAPMP* (2014–2018; see *Chapter 2*). By early March, when annual management and monitoring efforts were initiated, thousands of gulls were typically already present, and the first gull eggs were found in early April. The first gull chicks were observed in early May and the first gull fledglings in mid-June (*Figure 6.5*). Colony attendance generally peaked in early June, with most gulls dispersing from the Goose Island colony by late July (*Figure 6.6*).

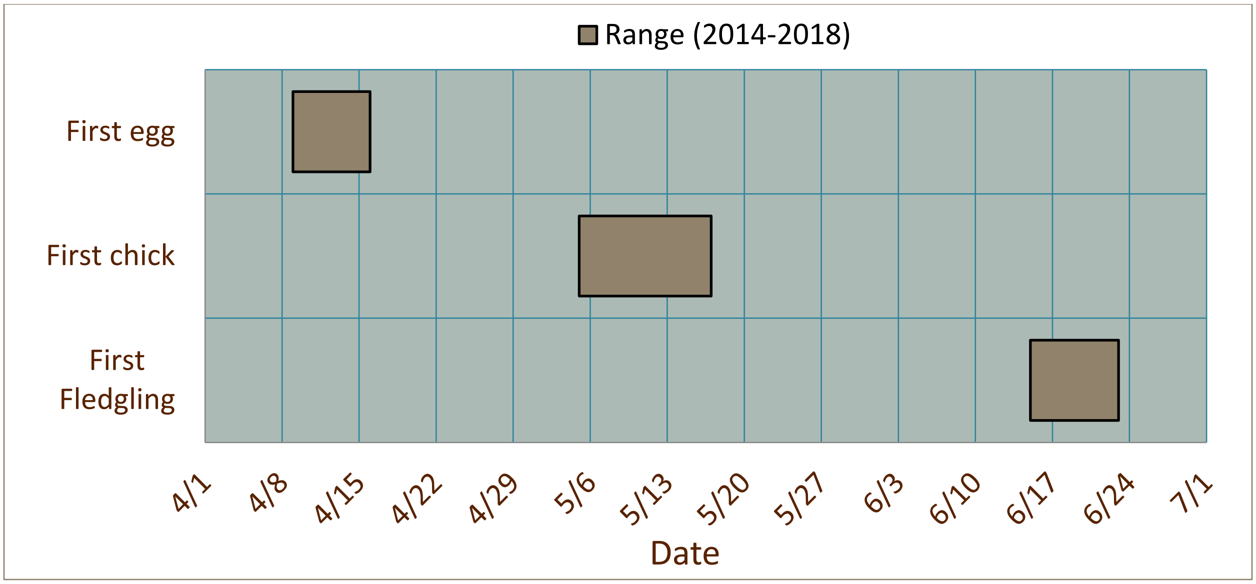

Figure 6.4. General nesting chronology for California and ring-billed gulls on Goose Island, Potholes Reservoir during 2014–2018.

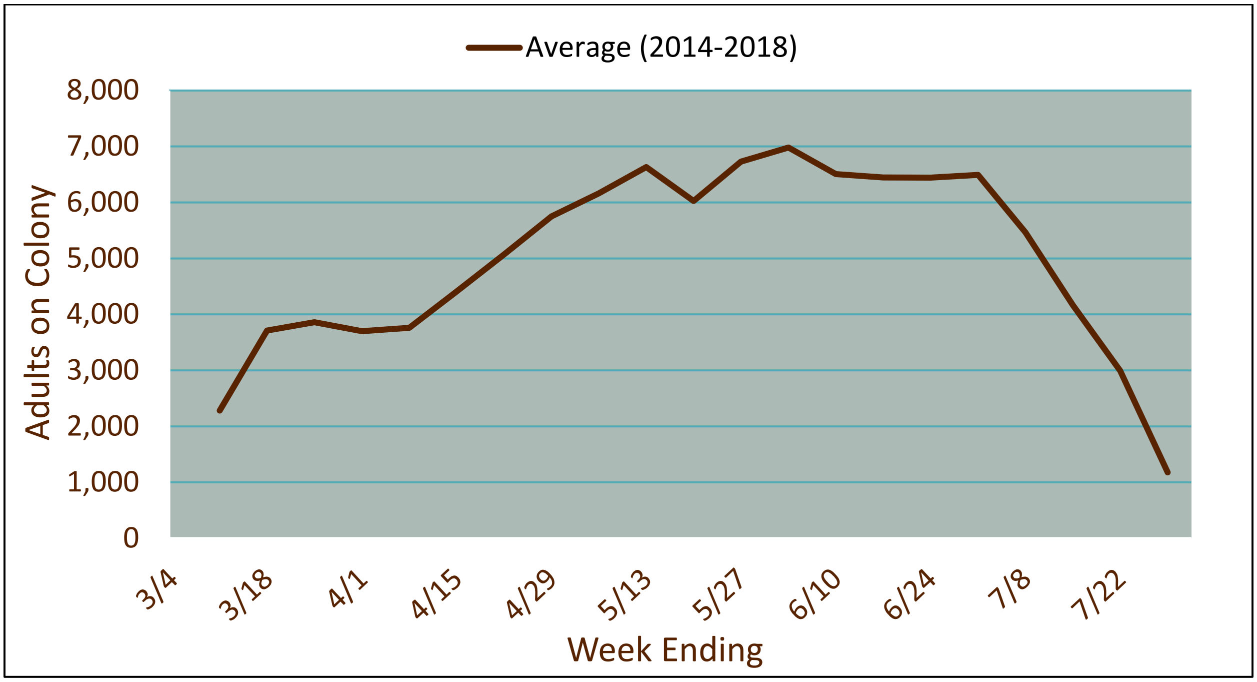

Figure 6.5. Average estimated number of adult California and ring-billed gulls attending the nesting colony on Goose Island, Potholes Reservoir during weekly boat-based surveys during the 2014–2018 breeding seasons.

Comparable data on nesting chronology and colony attendance at other gull colonies in the CPR were not collected because those colony sites were visited infrequently or constraints on access prevented such observations. Breeding chronology for gulls at colonies in the CPR were previously reported by Thompson and Tabor (1981) and Roby et al. (1997). Results herein are similar to those previously reported for other colonies, where egg-laying was first documented in mid-April and fledglings were first observed in late June. As part of the *IAPMP*, gull hazing to deter colony formation at Goose Island was attempted in some years. These efforts were unsuccessful and ultimately had little impact on the breeding chronology or size of the gull colony on Goose Island (see *Chapter 2; Table 6.2*). As such, it is believed that the nesting chronology observed for gulls nesting on Goose Island was generally representative of other gull colonies in the CPR. During 2006–2018, anecdotal observations at other gull colonies in the CPR (e.g., Miller Rocks, Crescent Island) indicated that gulls first arrived on-colony in mid-March, fledglings were first observed in mid-June, and colonies were vacated by late July.

In the CRE, nesting chronology data for gulls have not been collected recently. Roby et al. (1997) observed that breeding chronology for ring-billed gulls nesting in the CRE was similar to colonies in the CPR. In contrast, glaucous-winged/western gulls initiated nesting later (eggs first observed in mid-May) and chick fledging occurred into mid-August.

6.4.2.3 American white pelicans – Most of the American white pelicans nesting at the colony on Badger Island were concealed by vegetation during boat-based surveys and access to the island was not permitted during the nesting season because American white pelicans were listed as endangered by the State of Washington during most of the study period. During the study, pelicans were observed on Badger Island by late February and some pelicans remained present on the island until late October. Young-of-the-year were typically observed crèching along the shoreline of Badger Island beginning in early July.

6.4.2.4 Brandt's cormorants – Data on nesting chronology and colony attendance for Brandt's cormorants at the East Sand Island colony were collected opportunistically during research and monitoring of double-crested cormorants nesting on East Sand Island during 2006–2014. Prior to the initiation of double-crested cormorant management on East Sand Island in 2015, Brandt's cormorants generally initiated nesting somewhat later than double-crested cormorants, with the first eggs seen in early May, colony attendance peaking in mid-June, and first fledglings observed in late July (USACE 2015, see also *Chapter 4*).

6.4.3 Colony Size & Nesting Success

6.4.3.1 Double-crested cormorants – Complete census counts of the total number of double-crested cormorants nesting in the CPR were conducted during 2005-2013, and during this period the breeding population in the CPR was approximately stable, ranging from a low of about 1,150 breeding pairs in 2005 to a high of 1,575 breeding pairs in 2012 (*Figure 6.7, Table 6.1*). Limited double-crested cormorant research was conducted in the CPR after 2013, however, and precise estimates of cormorant colony size are not available for all colonies during 2014-2018. While precise colony size estimates after 2013 were generally limited to those colonies where smolt PIT tags were recovered as part of estimates of cormorant predation rates on juvenile salmonids (see *6.4.9 Per Capita Predation/Consumption Rates*) or at the request of resource managers, the rapid decline of the largest double-crested cormorant colony in the CPR at the north end of Potholes Reservoir (*Table 6.1*) suggests that the regional breeding population has recently declined. By 2018, 205 active cormorant nests were found at Potholes Reservoir, and in 2019 no active cormorant nests were found in the Potholes Reservoir area. Concurrent with the steep decline in the size of the Potholes Reservoir colony, cormorant colonies at Sprague Lake, Lenore Lake, and Crab Creek increased in size, but not commensurate with the decline at Potholes Reservoir (*Table 6.1*) Estimates of nesting success are not available for cormorant colonies within the CPR other than the colony on Foundation Island (see *below*). Brief summaries of nesting at cormorant colonies in the CPR where at least 50 active cormorant nests were counted are provided below.

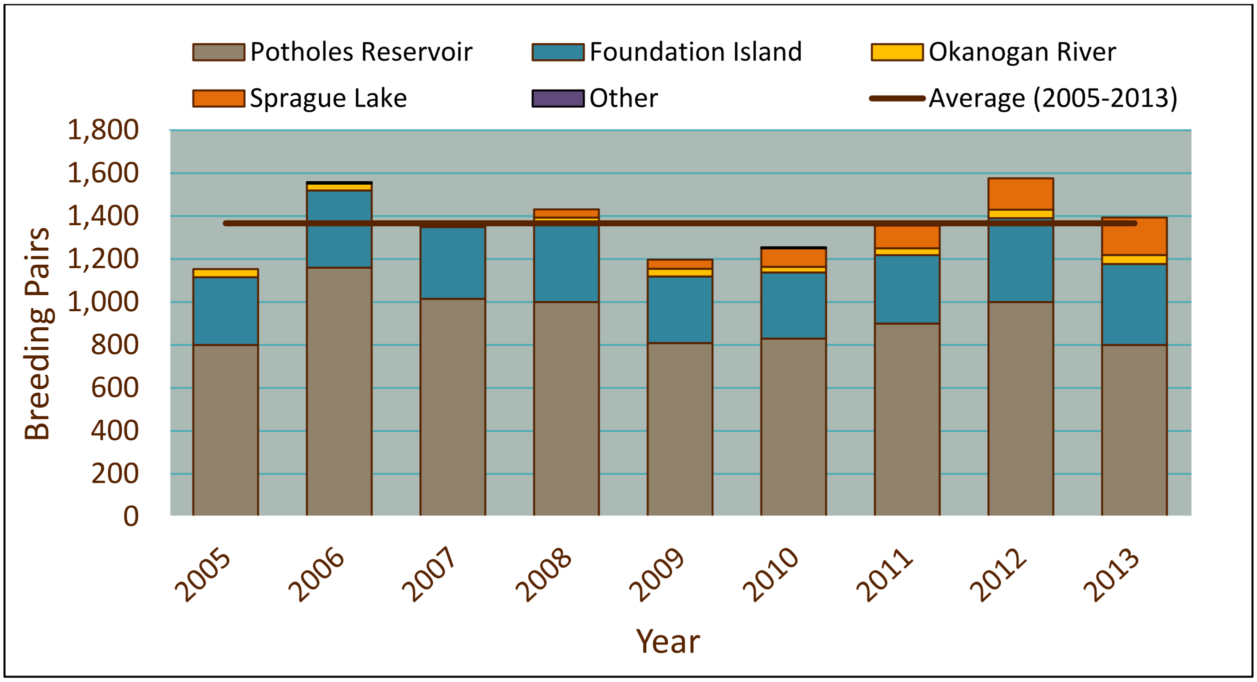

Figure 6.6. Total number of double-crested cormorant breeding pairs at colonies in the Columbia Plateau region during 2005–2013.

Table 6.1. Estimated peak number of breeding pairs of double-crested cormorants at colony sites in the Columbia Plateau region during 2003–2018.

	Lower Columbia River				**Mid-Columbia River**		**Lower Snake River**	**Potholes Reservoir**	**Sprague Lake**	**Lenore Lake**	**Crab Creek**
Year	Troutdale Towers	Miller Rocks	Foundation Island	Crescent Island	Hanford Reach - Locke Island Complex	Okanogan River Delta	Lyons Ferry RR Bridge	Colony Complex	Harper Is.	Shoal Is./ North Rock	Colony Complex
2003	--	--	200-300	0	--	--	--	--	--	0	--
2004	--	--	~300	0	--	20-30	--	300-500	--	0	--
2005	--	0	315	0	0	38	0	~800	0	0	--
2006	--	5	359	0	0	32	2	1,160	0	0	--
2007	--	0	334	0	8	10	0	1,015	0	0	--
2008	--	0	360	0	0	33	0	~1,000	38	0	--
2009	--	0	310	0	0	36	0	809	42	0	--
2010	--	0	308	5	0	26	0	~830	86	0	--
2011	--	2	318	15	0	32	0	~900	107	0	--
2012	--	0	390	0	0	40	0	~1,000	146	0	--
2013	B	0	377	0	0	42	0	~800	174	0	--
2014	80	0	B	0	6	49	0	B	266	5	--
2015	B	0	B	0[a]	25	50	0	B	336	39	--
2016	B	0	B	0[a]	35	46	0	146	420	66	--
2017	B	0	B	0[a]	B	B	0	346	302	B	--
2018	--	0	B	0[a]	68	B	0	205	B	111	387

[a] *= Active management used to completely dissuade or limit the number of Caspian terns breeding at the site; B = Breeding activity at the site but count not conducted; -- = Data not available.*

Lenore Lake: Double-crested cormorants first colonized a small, low-lying, rocky island referred to as “Shoal Island” in Lenore Lake near Soap Lake, Washington, in 2014, when 5 breeding pairs were observed (*Table 6.1* above). The colony grew in subsequent years and expanding to a second nearby rocky island referred to as “North Rock.” A high count of 111 breeding pairs of cormorants were observed nesting on the two islands at Lenore Lake in 2018.

Crab Creek: In 2018, 387 breeding pairs of double-crested cormorants were first observed nesting in several dozen inundated trees along a portion of Crab Creek near Wilson Creek, Washington. Prior nesting by cormorants at this location was not known, because this site was not surveyed during survey flights prior to 2018 (*Table 6.1* above).

Potholes Reservoir: The largest double-crested cormorant colony in the CPR has historically been located within the North Potholes Reserve at Potholes Reservoir, where cormorants have nested in dozens of inundated trees within the reserve. Cormorant nesting became increasingly dispersed among several sub-colonies in isolated patches of trees in four distinct areas (*Figure 6.1* above, *Appendix B.1*). The Potholes Reservoir colony was first established in the late 1970s and numbered 652 breeding pairs by 1997 (Finger and Tabor 1997). In more recent years, however, the numbers of cormorants nesting at this colony have varied considerably. The colony increased to approximately 1,160 breeding pairs by 2006. Although there was no clear trend in colony size between 2005 and 2013, the colony began to decline starting in 2014 and fewer than 350 breeding pairs were counted during the 2016–2018 nesting seasons (*Table 6.1* above). By 2019, no active cormorant nests could be found anywhere in the Potholes Reservoir area and the North Potholes colony had been completely abandoned.

Sprague Lake: In 2008, double-crested cormorants first colonized Harper Island at the southwest end of Sprague Lake near Sprague, Washington (*Figure 6.1* above, *Table 6.1* above). Nesting occurred on bare rocky outcrops at the north end of the island and, in some years, in at least one tree along the shoreline. In 2008, approximately 38 breeding pairs nested on Harper Island, and the colony subsequently increased to a high count of 420 breeding pairs in 2016.

Okanogan River mouth: During 2004–2018, double-crested cormorants nested in several trees at the mouth of the Okanogan River (Columbia River Rkm 859; *Figure 6.1* above). The colony ranged in size from 10 breeding pairs in 2007 to a high of 50 breeding pairs in 2015 (*Table 6.1* above).

Locke Island Complex: Double-crested cormorants have been documented nesting on islands in the Hanford Reach of the middle Columbia River since at least 1950 (Hanson 1968). During the present study, cormorants were first observed nesting in the Locke Island Complex in 2007, and again during 2014–2018. The colony consisted of a single tree on the downstream tip of an unnamed gravel island (Rkm 602; *Figure 6.1* above). Since 2014, the colony has grown from 6 breeding pairs to a high of 68 breeding pairs in 2018 (*Table 6.1* above).

Foundation Island: Foundation Island was consistently the largest double-crested cormorant colony on the Columbia River in the CPR and has remained an active colony since at least 1998. Foundation Island is a narrow, wooded island where cormorants nest in dozens of trees at the south end of the island. During 2004–2013, the Foundation Island colony fluctuated between approximately 300 and 390 breeding pairs, averaging 337 breeding pairs (*Table 6.1* above). Although colony size estimates were not available after 2013, the colony remained active and appeared to be within the colony size range listed above.

The average nesting success of cormorants at the Foundation Island colony during 2005–2012 was 2.16 fledglings raised per breeding pair and ranged from 1.37 fledglings/breeding pair in 2006 to 2.72 fledglings/breeding pair in 2010 (*Figure 6.8*). Estimates of nesting success at the Foundation Island cormorant colony were similar to those observed at the East Sand Island cormorant colony, where an average of 1.83 fledglings/breeding pair were produced during 1997–2013 (see also *Chapter 4*).

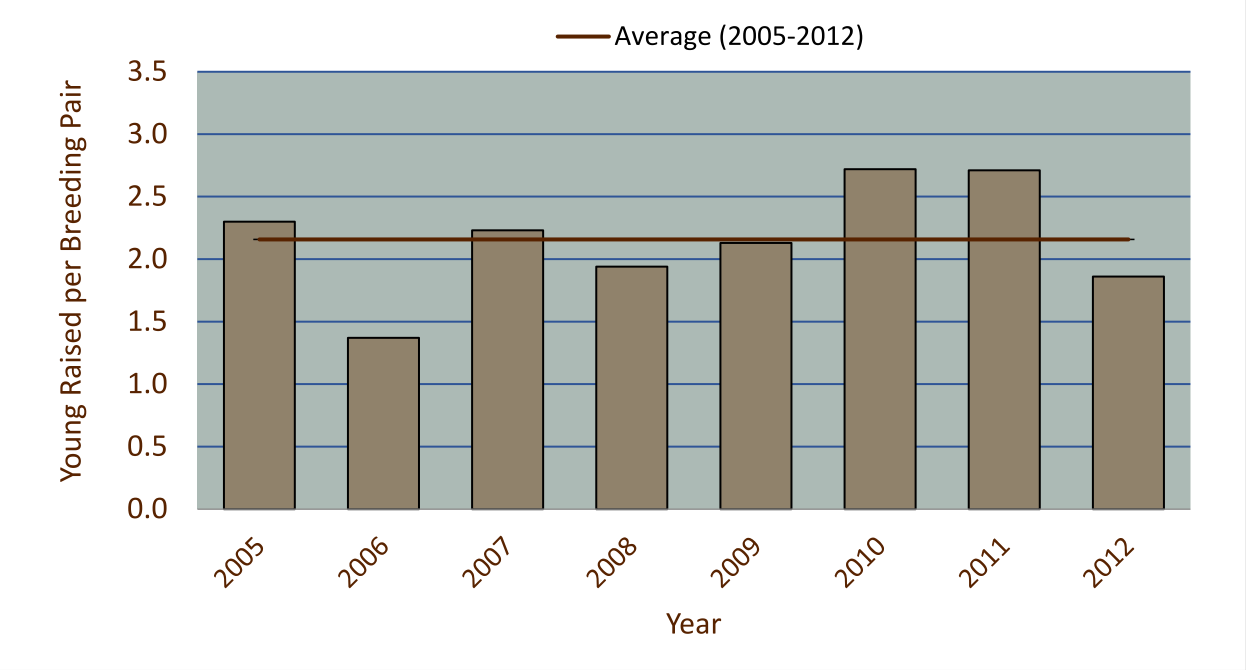

Figure 6.8. Estimated nesting success (average number of young raised per breeding pair) for double-crested cormorants nesting on Foundation Island during 2005–2012.

Troutdale Towers: Double-crested cormorants were first discovered nesting atop and within the rigid structure of several power transmission towers in the Columbia River near Troutdale, Oregon (Rkm 186; *Figure 6.1* above) during the 2013 breeding season. Nesting has been confirmed in all subsequent years, although the colony was only counted in 2014 when 80 breeding pairs were observed (*Table 6.1* above).

6.4.3.2 Gulls – Taken together, California and ring-billed gulls were the most abundant piscivorous colonial waterbirds in the CPR, where gull colonies were often at least an order of magnitude larger than colonies of Caspian terns, double-crested cormorants, or American white pelicans. Gulls, including glaucous-winged/western gulls, were confirmed nesting at up to 15 colony sites in the CRB during 2004–2018 (*Table 6.2* and *Table 6.3*). Most of these gull colonies were active in all years of the present study; however, these studies were not focused on gull colonies in the CRB because per capita consumption rates of juvenile salmonids by gulls were assumed to be comparatively low (Collis et al. 2002). Therefore, indices of breeding colony size are not available for all gull colonies in all years, and colony size indices are generally limited to those gull colonies where smolt PIT tags were recovered in order to estimate smolt consumption rates or at the request of resource managers (see also section *6.4.9 Per Capita Predation/Consumption Rates*). Due to intermittent observations at most gull colonies in the CRB, peak colony attendance was not always determined, with aerial surveys to estimate colony size generally limited to one or two occasions per year (see also *Chapter 9*). Also, diurnal patterns in colony attendance at gull colonies have been documented in the CPR, indicating that fewer gulls are present on-colony during the middle of the day (Conover and Miller 1980). As such, annual indices of peak breeding colony size for gull colonies in the CRB should be interpreted cautiously and generally limited to qualitative comparisons of relative abundance.

Table 6.2. Index counts of gulls (California gulls:ring-billed gulls) nesting at colony sites in the Columbia Plateau region during 1997–2018. Counts are of the number of individual gulls present on the colony at the peak of the nesting period. Total counts of gulls in bold; counts of two gull species in italics.

	Lower Columbia River						**Mid-Columbia River**		**Potholes Reservoir**	**Banks Lake**	**Sprague Lake**	**Lenore Lake**
Year	Little Memaloose Island	Three Mile Canyon Island	Miller Rocks	Blalock Islands Complex	Crescent Island	Badger Island	Island 20	Island 18	Goose Island	Twinning/ Goose Islands	Harper Island	Shoal Island/ North Rock
1997	**939** (*939:0*)	**13,305** (*nd*)	**3,783** (*nd*)	**0**	**5,769** (*5,769:0*)	**0**	**18,820** (*nd*)	**14,495** (*nd*)	--	--	--	--
1998	**357** (*357:0*)	**11,102** (*nd*)	**2,179** (*nd*)	**0**	**4,597** (*4,597:0*)	**0**	**22,348** (*nd*)	**12,669** (*nd*)	--	--	--	--
1999	--	--	--	--	--	--	--	--	--	--	--	--
2000	--	--	--	--	--	--	--	--	--	--	--	--
2001	--	--	--	--	--	--	--	--	--	--	--	--
2002	--	--	--	--	--	--	--	--	--	--	--	--
2003	--	--	--	--	--	--	--	--	--	--	--	--
2004	**0**	**B** (*nd*)	**B** (*nd*)	**0**	**B** (*nd*)	**0**	**B** (*nd*)	**B** (*nd*)	**B** (*nd*)	--	--	--
2005	**0**	**B** (*B:B*)	**B** (*B:B*)	**B** (*NB:B*)	**B** (*B:B*)	**0**	**B** (*B:NB*)	**B** (*NB:B*)	**B** (*B:B*)	**B** (*B:B*)	**B** (*B:B*)	--
2006	**0**	**B** (*B:B*)	**B** (*B:B*)	**B** (*NB:B*)	**B** (*B:B*)	**0**	**B** (*B:NB*)	**B** (*B:B*)	**B** (*B:B*)	**B** (*B:B*)	**B** (*B:B*)	--
2007	**0**	**B** (*B:B*)	**3,509** (*nd*)	**B** (*NB:B*)	**5,601** (*nd*)	**0**	**B** (*B:NB*)	**B** (*B:B*)	**B** (*B:B*)	**B** (*B:B*)	**B** (*B:B*)	--
2008	**0**	**B** (*nd*)	**4,443** (*nd*)	**B** (*NB:B*)	**8,567** (*nd*)	**0**	**20,999** (*nd*)	**B** (*nd*)	**B** (*B:B*)	**B** (*B:B*)	**B** (*B:B*)	--
2009	**0**	**6,161** (*5,868:293*)	**6,016** (*5,272:744*)	**1,631** (*0:1,631*)	**8,575** (*8,575:0*)	**0**	**19,341** (*9,946:9,395*)	**0**	**13,022** (*2,481:10,541*)	**6,602** (*3,737:2,865*)	**6,302** (*1,798:4,504*)	B (*nd*)
2010	**0**	**B** (*nd*)	**5,533** (*5,301:232*)	**B** (*nd*)	**B** (*nd*)	**0**	**B** (*nd*)	**0**	**B** (*nd*)	**B** (*nd*)	**B** (*nd*)	0
2011	**0**	**B** (*nd*)	**5,742** (*nd*)	**B** (*nd*)	**8,018** (*8,018:0*)	**0**	**B** (*nd*)	**0**	**11,392** (2,459:8,933)	**B** (*nd*)	**B** (*nd*)	--
2012	**0**	**0**	**4,509** (*nd*)	**8,989** (*B:B*)	**7,187** (*7,091:96*)	**0**	**B** (*nd*)	**0**	**12,005** (*3,684:8,321*)	**B** (*nd*)	**B** (*nd*)	--
2013	**0**	**0**	**4,807** (*4,757:50*)	**6,894** (*4078:2816*)	**5,707** (*5,555:152*)	**0**	**14,039** (*8,977:5,062*)	**0**	**12,797** (*3,007:9,790*)	**5,824** (*nd*)	**5,625** (*nd*)	804 (*nd*)
2014	**0**	**0**	**4,132** (*nd*)	**6,020** (*3,236:2,784*)	**6,404** (*5,652:752*)	**0**	**14,475** (*9,539:4,936*)	**0**	**14,334**[a] (*4,630:9,620*)	**B** (*nd*)	**B** (*nd*)	B (*nd*)
2015	**0**	**0**	**4,433** (*4,368:65*)	**7,377** (*4,756:2,621*)	**0**[a]	**3,740** (*nd*)	**16,557** (*12,009:4,548*)	**0**	**14,808**[a] (*5,180:9,628*)	**B** (*nd*)	**B** (*nd*)	743 (0:743)
2016	**0**	**0**	**3,734** (*nd*)	**6,741** (B:B)	**0**[a]	**4,126** (*nd*)	**14,316** (*nd*)	**0**	**13,273**[a] (*4,117:9,156*)	**B** (*nd*)	**B** (*nd*)	B (*nd*)
2017	**0**	**0**	**3,435** (*nd*)	**4,163** (*nd*)	**0**[a]	**4,505** (*nd*)	**11,176** (*nd*)	**0**	**11,225**[a] (*nd*)	**B** (*nd*)	**B** (*nd*)	B (*nd*)
2018	**0**	**0**	**4,284** (*nd*)	**3,408** (*nd*)	**0**[a]	**5,908** (*nd*)	**13,069** (*nd*)	**0**	**11,994**[a] (*nd*)	B (*nd*)	**B** (*nd*)	B (*nd*)

[a] = Active management to completely dissuade or limit the number of Caspian terns breeding at the site; B = Breeding activity observed but count not conducted; NB = No breeding activity confirmed at the site and count not conducted; Total counts of individuals presented in the table as an index of breeding activity; -- = Data not available; nd = gull species not differentiated in counts

Table 6.2. Index counts of gulls (glaucous-winged/western gulls:ring-billed gulls) nesting at colony sites in the Columbia River estuary from 1997–2018. Counts are of the number of individual gulls present on the colony at the peak of the nesting period. Total counts of gulls in bold; count of two gull species in italics.

Year	Rice Island	Miller Sands Spit	East Sand Island
1997	**1,583** (*1,583:0*)	**~1,460** (*1,268:< 200*)	**7,106** (*7,106:0*)
1998	**1,576** (*1,576:0*)	**~930** (*742:< 200*)	**5,496** (*5,496:0*)
1999	--	--	--
2000	--	--	--
2001	--	--	--
2002	--	--	--
2003	--	--	--
2004	**B** (*B:NB*)	**B** (*B:NB*)	**B** (*B:B*)
2005	**B** (*B:NB*)	**B** (*B:NB*)	**B** (*B:B*)
2006	**1,727** (*1,727:0*)	**704** (*704:0*)	**9,976** (*8,587:1,389*)
2007	**B** (*B:NB*)	**B** (*B:NB*)	**B** (*B:B*)
2008	**B** (*B:NB*)	**B** (*B:NB*)	**B** (*B:B*)
2009	**2,049** (*1,741:307*)	**160** (*160:0*)	**8,409** (*6,172:2,237*)
2010	**B** (*B:NB*)	**B** (*B:NB*)	**8,383** (*6,966:1,417*)
2011	**B** (*B:NB*)	**B** (*B:NB*)	**8,720** (*6,776:1,944*)
2012	**B** (*B:NB*)	**B** (*B:NB*)	**4,841** (*3,369:1,472*)
2013	**B** (*B:NB*)	**0**	**7,256** (*4,580:2,676*)
2014	**B** (*B:NB*)	**0**	**5,880** (*3,595:2,285*)
2015	**B** (*B:NB*)	**0**	**6,047** (*4,123:1,924*)
2016	**B** (*B:NB*)	**0**	**B** (*B:B*)
2017	**B** (*B:NB*)	**0**	**B** (*B:B*)
2018	**B** (*B:NB*)	**0**	**B** (*B:B*)

B = Breeding activity at the site but count not conducted; NB = No breeding activity confirmed at the site and count not conducted; Total counts of individuals presented in the table as an index of breeding activity; -- = Data not available

In the CPR, more comprehensive surveys of California and ring-billed gull colonies were conducted in 2009 and 2013. During these surveys, the total number of gulls counted on breeding colonies in the CPR was approximately 67,650 adults and 56,500 adults, respectively. In both years, the two largest gull colonies in the region were on Island 20 in the middle Columbia River (> 14,000 adults) and on Goose Island in Potholes Reservoir (> 11,000 adults). In 2009, roughly 62% of gulls nesting in the CPR were at colonies on islands in the Columbia River, with the remaining 38% nesting at colonies on islands in three inland waterbodies. Spatial distribution of nesting gulls was similar in 2013, when roughly 56% of all gulls nesting in the CPR were at colonies on islands in the Columbia River. Furthermore, based on these data and findings from Collis et al. (2002), a larger proportion of the California gulls that nested in the CPR used colonies on the Columbia River, while ring-billed gulls predominated at colonies located at off-river sites. Since 2009, the total number of gulls nesting at colonies on the Columbia River in the CPR has declined, from an index count of about 42,000 individuals in 2009 to a count of approximately 27,000 individuals in 2018 (*Figure 6.9*). It is unknown, however, whether a decline of this magnitude is also occurring at gull colonies in the CPR that are located off the Columbia River, where more recent data are lacking for most years.

Data on the species composition at gull colonies in the CPR (i.e. relative numbers of California gulls and ring-billed gulls) were limited to the two comprehensive regional surveys conducted in 2009 and 2013. In 2009, the total numbers of the two gull species counted on colonies in the CPR were ca. 37,680 California gulls and ca. 29,970 ring-billed gulls (*Figure 6.10*). Data on species composition at gull colonies in the CPR were incomplete in 2013 but were generally similar for colonies where the two species were counted separately in both years. At the two largest gull colonies in the CPR, California gulls predominated at the Island 20 colony, while ring-billed gulls predominated at the colony on Goose Island in Potholes Reservoir. The large gull colony on Island 20 was formerly about evenly divided into California and ring-billed gulls, but more recent declines in the numbers of ring-billed gulls nesting on Island 20 have resulted in a preponderance of California gulls nesting at this colony. The breeding populations of either species of gull in the CPR are clearly not increasing, and both species may be experiencing moderate regional declines.

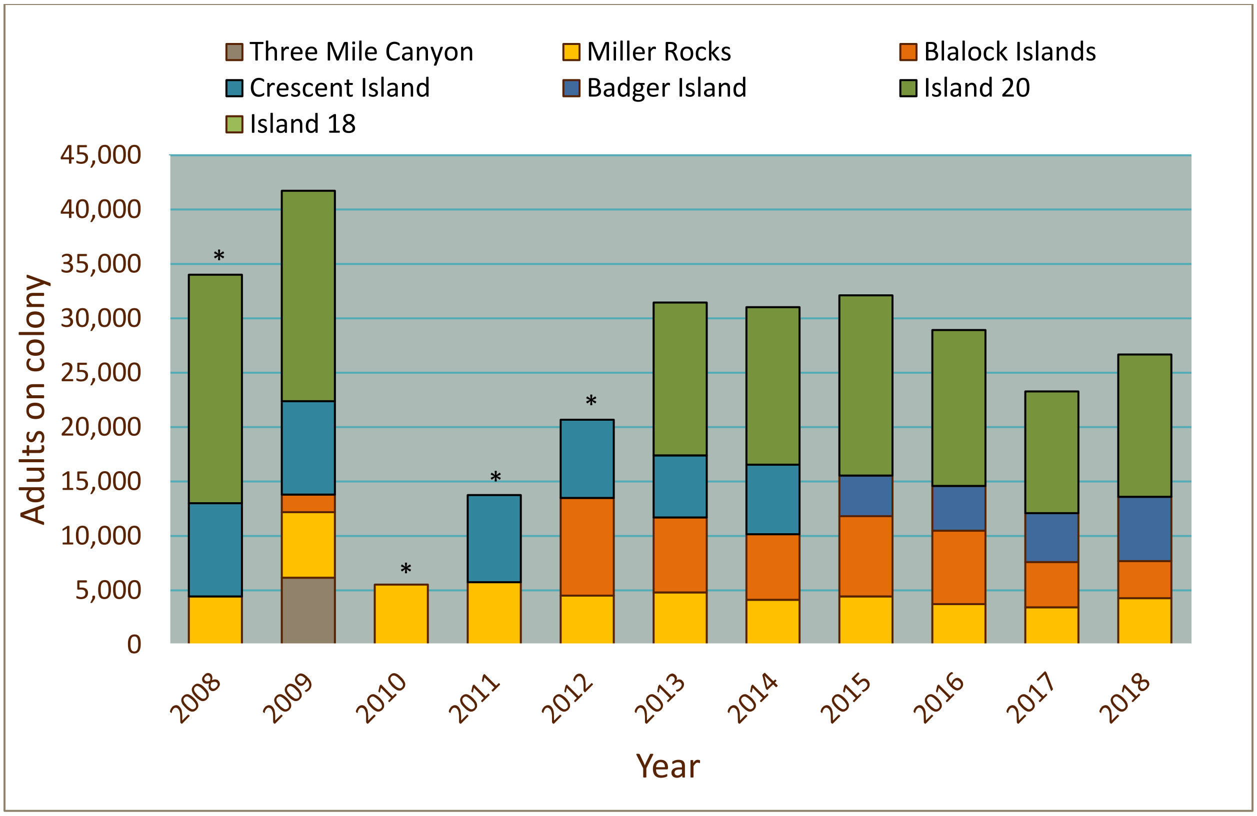

Figure 6.9. Total number of adult California and ring-billed gulls counted at colonies on the Columbia River in the Columbia Plateau region during 2008–2018. Counts of the number of adult gulls on-colony are an index to the number of breeding pairs and not adjusted to estimate colony size. Asterisks (*) denote years where partial counts of gull colonies were conducted (see also Table 6.2 above).

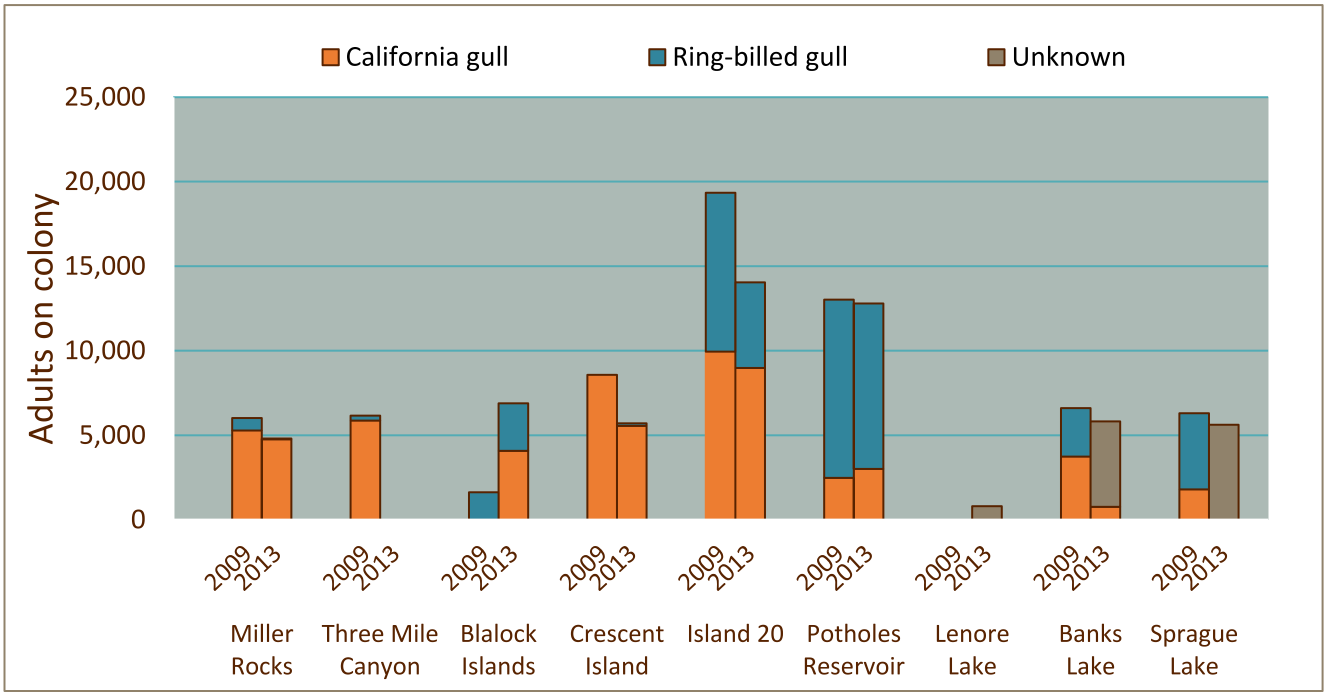

Figure 6.10. Relative species composition for gull colonies in the Columbia Plateau region during the two most recent complete colony censuses (2009 & 2013). Counts of the number of adult gulls on-colony are an index to the number of breeding pairs and are not adjusted to estimate colony size.

In the CRE, gulls, either glaucous-winged/western gulls, ring-billed gulls, or both, nested on three different islands during 2004–2018 (*Table 6.3* above). Comprehensive surveys of gulls nesting in the CRE were conducted in 2006 and 2009. During these surveys, the total number of nesting gulls counted in the CRE was approximately 12,400 and 10,600 adults, respectively. In both years, the largest gull colonies in the CRE were on East Sand Island, where large colonies of both glaucous-winged/western gulls and ring-billed gulls were present.

Precise estimates of nesting success are not available for gull colonies in the CPR or in the CRE, but anecdotal observations indicated that these colonies successfully fledged young in most years during the study period (see section *6.4.4 Limiting Factors for Colony Size & Nesting Success* for known exceptions).

A brief description of each gull nesting colony in the CRB is provided below:

Banks Lake: Nesting by California and ring-billed gulls at Banks Lake has occurred since at least the early 1970s (Conover et al. 1979). In the present study, nesting occurred in all years since 2005, typically on two rocky islands at the southern end of Banks Lake: Goose Island and the larger Twinning Island (*Figure 6.1* above). Gull colonies at Banks Lake were only enumerated in 2009 and 2013, and total counts of gulls were 6,602 and 5,824 breeding adults, respectively (*Table 6.2* above). In 2009, the only year in which the species composition of gulls nesting at

Banks Lake was determined, there were somewhat more California gulls (ca. 57%) than ring-billed gulls (ca. 43%; *Figure 6.10* above).

Lenore Lake: Gull nesting at Lenore Lake was first recorded in the late 1960s (Conover et al. 1979); however, gull nesting at Lenore Lake was not observed in this study until 2009. This colony was counted in 2013 and 2015, when a total of 804 and 743 adults on-colony were counted, respectively. The gull colony on Lenore Lake remained active through 2018 on both "Shoal" and "North Rock" islands. Data on species composition are not available for this mixed-species gull colony, but both California and ring-billed gulls were observed nesting on up to three different rocky islands in Lenore Lake during 2019.

Potholes Reservoir: Ring-billed gulls colonized the newly formed sand dune islands in northern Potholes Reservoir shortly after its formation in 1952 (Johnsgard 1954). California and ring-billed gulls were documented to be nesting on several islands in northern Potholes Reservoir again in 1977 (Conover et al. 1979). Nesting by gulls continued on various unnamed sand islands in Potholes Reservoir through 2005, notably on "Solstice Island" where Caspian terns also formerly nested; see also *Chapter 2*). Then in 2005 gulls first began nesting on Goose Island (Finger and Tabor 1997, BRNW 2006), a rock outcrop island in the southern part of the reservoir. Since 2005 all gull nesting at Potholes Reservoir has occurred on Goose Island; in some years, the Goose Island gull colony has been the largest gull colony in the CPR. During 2009–2018, an average of 12,7000 breeding gulls were counted on Goose Island during the peak of the nesting season, with a high of 14,808 counted in 2015 (*Table 6.2* above). Passive and active dissuasion used to deter Caspian terns from nesting at Goose Island during 2014–2019 do not appear to have negatively impacted the Goose Island gull colony because there was no apparent difference in the number of gulls nesting at this colony prior to and during the management period (see also *Chapter 2*). When measured in 2009 and 2013, species composition of gulls at the Goose Island colony was similar, with the colony dominated by ring-billed gulls in both years; about 81% of gulls attending the Goose island colony in 2009 were ring-billed gulls and about 77% of gulls attending the Goose Island colony in 2013 were ring-billed gulls (*Figure 6.10* above).

Sprague Lake: Nesting by gulls on Harper Island in Sprague Lake is thought to date back to the late 1940s (Conover et al. 1979). During the present study, nesting by gulls has occurred in all years since 2005, but the gull colony was only counted in 2009 and 2013, when a total of 6,302 individual adults and 5,625 individual adults, respectively, were enumerated. In 2009, the only year when data on the species composition of this mixed-species gull colony were available, the colony consisted predominately of ring-billed gulls (ca. 71%; *Figure 6.10* above).

Richland Islands: The Richland Islands complex (Rkm 543-551) is a series of low-lying sand and gravel islands typically surrounded by shoreline vegetation on the middle Columbia River near Richland, Washington (*Figure 6.1* above). Historically, these islands have supported large colonies of both California and ring-billed gulls. In the early 1970s, Conover et al. (1979) found gulls nesting on Island 18 (Rkm 549) and Island 20 (Rkm 545). These same colonies were again active during 1997–1998 when Collis et al. (2002) counted more than 30,000 gulls nesting on the two islands combined. During the present study, these two colonies, although not enumerated, remained active during 2004–2008. Following the abandonment of the gull colony on Island 18 in 2009, Island 20 was the only active gull colony in the Richland Islands complex during 2009–2018. Counts of gulls on Island 20 during the peak of the nesting season ranged from a low of 11,176 individuals in 2017 to a high of 21,000 in 2008, with large numbers of both California and ring-billed gulls present in all years (*Table 6.2* above*, Figure 6.10* above).

Badger Island: Gulls first colonized Badger Island (Rkm 512) in 2015 following the abandonment of the gull colony on nearby Crescent Island (see *below* and *Figure 6.1* above). The Badger Island gull colony has increased in size in each successive year, growing from ca. 3,740 individuals in 2015 to ca. 5,908 individuals in 2018 (*Table 6.2* above). Gull nesting has occurred in two primary areas of Badger Island: (1) along the eastern shoreline in the woody wrack line and (2) amongst nesting American white pelicans in openings in the densely vegetated shrubby interior of the island. Species composition of the gull colony on Badger Island has not been determined but is believed to be comprised almost entirely of California gulls, similar to the previous gull colony on Crescent Island (see *below*).

Crescent Island: Gulls colonized Crescent Island (Rkm 510) shortly after the island was built from dredged material in 1985; the colony was first counted in 1997 by Collis et al. (2002), when 5,769 California gulls were enumerated. The large California gull colony on Crescent Island occupied virtually all suitable gull nesting habitat surrounding the Caspian tern colony (see *Chapter 2*) and was active continuously through the 2014 breeding season. The size of the gull colony on Crescent Island peaked in 2009 at 8,575 individuals (*Table 6.2* above). Management implemented as part of the *IAPMP* during 2015–2019 was successful in not only achieving the objective of preventing Caspian terns from nesting on the island, but gulls also abandoned the Crescent Island colony (see also *Chapter 2*). A few ring-billed gulls have also historically nested on Crescent Island, with 152 individuals counted during the colony census in 2013.

Blalock Islands Complex: Nesting by gulls in the Blalock Islands complex near Boardman, Oregon was first documented in 2005, and counts during the peak of the nesting season increased from about 1,631 individuals in 2009 to more than 8,900 individuals in 2012 (*Figure 6.1* above, *Figure 6.9* above). Mixed-species colonies of California and ring-billed gulls have been active on up to three separate islands in the Blalock Islands complex. Relatively small colonies (< 1,700 individuals) have been observed on Straight Six Island (Rkm 440; mainly ring-billed gulls) and Rock Island (Rkm 441). The number of gulls nesting at the Blalock Island complex increased dramatically in 2012, coincident with the abandonment of the gull colony on Three Mile Canyon Island (Rkm 412; see *below*). Since 2012, most gulls nesting in the Blalock Islands have nested on the larger Anvil Island (Rkm 440), where roughly 83% of all gulls nesting in the Blalock Islands were counted in 2013 (BRNW 2014). During 2012–2016, the numbers of gulls nesting in the Blalock Islands appeared stable, with counts ranging from ca. 6,000 to ca. 8,990 individuals (*Table 6.2* above). Gull counts declined in subsequent years, however, with only 3,408 individuals counted in 2018. Although the cause of this decline is unknown, it could be due at least in part to vegetation encroaching on the sparsely vegetated portions of the islands where gulls prefer to nest; vegetation encroachment has been noted on both Straight Six and Anvil islands.

Three Mile Canyon Island: Nesting by California and ring-billed gulls on Three Mile Canyon Island (Rkm 412) in John Day Reservoir of the Columbia River near Boardman, Oregon has been documented since at least 1968 (Conover et al. 1979). This gull colony was active during 1997–1998, when 13,305 individual adults were counted during the peak of the 1997 nesting season (Collis et al. 2002). During the present study, nesting by gulls at Three Mile Canyon Island was documented from 2004 to 2011, when the gull colony was abandoned. During 2004-2011, the gull colony was only counted in 2009, when 6,161 individuals, predominately California gulls, were enumerated (*Table 6.2* above).

Miller Rocks: California and ring-billed gulls were first documented breeding at Miller Rocks (Rkm 331), just up-river of Miller Island in The Dalles Reservoir on the Columbia River, in 1977 by Conover et al. (1979) and confirmed in 1997 by Collis et al. (2002). Since 2004, the mixed-species gull colony has mainly consisted of California gulls, which have nested on several rocky islets. The size of the gull colony peaked in 2009, when 6,016 individuals were counted, but gull colony size at Miller Rocks has averaged about 4,500 breeding adults during 2007–2018 (*Table 6.2* above).

Little Memaloose Island: California gulls were first documented breeding on Little Memaloose Island (Rkm 315), just upstream from the Dalles Dam, in 1977 by Conover et al. (1979). The colony was confirmed active in 1997 by Collis et al. (2002), when 939 individuals were counted during the peak of the nesting season. The California gull colony on Little Memaloose Island was subsequently abandoned, however, and since the onset of annual aerial surveys in 2004 no gull nesting has been observed on the island.

Rice Island and Miller Sands Spit: Glaucous-winged/western gulls were first documented nesting at the western (downstream) end of Rice Island (Rkm 34; *Figure 6.1* above) in 1997 by Collis et al. (2002) and nested continuously there during 2004–2018. In 2009, 1,741 glaucous-winged/western gulls and 307 ring-billed gulls were counted on Rice Island during the peak of the nesting season, but no subsequent counts are available (*Table 6.3* above). Glaucous-winged/western gulls and ring-billed gulls were also observed nesting at the western (downstream) end of nearby Miller Sands Spit (Rkm 38) during 1997–1998 (Collis et al. 2002), when 1,268 glaucous-winged/western gulls and < 200 ring-billed gulls were counted. The ring-billed gull colony on Miller Sands Spit was subsequently abandoned in 2004, but glaucous-winged/western gulls continued to nest on Miller Sand Spit during 2004–2012, although the numbers of nesting gulls were in decline until the colony was completely abandoned in 2013 (*Table 6.3* above).

East Sand Island: Glaucous-winged/western gulls were documented nesting on East Sand Island (Rkm 8; *Figure 6.1* above) in the late 1980s (Bell 1997), and nesting was confirmed in 1997 when more than 7,000 individual adults were counted during the peak of the nesting season (Collis et al. 2002). Glaucous-winged/western gulls nested on East Sand Island each year during

2004–2018, using a variety of upland habitats, including sandy substrate, sparse vegetation, and the woody wrack line scattered along the shoreline of the island. Most nesting by glaucous-winged/western gulls on East Sand Island has been at the eastern end of the island near the large Caspian tern colony (see also *Chapter 1*). Ring-billed gulls first colonized the northeastern tip of East Sand Island in 2004, apparently by birds that had abandoned the colony on Miller Sands Spit (see *above*). This ring-billed gull colony expanded and persisted during 2004-2018, becoming by far the largest ring-billed gull colony in the Columbia River estuary. Although gull colony counts in *Table 6.3* (above) are not separated by gull species, species composition was determined in some years. During 2009–2015 glaucous-winged/western gulls comprised about 61-83% of all gulls counted on East Sand Island (*Figure 6.11*). During the most recent gull census on East Sand Island in 2015, ca. 4,123 glaucous-winged/western gulls and ca. 1,924 ring-billed gulls were counted on their respective colonies during the peak of the nesting season (Roby et al. 2016a). Numbers of glaucous-winged/western gulls nesting on East Sand Island had declined significantly during the previous decade, whereas the numbers of ring-billed gulls had increased substantially.

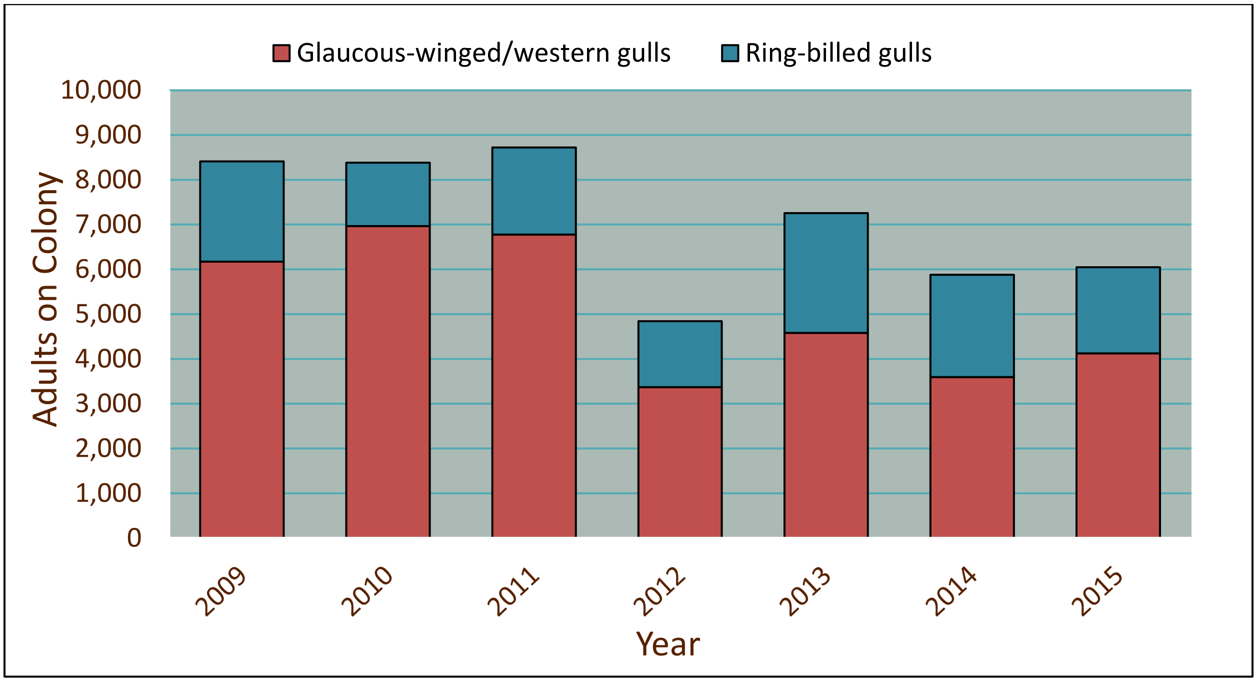

Figure 6.11. Approximate species composition for gulls nesting on East Sand Island in the Columbia River estuary during 2009–2015. Counts of the number of adult gulls on-colony are an index to the number of breeding pairs and are not adjusted to estimate colony size.

6.4.3.3 American white pelicans – During 2004–2018, American white pelicans successfully nested each year at the colony on Badger Island in the CPR (Rkm 512). The only other known location in the CPR where American white pelicans have attempted to nest was on nearby Crescent Island (Rkm 510) in 2010 (*Table 6.4*), but this nesting attempt failed (see section *6.4.4 Limiting Factors for Colony Size & Nesting Success*). No nesting by American white pelicans was recorded in the Columbia River estuary until a small colony (< 50 breeding pairs) was discovered on Miller Sands Spit (Rkm 38) in 2010, where pelicans continued to nest until 2016. Beginning in 2017, the American white pelican colony in the Columbia River estuary moved to Rice Island (Rkm 34), where they nested again the following year (*Table 6.4*). The total number of American white pelicans nesting in the CRB has grown substantially in the last 15 years, from over 1,057 individuals in 2005 to over 3,700 individuals in 2018 (*Table 6.4*).

Table 6.3. Estimated size of American white pelicans breeding colonies in the Columbia River basin during 2004-2018. The size of pelican colonies in the Columbia River estuary is expressed as the estimated number of breeding pairs at the peak of the nesting season, whereas the size of pelican colonies in the Columbia Plateau region is expressed as the number of individuals on-colony at the peak of the nesting season, an index to colony size.

	Columbia River Estuary		**Columbia Plateau Region**	
Year	Rice Island	Miller Sands Spit	Badger Island	Crescent Island
2004	0	0	B	0
2005	0	0	1057	0
2006	0	0	1310	0
2007	0	0	913	0
2008	0	0	1349	0
2009	0	0	1754	0
2010	0	42	2048	50
2011	0	194	2228	0
2012	0	122	2075	0
2013	0	104	2077	0
2014	0	376	2447	0
2015	0	B	3267	0[a]
2016	0	351	3118	0[a]
2017	102	76[b]	2513	0[a]
2018	398	246[b]	3330	0[a]

[a] = Active management used to completely dissuade or limit the number of Caspian terns breeding at the site; [b] = Colony failure occurred at Miller Sands Spit with subsequent re-nesting on Rice Island; B = Breeding activity at the site but count not conducted; -- = Data not available.

Badger Island: The American white pelican colony on Badger Island in McNary Reservoir on the Columbia River has grown substantially since monitoring began in 2004, with the number of adults counted on-colony at the peak of the nesting season increasing from ca. 1,060 in 2005 to ca. 3,330 in 2018 (*Figure 6.12*). The count of 3,330 pelicans recorded in 2018 is the highest total observed during the study period and suggests continued growth in the size of this colony is possible. Estimates of nesting success are not available for the Badger Island white pelican colony. Observations of young-of-the-year along the shoreline in late July and August indicate that pelicans have been successful in fledging at least some young in all years of this study.

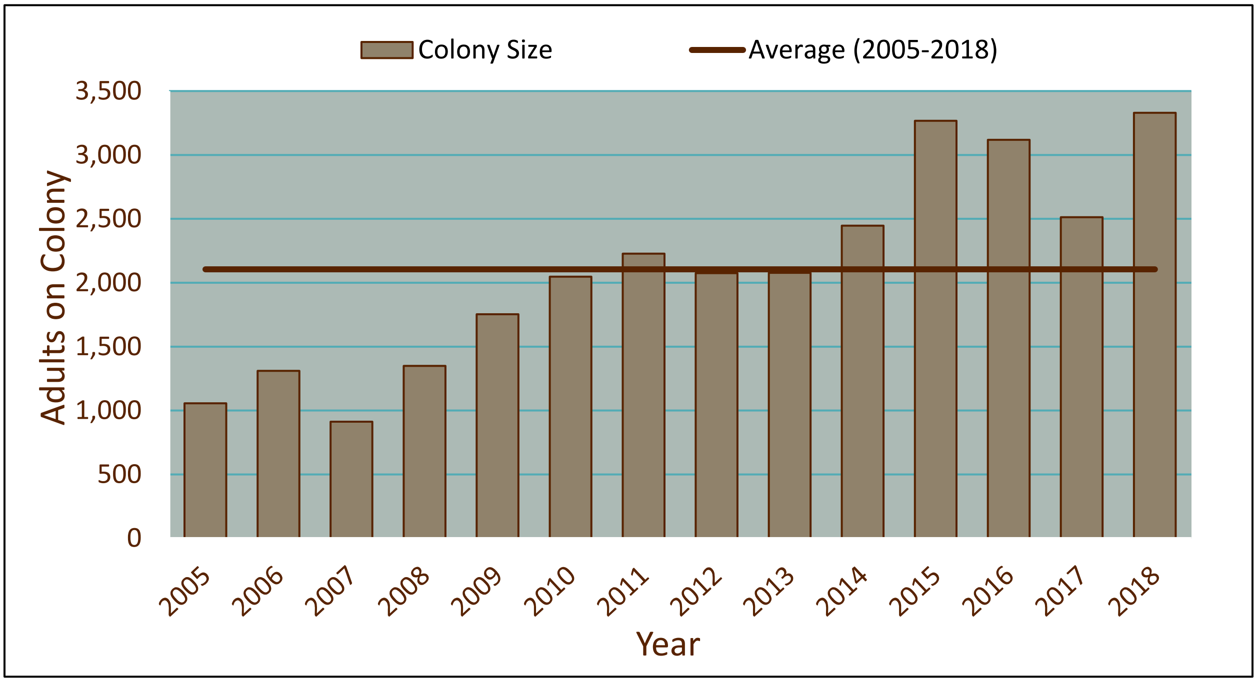

Figure 6.12. Total number of American white pelicans counted on the Badger Island colony at the peak of the nesting season during 2005–2018. Counts of the number of adult pelicans on-colony are an index to colony size (number of breeding pairs) and have not been adjusted to estimate the number of breeding pairs.

Rice Island & Miller Sands Spit: The first record of American white pelicans nesting in the CRE was at Miller Sands Spit in 2010, when 42 breeding pairs were counted near the western (downstream) end of this dredged material disposal island. Nesting attempts by white pelicans were confirmed on Miller Sands Spit in all subsequent years of the study, with estimated colony size ranging from 42 to 376 breeding pairs. In 2017, a breeding colony of American white pelicans was also detected on nearby Rice Island, following the mid-season abandonment of the colony on Miller Sands Spit. Nesting by white pelicans was again recorded on Rice Island in 2018, and resulted in the largest American white pelican colony recorded during the study period in the CRE (398 breeding pairs; *Table 6.4* above). While estimates of nesting success are not available, American white pelicans were successful in raising some young at the Miller Sands Spit colony in some years. In some years, the pelican colony was abandoned part-way through the chick-rearing period, apparently due to disturbance by boaters, campers, and fishers landing on the island. Nesting success at the pelican colony on Rice Island has not been estimated, but this colony appears to be at a site less prone to human disturbance than was the case at Miller Sands Spit.

6.4.3.4 Brandt's cormorants – Brandt's cormorants were first detected nesting at East Sand Island on a pile dike off the west end of the island in 1997 (Couch and Lance 2004; see also *Table 6.5*). A small colony remained active at this pile dike through the 2005 nesting season, before shifting in 2006 to East Sand Island itself, where Brandt's cormorants began nesting amongst the much larger double-crested cormorant colony (see also *Chapter 4*). During 2006–

2018, the Brandt's cormorant colony on East Sand Island grew from fewer than 50 breeding pairs to more than 2,100 pairs in 2018 (*Figure 6.13* and *Table 6.5)*, making it one of the largest Brandt's cormorant breeding colonies recorded along the coasts of Washington and Oregon (Speich and Wahl 1989, Naughton et al. 2007). In 2018 Brandt's cormorants were documented for the first time nesting on the cribs at the base of the Astoria-Megler Bridge, where double-crested cormorants had recently started to nest. Nesting on anthropogenic structures in estuaries is uncommon for Brandt's cormorants, with most breeding colonies found on isolated rocky islands or sea cliffs along the Pacific Coast of North America (Wallace and Wallace 1998).

Table 6.4. Estimated peak number of breeding pairs of Brandt's cormorants at colony sites in the Columbia River estuary during 1997–2018.

Year	Astoria-Megler Bridge	West Pile Dike – East Sand Island	East Sand Island
1997	0	B	0
1998	0	B	0
1999	0	B	0
2000	0	31	0
2001	0	30-50	0
2002	0	86	0
2003	0	97	0
2004	0	58	0
2005	0	62	0
2006	0	NA	44
2007	0	NA	288
2008	0	NA	508
2009	0	NA	684
2010	0	NA	985
2011	0	NA	1,491[a]
2012	0	NA	1,684[a]
2013	0	NA	1,523[a]
2014	0	NA	1,630
2015	0	NA	2,071[b]
2016	--	NA	1,515[b]
2017	--	NA	1,893[b]
2018	14	NA	2,120[b]

[a] = Active management (hazing) used to limit where double-crested cormorants could nest on East Sand Island; [b] = Active management (culling and egg-oiling) used to limit the size of the double-crested cormorant colony on East Sand Island; B = Breeding activity at the site but count not conducted; NA = Site not available as nesting habitat; -- = Data not available.

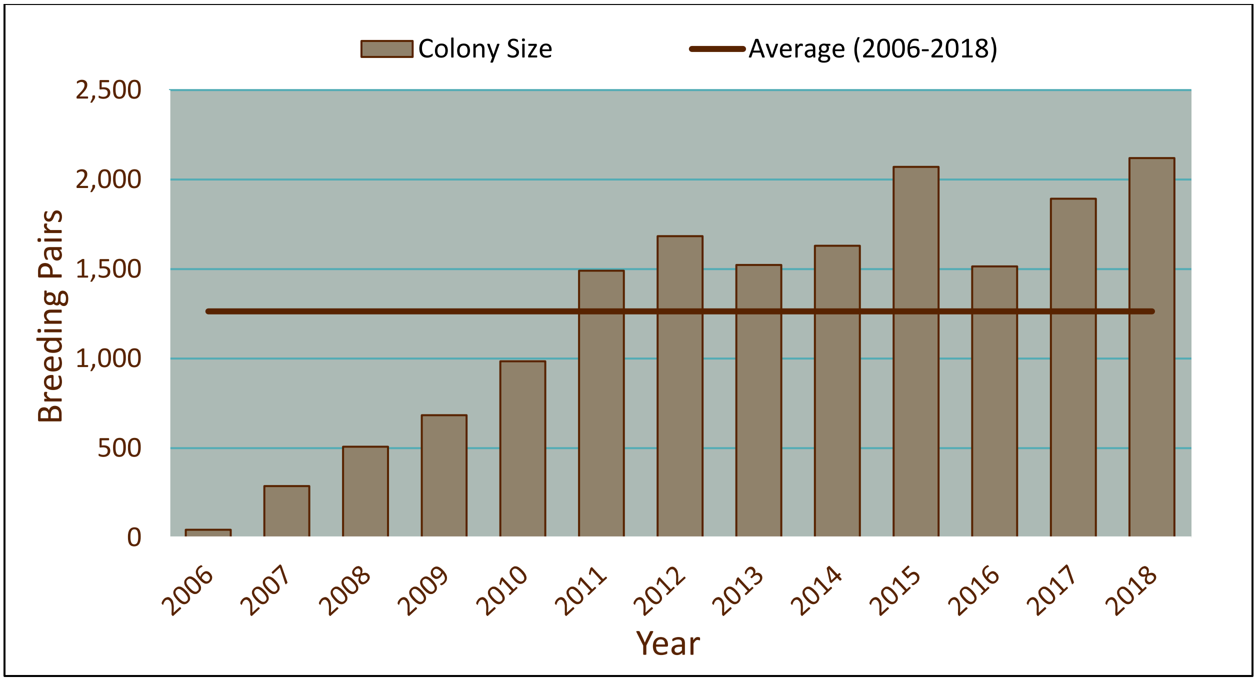

Figure 6.13. Number of breeding pairs of Brandt's cormorants counted in the cormorant colony at the west end of East Sand Island in the Columbia River estuary during 2006–2018.

6.4.4. Limiting Factors for Colony Size & Nesting Success

6.4.4.1 Double-crested cormorants – Unlike Caspian terns, California gulls, ring-billed gulls, and American white pelicans, which in the CRB nest exclusively on the ground and on islands, double-crested cormorants nest on a variety of substrates and not necessarily on islands. Double-crested cormorants frequently nest in trees, either on islands or on the mainland near or over water, especially in the CPR. Double-crested cormorants also nest on artificial structures over water, such as bridges, navigational markers, and transmission towers (see *above*; also see Hatch and Weseloh 1999). As such, suitable nesting habitat for double-crested cormorants is more readily available than for other piscivorous colonial waterbirds in the CRB. Factors limiting colony size and nesting success at double-crested cormorant colonies in the CPR are largely unknown, although windstorms have been known to destroy nests at the arboreal colony on Foundation Island (BRNW annual reports) and a wildfire destroyed an arboreal colony in the Hanford Reach in 2007 (Roby et al. 2008).

The double-crested cormorant colony in North Potholes Reserve was the largest cormorant colony in the CPR during each year of the study through 2013 (*Table 6.1* above). The colony was in decline since then and was completely abandoned during the 2019 breeding season. The cause of colony failure at this location is unknown; bald eagle disturbance, human disturbance, and low prey availability in some years may have been limiting factors at this colony.

Double-crested cormorants are susceptible to several infectious diseases. Newcastle disease and avian cholera have caused large die-offs of hatch-year cormorants in North America (Kuiken 1999). Newcastle disease was first diagnosed in juvenile double-crested cormorants at colonies in the CRE and the Great Salt Lake in Utah during 1997 (Wildlife Health Centre Newsletter 1997, Kuiken 1999); cormorant fledglings from East Sand Island have since been diagnosed with the disease in multiple years (2003, 2005, 2007, 2009). While these diseases have not been diagnosed in double-crested cormorants breeding in the CPR, they are a potential limiting factor for cormorant colony size and nesting success in the region.

6.4.4.2 Gulls – Human disturbance, nest predation, habitat loss, fluctuating water levels, and vegetation encroachment are the primary limiting factors for gulls nesting in the CRB. For example, the gull colony on Solstice Island in northern Potholes Reservoir was abandoned in 2007, apparently due to human disturbance (Roby et al. 2008). The Island 18 gull colony was abandoned following a disturbance caused by a coyote (*Canis latrans*) that swam to the island in 2008 (Collis et al. 2009). In 2011, all nesting attempts by gulls on Three Mile Canyon Island failed in early June; although the cause of this colony failure is unknown, fresh raccoon tracks and evidence of human disturbance on the island were discovered in late May, prior to colony abandonment (Roby et al. 2012). Habitat loss due to management initiatives that are intended to alter or restrict nesting habitat for other colonial waterbirds (i.e. Caspian terns and double-crested cormorants) may also negatively affect co-nesting gulls, either via direct loss of habitat or from the human disturbance often associated with implementation of these initiatives. Encroaching vegetation at the gull colonies on Three Mile Canyon Island, Solstice Island, and perhaps at the Blalock Islands complex may be an important factor that has limited colony size for gulls in the CPR. Because gulls are plastic in their food habits (Collis et al. 2002), they are less likely to be limited by food availability, as compared to other species that are strictly piscivorous (e.g., Caspian terns, cormorants, pelicans).

6.4.4.3 American white pelicans – Factors observed or suspected of limiting colony size and nesting success for American white pelicans in the CRB include mammalian nest predators at Badger Island, researcher disturbance at Crescent Island, and disturbance from recreationalists at the islands where pelicans have nested in the CRE. American white pelicans are highly susceptible to disturbance at the breeding colony, especially during the early stages of the nesting season, and human disturbance has been determined to be one of the most significant limiting factors for American white pelicans throughout North America (Evans and Knopf 1993). In the CRE, American white pelicans have nested on islands actively used for dredge material disposal. As such, nesting habitat at these sites is subject to being altered or destroyed during routine river channel maintenance. American white pelicans are also susceptible to disease, namely avian botulism and West Nile virus, which have caused large die-offs of pelicans throughout North America (Roche et al. 2005). While West Nile virus was diagnosed in an American white pelican from Potholes Reservoir in the fall of 2010 (Washington Department of Health 2010), no infectious diseases have been reported for white pelicans nesting on Badger Island. Based on the small number of breeding colonies for American white pelicans in the CRB, suitable undisturbed nesting habitat on islands may be the primary limiting factor for this species.

6.4.4.4 Brandt's cormorants – Factors that limit colony size or nesting success of Brandt's cormorants in the CRE, specifically those nesting on East Sand Island, are generally similar to those identified for double-crested cormorants (Adkins et al. 2014b, see also *Chapter 4*). In brief, predation and disturbance by increasing numbers of bald eagles (*Haliaeetus leucocephalus*) in the Pacific Northwest (Parrish et al. 2001, Isaacs and Anthony 2011, Hipfner et al. 2012), human disturbance during implementation of the *Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary* (hereafter referred to as the "*Cormorant Management Plan*"; USACE 2015), and nesting habitat loss (natural or anthropogenic) are all factors that have affected Brandt's cormorants nesting on East Sand Island. For example, during the 2017 and 2018 nesting seasons, human disturbance associated with implementation of the *Cormorant Management Plan*, along with bald eagle disturbance, were the likely causes for large scale dispersal/abandonment events by nesting double-crested and Brandt's cormorants on East Sand Island (Turecek et al. 2018, 2019). Following large-scale terrain modifications at the west end of East Sand Island in early 2019, which eliminated much of the cormorant nesting habitat as part of the *Cormorant Management Plan* (see *Chapter 4*), few Brandt's cormorants have initiated nests on the former colony site.

6.4.5 Dispersal & Colony Connectivity

6.4.5.1 Double-crested cormorants – Results from the present study on the size and distribution of double-crested cormorant colonies in the CPR suggest that there is connectivity among colonies in the CPR, but evidence for connectivity based on movements of marked individuals is lacking. For example, recent declines observed at the large colony in North Potholes Reserve were concurrent with growth of existing or incipient colonies in nearby waterbodies (i.e. Lenore Lake, Crab Creek, Sprague Lake; *Table 6.1* above). There is little evidence, however, to suggest connectivity between the double-crested cormorant colonies in the CRE and the CPR. During 2008–2013, 1,961 double-crested cormorants (adults and chicks) were banded on East Sand Island (see *Chapter 4*) and encounters of these banded cormorants in the CPR were rare; one individual banded on East Sand Island was recovered in the CPR during 2012. Results from other studies on the genetic structure of the Western North America population of double-crested cormorants (Mercer et al. 2013) and dispersal of cormorants from East Sand Island (Clark et al. 2006, Courtot et al. 2012, Peck-Richardson 2017) also indicated that there is limited connectivity between cormorant colonies in the CPR and those along the Pacific coast (see *Chapter 4*), far less than has been documented for Caspian terns (see *Chapter 1*).

Post-breeding dispersal and over-wintering use of the lower Snake River (LSR) by double-crested cormorants was investigated in 2008–2010. Cormorants were found throughout the LSR during October–February, averaging 256 individuals across the study area each month, or an average of about 1.2 cormorants per Rkm. About 22% of these cormorants were recorded within 0.25 Rkm of the four federal hydroelectric dams on the LSR. In addition, winter roost sites were identified at Foundation Island on the Columbia River, an unnamed island in the tailrace of Ice Harbor Dam, the Lyons Ferry and Central Ferry bridges over the LSR, and at Swallows Park on the Snake River above the confluence with the Clearwater River.

6.4.5.2 Gulls – Results from this study on the size and distribution of gull colonies in the CPR suggest that there is connectivity among gull colonies in the CPR, but evidence for connectivity based on movements of marked individuals is lacking. For example, following the abandonment of the Three Mile Canyon gull colony (Rkm 412) in 2012 there was a significant increase in the size of the gull colony in the Blalock Islands Complex (Rkm 440). Also, following the implementation of the *IAPMP* that resulted in the abandonment of the California gull colony on Crescent Island (Rkm 510) in 2015, a large California gull colony formed on nearby Badger Island (Rkm 512; *Table 6.2* above). Connectivity between gull colonies in the CRB and other gull colonies in the Pacific Flyway is unknown, however.

6.4.5.3 American white pelicans – Although few data exist linking American white pelicans breeding in the CRB to other colonies in the Pacific Flyway, there is anecdotal evidence of considerable connectivity. For example, two PIT tags implanted in suckers endemic to the Upper Klamath Basin, where several large white pelican colonies occur, have been recovered on the Badger Island pelican colony. Additionally, although few (20) white pelicans have been banded in Washington since 2002, 19 bands have been encountered in the state dating back to 1963 (USGS 2020). During the 2011–2012 nesting seasons, a total of 122 American white pelican chicks were banded with field-readable bands at the colony on Miller Sands Spit in the CRE, but there have been no subsequent resightings of these banded pelicans in the CPR.

6.4.5.4 Brandt's cormorants – Information about dispersal and colony connectivity for Brandt's cormorants nesting on East Sand Island was not collected during this study. In a recent study by researchers at Oregon State University, 10 Brandt's cormorants were captured in the CRE and fitted with solar-powered GPS tags to better understand foraging patterns and post-breeding dispersal. Preliminary results found that nine individuals traveled north to the Salish Sea and outer coast of Vancouver Island and one individual dispersed south along the California coast, and was tracked near several known Brandt's cormorant colony sites in California (A. Peck-Richardson, unpubl. data; see also Peck-Richardson et al. 2020).

6.4.6 Foraging Behavior

6.4.6.1 Double-crested cormorants – Unlike double-crested cormorant nesting in the CRE (see *Chapter 4*), no specific studies have been conducted to better understand the foraging behavior of double-crested cormorants nesting in the CPR. Double-crested cormorants forage by pursuit-diving, commonly in shallow water habitats less than 8 m deep (Coleman et al. 2005, Dorr et al. 2014, Peck-Richardson 2017), habitat that is widespread in the CPR. During the nesting season double-crested cormorants generally forage close to their breeding colony (Anderson et al. 2004, Coleman et al. 2005), but are known to forage more than 40 km from the nest site (Custer and Bunck 1992, Peck-Richardson 2017). Information on the foraging ecology of double-crested cormorants in the CPR is limited to weekly surveys conducted during the 2011 breeding season to assess piscivorous waterbird distribution in McNary Reservoir, and three JSATS studies which used last known live detections of acoustic- and PIT-tagged (i.e. double-tagged) juvenile salmonids and subsequent recoveries of tags on bird colonies to determine predation hotspots (Evans et al. 2016a, Roby et al. 2016, and Roby et al. 2017).

Weekly surveys of McNary Reservoir in 2011 indicated that approximately 80% of those double-crested cormorants detected were observed on or near the Foundation Island colony. Off-colony cormorants (n = 5–80) were primarily observed during May–July on or foraging near islands, sandbars, and secluded shorelines along the Columbia and Snake rivers, including Strawberry Island (Rkm 528), Goose Island below Ice Harbor Dam (Rkm 536), the tailrace at Ice Harbor Dam (Rkm 538), Crescent Island (Rkm 510), Badger Island (Rkm 512), Wade Island (Rkm 525), and the downstream tip of Island 20 (Rkm 545; see Roby et al. 2012 for complete results of this study).

Information derived from JSATS studies on the foraging behavior of double-crested cormorants nesting in the CPR is limited to birds nesting at the Foundation Island colony in 2012 and 2014, plus the small colony of cormorants at the Locke Island complex, Hanford Reach in 2014. Results of these JSATS studies indicated that cormorant predation rates on double-tagged smolts were generally highest in those river segments closest to each colony, with most predation occurring within a 40-km to 50-km radius of the colony (*Figure 6.14*). Results also indicated there were several foraging hotspots (areas of concentrated foraging within each river reach or segment); cormorants nesting on Foundation Island disproportionately commuted upstream of their breeding colony to forage in the lower Snake River (Evans et al. 2016a; *Figure 6.14*). Higher predation probabilities on smolts for cormorants foraging in the lower Snake River compared to those foraging in the middle Columbia River may be related to (1) the proximity of the Foundation Island colony to Ice Harbor Dam (19 Rkm downstream from the dam), (2) the relative abundance of smolts originating from the Snake River basin compared with smolts originating from the upper and middle Columbia River basin, and (3) environmental conditions resulting in favorable foraging conditions for cormorants in the lower Snake River, like reduced flows and higher turbidity (Hostetter et al. 2012; see also *Chapter 10*). There was no evidence of predation hotspots for double-crested cormorants nesting on an island in the Locke Island complex in Hanford Reach, with similar per km predation rates observed in 2015 and 2016 (Evans et al. 2016a, Roby et al. 2016, 2017; *Figure 6.15*). Cormorants from this colony foraged in relatively close proximity (less than 50 Rkm) to their nesting site, but small numbers of smolt PIT tags were recovered on this colony and results should be interpreted cautiously (see Evans et al. 2016a for a more detailed description).

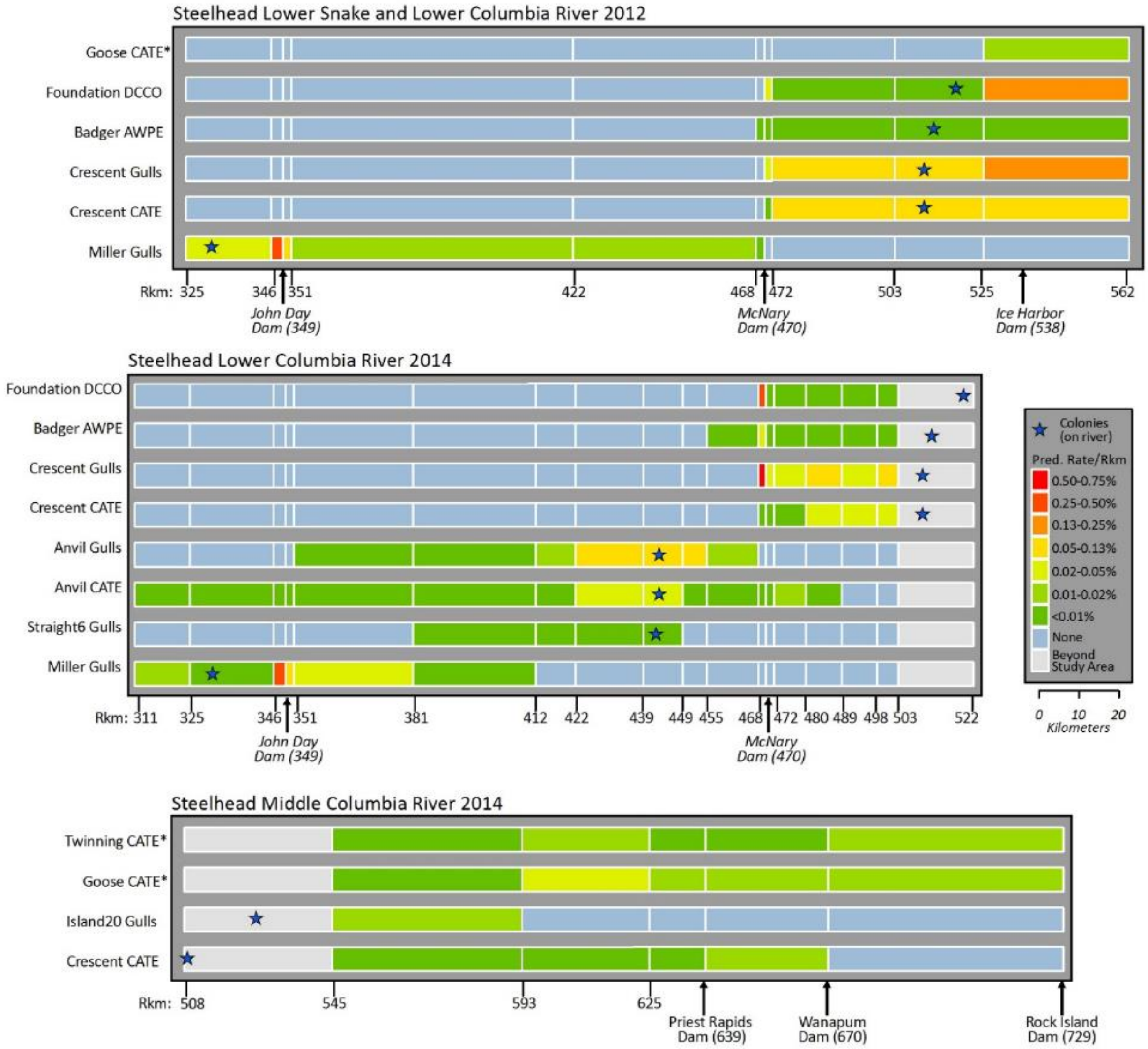

Figure 6.14. Bird colony-specific locations of predation/consumption of acoustic-tagged juvenile steelhead in sections of the lower Snake River, lower Columbia River, and middle Columbia River during 2012 and 2014. Results are depicted as predation/consumption rates per river km (Rkm). Species of colonial waterbirds evaluated include Caspian terns (CATE), double-crested cormorants (DCCO), California and ring-billed gulls (Gulls), and American white pelicans (AWPE). An asterisk next to a colony site denotes that the colony is located off the Snake or Columbia rivers. Results are from Evans et al. (2016a).

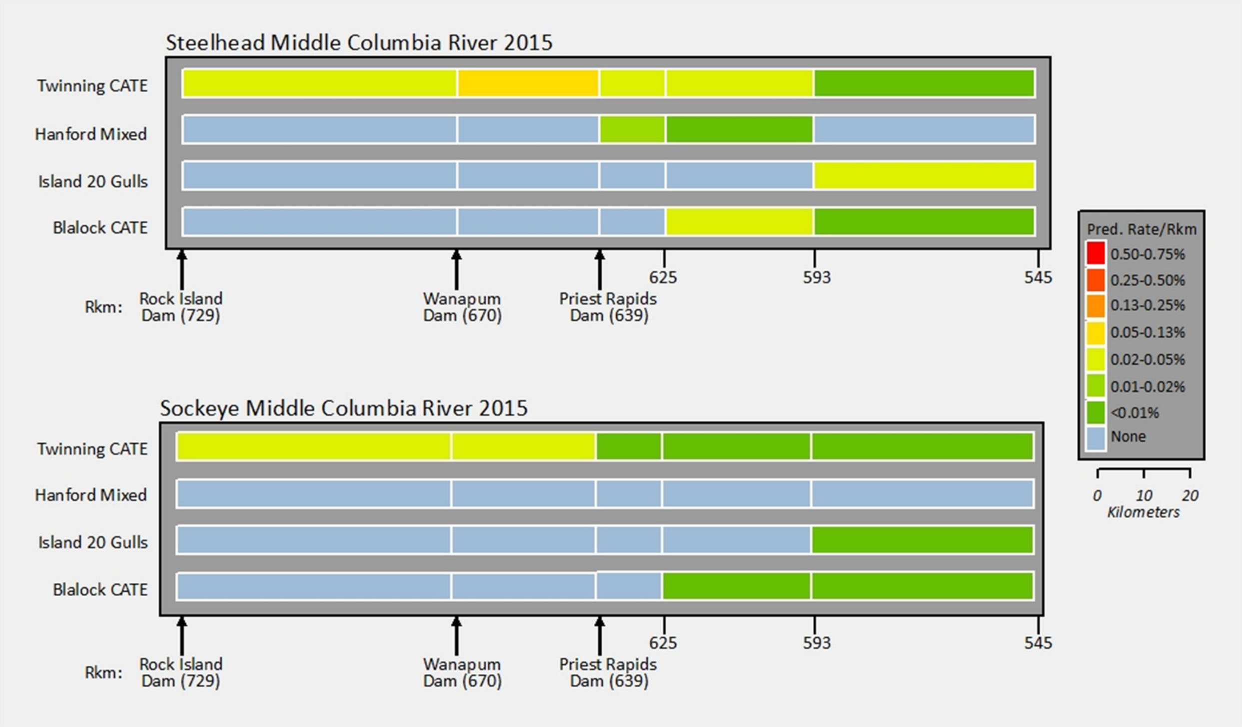

Figure 6.15. Bird colony-specific locations of predation/consumption of tagged juvenile steelhead and sockeye salmon in sections of the middle Columbia River in 2015. Results are depicted as predation/consumption rates per river km (Rkm). Species of piscivorous colonial waterbirds evaluated include Caspian terns (CATE), California and ring-billed gulls (Gulls), and double-crested cormorants and great blue herons at a mixed-species colony site (Mixed). Figure from Roby et al. 2016.

6.4.6.2 Gulls – Information on the foraging behavior of gulls nesting in the CRB is limited to JSATS studies conducted in 2012, 2014, and 2015 and to selected California and ring-billed gull colonies on the Columbia River (i.e. Miller Rocks, Blalock Islands, Crescent Island, and Island 20). Results of these studies indicated that the foraging range of gulls varied by colony location and year. In general, as was seen for double-crested cormorants, consumption rates were highest in those river segments closest to each gull colony, with most consumption occurring within a 40 km radius of the colony (*Figure 6.14* above). Several foraging hotspots (areas of concentrated foraging within each river reach or segment) were identified for particular gull colonies, where gulls disproportionately consumed smolts near dams in cases where the colony was located near a dam. For example, consumption rates by gulls nesting on Miller Rocks, which is located 18 Rkm downstream of John Day Dam, were significantly higher near the dam compared to in the open reservoirs away from the dam (*Figure 6.14* above). A similar relationship was observed for gulls nesting on Crescent Island, where birds disproportionately foraged on smolts near Ice Harbor and McNary dams. Gull consumption of tagged smolts, however, was not limited to foraging near dams, with consumption taking place within the open reservoirs as well, including consumption in the free-flowing section of the middle Columbia River in Hanford Reach, downstream of Priest Rapids Dam, by gulls nesting on Island 20 (Evans et al. 2016a, Roby et al. 2016).

Results of JSAT studies also provide evidence that California and ring-billed gulls nesting on Crescent Island disproportionately commuted upstream of their breeding colony to forage in the lower Snake River (Evans et al. 2016a; *Figure 6.14* above). Similar to findings for double-crested cormorants nesting on nearby Foundation Island, higher consumption probabilities on smolts by gulls foraging in the lower Snake River relative to the middle Columbia River may be related to (1) the proximity of the Crescent Island gull colony to Ice Harbor Dam (27 Rkm downstream from the dam), (2) the relative abundance of smolts originating from the Snake River compared with smolts originating from the upper and middle Columbia River, and (3) environmental conditions resulting in favorable foraging conditions for gulls in the lower Snake River, like reduced flows and higher turbidity (Hostetter et al. 2012; see also *Chapter 10*). Ruggerone (1986) and Zorich et al. (2011) also documented high rates of gull consumption of juvenile salmonids near dams and hypothesized that smolts may be more vulnerable near dams because of (1) increased smolt travel times or delayed migration in the forebay of dams, (2) injury or mortality associated with dam passage, or (3) smolts being temporarily stunned or disoriented by hydraulic conditions in the tailrace of dams. California and ring-billed gulls have also been documented foraging at structures (i.e. barrage and diversion dams) on the Yakima River, peaking in May during the smolt out-migration period (Major et al. 2005). There was no evidence of smolt consumption hotspots for gulls nesting on Island 20, with similar per km consumption rates observed in 2014 and 2015 (Evans et al. 2016a, Roby et al. 2016). Gulls from the Island 20 colony foraged in relative close proximity to their colony (less than 50 Rkm), but small numbers of PIT tags were recovered on these colonies and results should be interpreted cautiously (see also Evans et al. 2016a).

6.4.6.3 American white pelicans – Information on the foraging behavior of American white pelicans in the CRB is limited to weekly surveys documenting distribution and numbers of piscivorous waterbirds in the McNary Reservoir in 2011 and the aforementioned JSATS studies, which used last known live detections of acoustic- and PIT-tagged (Evans et al. 2016a). American white pelicans generally forage in shallow water less than three meters deep by dipping their bill into the water and scooping prey items (Knopf and Evans 2004). Such habitat is widespread in the CPR, both along the Columbia River and at inland waterbodies. Foraging in American white pelicans is often cooperative, with pelican flocks observed driving prey into shallow water where capture rates are likely greater (Anderson 1991, Mcmahon and Evans 1992). This behavior has been observed in the CPR on the mainstem Columbia and Snake rivers and at manmade diversion structures such as Horn Rapids Dam on the Yakima River near Richland, Washington (Stinson 2016).

During weekly surveys of McNary Reservoir in 2011, 48% of the American white pelicans detected were on or near Badger Island. However, boat-based counts of pelicans on Badger Island were obscured by the dense vegetation on the island and significantly underestimate the number of pelicans associated with the colony. Except for Wade Island (Rkm 525), pelicans were found at the same locations where double-crested cormorants were observed (see *above*). Similar to double-crested cormorants, pelicans were most abundant on the lower Snake River near Strawberry Island, Goose Island, and their associated sandbars during April–July (n = 7–95 loafing pelicans and 0–103 foraging pelicans, depending on the date; Roby et al. 2012).

Results of JSATS studies identified no hotspots for smolt predation by American white pelicans nesting on Badger Island, and that most pelicans that foraged on salmonid smolts did so within a 40-km to 50-km radius of the colony (Evans et al. 2016a; *Figure 6.14*). More recently, however, Payton et al. (2020) recovered PIT tags on the Badger Island pelican colony that were from sub-yearling Chinook salmon known to have survived migration to John Day Dam, over 150 Rkm downstream of Badger Island. American white pelicans have been documented consuming tagged fish over 300 km from their nest site at other colonies outside the CRB (Scoppettone et al. 2006), so it is possible that pelicans nesting on Badger Island commute long distances to forage both within and outside the CPR.

6.4.6.4 Brandt's cormorants – Information on the foraging ecology of Brandt's cormorants nesting in the CRE is limited to a study by Peck-Richardson et al. (2018), which found that, compared to double-crested cormorants nesting at the same colony on East Sand Island, Brandt's cormorants used foraging habitat closer to the colony and foraged mainly in deeper marine waters close to the mouth of the Columbia River (*Figure 6.16*; see also *Chapter 4*). Brandt's cormorants were also found to dive deeper than double-crested cormorants and forage more in benthic habitats where the foraging areas of the two cormorant species overlapped of the CRE (see Peck-Richardson 2017 for details associated with this study).

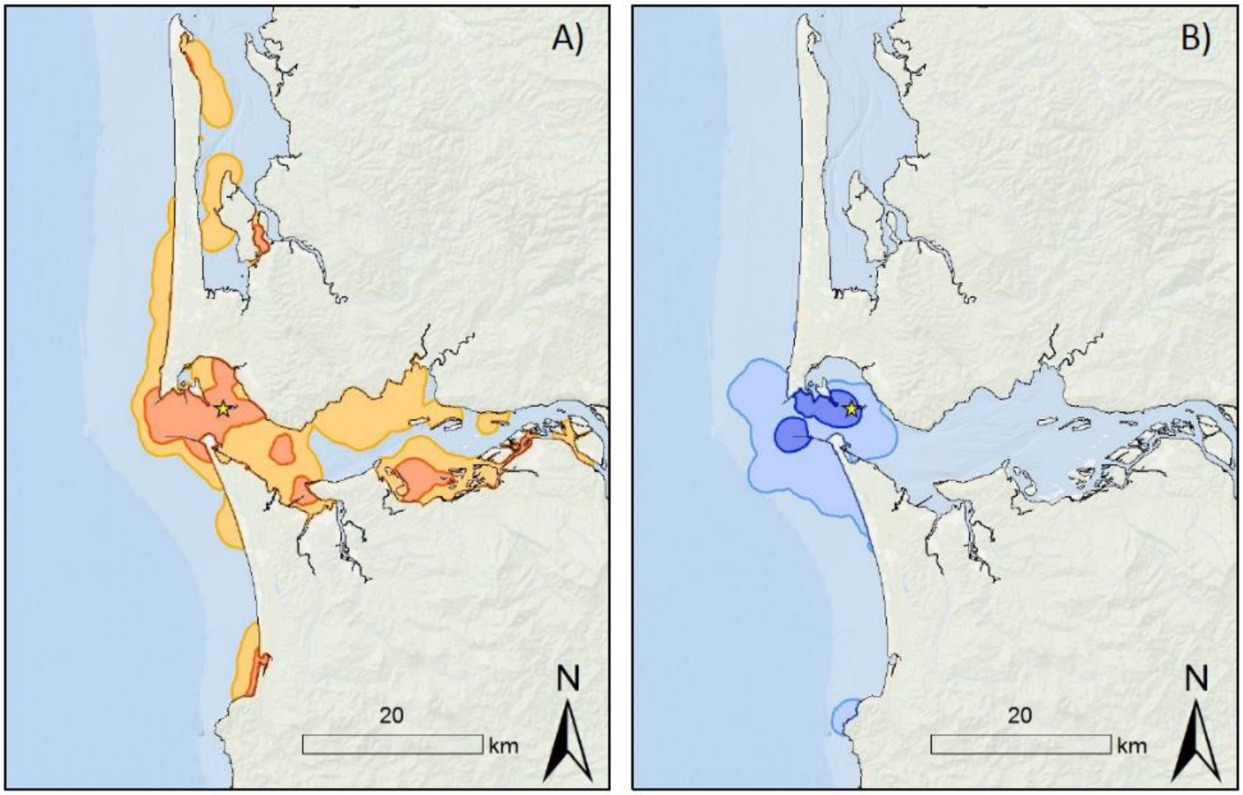

Figure 6.16. Foraging areas used by double-crested cormorants (A; n = 17 individuals) and Brandt's cormorants (B; n = 15 individuals) nesting on East Sand Island in the Columbia River estuary. Overall foraging area estimates (95% KDE) are indicated by lighter shaded areas and core foraging area estimates (50% KDE) are indicated by darker shaded areas. The location of the cormorant breeding colony on East Sand Island is indicated by the star. Figure is from Peck-Richardson (2017).

6.4.7 Diet Composition & Smolt Consumption

6.4.7.1 Double-crested cormorants – Based on identifiable fish tissue in samples of foregut contents, juvenile salmonids comprised 22% the diet (by biomass) of double-crested cormorants nesting at the Foundation Island colony during 2005–2010 (n = 140 adult foregut samples, or a total of 32.2 kg of identifiable fish tissue; *Figure 6.17*; Lyons et al. 2011). The peak in the salmonid proportion of the diet for cormorants nesting on Foundation Island coincided with the peak smolt out-migration period (i.e. late April to late May; *Figure 6.18*). Other prevalent prey types included centrarchids (bass and sunfish; 30%), ictalurids (channel catfish [*Ictalurus punctatus*]; 14%), cyprinids (carp and minnows; 9%), and percids (yellow perch [*Perca flavescens*]; 8%). These diet composition results should be interpreted cautiously, however, because they are based on relatively small sample sizes and are pooled across six breeding seasons. Information on diet composition was not available for double-crested cormorants nesting at other colonies in the CPR.

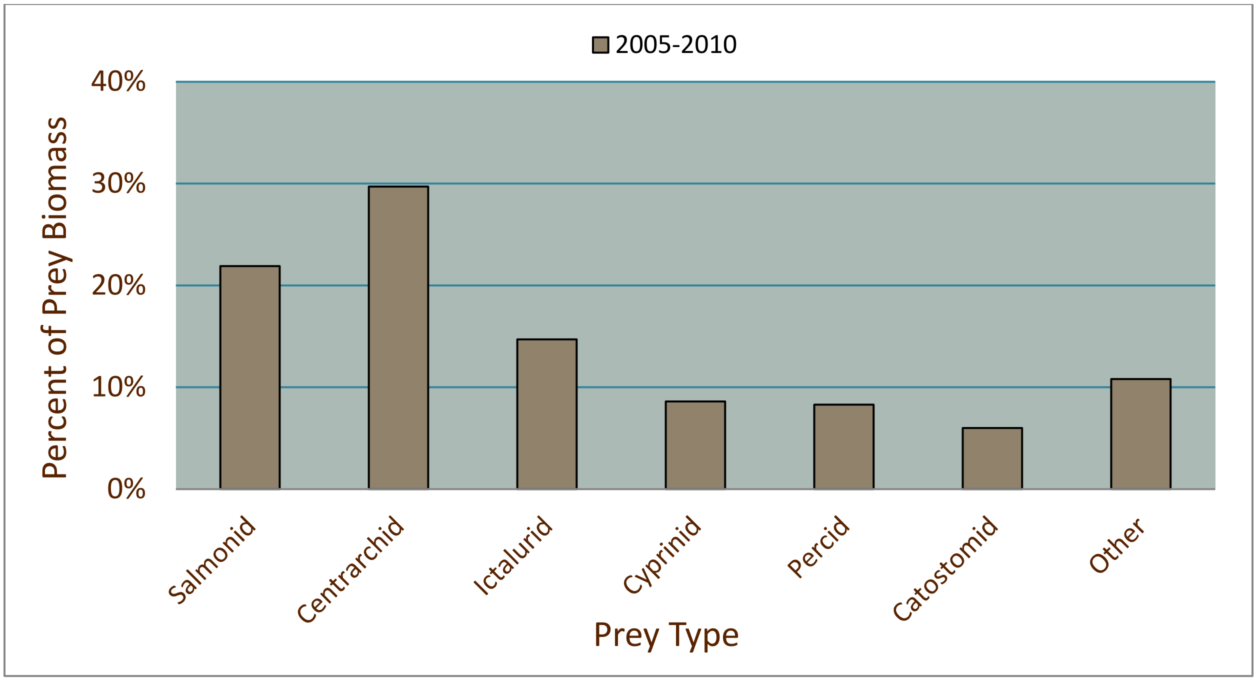

Figure 6.17. Combined diet composition by prey type (percent of prey biomass) for double-crested cormorants nesting on Foundation Island in the Columbia Plateau region during 2005–2010. Prey types in the Other category include lamprey, sticklebacks, sculpins, crustaceans, and unidentified non-salmonid fishes.

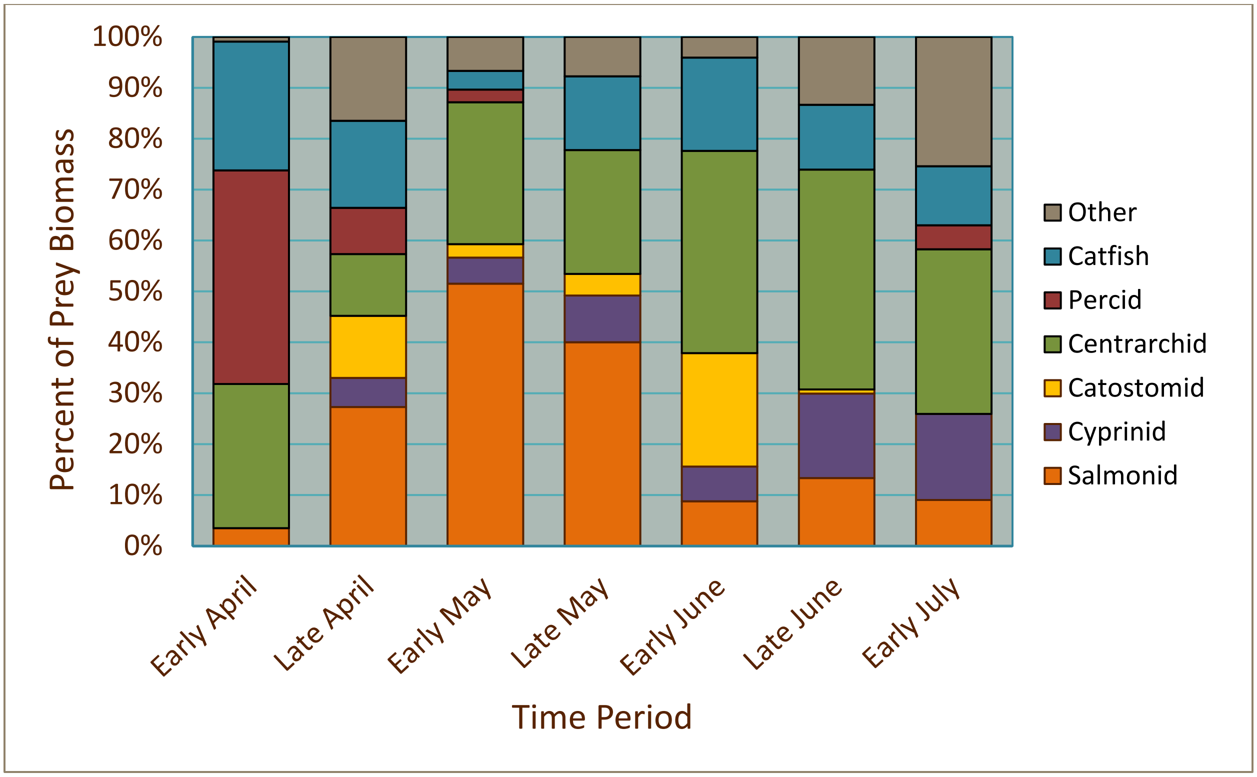

Figure 6.18. Seasonal pattern in diet composition (% biomass) of double-crested cormorants nesting at Foundation Island. Diet composition was based on identifiable soft tissue in samples of foregut contents from adult cormorants collected returning to the colony from foraging trips during 2005–2010. Prey types classified as "Other" include lamprey, sticklebacks, sculpins, crustaceans, and other unidentified non-salmonid fishes. Data are from Lyons et al. (2011).

During 2007–2009, a total of 130 samples of foregut contents were collected from double-crested cormorants shot along the lower Snake River (LSR) to determine whether over-wintering cormorants were consuming ESA-listed fall Chinook salmon that also over-winter in the LSR. Salmonids comprised an average of 11.7% (by biomass) of the diet, with both juvenile and adult (precocious male or jack) salmonids confirmed in the diet (Cramer et al. 2011). Fall Chinook salmon comprised an average of 3.4% of the diet by mass but were the most numerous salmonid found by number (individual fish). The most prevalent prey type found in foregut contents were centrarchids (bass and sunfishes), which accounted for 34.3% by biomass of the cormorant diet during the study. Juvenile American shad (*Alosa sapidissima*; 15.0%), cyprinids (minnows and carp; 11.7%), catostomids (suckers; 7.0%), and ictalurids (catfish; 6.3%) were also prominent non-salmonid prey types in the cormorant diet. Diet composition of cormorants over-wintering in the LSR was highly variable and changed throughout the season, but juvenile fall Chinook salmon appeared to be a small component of the overall diet.

Consumption of juvenile salmonid biomass by double-crested cormorants nesting on Foundation Island during 2004–2009 was estimated by Lyons et al. (2011b) and averaged 22,200 kg/yr, ranging from a minimum of 19,600 kg in 2005 (95% CI = 15,000–24,200 kg) to a maximum of 24,700 kg in 2008 (95% CI = 20,300–29,200 kg). Estimates of the number of juvenile salmonids consumed by Foundation Island cormorants depended greatly upon which approach was used to convert biomass consumed to numbers of smolts consumed. Of the foregut contents samples used for diet analyses that contained salmonid soft tissue (n = 53), 70% had at least a portion of that salmonid soft tissue genetically identified as *O. mykiss* (i.e. steelhead or resident rainbow trout). In contrast, annual estimates of relative availability derived from data on McNary Dam smolt passage indicated that steelhead ranged from 8% to 20% of all available salmonid smolts. Estimates of the total number of salmonids consumed by double-crested cormorants nesting on Foundation Island, using the genetics-based approach, ranged from 468,000 in 2004 (95% CI: 354,000–583,000) to 589,000 in 2008 (95% CI: 448,000–729,000). Using the availability-based approach, comparable estimates ranged from 711,000 in 2005 (95% CI: 498,000–925,000) to 883,000 in 2007 (95% CI: 620,000–1,146,000). Using either method, there was no discernable trend in the number of smolts consumed during the study period (*Figure 6.19*). Annual consumption of juvenile lamprey, a species of conservation concern, was approximately two orders of magnitude less than consumption of juvenile salmonids. Point estimates of lamprey consumption by Foundation Island cormorants averaged about 2,000 juvenile lamprey per year, ranging from 1,800 in 2004 (95% CI: 800–2,000) to 2,300 in 2008 (95% CI: 1,100–3,400; see Lyons et al. [2011b] for complete results from this study).

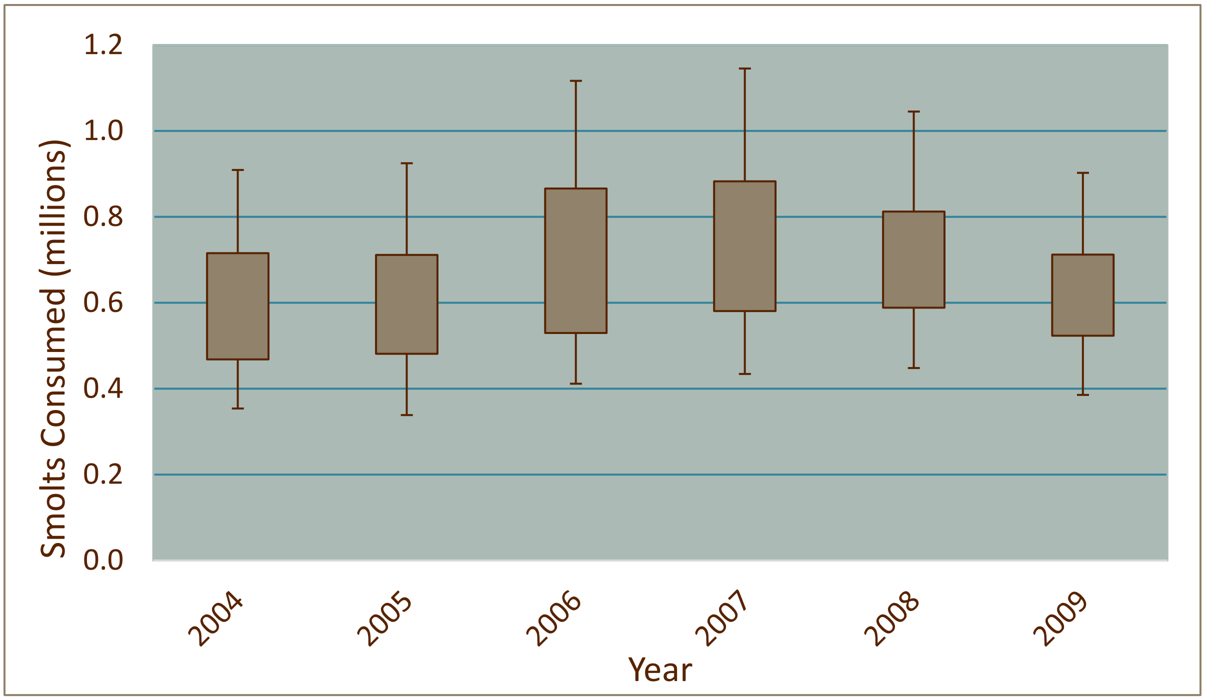

Figure 6.19. Annual estimates of numbers of juvenile salmonids consumed by double-crested cormorants nesting on Foundation Island during 2004–2009. The estimated number of salmonids consumed derives from two different approaches: (1) species composition of salmonids in the diet is based on genetic identification of salmonid tissues from samples of foregut content of collected cormorants (bottom of box and lower 95% CI bar) and (2) species composition of salmonids in the diet based on the proportions passing McNary Dam during out-migration and assuming salmonids were taken in proportion to availability (top of box and upper 95% CI bar). Results are from Lyons et al. (2011).

6.4.7.2 Gulls – Data on diet composition of the four gull species nesting at colonies in the CRB have not been collected since the late 1990s. During 1997–1998, Collis et al. (2002) found that the proportion of fish and salmonids in the diet varied considerably by gull species and colony location. Results indicated that for some California gull colonies (i.e. Three Mile Canyon Island, Island 18) and for the only ring-billed gull colony sampled (Island 18) the diet consisted mainly of anthropogenic food sources, with fish accounting for less than 3% of the biomass of gull diets. Juvenile salmonids were found in the diet of California gulls nesting on Little Memaloose Island (15% of total food biomass) and Miller Rocks (3% of total food biomass). In contrast to California and ring-billed gulls nesting in the CPR, glaucous-winged/western gulls nesting in the CRE consumed primarily fish. In general, glaucous-winged/western gulls nesting on Rice Island and Miller Sands Spit consumed mostly riverine fishes (e.g., salmonid smolts, cyprinids), whereas glaucous-winged/western gulls nesting on East Sand Island consumed more marine and estuarine fishes, primarily Pacific herring (*Clupea pallasii*). Juvenile salmonids comprised 10.9% of the diet (by mass) of glaucous-winged/western gulls nesting on Rice Island and Miller Sands Spit, whereas juvenile salmonids comprised 4.2% of the diet of glaucous-winged/western gulls nesting on East Sand Island.

More recently, gulls nesting at the colony on Miller Rocks have been documented consuming juvenile salmonids at nearby The Dalles and John Day dams (Zorich et al. 2010, 2011). In contrast to the study by Collis et al. (2002), these studies found that for California gulls collected at the dams, juvenile salmonids, and at times juvenile lamprey, were a primary component of identifiable diet biomass. The seasonal proportion of salmonids identified in the diet ranged from ca. 46% to 83%, depending on the year and the dam. Data on diet composition for gulls from other colonies in the CPR are lacking, but a similar prevalence of salmonids in the diet of gulls foraging at dams may occur if smolts are available to gulls in shallow pools, near the surface at hydroelectric dams, or when out-migration has been slowed or altered by anthropogenic structures (Ruggerone 1986, Jones et al. 1996, Walter et al. 2005, Evans et al. 2016a). Historical records also indicate that gulls were known to forage at dams. Merrell (1959) collected California and ring-billed gulls foraging below McNary Dam and found that juvenile lamprey comprised approximately 70% of gull diets by volume during a single sampling period.

6.4.7.3 American white pelicans – No standardized data on diet composition for this species were collected during the present study.

6.4.7.4 Brandt's cormorants – No standardized data for this species were collected as part of this study. Based on a study in 2000 of fish otoliths found in cormorant pellets (Couch and Lance 2004), diet composition of Brandt's cormorants was diverse with more than 10 prey families represented. The frequency of occurrence of in the diet of Brandt's cormorants nesting on East Sand Island was 74.1% for sculpins (Cottidae), 68.5% for flatfishes (Pleuronectidae), 61.1% for sand lance (Ammodytidae), and 42.6% for anchovy (Engraulidae); the frequency of occurrence of juvenile salmonids in the diet was only 7.4% (Couch and Lance 2004).

6.4.8 Predation/Consumption Rates of Juvenile Salmonids

Since 1999, the first year of studies to determine salmonid predation/consumption rates (see *below*) by piscivorous colonial waterbirds using on-colony smolt PIT tag recoveries (Ryan et al. 2001), over 100,000 PIT tags (detections of individual fish) have been recovered on waterbird colonies in the CPR. This includes 76,546 smolt PIT tags on California and ring-billed gull colonies, 14,566 smolt PIT tags on double-crested cormorant colonies, and 17,206 smolt PIT tags on American white pelican colonies. Similar to smolt PIT tags found on Caspian tern colonies in the CPR, the majority of these tags (ca. 97%) were from anadromous juvenile salmonids, but tags from other fish species were also recovered (Pacific lamprey [*Lampetra tridentata*], bull trout [*Salvelinus confluentus*], white sturgeon [*Acipenser transmontanus*], northern pikeminnow [*Ptychocheilus oregonensis*], smallmouth bass [*Micropterus dolomieu*], and American shad [*Alosa sapidissima*]). PIT tags have also been recovered from loafing or roosting sites for piscivorous colonial waterbirds in the CPR (n = 18,288), but these tags were not directly incorporated into colony-specific estimates of predation/consumption rates because the species of consumer that deposited the tag was unknown. Also, some proportion of tags recovered at loafing/roosting sites were presumably consumed by birds that were nesting at colonies where tags were being recovered, tags that were incorporated into colony-specific estimates of predation/consumption rates based on deposition probabilities (see Hostetter et al. 2015 and *Appendix A.2* for additional details).

6.4.8.1 Double-crested cormorants – Until recently, most previously reported estimates of predation rates on juvenile salmonids by double-crested cormorants nesting in the CPR were reported as minimum estimates of predation rates (e.g., Evans et al. 2012, USACE 2014, *Appendix A.2*) due to a lack of information on smolt PIT tag deposition probabilities for double-crested cormorants (Hostetter et al. 2015). Standardized, accurate (unbiased) estimates of predation rates for double-crested cormorant nesting at colonies that were scanned for smolt PIT tags beginning in 2008 are provided below.

Lenore Lake: Only one smolt PIT tag has been recovered from the double-crested cormorant colony on Lenore Lake, despite scanning for PIT tags on-colony after the 2018 and 2019 nesting seasons, when 111 and 138 breeding pairs, respectively, nested at the colony (*Table 6.1* above). These results provide strong evidence that, unlike Caspian terns that are also nesting at Lenore Lake (see *Chapter 2*), cormorants nesting at Lenore Lake do not commute frequently to the middle Columbia River to forage on juvenile salmonids and consequently pose little or no threat to smolt survival. The middle Columbia River is at least 45 km from Lenore Lake and the lower Snake River is at least 120 km from Lenore Lake, and studies of double-crested cormorant foraging ecology indicate that cormorants primarily forage closer to their nesting colony (see section *6.4.6 Foraging Behavior* above).

Potholes Reservoir: The double-crested cormorant colony in North Potholes Reserve near Potholes Reservoir was scanned for smolt PIT tags following the 2012 breeding season. Predation rate estimates were < 0.1% for Upper Columbia River spring Chinook salmon and 0.3% (95% credible interval = < 0.1–0.8%) for Upper Columbia River steelhead, indicating that cormorants nesting at this colony posed little threat to smolt survival in the middle Columbia River during the 2012 out-migration (*Table 6.6*). Similar to the cormorant colony on Lenore Lake, the colony in North Potholes Reserve is at least 50 km from the middle Columbia River and at least 85 km from the lower Snake River, where anadromous salmonids are potential prey.

Table 6.5. Average annual predation/consumption rates (95% Credible Intervals) of PIT-tagged juvenile salmonids by California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), and American white pelicans (AWPE) nesting at colonies in the Columbia Plateau region during 2007–2019. ESA-listed salmonid populations (ESUs/DPSs) include those from the Snake River (SR) and the Upper Columbia River (UCR), with runs of spring (Sp), summer (Su), and fall (Fall) fish evaluated. See Appendix B, Tables B9-B13 for annual estimates of predation/consumption by birds from each colony.

ESU/DPS	Island 20 LAXX 2013-2019	Badger LAXX 2015-2019 [1]	Crescent LAXX 2007-2014	Blalock LAXX 2013-2019	Miller Rocks LAXX 2007-2019
SR Sockeye	0.9% (0.3-1.7)	2.7% (1.2-4.8)	2.2% (1.3-3.2)	2.0% (1.0-3.0)	6.2% (4.8-7.7)
SR Spr/Sum Chinook	0.2% (0.1-0.3)	0.5% (0.3-0.6)	1.0% (0.8-1.2)	0.2% (0.1-0.2)	1.2% (1.1-1.4)
UCR Spr Chinook	0.5% (0.1-1.2)	1.6% (0.7-2.9)	1.2% (0.4-2.3)	0.4% (0.2-0.6)	2.1% (1.7-2.4)
SR Fall Chinook	0.2% (0.1-0.4)	0.9% (0.5-1.4)	0.6% (0.4-0.9)	0.4% (0.3-0.6)	2.0% (1.8-2.4)
SR Steelhead	1.6% (1.2-1.9)	3.3% (2.5-4.3)	4.8% (4.1-5.6)	2.4% (2.0-3.0)	7.2% (6.5-8.1)
UCR Steelhead	4.1% (3.3-4.9)	5.0% (3.5-6.9)	5.8% (5.0-6.9)	3.9% (3.0-4.8)	8.2% (6.9-9.3)

ESU/DPS	Foundation DCCO 2007-2014	Lenore DCCO 2017-2019	Potholes DCCO 2012	Hanford DCCO 2018
SR Sockeye	3.4% (2.4-4.5)	< 0.1%	< 0.1%	< 0.1%
SR Spr/Sum Chinook	1.7% (1.5-2.1)	< 0.1%	< 0.1%	< 0.1%
UCR Spr Chinook	0.4% (0.1-0.9)	< 0.1%	< 0.1%	0.2% (0.1-0.7)
SR Fall Chinook	1.2% (0.9-1.5)	< 0.1%	< 0.1%	< 0.1%
SR Steelhead	4.0% (3.4-4.7)	< 0.1%	< 0.1%	< 0.1%
UCR Steelhead	0.3% (0.2-0.4)	< 0.1%	0.3% (<0.1-0.8)	0.2% (0.1-0.4)

ESU/DPS	Badger AWPE 2007-2014
SR Sockeye	0.2% (0.2-0.3) [2]
SR Spr/Sum Chinook	0.1% (0.1-0.1) [2]
UCR Spr Chinook	0.2% (0.1-0.5) [2]
SR Fall Chinook	0.1% (0.1-0.1) [2]
SR Steelhead	0.2% (0.2-0.3) [2]
UCR Steelhead	0.1% (0.1-0.2) [2]

[1] *AWPE deposited some PIT tags on the LAXX colony, but the area scanned for tags was numerically dominated by LAXX*

[2] *Minimum estimates of predation rates due to the lack of PIT tag deposition probabilities (see Appendix B, Table B.13).*

Locke Island Complex: The colony on Locke Island in Hanford Reach on the Columbia River was scanned for PIT tags after the 2018 nesting season, when 68 breeding pairs of double-crested cormorants were counted at the colony, the highest number observed since cormorants were first confirmed nesting at the site in 2007. The results indicated that predation rates were less than 0.3% for all salmonid ESUs/DPSs evaluated, with the highest rates observed on Upper Columbia River steelhead at 0.2% (95% CI = 0.1–0.4%) and on Upper Columbia River spring Chinook salmon at 0.2% (95% CI = 0.1–0.7%; *Table 6.6* above). The small size of this cormorant colony is presumably a factor in with the low rates of predation on juvenile salmonids.

Foundation Island: Estimates of predation rates on juvenile salmonids by double-crested cormorants nesting on Foundation Island are available during 2007–2014. Results indicated that predation rates on some salmonid ESUs/DPSs by cormorants nesting at Foundation Island were substantial; estimated predation rates were the highest of all cormorant colonies evaluated in the CPR (*Table 6.6* above). For example, average annual predation rates of 4.0% (95% CI = 3.4–4.7%) and 3.4% (95% CI = 2.4–4.5%) were observed on Snake River steelhead and Snake River sockeye salmon, respectively. Average annual predation rates on other ESUs/DPSs ranged from 0.3% (95% CI = 0.2–0.4%) on Upper Columbia River steelhead to 1.7% (95% CI = 1.5–2.1%) on Snake River spring/summer Chinook salmon (*Table 6.6* above; see *Appendix B.2, Table B.25* for annual estimates). Unlike Caspian terns and California and ring-billed gulls, there was less evidence that cormorants disproportionately consumed smolts from steelhead DPSs relative to smolts from salmon ESUs. For example, of the salmonid ESUs/DPSs evaluated, average annual predation rates were consistently the lowest on Upper Columbia River steelhead, at 0.3% of available fish. A similar lack of a strong preference for steelhead relative to salmon was also observed at the East Sand Island double-crested cormorant colony (see *Chapter 4*) and results suggest that, compared to Caspian terns, cormorants consumed fish more in proportion to their availability (see *Chapter 10* for a more detailed discussion of factors that influence smolt susceptibility to cormorant predation). Results of diet composition studies for double-crested cormorants nesting at the Foundation Island and East Sand Island colonies support this hypothesis, with Chinook salmon the most common salmonid species found in the diet and the most numerous species of juvenile salmonids in the Columbia River (see *above*).

Predation rate estimates for double-crested cormorants nesting at Foundation Island during 2007–2014 were similar to those of Caspian terns nesting on nearby Crescent Island, a colony that was managed as part of the *IAPMP* to reduce predation on juvenile salmonids starting in 2015 (see *Chapter 2*). When management actions for double-crested cormorants nesting on Foundation Island were considered in 2014 as part of *IAPMP*, however, accurate estimates of cormorant predation rates were not yet available, and the reported estimates were minimums (USACE 2014). The Foundation Island cormorant colony remains an active colony but estimates of colony size and predation rates on salmonid smolts have not been generated since 2014. Based on the estimates of predation rates presented herein, this colony posed a greater threat to smolt survival than previously documented and new data collection efforts at this cormorant colony may now be warranted.

Other colonies: There are several other double-crested cormorant colonies in the CPR within foraging distance of water bodies containing anadromous juvenile salmonids, including colonies at the mouth of the Okanogan River, on Harper Island in Sprague Lake, and, more recently, along sections of Crab Creek (see also section *6.4.3 Colony Size & Nesting Success* above). Similar to cormorant colonies in Potholes Reservoir and on islands in Lenore Lake, colonies at Sprague Lake and Crab Creek are a considerable distance from the Columbia and Snake rivers and thus unlikely to pose a significant risk to smolt survival. The colony at the mouth of Okanagan River, however, is very close to the Columbia River and anadromous juvenile salmonids from Upper Columbia River ESUs/DPSs. While smolt PIT tags have been recovered from this colony (S. Hopkins, Chelan County Public Utility District, pers. comm.), the colony has remained small (20 to 50 breeding pairs; *Table 6.1* above). Accurate estimates of smolt predation rates – those corrected for detection and deposition probabilities – are not currently available for cormorants nesting at the colony at the mouth of the Okanagan River.

6.4.8.2 Gulls – Unlike Caspian terns, double-crested cormorants, and American white pelicans, gulls are not strictly piscivorous and are considered omnivorous and generalist predators (Winkler 1996). Gulls are also known to consume dead or moribund fish and to kleptoparasitize (steal fish from) other piscivorous waterbirds, such as Caspian terns. Consequently, smolt PIT tag recoveries on gull colonies may be more indicative of consumption rates of smolts, rather than predation rates (Roby et al. 2016; see *below*).

Similar to estimates of smolt predation rates by double-crested cormorants nesting at colonies in the CPR, estimates of smolt consumption rates by gulls nesting at colonies in the CPR reported herein are higher than those previously reported (e.g., Evans et al. 2012, USACE 2014) because previous estimates did not include an accurate estimate of on-colony PIT tag deposition probabilities for gulls, probabilities that are low (ca. 0.15) relative to those of other piscivorous colonial waterbird species (Hostetter et al. 2015; see also *Appendix A.2*). Data from Zorich et al. (2011) and Roby et al. (2014) indicated that gulls macerate PIT tags in their gizzards following ingestion, resulting in a smaller fraction of ingested PIT tags being egested on-colony in detectable condition. Standardized, colony-specific estimates of gull consumption rates of juvenile salmonids are reported below for those gull colonies scanned for smolt PIT tags beginning in 2008.

Goose Island: Smolt PIT tags were recovered from the mixed California and ring-billed gull colony on Goose Island in Potholes Reservoir in 2012 (Roby et al. 2013). These smolt PIT tag recoveries on the Goose Island gull colony indicated that consumption rates were less than 0.1% for all available salmonid ESUs/DPSs evaluated, except for Upper Columbia River steelhead; Goose Island gulls consumed an estimated 2.8% (1.1–5.6%) of available Upper Columbia River steelhead smolts (*Appendix B.2, Table B.27*). Despite the relatively high consumption rate of Upper Columbia River steelhead, an unknown proportion of smolt PIT tags recovered on the Goose Island gull colony in 2012 were from smolts that were kleptoparasitized from Caspian terns that were also nesting on Goose Island (see *Chapter 2*). Gulls selectively kleptoparasitize larger-sized fish from Caspian terns (Adkins et al. 2011), and steelhead smolts are, on average, significantly larger than salmon smolts, which may explain the substantially higher rates at which gulls nesting on Goose Island consumed steelhead compared to smolts of other salmonid species. As such, the actual percentage of steelhead directly captured or procured (i.e. not kleptoparasitized) by gulls nesting on Goose Island is likely lower than 2.8%, but additional research is needed to obtain more accurate estimates of gull consumption rates at this colony.

Island 20: Gull consumption rates of juvenile salmonids based on smolt PIT tag recoveries were estimated on the Island 20 mixed-species California and ring-billed gull colony during 2013–2018. Consumption rates of salmon ESUs were less than 1.0% in all study years (*Table 6.6* above), with slightly higher average annual consumption rates observed for Snake River sockeye salmon (ca. 0.9%) compared to other salmon ESUs (ca. 0.2–0.5%; *Table 6.6* above*;* see *Appendix B.2, Table B.27* for annual estimates). Consumption rates were significantly higher for steelhead DPSs, however, with average annual consumption rates for Upper Columbia River steelhead and Snake River steelhead of 4.1% (3.3–4.9%) and 1.6% (1.2–1.9%), respectively. Estimated consumption rates were as high as 7.9% (5.3–12.0%) and 3.6% (1.7–6.9%) for Upper Columbia River and Snake River steelhead DPSs, respectively, in 2015 (see *Appendix B.2, Table B.27*). Unlike the Goose Island gull colony (see *above*), there was no Caspian tern colony associated with the gull colony on Island 20, so consumption estimates are of smolts depredated or obtained by gulls initially, rather than those kleptoparasitized from terns. The difference in consumption rates between Upper Columbia River and Snake River steelhead by Island 20 gulls was presumably associated with the location of Island 20 on the middle Columbia River, where gulls must commute further from the colony site to forage on smolts from Snake River populations compared to those from Upper Columbia River populations (*Figure 6.1* above).

Inter-annual comparisons of consumption rates for individual ESUs/DPSs indicated that rates were consistently higher in 2015 relative to other years (see *Appendix B.2, Table B.27*). We hypothesize that that the larger than average size of the Island 20 gull colony in 2015 (*Table 6.2* above), coupled with poor river conditions for smolt out-migration (high temperatures and reduced flows), were associated with higher-than-average smolt consumption rates by Island 20 gulls in 2015 compared to other years (Roby et al. 2016; see also *Chapter 10* for an in-depth review of factors that influence smolt susceptibility to gull consumption). In 2019, estimates of consumption rates by Island 20 gulls were again low for salmon ESUs (less than 0.3%), but comparatively high on Upper Columbia River steelhead (7.4% [4.8–11.7%]), yet river conditions in 2019 were near the 10-year average in terms of flow and temperature (FPC 2019). Collectively, consumption rate estimates indicate that, in most years, gulls nesting on Island 20 posed little risk to survival of salmon smolts, particularly those belonging to Snake River salmon ESUs. Consumption rates for steelhead DPSs, however, were comparable to those for other gull colonies in the region (see *below*) and, in the case of Upper Columbia River steelhead, consumption rates were similar to or greater than predation rates measured at nearby Caspian tern and double-crested cormorant colonies in the CPR (see *Appendix B.2, Tables B.21-25*).

Badger Island: Similar to consumption rates by Island 20 gulls, consumption rates by California gulls nesting on Badger Island were generally lower on salmon ESUs compared with steelhead

DPSs. Average annual consumption rates on salmon ESUs were less than 1% for all populations, except for Snake River sockeye salmon, where an average of 2.7% (1.2–4.8%) of available smolts were consumed annually during 2015–2019 (*Table 6.6* above; see also *Appendix B.2, Table B.27* for annual estimates). Average annual consumption rates for steelhead DPSs were significantly higher than those for salmon ESUs in most years, with rates averaging 5.0% (3.5–4.3%) and 3.3% (2.5–4.3%) for Snake River and Upper Columbia River steelhead, respectively (*Table 6.6* above). The difference in consumption rates between Upper Columbia and Snake River steelhead DPSs was less than for those gull colonies located on the middle Columbia River, presumably because Badger Island is located downstream of the confluence of the Snake and Columbia rivers (*Figure 6.1* above).

Among steelhead DPSs, consumption rate estimates by Badger Island gulls were highly variable, with annual estimates ranging from a low of 1.3% (0.6–2.6%) in 2017 to a high of 10.9% (6.7–17.7%) in 2019 (see *Appendix B.2, Table B27*). There is some evidence that consumption rates have increased as the size (number of breeding adults) of the Badger Island gull colony has increased (*Table 6.2* above). It should also be noted that small samples sizes of available PIT-tagged smolts in recent years (i.e. smolts of Snake River sockeye salmon and Upper Columbia River steelhead) have resulted in imprecise estimates of consumption rates (based on the size of the 95% credible intervals; see *Appendix B.2, Table B.27*). Furthermore, since the gull colony became established on Badger Island in 2015, the extent of over-lap in the areas where American white pelicans and gulls are nesting on Badger Island has increased. As such, it is likely that some of the PIT tags recovered on gull nesting areas were deposited by pelicans and erroneously attributed to gull consumption. Estimated predation rates by American white pelicans nesting on Badger Island prior to 2015 – when only pelicans nested on the island – however, indicated that pelicans consumed a very small percentage of available steelhead smolts, particularly Upper Columbia River steelhead (Evans et al. 2012). As such, tags deposited by pelicans on gull nesting areas at Badger Island are unlikely to explain the recent high estimates of consumption rates by gulls for steelhead DPSs.

Crescent Island: Smolt PIT tags and data on PIT tag detection and deposition probabilities that are necessary to accurately estimate consumption rates by gulls nesting on Crescent Island were available during 2007–2015. Average annual consumption rates indicated that, like other gull colonies in the CPR, consumption rates were generally low (less than 1%) for salmon ESUs (*Table 6.6* above*;* see also *Appendix B.2, Table B.27* for annual estimates). Of those salmon ESUs evaluated, consumption rate estimates were often higher on Snake River sockeye salmon, with an average annual estimate of 2.2% (*Table 6.6* above). Also, similar to other gull colonies in the CPR, consumption rates for PIT-tagged smolts were consistently the highest for steelhead DPSs, with average annual consumption rates of 4.8% (4.1–5.6%) and 5.8% (5.0–6.9%) for Snake River and Upper Columbia River steelhead, respectively. Consumption rate estimates by gulls nesting on Crescent Island were similar to or higher than predation rates by Caspian terns nesting on Crescent Island (see *Appendix B.2, Table B21*). Again, these estimates of gull consumption rates are higher than previously reported because, prior to 2014, estimates of gull consumption rates of salmonid smolts in the CPR were minimums (Evans et al. 2012, USACE 2014).

Higher consumptions rates of steelhead smolts by Crescent Island gulls may have been associated with the larger average size of steelhead smolts compared to salmon smolts and the finding that gulls disproportionately consume larger-sized smolts (Evans et al. 2019b; see also *Chapter 10* for a more detailed discussion of factors that influence smolt susceptibility to gull consumption). Furthermore, analogous to consumption rate estimates for gulls nesting on Goose Island in 2012 (see *above*), some fraction of the smolt PIT tags deposited on the Crescent Island gull colony were likely kleptoparasitized from Caspian terns that also nested on Crescent Island. Adkins et al. (2011) observed that kleptoparasitism rates by Crescent Island gulls ranged from 10 to 17% of smolts delivered as Caspian tern bill loads and that steelhead smolts were more likely to be kleptoparasitized than salmon smolts due to the larger average size of steelhead smolts.

Blalock Islands Complex: Since 2012, there have been at least two separate gull nesting areas within the Blalock Islands complex in John Day Reservoir, one on Anvil Island and one on Straight Six Island (*Figure 6.1* above). Small numbers of gulls have also nested on Rock Island in some years. Of the two primary gull nesting areas, smolt consumption rates have been significantly higher for gulls nesting on Anvil Island compared to gulls nesting on Straight Six Island (see section *6.4.3 Colony Size & Nesting Success* above). Differences in consumption rates were attributed to both a difference in the number of gulls nesting on the two islands, with substantially more gulls nesting on Anvil Island, and a difference in the gull species composition, with Anvil Island dominated by nesting California gulls and Straight Six Island dominated by nesting ring-billed gulls. Data from Hostetter et al. (2015) indicated that per capita (per bird) consumption of juvenile salmonids was greater for gull colonies dominated by California gulls compared to those dominated by ring-billed gulls. This difference in smolt consumption rates between the gull species is likely due to differences in body size and energy requirements (Winkler 1996), as well as the proportion of the diet that consists of fish (Collis et al. 2002), both of which are greater for California gulls compared with ring-billed gulls. For the purpose of this and other consumption rate studies in the CPR (Evans et al. 2016a, Evans et al. 2019a), consumption estimates were combined for gulls nesting on Anvil and Straight Six islands (collectively referred to as “Blalock Islands gulls”). Estimates of smolt consumption rates by gulls nesting on the Blalock Islands were available during 2013–2019.

Results indicated that consumption rates, on average, were less than 2% on all PIT-tagged salmon ESUs and less than 4% of all PIT-tagged steelhead DPSs, with the highest estimated consumption rate observed for Upper Columbia River steelhead at 3.9% (3.0–4.8%; *Table 6.6* above*; see* *Appendix B, Table B.26*). Average annual consumption rates by Blalock Islands gulls were similar to those of gulls nesting on Island 20, but consistently lower than those of gulls nesting on Badger Island, Crescent Island, and Miller Rocks (see *below* and *Table 6.6* above). Analogous to gulls nesting on Crescent Island, some fraction of smolt PIT tags deposited on the Blalock Islands gull colony could have been kleptoparasitized from Caspian terns, but Caspian terns did not nest on the same islands as gulls within the Blalock Islands.

Miller Rocks: Most gulls nesting on Miller Rocks are California gulls, with very few, if any, ring-billed gulls nesting there in most years (see section *6.4.3 Colony Size & Nesting Success* above).

Smolt consumption rates by gulls nesting on Miller Rocks were, on average, among the highest of any piscivorous waterbird colony in the CPR during 2007–2019, including colonies of Caspian terns and double-crested cormorants (*Table 6.6* above; see also *Appendix B.2, Table B.26* for annual estimates of consumption). The highest estimate of average annual ESU/DPS-specific consumption rate by gulls nesting at Miller Rocks was on Upper Columbia River steelhead at 8.2% (6.9–9.3%), with rates as high as 13.2% (8.3–21.1%) observed in one year (see *Appendix B.2, Table B.26*). Unlike the other gull colonies in CPR, where consumption rates for salmon ESUs were generally less than 1.0%, an estimated 6.2% (4.8–7.7%) of Snake River sockeye salmon were consumed, on average, by Miller Rocks gulls, the highest average annual consumption rate for sockeye salmon observed at any piscivorous waterbird colony in the CRB, including the large Caspian tern and double-crested cormorant colonies on East Sand Island in Columbia River estuary (see *Chapter 1* and *Chapter 4*). There is evidence that, relative to other salmon ESUs, sockeye salmon were disproportionately consumed at most gull colonies evaluated in the CPR (see also *Chapter 7).*

As was the case at other gull colonies in the CPR, high consumption rates of steelhead DPSs by Miller Rocks gulls were presumably associated with the larger average size of steelhead smolts compared to salmon smolts (see *above*). Evans et al. (2016, 2019a) also hypothesized that steelhead smolts may be more susceptible to plunge-diving waterbirds, like California gulls and Caspian terns, because juvenile steelhead migrate near the surface (see *Chapter 10* for a more detailed discussion of factors linked to smolt susceptibility to gull consumption). The reason why sockeye salmon smolts were especially susceptible to gull consumption, however, is unknown (see *Chapter 7* for a more detailed discussion). The Miller Rocks gull colony, however, is near both the John Day Dam and The Dalles Dam (18 Rkm and 23 Rkm, respectively), where smolts maybe especially susceptible to consumption by gulls (see section *6.4.6 Foraging Behavior* above and Evans et al. 2016a). Gull consumption of smolts, however, is not limited to foraging on dead, moribund, or stunned fish near dams, with gulls consuming substantial numbers of smolts in good-condition from open reservoirs and free-flowing sections of the river as well (see Evans et al. 2016a and *Chapter 10* for a detailed discussion).

Other colonies: Several other gull colonies in the CPR are either no longer extant (i.e. the colony site was abandoned by nesting gulls) or were active, but standardized estimates of smolt consumption rates based on PIT tag recoveries on-colony are not available. For example, Island 18 was formerly the site of the largest mixed-species California and ring-billed gull colony in the CPR, with over 20,000 breeding adults counted on-colony in some years (*Table 6.2*). The Island 18 colony, however, was abandoned in 2009. Three Mile Canyon Island in John Day Reservoir also formerly supported a large mixed-species gull colony until it was abandoned in 2012 (see section *6.4.4 Limiting Factors for Colony Size & Nesting Success* above). Other gull colonies within foraging distance of the Columbia or Snake rivers in the CPR currently exist on Banks Lake, Lenore Lake, and Sprague Lake (*Table 6.2* above). These gull colonies, however, are located on islands that are a considerable distance from the Columbia and Snake rivers (generally greater than 60 km, depending on the colony). Given the finding that gulls generally forage in closer proximity to their nesting site (less than 50 km; see section *6.4.6 Foraging Behavior* above), it is unlikely that gulls from these colonies pose a significant risk to smolt survival in the Columbia and Snake rivers, but data to support or refute this hypothesis are lacking. In the CRE, extant colonies of glaucous-winged/western gulls and ring-billed gulls exist on East Sand Island and Rice Island (see section *6.4.3 Colony Size & Nesting Success* above). Smolt PIT tag recovery using standardized procedures has not occurred at these colonies, however, and no accurate estimates of consumption rates are therefore available. Given the consumption rates observed at some gull colonies in the CPR, evaluating current consumption rates by gulls nesting at these colonies may be warranted. On-colony PIT tag deposition probabilities have not been determined for glaucous-winged/western gulls, however, and it is unknown whether deposition probabilities for California gulls, as measured by Hostetter et al. (2015), could be used as an accurate surrogate given the differences in diet composition of the two gull species observed by Collis et al. (2002).

6.4.8.3 American white pelicans – Unlike the other piscivorous waterbird species evaluated herein, American white pelicans are capable of consuming both juvenile and adult-sized salmonids due to their large size and gape-width (size and width of mouth). For example, adult-sized fish greater than 700 mm fork-length have been confirmed in the diet of American white pelicans (Scoppettone et al. 2006; this study), including adult salmonids and common carp (*Cyprinus carpio)* found on the Badger Island pelican colony following the breeding season (see *below*). American white pelicans capture fish primarily within the top meter of the water column, limiting foraging to surface-oriented fish or to fish occupying shallow water habitats (Knopf and Evans 2004).

No estimates of PIT tag deposition probabilities on-colony have been developed for American white pelicans, so even after adjusting for detection probabilities, the estimated predation rates presented herein represent minimum estimates of smolt losses to pelicans. Because the maximum reported foraging distance of American white pelicans (300 km; Scoppettone et al. 2006) is greater than that of Caspian terns, double-crested cormorants, and California and ring-billed gulls (see section *6.4.6 Foraging Behavior* above), it is possible that nesting American white pelicans spend a greater proportion of their time off-colony as opposed to on-colony. If true, PIT tag deposition probabilities for pelicans may differ from those of other piscivorous colonial waterbirds in the present study (Payton et al. 2020). Minimum predation rates on salmonids by American white pelicans nesting on Badger Island, the only pelican colony in the CRB that has been scanned for smolt PIT tags, are presented below.

Badger Island: Data to estimate minimum predation rates are available for the Badger Island pelican colony during 2007–2019. Results indicated that minimum predation rates by American white pelicans nesting on Badger Island were, on average, less than 0.3% for all ESA-listed ESUs/DPSs evaluated (*Table 6.6* above*; see Appendix B, Table B.29*). Despite a steady increase in the size of the Badger Island pelican colony, from 913 adults counted in 2007 to 3,330 adults in 2018 (*Table 6.4* above), estimates of predation rates remained low throughout the 13-year time series, with predation rates less than 0.6% in all study years and for all ESA-listed ESUs/DPSs of salmonids evaluated. In a study of non-ESA-listed subyearling Chinook salmon, however, Payton et al. (2020) estimated that predation rates by American white pelicans nesting on Badger Island were substantial in some but not all years, with estimates as high as 10% reported in

2016. Due to persistent uncertainties in PIT tag deposition probabilities for white pelicans and evidence that fish from particular stocks may be more susceptible to pelican predation than others, additional research regarding predation by pelicans nesting at Badger Island and at other colonies in the CRB may be warranted (Payton et al. 2020).

In addition to PIT tags from juvenile salmonids, PIT tags from adult salmonids have been recovered on the American white pelican colony on Badger Island. For example, during 2007–2019, 470 PIT tags from adult-sized salmonids (detections of individual fish) have been recovered on the Badger Island pelican colony. PIT tags were from adult steelhead (n = 339, including both pre- and post-spawned [kelt] adults), adult sockeye salmon (n = 95), adult Chinook salmon (n = 32, including jack salmon), and adult-sized bull trout (n = 4). Except for the bull trout, most fish were tagged as adult salmonids at the Bonneville Dam fishway. Adult salmonids recovered on Badger Island ranged in size from 380 mm to 770 mm at the time of tagging. Standardized estimates of predation rates on adult salmonids by Badger Island pelicans, however, are not available.

6.4.8.4 Brandt's cormorants – Smolt predation rates were measured for Brandt's cormorants nesting on East Sand Island, formerly the sole breeding colony for the species in the Columbia River estuary (*Figure 6.1* above). Brandt's cormorants nested in a mixed-species colony with double-crested cormorants on East Sand Island; predation rates based on smolt PIT tag recoveries and corrected for both PIT tag detection and deposition probabilities were estimated for the 2012-2015 breeding seasons by Evans et al. (2016c). All estimates of predation rates were made under the assumption that deposition probabilities for Brandt's cormorants were the same as those measured for double-crested cormorants (see *Appendix A.2* and *below*). Results indicated that predation rates on smolts by Brandt's cormorants nesting on East Sand Island were less than 1.0% for all available ESA-listed salmonid ESUs/DPSs in each year (see *Appendix B.2, Table B.28*). For most ESUs/DPSs in most years, predation rates were below 0.5%. Predation rates by Brandt's cormorants were so low that differences among salmonid species and among ESUs/DPSs were not readily discernible, and likely not biologically meaningful (Evans et al. 2016c). As noted above, estimates of PIT tag deposition rates for double-crested cormorants were used to infer deposition rates for Brandt's cormorants, as there are currently no estimates of PIT tag deposition rates for Brandt's cormorants in the literature. Although there are similarities in the foraging behavior, nesting behavior, and general life history between Brandt's cormorants and double-crested cormorants (Couch and Lance 2004, Peck-Richardson et al. 2018), it is possible that deposition rates differ between the two species and, if so, estimates of predation rates by Brandt's cormorants may be biased to some degree. Regardless, given how consistently low the estimates of predation rates were, in part due to the paucity of smolt PIT tags recovered on or near Brandt's cormorant nests, differences in deposition rates between double-crested and Brandt's cormorants would presumably have little influence on the accuracy of predation rate estimates for Brandt's cormorants (see *Appendix B.2, Table B.28*).

6.4.9 Per Capita Predation/Consumption Rates

Several studies have used information on the size of piscivorous waterbird colonies (number of breeding pairs) and smolt predation/consumption rates (as estimated by smolt PIT tags recovered on-colony) to estimate per capita (per bird) predation/consumption rates (Evans et al. 2012, Collis et al. 2016, Evans et al. 2016c). These studies can provide valuable information on the foraging ecology of piscivorous waterbirds from specific nesting colonies and have also been used to help prioritize colonies for potential management to enhance smolt survival (USACE 2014). For instance, the decision to manage specific Caspian tern breeding colonies in the CPR as part of the *IAPMP* (see *Chapter 2*) were due to high per capita predation rates on steelhead DPSs by Caspian terns nesting at those colonies, coupled with the relatively large size of the colonies compared with other Caspian tern colonies in the CPR (USACE 2014).

Investigations of per capita predation/consumption rates indicated that smolt losses varied considerably by piscivorous waterbird species (tern, cormorant, gull), salmonid ESU/DPS, and colony location (*Figure 6.20*). For steelhead DPSs, per capita predation rates were often the highest for Caspian terns, followed by double-crested cormorants. For salmon ESUs/DPSs, however, per capita predation rates were often highest for double-crested cormorants, followed by Caspian terns. Estimates of per capita predation rates for Brandt's cormorants nesting in the CRE were the lowest of all piscivorous waterbird species evaluated, regardless of the ESU/DPS or the year. Due to a lack of measurements of PIT tag deposition probabilities, no accurate estimates of per capita predation rates could be calculated for American white pelicans, so inter-specific comparisons of per capita predation/consumption rates are limited to Caspian terns, double-crested cormorants, Brandt's cormorants, and California and ring-billed gulls (*Figure 6.20*).

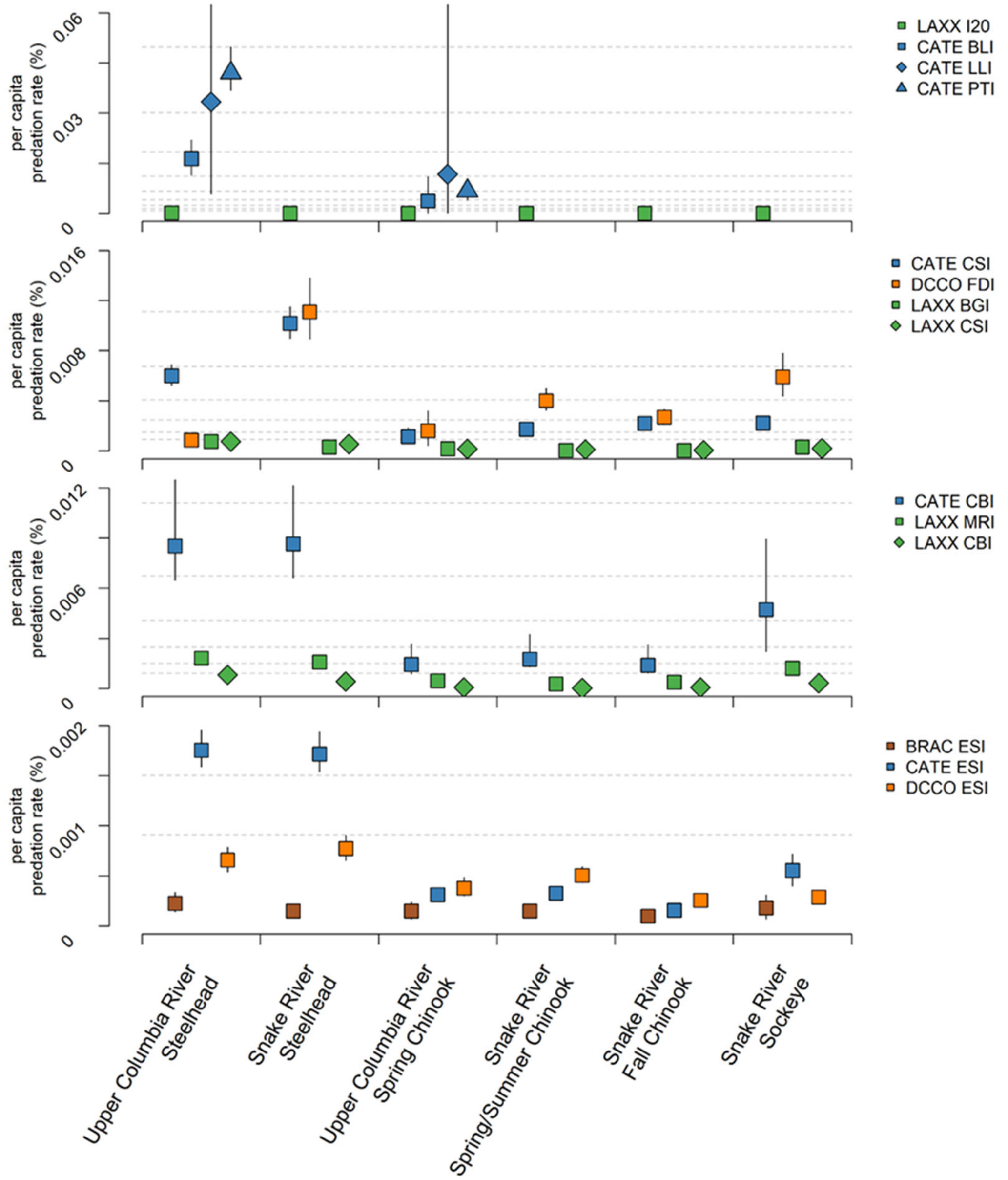

Figure 6.20. Average annual estimates of per capita predation/consumption rates by colony and ESA-listed salmonid population during 2007–2018. Error bars denote 95% credible intervals. Avian species include Caspian terns (CATE), California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), and Brandt's cormorants (BRAC). Colony sizes were based on the peak number of breeding pairs for CATE, DCCO, and BRAC or the peak number of adults for LAXX (see Appendix A for details). Colonies include Potholes Reservoir islands (PTI), Banks Lake islands (BLI), Lenore Lake islands (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks (MRI), and East Sand Island (ESI). Comparisons were limited to salmonid populations that were susceptible to predation/consumption by colonies foraging within the same river reach and where data were available in two or more years. Logarithmic scales (dotted line) are provided for relative comparisons across river reaches.

Results indicated that the high smolt consumption rates by gulls nesting at some colonies in the CPR (*Table 6.6* above) were associated with the large size of these colonies (several thousand breeding adults), rather than high per capita consumption rates. For example, gull colonies on Miller Rocks, Island 20, Crescent Island, and the Blalock Islands were all an order of magnitude greater in size than nearby Caspian tern and double-crested cormorant colonies. As such, although consumption rate estimates for gulls nesting at these colonies were similar to or greater than those of tern and cormorant colonies, per capita consumption rate estimates for gulls were significantly lower than predation rates for terns and cormorants (Evans et al. 2012; *Figure 6.20* above). Previous research indicates that fish in general, and salmonids in particular, comprised a small proportion of the diets of California and ring-billed gulls nesting on islands in the CPR. For instance, foregut contents analysis of gulls nesting at Miller Rocks and Crescent Island (Collis et al. 2002) indicated that juvenile salmonids comprised 3% (by mass) of food biomass. In contrast, salmonids comprised 74% (by mass) of the diets of Caspian terns nesting on Crescent Island (Antolos et al. 2005) and 22% (by mass) of the diets of double-crested cormorants nesting on Foundation Island (Lyons et al. 2011b; this study). As such, large colony size and behavioral flexibility to exploit temporally available food sources resulted in high rates of consumption by gulls nesting at specific colonies in the CPR, despite low per capita consumption rates.

Comparisons of per capita smolt predation rates among colonies of Caspian terns and double-crested cormorants indicated that per capita smolt losses varied by not only the size of the colony, but also the location of the colony (Roby et al. 2003, Evans et al. 2012; *Figure 6.20* above). For example, within the same piscivorous waterbird species, per capita predation rates on steelhead DPSs by Caspian terns and double-crested cormorants foraging in strictly freshwater (e.g., mainstem Columbia and Snake rivers) were significantly higher than those of Caspian terns and double-crested cormorants foraging on smolts in the Columbia River estuary, particularly colonies located near the mouth of Columbia River in the marine zone of the estuary (i.e. East Sand Island). Per capita predation rates by Caspian terns and double-crested cormorants nesting in the CPR were higher due to the higher prevalence of juvenile salmonids in the diet of terns and cormorants nesting at inland colonies compared with their estuary counterparts (Collis et al. 2002, Roby et al. 2003, Antolos et al. 2005). *Chapter 5* presents a more detailed discussion and a new analysis of how colony location is linked to foraging habitat and associated per capita predation rates on juvenile salmonids.

6.4.10 Factors Affecting Smolt Susceptibility to Predation

As noted throughout this summary, multiple biotic factor (e.g., colony size, fish abundance, fish condition) and abiotic factors (e.g., river flow rate and turbidity) have been linked to the susceptibility of juvenile salmonids to predation/consumption by piscivorous colonial waterbirds. *Chapter 10* provides a detailed summary and synthesis of this topic.

6.5 Discussion

During the last two decades breeding colonies of Caspian terns, double-crested cormorants, Brandt's cormorants, glaucous-winged/western gulls, California gulls, ring-billed gulls, and American white pelicans have been monitored throughout much of the Columbia River basin. These piscivorous colonial waterbirds nest abundantly but locally in the basin during April–August, a period that coincides with the annual out-migration of millions of juvenile salmonids to the Pacific Ocean (Adkins et al. 2014a, Evans et al 2016a, see also *Chapter 10*). Numerous studies have demonstrated that Caspian terns and double-crested cormorants nesting at colonies in the CRB can consume substantial numbers of juvenile salmonids and may therefore limit the survival of some ESA-listed salmonid populations in the CRB (Roby et al. 2003, Lyons 2010, Evans et al. 2012, Evans et al. 2019a). In aggregate, the numbers of unmanaged piscivorous colonial waterbirds nesting in the CRB (i.e. double-crested cormorants in the Columbia Plateau region, gulls, American white pelicans, and Brandt's cormorants) exceed the numbers of Caspian terns and double-crested cormorants that formerly nested at managed colonies in the CRB (see *Appendix B.1*), yet the impacts of these unmanaged piscivorous waterbirds on smolt survival are not well understood.

Available data indicate that regional populations of some of these unmanaged piscivorous waterbird species and colonies in the CRB are growing (i.e. American white pelicans), while others are apparently stable or declining (i.e. Brandt's cormorant and glaucous-winged/western gulls in the CRE, double-crested cormorants in the CPR, California and ring-billed gulls throughout the basin). Historically, these other piscivorous waterbird colonies have not been the subject of the same level of research and monitoring as the Caspian tern and double-crested cormorant colonies that have been subjected to management to reduce impacts on smolt survival, nor have they received much management consideration. New information provided by Hostetter et al. (2015) regarding smolt PIT tag deposition probabilities for double-crested cormorants and California gulls, along with more recent studies that have looked at the system-wide impacts of avian predation/consumption on smolt survival (Evans et al. 2016a, Evans et al. 2019b, Payton et al. 2019, Payton et al. 2020) have focused new attention on the potential impacts on smolt survival of piscivorous waterbirds from unmanaged colonies in the CRB. This synthesis of available data on unmanaged colonies of piscivorous waterbirds presented here was assembled to identify which unmanaged species and colonies may pose a threat to survival of juvenile salmonids in the CRB and which piscivorous waterbird colonies may warrant additional monitoring and evaluation for potential management.

6.5.1 Size of Unmanaged Piscivorous Waterbirds Colonies

Results from this study demonstrate that the demography and distribution of piscivorous colonial waterbirds nesting in the CRB is dynamic. Habitat loss (natural or induced), colony disturbance, immigration, and other stochastic environmental factors influenced the number of breeding birds at both the species and colony levels. As such, no clear trend in numbers of breeding birds could be identified across all species. California and ring-billed gulls were the most abundant and widespread piscivorous colonial waterbirds in the CPR, with the number of breeding gulls counted on colonies often an order of magnitude greater than the numbers of Caspian terns, double-crested cormorants, and American white pelicans nesting at colonies in the region. For example, in some years more than 50,000 gulls were counted collectively at as many as eight different colonies in the CPR. In comparison, the total number of double-crested cormorants nesting in the CPR has not exceeded 1,600 breeding pairs in the last quarter century and the total number of American white pelicans has not exceed 3,500 breeding adults in the same period. In the CPR, gulls have utilized diverse nesting habitats on islands, from small, temporary colonies on low-lying gravel islands to large colonies on perennial islands with a history of nesting that spans several decades.

There is evidence that the total number of gulls breeding in the CPR, particularly those at colony sites along the Columbia River, have declined in the past two decades. The most recent comprehensive survey (2013) counted a total of approximately 56,500 gulls at colonies in the CPR, with roughly 31,400 (56%) counted on colony sites on islands in the Columbia River. These totals are considerably less than those observed during other comprehensive surveys in 1997 (ca. 57,000 gulls on Columbia River colonies; Collis et al. 2002) and 2009 (ca. 67,650 total gulls at colonies in the CPR and ca. 41,700 gulls at colonies on the Columbia River; Adkins et al. 2014a). The total number of gulls nesting at colonies on the Columbia River has apparently stabilized at approximately 29,000 gulls during 2013–2018; however, this represents about a 50% decline from numbers counted in 1997 and a 30% decline from numbers counted in 2009. The cause(s) of this apparent decline is not understood, and it remains unclear whether a similar decline occurred at all colonies throughout the CPR, not just gull colonies on the Columbia River where the majority of gulls nest, but recent comprehensive survey data for gull colonies off the Columbia River are lacking. During the study period, declines or abandonment events at gull colonies were linked to human disturbance (e.g., Island 18, Crescent Island), mammalian predators (e.g., Island 18, Three Mile Canyon Island), and encroaching vegetation that reduced or eliminated suitable nesting habitat (e.g., Solstice Island on Potholes Reservoir, Blalock Islands Complex). In contrast, new and/or growing gull colonies were also documented during the present study, including at the Blalock Islands Complex, Lenore Lake, and, most notably, at Badger Island, which was first colonized in 2015 following the abandonment of the gull colony on nearby Crescent Island (see also *Chapter 2*).

Because data on gull colony size collected during this study were often incidental or collected infrequently, inter-annual variation in the number of gulls counted on colonies in the CPR should be interpreted cautiously. For example, the census of gull colonies as part of this study may not have consistently been timed during the peak of colony attendance. Also, the census of gull colonies did not always occur at a particular time of day, and diurnal patterns in colony attendance by gulls have been observed in the CPR, whereby fewer gulls are typically present on-colony during mid-day (Conover and Miller 1980). Furthermore, colony connectivity for California gulls and ring-billed gulls is not well understood (Winkler 1996, Pollet et al. 2012). It is thus difficult to make strong inferences about gull population trends in the CPR based on the results presented here. Although the number of gulls breeding in the CPR is apparently lower than in 1997 (Collis et al. 2002), gull consumption of juvenile salmonids may still negatively influence smolt survival in the CPR. Results from this study and others demonstrate that colony size, particularly for gulls, is not always a reliable indicator of smolt predation/consumption rates (Collis et al. 2002, Hostetter et al. 2015, Evans et al. 2019a). The location of gull colonies (e.g., proximity to hydropower dams) and the species composition of the gull colony (i.e. California gull vs. ring-billed gull) were apparently more important as factors influencing smolt consumption rates by gulls in the CPR (Collis et al. 2002, Zorich et al. 2010, Evans et al 2016a, Evans et al. 2019b).

Few estimates of colony size for double-crested cormorant colonies in the CPR are available after 2013, but the stable trend in the CPR breeding population that was documented by Adkins et al. (2014a) may have persisted in subsequent years. Between 1,150 breeding pairs and 1,575 breeding pairs of double-crested cormorants were counted nesting in the CPR during 2004–2013. Abandonment of formerly the largest cormorant colony in the CPR (i.e. the Potholes Reservoir colony) is not well understood. Nesting bald eagles were observed near this colony in some years; disturbance from bald eagles does not appear to be the sole cause of abandonment, however, as this location still supports colonies of great egrets and great blue herons. Although no evidence has been found of human disturbance or persecution of cormorants at the Potholes Reservoir colony has been found, it may have been a factor in colony abandonment.

Double-crested cormorants that previously nested at Potholes Reservoir likely dispersed to incipient colonies less than 100 km from Potholes Reservoir (i.e. Locke Island Complex [38 km], Crab Creek [45 km], Lenore Lake [48 km], or contributed to colony growth at Harper Island on Sprague Lake [103 km]). Although colony connectivity for double-crested cormorants nesting in the CPR is poorly understood, regional redistribution of this magnitude is common for double-crested cormorants (Carter et al. 1995, Duerr et al. 2009, Adkins et al. 2014b). Given the small size of most extant double-crested cormorant colonies in the CPR (e.g., Locke Island Complex, Okanogan River mouth) or the colony's distance from the Federal Columbia River Power System where predation on juvenile salmonids may occur (e.g., Sprague Lake, Lenore Lake, Crab Creek), only the Foundation Island cormorant colony apparently poses a significant risk for smolt survival in the CPR. Additional research and monitoring may be warranted at the Foundation Island cormorant colony, where colony size is thought to be stable, but the colony has not been counted since 2014.

Badger Island remains the only known breeding colony of American white pelicans in the CPR. American white pelicans were the only species of piscivorous colonial waterbirds whose numbers dramatically increased during the study period, growing by more than 300% since 2005. It is unknown whether this increase was due to intrinsic growth (because data on pelican nesting success at the Badger Island colony are not available) or via recruitment of breeding adults to the Badger Island colony from other colonies in western North America. Limited data exist linking pelicans in the CRB to the western population at large. For example, no American white pelicans were tracked to Badger Island in a recent telemetry study of pelicans dispersing from Utah, nor to any other extant colonies in southern Oregon and northeast California

(UDWR 2020). Nesting habitat for pelicans on Badger Island does not appear to be a limiting factor and continued growth of this colony is possible.

In addition to the well-studied Caspian tern and double-crested cormorant colonies in the Columbia River estuary, the estuary also supports sizeable colonies of glaucous-winged/western gulls, ring-billed gulls, and Brandt's cormorants, plus a small and recently established colony of American white pelicans. These colonies have not been rigorously monitored and much of the information presented here has been collected incidentally as part of other studies. The total number of gulls breeding in the CRE is in decline, with an estimated 10,000 and 6,000 breeding adults counted on-colony at the peak of nesting in 1997 and 2015, respectively. While some movement among gull colonies in CRE has apparently occurred, most gull nesting in the CRE occurs on East Sand Island, where both glaucous-winged/western gulls and ring-billed gulls continue to breed.

The numbers of Brandt's cormorants nesting on East Sand Island increased dramatically during 2006–2018. In 2018, the Brandt's cormorant colony on East Sand Island was more than 50 times larger than when nesting by Brandt's cormorants was first detected in the CRE in 1997 and was one of the largest colonies for the species recorded in the coastal regions of Washington and Oregon (Couch and Lance 2004, Speich and Wahl 1989, Naughton et al. 2007). Post-breeding dispersal and colony connectivity for Brandt's cormorants are not well understood; hence, the emigration patterns from other Brandt's cormorant colonies along the Pacific Coast that led to this growth of the East Sand Island colony are unknown. Following the 2018 breeding season, however, much of nesting habitat used by both double-crested and Brandt's cormorants was eliminated to reduce the number of double-crested cormorants nesting on East Sand Island (see also *Chapter 4).* This management action, along with predation and disturbance by increasing numbers of bald eagles, was presumably responsible for causing Brandt's cormorants to largely abandon the East Sand Island colony in 2019; no nesting on East Sand Island by Brandt's cormorants was observed in 2020. The recent abandonment of the large Brandt's cormorant colony on East Sand Island has led to growth of the incipient colony of Brandt's cormorants on the Astoria-Megler Bridge; Brandt's cormorants were first detected nesting on the Astoria-Megler Bridge in 2018.

Nesting by American white pelicans in the CRE is apparently a recent development, with no historical records prior to this study (Stinson 2016). The number of breeding adult American white pelicans observed in the CRE appeared to be increasing through 2018, but like American white pelicans nesting on Badger Island, it is not known to what extent this increase was due to intrinsic growth vs. recruitment of breeding birds from other colonies in western North America, including the Badger Island colony. So far the American white pelican colony in the CRE has remained an order of magnitude smaller than the Badger Island colony, and has moved from Miller Sand Spit to Rice Island, apparently due to human disturbance.

6.5.2 Predation by Unmanaged Piscivorous Waterbirds on Juvenile Salmonids

Studies of avian predation/consumption of juvenile salmonids based on recoveries of smolt PIT tags on-colony have been conducted at numerous unmanaged colonies of piscivorous waterbirds throughout the CRB for more than two decades (Ryan et al. 2001, Collis et al. 2001, Lyons et al. 2011a, Evans et al. 2012). Despite such a long time series, many of the initial estimates of predation/consumption rates on ESA-listed salmonid ESUs/DPSs were reported as minimum estimates of smolt losses (e.g., Antolos et al. 2005, Lyons et al. 2011, Evans et al. 2012, USACE 2014) due to a lack of information on smolt PIT tag deposition probabilities (i.e. proportion of consumed tags deposited on-colony; Hostetter et al. 2015, see also *Appendix A.2*). Consequently, the full impact on salmonid smolt survival of piscivorous waterbirds from most unmanaged colonies were unknown during the development of management plans to reduce avian predation rates on juvenile salmonids in the CRB and has not been fully considered to date. The accurate, standardized estimates of smolt predation/consumption rates presented here indicate that some, but not all, unmanaged colonies of piscivorous waterbirds pose a potential threat to the survival of some ESA-listed populations of juvenile salmonids. For example, smolt predation/consumption rates by gulls and cormorants from some colonies in the CPR were similar to or greater than those of Caspian terns and double-crested cormorants from colonies in CRB that were subjected to managed reductions due to their impact on smolt survival rates (see *below*).

Despite low per capita smolt consumption rates, overall consumption rates of some ESUs/DPSs of salmonids by California and ring-billed gulls nesting at certain breeding colonies in the CPR were substantial. Of the extant gull colonies evaluated, consumption rates were consistently highest for the California gull colony on Miller Rocks in The Dalles Reservoir, where average consumption rates of salmon ESUs were consistently greater than 2% and average consumption rates of steelhead DPSs were greater than 7% during 2007–2019. These estimates of consumption rates were among the highest for any colony of piscivorous waterbirds in the CPR that was evaluated during 2007–2019 and exceeded those observed for all ESUs/DPSs of salmonids at the nearest extant colony of Caspian terns in the Blalock Islands Complex. For example, during this study consumption rates of Upper Columbia River steelhead by gulls nesting at Miller Rocks averaged 8.2% (6.9–9.3%) compared to 4.3% (3.6–5.6%) for Caspian terns nesting in the Blalock Islands Complex during 2014–2019. The average annual consumption rate of Snake River sockeye salmon by Miller Rocks gulls (ca. 6.2%) was also higher than at any other colony of piscivorous waterbirds in the CPR (see also *Chapter 7*). Elevated consumption rates relative to other waterbird colonies in the CPR may be partially attributable to the location of the Miller Rocks gull colony near John Day Dam and The Dalles Dam (18 Rkm and 23 Rkm, respectively) where smolts may be especially susceptible to consumption by gulls (Evans et al. 2016a). As discussed above, the comparatively high consumption rates of steelhead DPSs by gulls nesting at Miller Rocks were presumably associated with the larger average size of steelhead smolts compared to salmon smolts, and greater susceptibility of steelhead smolts to surface feeding predators, such as gulls, because juvenile steelhead tend to migrate near the surface (see also *Chapter 10*).

Other extant California and ring-billed gull colonies in the CPR that may pose a significant risk to smolt survival include the colonies at the Blalock Islands Complex in John Day Reservoir and at Badger Island and Island 20 in McNary Reservoir. Although both per capita consumption rates and colony-based consumption rates at these gull colonies were, on average, lower than those observed at the Miller Rocks gull colony, average consumption rates of Snake River steelhead and Upper Columbia River steelhead were generally high at these three gull colonies (greater than 2%), and similar to those observed for Caspian terns nesting at the most proximate colony in each reservoir. For gulls nesting in the Blalock Islands, average consumption rates were less than 2% for salmon ESUs and less than 4% for steelhead DPSs. These rates were similar, however, to those of Caspian terns nesting in the Blalock Islands during 2014–2019. In McNary Reservoir, consumption rates by gulls nesting on Badger Island and Island 20 were generally similar to or greater than those observed for Caspian terns nesting on Crescent Island prior to implementation of management under the *IAPMP* (see *Chapter 2*). Other extant gull colonies in the CPR (i.e. Potholes Reservoir, Lenore Lake, Sprague Lake, Banks Lake) are within foraging distance of ESA-listed salmonid populations in the Federal Columbia River Power System, but because these colonies are located on islands that are a considerable distance (> 50 km) from the Columbia and Snake rivers, and given the finding that gulls generally forage in close proximity to their colonies, it is unlikely that gulls from these other colonies pose a significant risk to smolt survival.

It is important to note that gulls are known to consume dead or moribund fish, and to kleptoparasitize (steal fish from) other piscivorous waterbirds. As such, some fraction of fish consumed by gulls either died or were injured due to factors other than predation by the gull itself (see also *Chapter 8*). Gulls frequently forage in the tailrace of hydropower dams, where smolts that were stunned, injured, or killed during dam passage are susceptible to being consumed by gulls. Research by Evans et al. (2016a, 2019b), however, indicated that gulls consumed a large proportion of smolts that were previously judged to be in good external condition. Gulls also disproportionately consumed larger-sized smolts (Evans et al. 2019b; *Chapter 10*) and larger smolts may have higher survival probabilities than relatively small smolts (Zabel et al. 2005, Evans et al. 2014, Hostetter et al. 2015b). These results suggest that gull consumption rates may be at least a partially additive source of smolt mortality.

For double-crested cormorants nesting in the CPR, results from this study indicate that only those nesting at the colony on Foundation Island in the middle Columbia River may pose a risk to smolt survival comparable to the risk from piscivorous waterbird colonies in the CPR that have been managed under the *IAPMP* (see *Chapter 2*). For instance, smolt PIT tag recoveries indicated that Foundation Island cormorants consumed an estimated 4.0% (3.4–4.7%) and 3.4% (2.4–4.5%) of Snake River steelhead and Snake River sockeye salmon, respectively. Estimates of predation rates on all other salmonid ESUs/DPSs by Foundation Island cormorants, however, were less than 2% of available smolts. Predation rates on Snake River populations of salmonids were significantly higher than those on Upper Columbia River populations; data on foraging behavior of Foundation Island cormorants indicated that they disproportionately foraged in the lower Snake River compared with the middle Columbia River (Evans et al. 2016a). Unlike Caspian terns and gulls, however, there was little evidence that Foundation Island cormorants disproportionately consumed steelhead smolts relative to salmon smolts. A similar lack of a strong preference for steelhead smolts relative to salmon smolts was also observed for double-crested cormorants nesting at the East Sand Island colony in the Columbia River estuary (see *Chapter 4*), suggesting that cormorants, as pursuit-diving predators, do not focus on depredating smolts near the water surface and instead consume smolts more in proportion to their relative availability throughout the water column (see also *Chapter 9*). The results of diet composition studies at both the Foundation Island and East Sand Island cormorant colonies support this hypothesis, with Chinook salmon being the most common juvenile salmonid found in the cormorant diet and numerically the most abundant species of juvenile salmonid in the Columbia River (see *Chapter 4*). Estimates of predation rates on Chinook salmon ESUs by Foundation Island cormorants during 2007–2014 were similar to those of Caspian terns nesting on nearby Crescent Island prior to management (see *Chapter 2*). Taken together, results from this study indicate that predation by double-crested cormorants nesting on Foundation Island may be a factor limiting smolt survival for some ESA-listed salmonid populations. Although data from other double-crested cormorant colonies in the CPR (e.g., Locke Island, Lenore Lake, North Potholes Reservoir) were limited, the available data suggested that, at current colony sizes, cormorants nesting on these inland waterbodies posed little or no threat to smolt survival.

Estimates of predation rates on juvenile salmonids by American white pelicans nesting on Badger Island were consistently low throughout the present study, despite the observed growth of the Badger Island pelican colony. Predation rates were less than 0.6% on all ESA-listed salmonid ESUs/DPSs evaluated and in all study years. Estimates of predation rates on smolts by Badger Island pelicans represent minimum estimates of smolt losses, however, due to the lack of data on smolt PIT tag deposition probabilities by breeding pelicans. In a study of predation on trout by American white pelicans breeding in Idaho, Teuscher et al. (2015) estimated that on-colony recovery probabilities (detection and deposition probabilities combined) of PIT-tagged trout consumed by pelicans was approximately 0.30 (range = 0.10–0.60). Applying this correction factor to the raw, unadjusted number of PIT-tagged smolts recovered on the Badger Island pelican colony would not, however, dramatically increase estimates of ESU/DPS-specific predation rates, as predation rates would still be < 2% for most ESA-listed ESUs/DPSs evaluated herein. There was some evidence, however, that predation rates by American white pelicans on smolts originating from the middle Columbia River, particularly subyearling Chinook salmon, may have been significantly higher than predation rates on smolts originating from the upper Columbia River and Snake River. For instance, Payton et al. (2020) detected predation rates as high as 10% on wild subyearling Chinook salmon that were PIT-tagged in the Hanford Reach of the middle Columbia River, just upstream of Badger Island, and hypothesized that the abundance of shallow water habitat, coupled with prolonged rearing and residence times of wild subyearling Chinook salmon, resulted in higher rates of predation by white pelicans in some years. Additional research is needed to quantify PIT tag deposition probabilities at American white pelican colonies in order to estimate predation rates more accurately. Without this information, estimates of pelican predation rates will continue to represent minimum estimates.

No standardized estimates of consumption rates based on smolt PIT tag recoveries have been generated for gulls nesting at colonies in the CRE. Although impacts on smolt survival from consumption by gulls nesting at colonies in the CRE are expected to be low based on limited data on diet composition, evaluating consumption rates using corrections for PIT tag deposition and detection probabilities seems warranted. Diet composition data for gulls nesting in the CRE were last collected during 1997–1998 by Collis et al. (2002). Relative to Caspian terns and double-crested cormorants nesting in the CRE, glaucous-winged/western gulls consumed fewer juvenile salmonids and were not thought to pose a significant risk to smolt survival. Given that glaucous-winged/western gulls are opportunistic foragers that exploit locally abundant food resources, however, it is possible that changes in their diet composition have occurred over time. Davis et al. (2015) found significant differences in the diet composition of glaucous-winged gulls nesting at colonies in the Salish Sea when comparing historical data from 1980 (Vermeer 1982) to data collected in 2010. For colonies in the Strait of Georgia the frequency of human refuse in adult gull pellets declined, while the proportion of fish in the diet increased over time. High consumption rates of out-migrating steelhead and coho salmon smolts have also been documented for western gulls nesting on the central California coast, with most consumption occurring at the mouths of coastal creeks close to the colony site (Osterback et al. 2013, Frechette et al. 2015).

While the size of the Brandt's cormorant colony on East Sand Island increased dramatically from 2006 to 2018, estimates of Brandt's cormorant predation rates on juvenile salmonids have remained low throughout. Evans et al. (2016b) hypothesized that several factors likely accounted for the large differences in predation rates on smolts between Brandt's cormorants, double-crested cormorants, and Caspian terns nesting on East Sand Island. First, Brandt's cormorants nesting on East Sand Island generally use foraging habitats closer to the colony, mainly foraging in marine waters close to the Columbia River mouth where non-salmonid prey types (e.g., anchovy, herring, smelt) are more common (Couch and Lance 2004, Peck-Richardson 2017). Consequently, salmonids comprise a smaller proportion of the diet of Brandt's cormorants compared to that of double-crested cormorants and Caspian terns, which forage in the freshwater portions of the estuary, in addition to marine waters (see also *Chapter 2* and *Chapter 3*). Second, the nesting chronology of Brandt's cormorants differs somewhat from that of double-crested cormorants and Caspian terns in the Columbia River estuary, with colony attendance peaking in late June, compared to late May for Caspian terns and early June for double-crested cormorants. This slight difference may be important because by June the peak of the salmonid run has usually passed, especially for the large groups of PIT-tagged steelhead and yearling Chinook salmon smolts. Finally, relative to double-crested cormorants, Brandt's cormorants are slightly smaller (by body mass) and thus presumably have lower daily food requirements (Couch and Lance 2004). Taken together, the available data suggest that Brandt's cormorants that formerly nested on East Sand Island posed little risk to survival of juvenile salmonids in the CRE. In 2019, however, very few Brandt's cormorants nested on East Sand Island, and by 2020 the Brandt's cormorant colony on East Sand Island had been abandoned. It seems unlikely that a Brandt's cormorant colony will become re-established on East Sand Island until double-crested cormorants again nest on the island.

No standardized estimates of predation rates on juvenile salmonids by American white pelicans nesting in the CRE are available because smolt PIT tags have not been recovered from the pelican colonies on Miller Sands Spit and Rice Island. Based on the relatively small size of these colonies to date (< 400 individuals counted on-colony annually), the small number of smolt PIT tags (n = 15) recovered from the Miller Sands Spit colony site in 2011, and the current understanding of predation rates on juvenile salmonids by American white pelicans nesting at the large Badger Island colony, white pelicans nesting in the CRE are unlikely to pose a substantial risk to survival of ESA-listed salmonids. As noted above for American white pelicans nesting on Badger Island, however, data on PIT tag deposition probabilities will be necessary to accurately estimate smolt losses to American white pelicans nesting in the CRE.

6.5.3 Management Considerations

There remain several uncertainties associated with predation/consumption rates on juvenile salmonids by piscivorous waterbirds from unmanaged colonies in the CRB that are relevant to management decisions. The following is a list of the management considerations and the information that is important for resolving these uncertainties and gaining a better understanding of the impacts on ESUs/DPSs of salmonids from avian predation in the CRB. By comparing the impacts on smolt survival from unmanaged species and colonies of piscivorous waterbirds in the CRB to those of species and colonies that have been the object of management to enhance smolt survival, resource managers will have the necessary information for developing a comprehensive, systemwide management plan for reducing avian predation to acceptable levels. In addition, managers would have the required information for identifying new and emerging issues of avian predation on juvenile salmonids in the CRB that warrant management consideration, both in the short-term and the long-term.

1. Predation on juvenile salmonids by piscivorous colonial waterbirds in the CRB is a dynamic and complex predator-prey system that involves multiple predator species and breeding colonies, as well as multiple populations of salmonids. Maximizing the benefits to ESA-listed juvenile salmonids from management of piscivorous colonial waterbirds depends on developing a holistic, system-wide approach to the problem. Research has demonstrated that the population dynamics, diet composition, and predation/consumption rates of juvenile salmonids by piscivorous colonial waterbirds can change rapidly, especially when breeding colonies first become established or are abandoned, or extant colonies rapidly increase decrease in size. Some level of baseline monitoring is needed to detect changes in the distribution and size of piscivorous waterbird colonies, as well as changes in predation/consumption rates, changes that could potentially dramatically alter current impacts to smolt survival from avian predation in the CRB.

2. The piscivorous colonial waterbirds included in this chapter are all native migratory species protected under the Migratory Bird Treaty Act (MBTA). Prospective management actions that affect the survival or reproduction success of these species

must be permitted by the U.S. Fish and Wildlife Service under the MBTA, and permitted management actions must not compromise the conservation status of managed species. Few data exist on the connectivity of breeding colonies for these species in the CRB; therefore, direct management of any of these species or breeding colonies, or incidental management of non-target species (e.g., displacement of Brandt's cormorants nesting on East Sand Island while managing double-crested cormorants nesting at the same site), may negatively affect regional populations of these species to an unknown degree.

3. Results indicate that California gulls consumed smolts that were tagged and released in apparently good condition and preferentially consumed large smolts, attributes that suggest that gull consumption can be a partially additive source of mortality for juvenile salmonids. Nevertheless, the proportion of smolts consumed by gulls that were dead or moribund when consumed, rather than depredated in an alive and healthy state, is currently unknown. For example, we do not understand what proportion of the salmonid smolts consumed by gulls nesting at colonies in the CPR are taken in the tailraces of hydroelectric dams, where smolts killed or injured during dam passage, or stunned and disoriented by turbulence and pressure changes, are highly susceptible to being consumed by foraging gulls. Addressing these questions is paramount to understanding what proportion of salmonid smolts consumed by gulls were depredated, and to what degree the mortality of smolts that were depredated by gulls was additive mortality, and therefore limiting to smolt survival in the CRB (see also *Chapter 7* and *Chapter 8).*

4. Few data are available to evaluate the potential impacts on smolt survival from consumption by glaucous-winged/western gulls nesting in the CRE, or by California gulls nesting at inland waterbodies in the CPR (i.e. Banks Lake, Sprague Lake, Lenore Lake, Potholes Reservoir). Some baseline data should be collected to validate current hypotheses that these gull colonies pose little risk to ESA-listed salmonid smolts in the CRB. Similarly, determining how consumption rates of salmonid smolts differ between California gulls and ring-billed gulls nesting at mixed-species colonies in the CPR is needed. Data presented in this chapter suggest that per capita (per bird) consumption of juvenile salmonids was greater for mixed-species gull colonies that were dominated by California gulls compared to those dominated by ring-billed gulls. If management of gull colonies to enhance salmonid smolt survival in the CPR is considered, additional research should be conducted to estimate discrete smolt consumption rates by each gull species.

5. The full impact on survival of ESA-listed salmonids from predation by American white pelicans nesting in the CRB remains unknown. Future research should quantify PIT tag deposition probabilities for tagged smolts consumed by American white pelicans and identify factors that influence the susceptibility of specific stocks of salmonids to pelican predation (e.g., flow rates, presence of diversion dams, tendency for fish to aggregate in shallow water habitats, fish-size, and run-timing; see also *Chapter 10*).

6. An understanding of the degree to which predation/consumption by piscivorous waterbirds from the colonies discussed in this chapter is an additive source of smolt mortality (i.e. predation adds to mortality) versus a compensatory source of smolt mortality (i.e. predation is compensated for by other sources of mortality) is crucial for evaluating the efficacy of prospective management actions aimed at increasing smolt survival by decreasing the size of unmanaged colonies. Recent advances in mark-recapture-recovery models make it possible to jointly estimate predation rates and survival rates of PIT-tagged smolts and thereby measure the strength and magnitude of the relationship between predation and survival rates (see also *Chapter 8*). Such studies are especially critical for addressing uncertainties over the degree to which gull consumption of juvenile salmonids is an additive source of smolt mortality and the degree to which prey-switching, predator swamping, and other density-dependent functional responses may be related to fish survival both within and across river reaches and amongst different predator species and colonies (see also *Chapter 9*).

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CHAPTER 7: Cumulative Effects of Avian Predation on Juvenile Salmonids in the Columbia River Basin

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7.1 Summary

We investigated the cumulative effects of piscivorous colonial waterbird (avian) predation on juvenile salmonids (smolts; *Oncorhynchus* spp.*)* in the Columbia River basin (CRB) and sought to determine what proportion of all sources of smolt mortality were due to avian predation. To achieve these objectives, we conducted a mark-recapture-recovery analysis on six different groups of passive integrated transponder (PIT)-tagged smolts that originated from the Upper Columbia River (UCR) or Snake River (SR) basins: (1) UCR steelhead trout (*O. mykiss*), (2) UCR yearling Chinook salmon (*O. tshawytscha),* (3) SR steelhead, (4) SR yearling Chinook, (5) SR sub-yearling Chinook, and (6) SR sockeye salmon (*O. nerka*). Predation probabilities (proportion of available fish consumed) were derived from the number of PIT-tagged smolts detected alive passing Rock Island Dam on the middle Columbia River or Lower Granite Dam on the lower Snake River that were subsequently consumed by Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), California gulls (*Larus californicus*)*,* and ring-billed gulls (*L. delawarensis*) during out-migration to the Pacific Ocean over an 11-year study period (2008–2018). Data were analyzed using a state-space Bayesian model that accounted for smolt survival to spatially explicit downstream river reaches and that partitioned predation by avian predator species and specific breeding colonies of avian predators within each river reach.

Results indicated that the effects of avian predation were highly variable depending on the salmonid species and population, the avian predator species and breeding colony, and the river reach and year. Of the salmonid species evaluated, cumulative predation probabilities (predation by birds from all avian colonies combined) were consistently the highest on steelhead smolts, with estimates ranging annually from 0.31 (95% credible interval = 0.26–0.37) to 0.53 (0.44–0.63) and from 0.18 (0.16–0.21) to 0.46 (0.39–0.55) of available UCR steelhead and SR steelhead, respectively. The cumulative effects of avian predation were significantly lower on Chinook smolts, with predation probabilities ranging annually from 0.04 (0.07–0.07) to 0.10 (0.7–0.15) of available SR sub-yearling Chinook, from 0.05 (0.04–0.06) to 0.16 (0.13–19) of available SR yearling Chinook, and from 0.06 (0.3–0.09) to 0.15 (0.10–0.23) of available UCR yearling Chinook. The cumulative effects of avian predation on SR sockeye smolts were generally higher than those on Chinook, but lower than those observed on steelhead, ranging annually from 0.08 (0.03–0.22) to 0.25 (0.14–0.44) of available sockeye.

Of the avian predator species and breeding colonies evaluated, predation by Caspian terns was significantly higher on steelhead smolts compared to Chinook and sockeye smolts. Of the Caspian tern breeding colonies evaluated, predation probabilities were often the highest for terns nesting on East Sand Island in the Columbia River estuary, on Goose Island in Potholes Reservoir, on the Blalock Islands in John Day Reservoir, and on Crescent Island in McNary Reservoir. Predation by California and ring-billed gulls was generally higher on steelhead and sockeye smolts compared to Chinook smolts. Of the mixed gull species breeding colonies evaluated, predation probabilities were consistently the highest for gulls nesting on Miller Rocks in The Dalles Reservoir and on Crescent and Badger islands in McNary Reservoir.

Predation probabilities by double-crested cormorants were similar amongst the three salmonid species (steelhead, Chinook, sockeye) and two age-classes (yearling, sub-yearling) of smolts evaluated. Of the double-crested cormorant breeding colonies evaluated, predation by cormorants nesting on East Sand Island in the Columbia River estuary were consistently higher than those of cormorants nesting on Foundation Island in McNary Reservoir. Although predation probabilities at other tern, cormorant, and gull colonies in CRB were substantial in some years, estimates were generally less than 0.02 of available smolts per year, per salmonid population, indicating that not all avian colonies posed a substantial risk to smolt mortality. Predation probabilities were often the lowest for smaller-sized breeding colonies, for Caspian tern colonies located at a considerable distance from the Columbia and Snake rivers (e.g., Lenore Lake and Banks Lake) or at certain mixed species gull colonies (e.g., Island 20 in McNary Reservoir and the Blalock Islands in John Day Reservoir), particularly impacts on Chinook smolts, which were often less than 0.01 per colony.

Comparisons of total smolt mortality (1 – survival) and mortality due to avian predation indicated that avian predation ranged from a relatively minor source of total smolt mortality (< 10% for Chinook) to the single greatest source of smolt mortality (often > 50% for steelhead) during out-migration to Bonneville Dam, the lowest hydroelectric dam on the Columbia River encountered by SR and UCR smolts. Avian predation on steelhead smolts was one of the greatest sources of mortality, and often the single greatest source of mortality during out-migration, with mortality due to avian predation accounting for from 42% (30–56%) to 70% (53–87%) and from 37% (33–46%) to 89% (80–98%) of all mortality in UCR steelhead and SR steelhead, respectively. Results suggested that more UCR and SR steelhead were consumed by avian predators than died from all other mortality sources combined during smolt passage from Rock Island Dam to Bonneville Dam (for UCR steelhead) or from Lower Monumental Dam to Bonneville Dam (for SR steelhead) in 9 of the 11 years evaluated. In contrast, comparisons of total smolt mortality and mortality due to avian predation in UCR and SR Chinook smolts indicated that avian predation was often a small component of all mortality during out-migration. For example, avian predation accounted for less than 20% of all mortality in UCR yearling Chinook, SR yearling Chinook, and SR sub-yearling Chinook upstream of Bonneville Dam in most, but not all, years. For SR sockeye, the proportion of total smolt mortality upstream of Bonneville Dam that was due to avian predation was highly variable, ranging annually from less than 10% to over 70% of all mortality. Small sample sizes of PIT-tagged SR sockeye, UCR yearling Chinook, and SR sub-yearling Chinook smolts, however, resulted in imprecise estimates of predation and survival probabilities, making it difficult to determine to what degree avian predation might have limited smolt survival in these species and populations relative to other sources of mortality.

Results of this study suggest that avian predation, although not the original cause of salmonid declines in the Columbia River basin, in now a factor limiting the survival of some salmonid populations that are listed under the U.S. Endangered Species Act, in particular UCR and SR steelhead. For UCR and SR Chinook, however, avian predation represented a relatively small component of total smolt mortality during out-migration. These results suggest that the potential benefits of managing avian predators to reduce mortality of smolts during out-migration will vary widely depending on the species of avian predator, the location of the breeding colony, and the salmonid species and population that is being depredated.

7.2 Introduction

Identifying factors that affect the survival of juvenile salmonids (smolts; *Oncorhynchus* spp.), particularly populations listed under the U.S. Endangered Species Act (ESA), is necessary to develop effective recovery plans. Previous research has identified predation by Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), California gulls (*Larus californicus*) and ring-billed gulls (*L. delawarensis*) as a significant mortality factor for some salmonid populations in the Columbia River basin (CRB; Collis et al. 2002, Evans et al. 2012, Evans et al. 2016). Previous research has largely focused on quantifying the effects of specific breeding colonies but some salmonid populations, like those originating from the Upper Columbia River (UCR) and Snake River (SR), must migrate through the foraging ranges of breeding birds from multiple colonies during out-migration to the Pacific Ocean (Evans et al. 2019, Payton et al. 2019). In addition to predation from piscivorous colonial waterbirds, salmonids are subject to numerous other non-avian sources of mortality during out-migration. For example, mortality associated with hydroelectric dam passage, predation by piscivorous fish, disease, and other factors that are well documented (Ward et al. 1995, Muir et al. 2001, Dietrich et al. 2011). Determining to what extent avian predation limits smolt survival relative to other sources of mortality may be critical for prioritizing recovery actions for ESA-listed salmonid populations in the CRB (Evans et al. 2016).

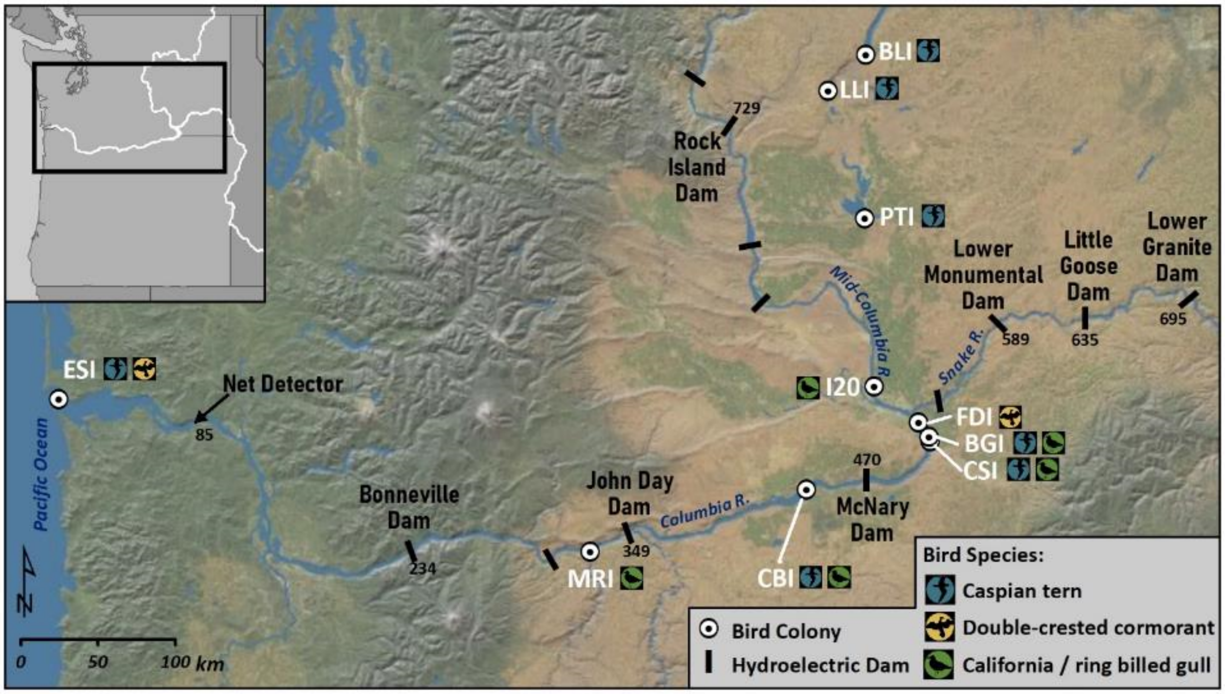
Mark-recapture-recovery studies have been used to identify and quantify specific sources of mortality in anadromous juvenile salmonids (Mathur et al. 1996, Muir et al. 2001, Hostetter et al. 2012). These studies rely on marking (tagging) fish and then using subsequent recapture and recovery events to estimate survival and cause-specific mortality (e.g., harvest, dam passage, predation). Using this analytical framework, Evans et al. (2019) provided evidence that the cumulative effects of predation by multiple bird species and colonies on ESA-listed Upper Columbia River (UCR) steelhead trout *(O. mykiss)* were substantial, with predation probabilities (proportion of available fish) in excess of 0.25 (or 25%) of available smolts observed in most years during 2008–2018. Comparisons of total steelhead mortality (1 – survival) to mortality due to colonial waterbird predation indicated that predation accounted for 42–70% of all mortality sources during smolt passage from Rock Island Dam on the middle Columbia River to Bonneville Dam on the lower Columbia River, a distance of 495 river km (Rkm) where up to 12 different avian colonies foraged on steelhead smolts (Evans et al. 2019). Several previously published studies indicate that steelhead smolts are particularly susceptible to colonial waterbird predation, with predation impacts on steelhead significantly higher than those observed on salmon smolts at the same colony and year (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Evans et al. 2016). Given the greater susceptibility of steelhead smolts to colonial waterbird predation observed in these studies, it is likely that the cumulative effects of avian predation on salmon smolts (e.g., Chinook [*O. tshawytscha*] and sockeye [*O. nerka*]) are lower than those on steelhead smolts. Research to quantify cumulative predation probabilities in salmon species, however, are generally lacking in the published literature (Evans et al. 2016, Evans et al. 2019).

To investigate the cumulative effects of avian predation and to determine what proportion of all sources of smolt morality were due to avian predation, we conducted mark-recapture-recovery analyses on six different groups of PIT-tagged juvenile salmonids originating from the UCR and Snake River (SR) basins: (1) UCR steelhead, (2) UCR yearling Chinook, (3) SR steelhead, (4) SR yearling Chinook, (5) SR subyearling Chinook, and (6) SR sockeye. Survival and predation rates were evaluated during an 11-year study period (2008–2018) across multiple river reaches (spatial scales) where piscivorous colonial waterbirds (Caspian terns, double-crested cormorants, California gulls, and ring-billed gulls) that nested at 14 different breeding colonies foraged on smolts during out-migration to the Pacific Ocean. Results provide a unique, comprehensive, and system-wide evaluation of avian predation and the cumulative effects of avian predation on smolt survival across multiple ESA-listed salmonid species and populations in the CRB.

7.3 Methods

7.3.1 Study Area

Survival and predation probabilities were estimated during smolt out-migration through multiple river reaches or segments of the Snake and Columbia rivers. For UCR steelhead and yearling Chinook, survival and predation were estimated during smolt out-migration from (1) Rock Island Dam to McNary Dam, a 259 Rkm section of the middle and lower Columbia rivers, (2) McNary Dam to John Day Dam, a 121 Rkm section of the lower Columbia River (3) John Day Dam to Bonneville Dam, a 115 Rkm section of the lower Columbia River. Predation was also estimated from (4) Bonneville Dam to the Pacific Ocean, a 234 Rkm section of the lower Columbia River (*Map 7.1*) but estimates of smolts survival in this river reach were available due to a lack of PIT tag recapture sites at the mouth of the Columbia River (see below). For SR steelhead, yearling Chinook, subyearling Chinook, and sockeye, survival and predation probabilities were estimated during out-migration from (1) Lower Granite Dam to Little Goose Dam, 60 Rkm section of lower Snake River, (2) Little Goose Dam to Lower Monumental Dam, a 46 Rkm section of lower Snake River, (3) Lower Monumental Dam to McNary Dam, a 119 Rkm section of the lower Snake River and lower Columbia River, (4) McNary Dam to John Day Dam, (5) John Day Dam to Bonneville Dam, and predation was estimated from (6) Bonneville Dam to the Pacific Ocean (*Map 7.1*). The number of UCR and SR smolts surviving to adulthood (smolt-to-adult returns or SARs) were estimated based on detections of returning adults to Bonneville Dam, the first hydroelectric dam UCR and SR salmonids must pass following ocean residency (*Map 7.1*).



Map 7.1. Mark-recapture-recovery locations of PIT-tagged smolts interrogated/released at Rock Island Dam and Lower Granite Dam. Downstream recapture locations include Little Goose Dam, Lower Monumental Dam, McNary Dam, John Day Dam, Bonneville Dam, and a towed net detection system in the Columbia River estuary (Net Detector). Recovery locations include bird colonies on Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns, double-crested cormorants, and California and ring-billed gulls. Numbers represent the distance in river km (Rkm) from the Pacific Ocean.

7.3.2 Mark-Recapture-Recovery

Using the previously published methods of Payton et al. (2019), we integrated multiple sources of data to estimate avian predation and survival probabilities on tagged UCR and SR smolts. Data sources included fish marked at or interrogated at Lower Granite and Rock Island dams, subsequent detections of fish passing downstream recapture sites, and recoveries of tags from depredated fish on bird colonies. Smolts were marked (tagged) with Passive Integrated Transponder (PIT) tags as part of independent and on-going juvenile salmonid behavior and survival studies in the CRB (e.g., Evans et al. 2014, Marsh 2017, McCann et al. 2019). At Lower Granite and Rock Island dams smolts were tagged or interrogated (if previously tagged) from early-April to late-July each year, with the duration of tagging dependent on the run-timing of fish at each location, each year. Fish were grouped based on their river-of-origin (UCR, SR), species (steelhead, Chinook, sockeye), and age-class (yearling, subyearling for Chinook only) to represent a unique group or population of PIT-tagged fish. Both hatchery and wild (natural-origin) fish were included but not all tagged fish from a given age-class and rear-type were from a known ESA-listed salmonid population because an unknown number of unlisted hatchery fish were tagged and because ESA-listed fall Chinook salmon exhibit both a yearling and subyearling life history (NOAA 2014).

Following interrogation/release (hereafter "release") at Rock Island Dam or Lower Granite Dam, tagged fish could be volitionally detected passing downstream recapture sites outfitted with in-stream PIT tag arrays (see *Map 7.1* above). For fish released at Lower Granite Dam, only live fish that remained in-river when passing downstream dams were retained (i.e. known mortalities or fish that were collected and subsequently transported on barges were excluded). Adults returning to the Columbia River following ocean residency were also detected at PIT tag arrays located in fishways at Bonneville Dam. Release and recapture records were retrieved from the PIT Tag Information System, a regional mark, recapture, recovery database maintained by Pacific States Marine Fisheries Commission (PSMFC 2018).

Smolt PIT tags were also recovered (detections of dead fish) on piscivorous waterbird colonies located throughout the CRB. Recoveries from a total of 14 different bird colonies were included in the study, colonies that were previously identified as potential sources of smolt mortality (see *Map 7.1* above; see also Evans et al. 2019). The methods of Evans et al. (2012) were used to recover PIT tags from each bird colony. In brief, portable PIT tag antennas were used to detect tags after birds dispersed from their breeding colonies at the end of the nesting season (August–October). The entire land area occupied by nesting birds was scanned for tags following each nesting season, with a minimum of two complete sweeps or passes of each colony site conducted each year. The land area occupied by birds during each nesting season was determined based on aerial photography surveys and/or ground surveys of the colony carried out during the peak of the nesting season in late-May to early-June.

Not all fish PIT tags ingested by birds are deposited on the bird's nesting colony (i.e. deposition probabilities for consumed fish tags were less than 1.0) and not all tags deposited on the colony are detected by researchers after the nesting season (i.e. detection probabilities for deposited fish tags were less than 1.0; Hostetter et al. 2015). We followed previously published methods for estimating colony-specific PIT tag deposition and detection probabilities, those of Hostetter et al. (2015) and Payton et al. (2019). In brief, recoveries of PIT-tagged smolts that were intentionally fed to nesting Caspian terns, double-crested cormorants, and California and ring-billed gulls at multiple colonies and across multiple years were used to estimate PIT tag deposition probabilities (Hostetter et al. 2015). To estimate detection probabilities, PIT tags were sown on each bird colony by researchers prior to, during (when possible), and following the nesting season. Recoveries of these tags during scanning efforts after the nesting season were then used to model the probability of detecting a tag that was deposited on the colony during the nesting season. Colony-specific PIT tag recovery probabilities were those previously reported in peer-reviewed literature (see also *Appendix B, Table B.17*).

Finally, birds nesting at some of the colonies included in the study were capable of foraging in multiple river reaches (i.e. upstream and downstream of a given PIT-tag array), which required predation rates be partitioned by colony and river reach. The methods of Payton et al. (2019) were used to achieve this partitioning (see *below* for additional details).

7.3.3 Predation & Survival Estimation

The joint mortality and survival (JMS) estimation methods of Payton et al. (2019) were used to independently estimate reach-specific and cumulative predation and survival probabilities for each salmonid species evaluated. This hierarchal state-space Bayesian model incorporated both live and dead detections of PIT-tagged fish in space and time to simultaneously estimate predation and survival through *J* total river segments (where *J*=5 for UCR smolts and *J*=7 for SR smolts). In brief, the model used two vectors, $\boldsymbol{y}$ and $\mathbf{r}$, to describe each fish's recapture history following tagging and release at each of the five (5) downstream recapture sites and each of the 14 avian recovery sites under consideration. Each vector $\mathbf{y}$ was a *J*-length vector, where y_j was an indicator variable of a fish's recapture at recapture opportunity j for $j \in \{1,2, \dots ,J\}$, and $\mathbf{r}$ was a 15-length vector where r_d was an indicator variable of recovery on colony *d* for $d \in \{1,2, \dots ,14\}$, and r_{15} indicated a fish was unrecovered. Implicitly, the model provided inference about each fish's state, represented by an unobserved 5-length vector $\mathbf{z}$, were z_j was an indicator variable of whether the fish was alive at recapture opportunity j.

Parameters used in the model were:

$\boldsymbol{\Theta}$**,** a 15x5 matrix where $\Theta_{k,d}$ represented the probability a released fish survived to recapture opportunity k and then subsequently succumbed to mortality cause *d* prior to arrival at recapture opportunity $k + 1$

p, a *J*-length vector where p_k represented the probability that a fish alive at recapture opportunity *k* was successfully recaptured

$\boldsymbol{\gamma}$, a 15-length vector whereγ_d represented the probability of recovering a fish which died due to mortality cause *d* for $d \in \{1,2, \dots ,14\}$, and γ_{15} = 0 represented the lack of recoveries of fish which died from all other unspecified causes.

The recovery parameters, γ_d, represent the combined probability that a consumed tag was deposited on colony and the probability that the tag is subsequently recovered by researchers following the breeding season given deposition on a colony. Deposition probabilities were identified through informative priors (see Hostetter et al. 2015). Recovery probabilities were colony and year specific and informed through data collected in each year.

Low recapture rates inhibit precision in partitioning the morality impacts of colony d among the river reaches where that colony was assumed to forage. Previous research indicates that predation impacts by individual colonies were spatially proportionate amongst river reaches and years (Evans et al. 2016, Hostetter et al. 2018). Therefore, we implemented an "informed partitioning" method to share information among years based on the assumption that the odds of being consumed by a colony foraging among multiple river segments were similar among years. Specifically, informed partitioning involved first defining $\boldsymbol{\theta}_{\mathbf{y}}^{cumulative}$ to be a 15-sized vector where ${\theta_y}_d^{cumulative}$ represented the probability a released fish succumbed to mortality cause *d,* in year y. Then, for each colony d in year y, J -length vector $\boldsymbol{\rho}_{y_d}$ defined the partitioning of $\theta_{y_d}^{cumulative}$ into $[\Theta_{y_{\mathbf{1},d}}, \Theta_{y_{\mathbf{2},d}}, \dots, \Theta_{y_{\boldsymbol{J},d}}]$. That is,

$$[\Theta_{y_{\mathbf{1},d}}, \Theta_{y_{\mathbf{2},d}}, \dots, \Theta_{y_{\boldsymbol{J},d}}] = \theta_{y_d}^{cumulative} \boldsymbol{\rho}_{\mathbf{y}_d}$$

where

$$\boldsymbol{\rho}_{\mathbf{y}_d} \sim dirichlet(\boldsymbol{\alpha}^d).$$

The spatially explicit nature of the $\boldsymbol{\Theta}$ probabilities enabled the construction of survival probabilities across river reaches could be derived. $\boldsymbol{\Phi}$ was defined to be a J -length vector where Φ_k represented the probability a fish released at Rock Island dam survives through river reach *k* (i.e., $\Phi_\mathrm{k} = 1 - \sum_{i \le k} \sum_d \Theta_{k,d}$).

It follows that an individual fish's life can be expressed with the following state-space interpretation:

$$z_j \sim bernoulli\big(z_{(j-1)} * \Phi_j\big),$$
$$y_j \sim bernoulli\big(z_j * p_j\big),$$

and

$$r_d \sim bernoulli(\sum_{j=1}^{M-1} \big(z_{j+1} - z_j\big) * \Theta_{j,d} * \gamma_d).$$

Temporal variation was assumed to be inherent to all rates of mortality/survival (Evans et al. 2014, Hostetter et al. 2015), recapture (Sandford and Smith 2002), and recovery (Ryan et al. 2003, Evans et al. 2012). Under the assumption that fish released within the same week experienced similar rates of mortality/survival, recapture, and recovery, smolts from each salmonid population were grouped into weekly release cohorts. The week specific rates were accordingly denoted $\boldsymbol{\Theta}_{\mathrm{y,w}}$, $\mathbf{p}_{y,w}$, and $\boldsymbol{\gamma}_{y,w}$. Probabilities of mortality, recapture, and recovery from weeks closer in time were assumed to be more alike than those temporally further apart. Serial correlation in survival/mortality and recapture rates were assumed and accounted for as described by Payton et al. (2019). Temporal variation in detection rates were estimated more directly by intentionally sown PIT tags with known tag codes on each colony before, after, and, in some instances, within each breeding season (see Hostetter et al. 2015). Estimated detection probabilities at each colony were then interpolated from the logistic curve estimated from the intentionally sown tags. In some uncommon instances, researchers were unable to sow PIT tags prior to the nesting season. In these cases, the methods of Payton et al. (2019) were used to infer an estimation of inner-seasonal variation in recovery rates using information from similar colonies in the same year or using information from the same colony in different years.

Weakly informative priors (as suggested by Vehtari et al. [2017]) were assigned to most of the parameters of the model. The prior for the initial week's detection probability in each year was defined to be $uniform(0,1)$. Analogously, the prior distribution assigned for the life paths simplexes in the initial week of each year was assumed to be $Dirichlet(\mathbf{1})$, where $\mathbf{1}$ was an appropriately sized vector of ones. Weakly-informative priors of $half\ normal(0, 5)$ were also implemented for all variance parameters.

Models were analyzed using the software STAN (Stan Development Team 2020), accessed through R version 3.6.2 (R Development Core Team 2014), and using the rstan package (version 2.19.3; Stan Development Team 2020). To simulate random draws from the joint posterior distribution, we ran four Hamiltonian Monte Carlo (HMC) Markov Chain processes. Each chain contained 4,000 warm-up iterations followed by 4,000 posterior iterations thinned by a factor of 4. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013); only chains with zero reported divergent transitions were accepted. Posterior predictive checks compared simulated and observed annual aggregate raw recapture and recovery numbers to ensure model estimates reflected the observed data. Reported estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% Credible Interval [CRI]) calculated using the HDInterval package (version 0.2.0; Meredith and Kruschke 2016).

7.3.3.1 Model assumptions – The accuracy and precision of survival and predation estimates depend, in part, on the validity of the following assumptions (see also Payton et al. 2019):

A1. Smolt survival, predation, and recapture/recovery probabilities were independent.

A2. Fish released within the same week had identical recapture/recovery probabilities.

A3. Intra-annual variation in survival, predation, and recapture/recovery probabilities can be described as a 'random walk' process.

A4. Sampled fish were representative of all fish (tagged and un-tagged) in the population at-large.

The fate of each tagged fish was assumed to be independent of the fate of other tagged fish in the sample (A1). This assumption is ubiquitous amongst mark-recapture studies. Lack of independence would likely overstate precision and bias predation and survival estimates to an unknown degree. Detection probabilities did not change dramatically on a weekly basis and there was no evidence of inter- or intra-annual changes in deposition probabilities across colonies of the same species of avian predator (A2; Hostetter et al. 2015). The random walk framework allowed for inter-weekly fluctuations in survival, predation, and recapture/recovery probabilities, so assumption A3 only needs to be approximately true for rates to be unbiased. Assumption A4 rests on the representative selection of smolts for PIT-tagging. In case of UCR steelhead and spring Chinook smolts in some years (2013–2016), fish were selected for tagging at random and were tagged in proportion to the number available each week, each year (see also Evans et al. 2014). In the case of SR smolts and UCR spring Chinook smolts in select years (2008–2012, 2017–2018), tagging efforts were part of regional studies that including both hatchery and wild fish and fish originating from different sub-basins (e.g., Salmon River, Clearwater River, etc.). Run-of-river fish were also captured and tagged at Lower Granite Dam throughout the peak smolt outmigration period, with tagging efforts commensurate with the relative abundance and run-timing of smolts passing Lower Granite Dam each year. Smolts, however, were generally culled for PIT-tagging on the SR, with fish greater than 80 mm and in good condition tagged (i.e., fish without severe descaling; body injuries; or signs of disease). For all releases (UCR and SR) the effects of handling and PIT-tagging each fish were inestimable. A significant number of losses due to handling/tagging would result in an overstatement of fish availability and consequently underestimate predation and survival to an unknown degree.

7.4 Results

7.4.1 Mark-Recapture-Recovery

Sample sizes of PIT-tagged smolts used in predation and survival analyses varied considerably by release location, salmonid species and population, and year (*Table 7.1*). In total, 78,409 UCR steelhead, 34,725 UCR yearling Chinook, 713,877 SR steelhead, 1,044,755 SR yearling Chinook, 150,351 SR subyearling Chinook, and 31,220 SR sockeye were released at either Rock Island Dam (for UCR fishes) or Lower Granite Dam (for SR fishes) and included in analyses during 2008–2018. Annual sample sizes ranged from 298 SR sockeye in 2008 to 167,925 SR yearling Chinook in 2009 (*Table 7.1*). In general, sample sizes of SR smolts were greater than those of UCR smolts, except for Snake River sockeye, where sample sizes were the smallest of the six populations evaluated. Sample sizes of UCR yearling Chinook were also amongst the lowest, ranging annually from 756 in 2009 to 5,908 in 2015 (*Table 7.1*).

The number of smolts released at Rock Island Dam or Lower Granite Dam that were subsequently recapture alive at downstream PIT tag arrays or recovered dead on bird colonies also varied considerably by location, salmonid population, and year (*Table 7.1*). In total, 8,097 UCR steelhead, 812 UCR yearling Chinook, 57,433 SR steelhead, 27,444 SR yearling Chinook, 2,525 SR subyearling Chinook, and 723 SR sockeye tags were recovered on bird colonies during 2008–2018. Only a small number and proportion of PIT-tagged smolts from Rock Island and Lower Granite dams returned to Bonneville Dam as adults, with smolt-to-adult (SAR) return rates ranging from 0.2–3.0%, depending on the salmonid species and year (*Table 7.1*).

Not all colony sites had nesting birds in all study years, nor were all sites scanned for smolt PIT tags in all years. Specifically, California and ring-billed gull colonies on Island 20 and the Blalock Islands were not scanned for PIT tags during 2008–2012, preventing estimation of predation by birds from those colonies in those years. The Foundation Island double-crested cormorant colony was not scanned for smolt PIT tags during 2013 or 2015–2018, preventing estimation of predation rates by cormorants in those years. Double-crested cormorants temporarily dispersed (abandoned) their breeding site on East Sand Island during the peak of the smolt out-migration period annually during 2016–2018 (Turecek et al. 2018, 2019). As such, although the cormorant colony was scanned for PIT tags in all years following each nesting season, PIT-tagged smolts that were consumed by cormorants during dispersed events were presumably deposited off-colony, resulting in minimum annual estimates of predation in those years. Unlike gull and cormorant colonies, all large Caspian tern colonies (those greater than 20 breeding pairs) were scanned for smolt PIT tags in all study years (see also Evans et al. 2019).

Table 7.1. Numbers of tagged juvenile salmonids interrogated/released (Rel) at Lower Granite Dam (for Snake River [SR] smolts; top) or Rock Island Dam (for Upper Columbia River [UCR] smolts; bottom) subsequently recaptured downstream alive (live), recovered on a bird colony (dead), or detected returning as an adult to Bonneville Dam (SAR). Species include steelhead, Chinook, and sockeye. Chinook age-classes include yearling (1) and subyearling (0). NA denotes that complete adult returns were not available. The same fish could be recaptured multiple times at downstream recapture sites, but only one detection was possible at a recovery site.

Year	SR Steelhead (Rel–Live–Dead–SAR)	SR Chinook 1 (Rel–Live–Dead–SAR)	SR Chinook 0 (Rel–Live–Dead–SAR)	SR Sockeye (Rel–Live–Dead–SAR)
2008	53155–50934–6822–1481	156901–131520–4336–2471	22032–12284–469–342	298–265–17–2
2009	89445–100983–12467–1404	167925–160002–8246–1706	21347–12705–444–76	3009–2407–111–45
2010	45348–29304–5318–615	136085–79256–4501–803	21075–14605–314–309	1656–806–58–9
2011	80903–86392–6427–606	134728–136283–3208–714	36537–19296–596–604	8250–4543–169–26
2012	78040–74315–5085–1336	96343–93182–1439–1831	25286–12486–429–318	4322–3326–113–16
2013	42953–25868–2859–514	42410–30711–649–543	2529–1132–47–30	5521–2336–92–120
2014	64725–47813–5285–799	75662–69988–1896–483	3009–1859–68–15	1656–837–69–14
2015	40715–16950–3198–98	27792–12472–1046–118	3718–980–39–NA	1019–331–16–2
2016	79262–66351–3783–NA	102858–97031–1348–NA	5203–1694–32–NA	1721–823–14–NA
2017	75019–57734–3744–NA	56829–42363–445–NA	5251–2386–26–NA	451–170–5–NA
2018	64312–52118–2445–NA	47222–34233–330–NA	4364–2544–61–NA	3317–1440–59–NA

Year	UCR Steelhead (Rel–Live–Dead–SAR)	UCR Chinook 1 (Rel–Live–Dead–SAR)
2008	7271–1107–1034–220	907–246–27–8
2009	7114–1205–1098–77	756–224–29–6
2010	7365–1447–969–88	1076–269–30–13
2011	7757–583–794–46	1705–364–30–10
2012	6711–845–570–67	1885–537–38–13
2013	5893–846–680– 61	5826–1532–118–111
2014	7663–1017–775–65	5551–1375–159–54
2015	7069–1189–792–5	5908–1050–197–NA
2016	6764–1577–533–NA	5781–1742–130–NA
2017	7436–797–514–NA	2962–693–35–NA
2018	7366–912–338–NA	2368–385–19–NA

7.4.2 Avian Predation

Predation probabilities based on smolt PIT tag recoveries from the 14 bird colonies included in the study varied considerably by avian species and colony and by salmonid species (steelhead, Chinook, sockeye), population (UCR, SR), river reach, and year. Results are summarized below by salmonid species and population.

7.4.2.1 Upper Columbia River steelhead trout – Of the individual colonies evaluated, predation probabilities on UCR steelhead smolts during passage from Rock Island Dam to McNary Dam were the greatest by Caspian terns nesting on Goose Island in Potholes Reservoir during 2008–2013, with estimates as high 0.26 (95% credible interval = 0.18–0.34) observed in some years (see *7.8 Supplements*, *Figure 7.S1*). These impacts occurred prior to implementation of management actions to reduce the size of the Goose Island tern colony as part of the *Inland Avian Predation Management Plan* (*IAPMP*; see *Chapter 2*). During passage from McNary Dam to John Day Dam, UCR steelhead predation probabilities were the greatest by the Blalock Island tern colony during 2015–2018, with probabilities as high as 0.11 (0.07–0.19) observed in some years (see *7.8 Supplements*, *Figure 7.S1*). During passage from John Day Dam to Bonneville Dam nearly all predation was due to the gull colony on Miller Rocks Island, with estimates in excess of 0.10 observed in multiple years (see *7.8 Supplements*, *Figure 7.S1*). Predation by colonies in the Columbia River estuary on UCR steelhead were consistently the greatest by Caspian terns breeding on East Sand Island, with probabilities often in excess of 0.10, and as high as 0.21 (0.16–0.31) in some years (see *7.8 Supplements*, *Figure 7.S1)*.

Cumulative predation probabilities (predation by all 14 colonies combined) on UCR steelhead during outmigration from Rock Island Dam to the Pacific Ocean were substantial and the highest of the salmonid populations evaluated, ranging annually from 0.31 (0.26–0.37) to 0.53 (0.44–0.63; *Figure 7.1*). Of the avian species evaluated, aggregate predation probabilities (predation by all predators of the same species) on UCR steelhead were often the highest by Caspian tern colonies, with terns consuming 0.11 (0.09–0.14) to 0.38 (0.29–0.47) of all steelhead smolts, per year (*Figure 7.1*). Predation of UCR steelhead by gull colonies that foraged upstream of Bonneville Dam were also substantial, ranging from 0.07 (0.05–0.10) to 0.31 (0.25–0.39). There was some evidence that gull consumption rates increased during 2015–2018 (*Figure 7.1)* but gull consumption could not be fully evaluated across all study years due to a lack of tag recoveries from the Island 20 and Central Blalock Islands gull colonies during 2008–2012. Predation by double-crested cormorant colonies were generally less than that of tern and gull colonies, ranging from 0.01 (0.01–0.02) to 0.04 (0.03–0.07). Analogous to consumption by gull colonies, however, estimates of predation by cormorants nesting on Foundation Island were not available in all study years and, in the case of the large cormorant colony on East Sand Island in the estuary, predation probabilities from PIT tag recoveries during 2016–2018 likely represent minimum losses due to colony dispersal events during the peak smolt out-migration period in those years. Caveats regarding gull and cormorant predation impacts being biased low in some years apply to all salmonid species (steelhead, Chinook, sockeye) and populations (UCR, SR) evaluated herein, to varying degrees.

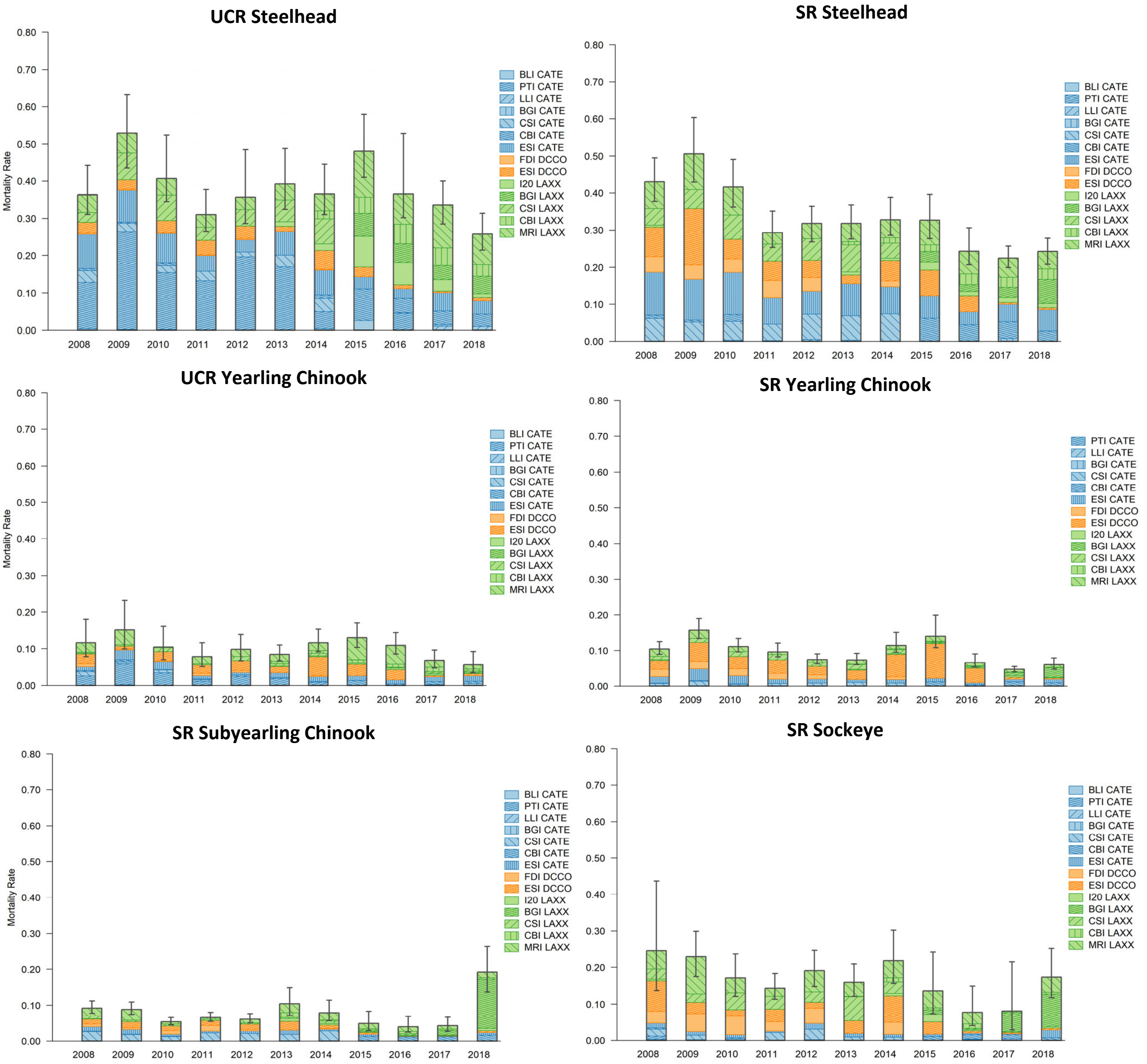


Figure 7.1. Cumulative avian predation probabilities on Upper Colombia River (UCR) steelhead and yearling Chinook smolts during passage from Rock Island Dam to the Pacific Ocean and Snake River (SR) steelhead, yearling Chinook, subyearling Chinook, and sockeye during passing from Lower Monumental Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CBI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals of cumulative avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, the FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods).

7.4.2.2 Snake River steelhead trout – Predation effects for SR steelhead (and the other smolts originating from the SR; see *below*) were measured starting at Lower Monumental Dam as opposed to Lower Granite Dam because this is where avian predation by the colonies included in this study first occurred. Estimates account for smolt survival from Lower Granite Dam to Lower Monumental Dam (i.e. based on the number of fish released at Lower Granite that subsequently survived out-migration to Lower Monumental Dam). Of the individual colonies evaluated, predation probabilities on SR steelhead during passage from Lower Monumental Dam to McNary Dam were the greatest by Caspian terns breeding on Crescent Island during 2008–2014 (prior to management actions associated with *IAPMP*; see *Chapter 2*) and by double-crested cormorants breeding on Foundation Island, with impacts often in excess 0.04 (see *7.8 Supplements*, *Figure 7.S2*). Analogous to UCR steelhead, predation probabilities on SR steelhead during passage from McNary Dam to John Day Dam were the greatest by the Central Blalock Island tern colony during 2015–2018, with probabilities as high as 0.07 (0.05–0.10; see *7.8 Supplements*, *Figure 7.S2*). Also analogous to UCR steelhead, nearly all predation on SR steelhead during passage from John Day Dam to Bonneville Dam was due to the gulls breeding on Miller Rocks Island. Predation by terns breeding on East Sand Island in the Columbia River estuary was the highest of the 14 colonies evaluated, with probabilities consistently greater than 0.10, and as high as 0.19 (0.15–0.27) observed in some years (see *7.8 Supplements*, *Figure 7.S2*).

Similar to predation on UCR steelhead, the cumulative effects of avian predation on SR steelhead were substantial and were the second highest of the salmonid species and populations evaluated, ranging annually from 0.18 (0.16–0.21) to 0.46 (0.39–0.55; *Figure 7.1* above). Predation probabilities on SR steelhead were often the highest by Caspian tern colonies, with terns consuming 0.07 (0.05–0.09) to 0.17 (0.13–0.23) of all SR steelhead smolts, per year. The effects of predation from all double-crested cormorant colonies on SR steelhead were similar to that of tern colonies, ranging from 0.06 (0.04–0.07) to 0.17 (0.11–0.26). The cumulative effects of predation on SR steelhead by gull colonies foraging upstream of Bonneville Dam were also substantial, but generally less than that of tern and cormorant colonies, ranging from 0.07 (0.05–0.10) to 0.13 (0.10–0.17).

7.4.2.3 Upper Columbia River yearling Chinook salmon – Of the individual colonies evaluated, predation probabilities on UCR yearling Chinook during passage from Rock Island Dam to McNary Dam were the highest by Caspian terns breeding on Goose Island in Potholes Reservoir during 2008–2013, with estimates peaking at 0.05 (0.02–0.11) in 2009 (see *7.8 Supplements*, *Figure 7.S3*). Analogous to UCR steelhead, impacts on UCR yearling Chinook were highest prior to implementation of management actions to reduce the size of the Goose Island tern colony as part of the *IAPMP* (see *Chapter 2*). Predation by all other individual colonies in this reach, however, were low, with most colonies consuming less than 0.02 of available smolts, per year (see *7.8 Supplements*, *Figure 7.S3*). During passage from McNary Dam to John Day Dam, UCR yearling Chinook predation probabilities were again generally less than 0.02 per colony, per year (see *7.8 Supplements*, *Figure 7.S3*). During passage from John Day Dam to Bonneville Dam nearly all predation was due to the gull colony on Miller Rocks Island, with estimates often greater than 0.02, and as high as 0.08 (0.03–0.15; see *7.8 Supplements*, *Figure 7.S3*). Predation probabilities were consistently the highest by double-crested cormorants breeding on East Sand Island, with estimates during 2008–2015 (years with typical breeding behavior; see *above*) being consistently greater than 0.05, and as high as 0.09 (0.07–0.14; see *7.8 Supplements*, *Figure 7.S3*). Small sample sizes of UCR yearling Chinook, however, resulted in imprecise estimate of predation probabilities for most colonies and years included in the study, as evident by the large credible intervals.

The cumulative effects of avian predation on UCR yearling Chinook by all 14 colonies were significantly lower than those observed on UCR and SR steelhead, ranging annually from 0.06 (0.03–0.09) to 0.15 (0.10–0.23; *Figure 7.1* above). Of the avian species evaluated, predation probabilities on UCR yearling Chinook were generally the highest by Caspian tern colonies prior to implementation of management actions at Goose Island, Crescent Island, and East Sand Island (see also *Chapters 1-2*), but predation by tern colonies was similar to those of cormorant and gulls colonies, predation probabilities that were amongst the lowest of the six salmonid populations evaluated herein (*Figure 7.1*).

7.4.2.4 Snake River yearling Chinook salmon – Of the individual colonies evaluated, predation probabilities on SR yearling Chinook during passage from Lower Monumental Dam to McNary Dam were generally the greatest by double-crested cormorants breeding on Foundation Island. Impacts by colonies in this reach, however, were low, with most colonies consuming less than 0.02 of available smolts, per year (see *7.8 Supplements*, *Figure 7.S4*). Predation probabilities on SR yearling Chinook during passage from McNary Dam to John Day Dam were also low, with less than 0.02 of available fish consumed per colony, per year (see *7.8 Supplements*, *Figure 7.S4*), Analogous to the other salmonid populations evaluated, nearly all predation on SR yearling Chinook during outmigration from John Day Dam to Bonneville Dam were due to gulls breeding on Miller Rocks Island. Predation probabilities on SR yearling Chinook by Miller Rocks Island gulls were among the lowest colony-specific impact observed at just 0.01 in most years (see *7.8 Supplements*, *Figure 7.S4*). Despite the small colony-specific predation impacts observed at colonies that forage upstream of Bonneville Dam, predation probabilities on SR yearling Chinook smolts were more substantial by colonies in the Columbia River estuary, particularly predation by the East Sand Island double-crested cormorant colony, where impacts were consistently higher than 0.05, and as high as 0.18 (0.13–0.28) during 2008–2015 (see *7.8 Supplements*, *Figure 7.S4*), losses that occurred prior to the mass dispersal events that resulted in minimum estimates of predation during 2016–2018.

The cumulative effects of avian predation by all 14 colonies on SR yearling Chinook during outmigration from Lower Monumental Dam to the Pacific Ocean were similar to those on UCR yearling Chinook, ranging annually from 0.05 (0.04–0.06) to 0.16 (0.13–0.19; *Figure 7.1* above). Of the predator species evaluated, predation effects were the highest by double-crested cormorant colonies, particularly the colony on East Sand Island in the Columbia River estuary (*Figure 7.1* above).

7.4.2.5 Snake River subyearling Chinook salmon – Of the individual colonies evaluated, predation probabilities on SR subyearling Chinook during passage from Lower Monumental Dam to McNary Dam were less than 0.03 by all colonies in all years, with the exception of the gull colony on Badger Island where an estimated 0.13 (0.08–0.20) of available fish were consumed in 2018 (see *7.8 Supplements*, *Figure 7.S5*). The Badger Island gull colony formed in 2015, coincident with abandonment of the gull colony on nearby Crescent Island in 2015 (see also *Chapter 6*). Similar to UCR and SR yearling Chinook, predation probabilities by individual colonies on SR subyearling Chinook during passage from McNary Dam to John Day Dam and John Day Dam to Bonneville Dam were less than 0.02 per colony, per year, impacts that were amongst the lowest reach-specific estimates observed (see *7.8 Supplements*, *Figures 7.S1-S6*). Analogous to other salmonid populations, predation by SR subyearling Chinook during passage from John Day Dam to Bonneville Dam was the highest by gulls breeding on Miller Rocks Island, with nearly all avian predation documented associated with this colony (see *7.8 Supplements*, *Figure 7.S5*). Similar to predation on other Chinook populations, predation by double-crested cormorants breeding on East Sand Island in the Columbia River estuary were the highest, with upwards of 0.04 (0.02–0.08) of available SR subyearling Chinook consumed in some years (see *7.8 Supplements*, *Figure 7.S5*). Predation impacts by Caspian terns breeding on East Sand Island, however, were consistently less than 0.02 (see *7.8 Supplements*, *Figure 7.S5*).

The cumulative effects of avian predation on SR subyearling Chinook during outmigration to Pacific Ocean were similar to and slightly lower than those on UCR and SR yearling Chinook, ranging annually from 0.04 (0.03–0.07) to 0.10 (0.07–0.15) during 2008–2017 (*Figure 7.1* above). Again, the one notable exception were impacts in 2018 from birds breeding on Badger Island, birds that consumed significantly more SR subyearling Chinook, which resulted in a cumulative predation estimate of 0.19 (0.14–0.26) in 2018. Excluding estimates from 2018, cumulative predation effects on SR subyearling Chinook were amongst the lowest of any salmonid species and population evaluated (*Figure 7.1* above).

7.4.2.6 Snake River sockeye salmon – Of the individual colonies evaluated, predation probabilities on SR sockeye during passage from Lower Monumental Dam to McNary Dam were consistently the highest by double-crested cormorants on Foundation Island and, more recently, by gulls on Badger Island. Predation probabilities by Foundation Island cormorants were consistently greater than 0.03, and as high as 0.05 (0.03–0.08; see *7.8 Supplements*, *Figure 7.S6*). Predation by gulls on Badger Island and other nearby gull colonies (Crescent Island and Island 20) were less than 0.02 in most years, except in 2017–2018, when upwards of 0.09 (0.04–0.16) of available fish were consumed by the gull colony on Badger Island (see *7.8 Supplements*, *Figure 7.S6*). Predation by Caspian terns breeding on Crescent Island were as high 0.03 prior to implementation of management actions associated with the *IAPMP* that eliminated the Crescent Island tern colony in 2015 (see also *Chapter 2*). Predation probabilities on SR sockeye during passage from McNary Dam to John Day Dam and John Day Dam to Bonneville Dam were almost exclusively by gulls nesting on Central Blalock Island and Miller Rocks Island. Unlike the predation on SR and UCR steelhead, predation by Central Blalock Island terns on SR sockeye were extremely low (less than 0.01; see *7.8 Supplements*, *Figure 7.S6*), even during 2015–2018 when upwards of 0.11 of UCR steelhead were consumed by terns breeding a Central Blalock Islands (see *7.8 Supplements*, *Figure 7.S1*). Similar to predation on Chinook populations, predation by double-crested cormorants breeding on East Sand Island in the Columbia River estuary were amongst the highest of the 14 colonies evaluated, with estimates as high 0.11 (0.04–0.34) in 2008 and 0.09 (0.05–0.17) in 2014, estimates that were documented prior to the mass colony dispersal events during 2016–2018 (see *7.8 Supplements*, *Figure 7.S6*). Small sample sizes of PIT-tagged SR sockeye (*Table 7.1* above) resulted in imprecise estimates of predation relative to other SR salmonid populations, as noted by the width of 95% creditable intervals.

Cumulative predation probabilities on SR sockeye were consistently greater than those observed in Chinook populations and were substantial in some, but not all years, ranging annually from 0.08 (0.03–0.22) to 0.25 (0.14–0.44; *Figure 7.1* above). Of the avian species evaluated, cumulative predation probabilities were often the highest by gull colonies, followed closely by double-crested cormorant colonies, with probabilities for gulls ranging annually from 0.05 (0.02–0.11) to 0.14 (0.08–0.22) and for cormorants from 0.02 (0.01–0.04) to 0.10 (0.06–0.17). The cumulative effects of Caspian tern predation on SR sockeye were generally low compared with gulls and cormorants, ranging annually from 0.02 (0.01–0.03) to 0.05 (0.02–0.10).

7.4.2.7 Average annual avian predation – Since 2008, there have been substantial changes regarding the numbers of piscivorous colonial waterbirds and locations of their colonies in the CRB (see *Appendix B1, Colony Sizes*). Some of these changes were due to naturally occurring events and some were associated with management plans aimed at reducing the size of Caspian tern colonies on Goose Island, Crescent Island, and East Sand Island (see *Chapters 1-2*) and the double-crested cormorant colony on East Sand Island (see *Chapter 4*). To depict the most recent predation losses, *Figure 7.2* and *Figure 7.3* depict average annual, reach-specific predation probabilities on UCR and SR smolts during 2015–2018, a period that coincides with management actions at the aforementioned tern and cormorant colonies. As previously noted, predation impacts in the estuary were likely higher than what is reported during 2016–2018 due to dispersal (abandonment) events of cormorants from East Sand Island during the peak smolt out-migration period. No data were available for cormorants nesting on Foundation Island in McNary Reservoir during 2015–2018, but an active colony was confirmed during these years (*Appendix B, Table B.11*). To account for smolt losses by Foundation Island cormorants in estimates of average annual predation probabilities, we used the average annual predation probability observed during 2008–2014 as a surrogate for annual average probabilities during 2015–2018. This approach was used only for cormorants on Foundation Island for the purpose of depicting average annual cumulative predation probabilities during 2015–2018; all other estimates of predation presented in this study were based solely on empirically-derived estimates of predation.

Average annual estimates indicated that for UCR steelhead predation probabilities were greater than 0.10 in all river reaches between Rock Island Dam and the Pacific Ocean during 2015–2018, with the highest estimates observed between McNary Dam and John Day Dam at 0.17 (0.13–0.21; *Figure 7.2*, see *7.8 Supplements*, *Figure 7.S1* for colony- and year-specific estimates). For UCR yearling Chinook, predation probabilities were, on average, estimated to be less than 0.02 and less than 0.05 between Rock Island Dam and McNary Dam and McNary Dam and Bonneville Dam, respectively, and were the highest in Columbia River estuary at 0.06 (0.05–0.08; *Figure 7.2* and see *7.8 Supplements*, *Figure 7.S3*).

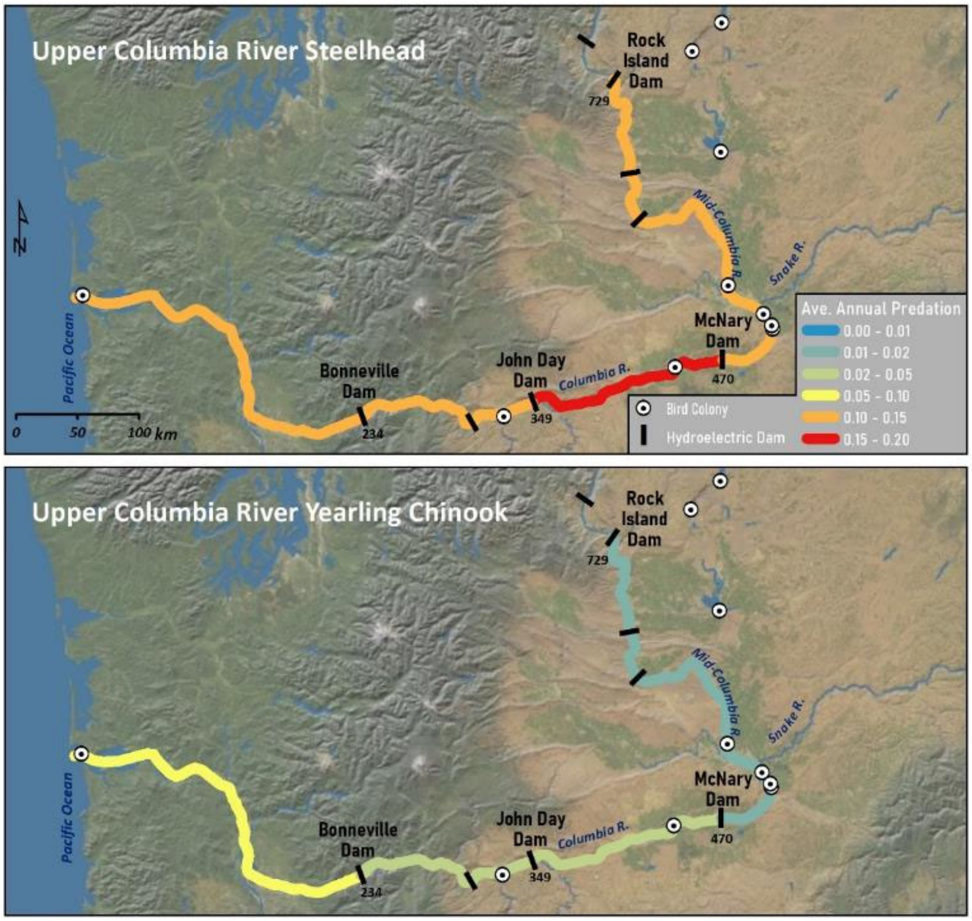


Figure 7.2. Average annual, reach-specific predation probabilities on Upper Columbia River steelhead and yearling Chinook by colonial waterbirds during 2015–2018. Predation is depicted as the proportion of available fish passing through each river reach that were consumed by all colonies combined foraging within that river reach (see 7.8 Supplements, Figures 7.S1–7.S6 for colony- and reach-specific predation probabilities).

For Snake River steelhead, average annual predation probabilities during 2015–2018 were greater than 0.05 in all river reaches between Lower Monumental Dam and the Pacific Ocean, with the highest rates observed in the Columbia River estuary at 0.14 (0.12–0.17; *Figure 7.3* and see *7.8 Supplements*, *Figure 7.S2*). Average annual estimates were consistently less than 0.05, and often less than 0.02, for both yearling and subyearling SR Chinook. The highest average annual predation estimates were 0.07 (0.06–0.09) for SR yearling Chinook in the Columbia River estuary and 0.06 (0.04–0.07) for SR subyearling Chinook between Lower Monumental Dam and

McNary Dam (*Figure 7.3* and see *7.8 Supplements*, *Figures 7.S4-7.S5*). Estimates for SR subyearling Chinook were greatly influenced by unusually high estimates of predation by the gull colony on Badger Island in 2018; predation probabilities in this river reach were less 0.02 during 2015-2017 (see *7.8 Supplements*, *Figure 7.S5*). For SR sockeye, average annual predation probabilities were less than 0.05 in all river reaches, except for Lower Monumental Dam to McNary Dam where an estimated 0.09 (0.07–0.13) of available sockeye were consumed during 2015–2018. Again, due to cormorant dispersal events from the East Sand Island colony and the growth of colonies in the upper estuary (i.e. the Astoria-Megler bridge colony; see *Chapter 4*) that occurred during 2016–2018, predation impacts in the estuary were likely greater than those observed, particularly on SR sockeye, a species that was especially susceptible to cormorant predation in the estuary in years when typical cormorant nesting behavior on East Sand Island occurred (see *7.8 Supplements*, *Figure 7.S6*).

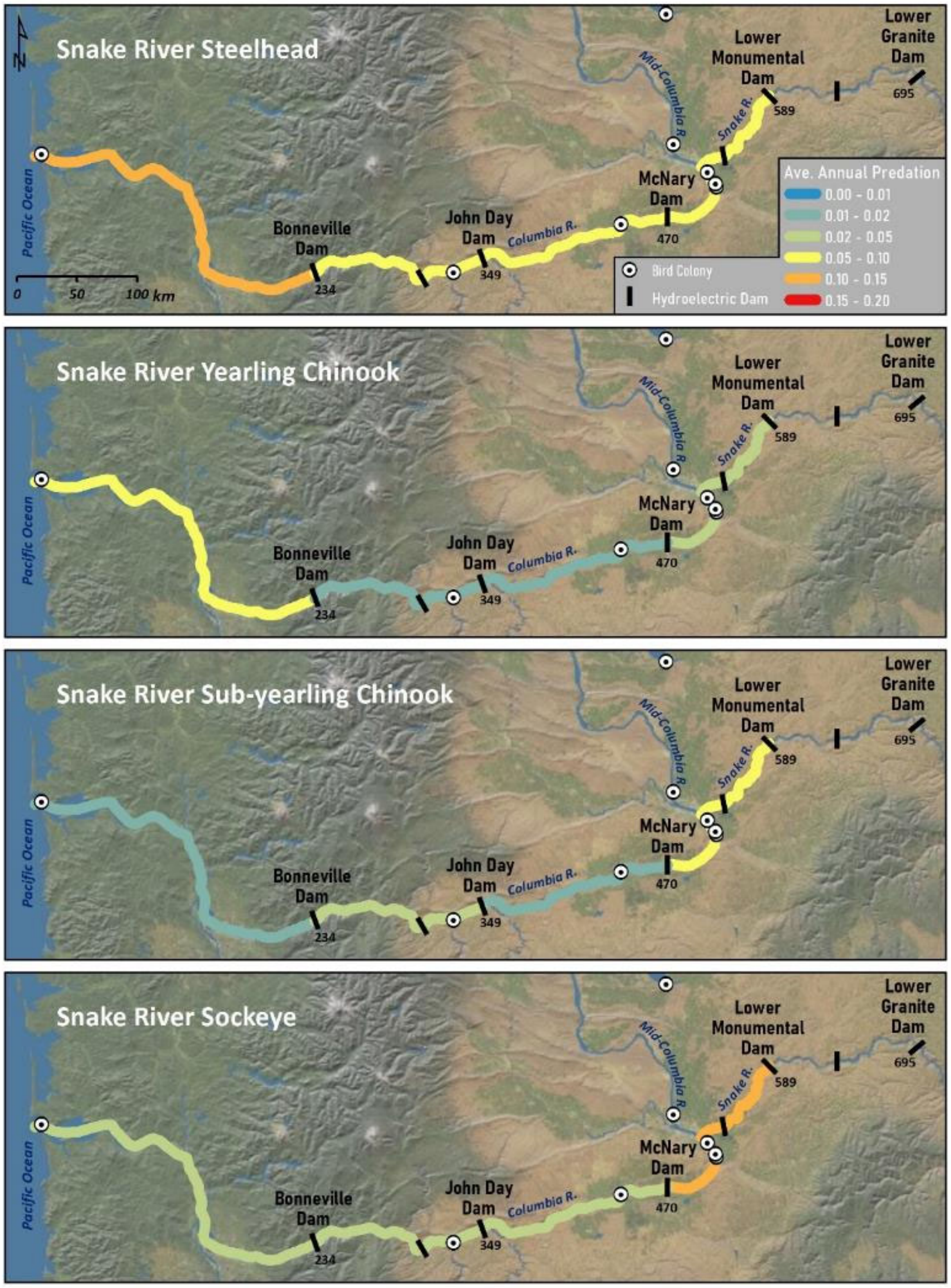


*Figure 7.3. Average annual, reach-specific cumulative predation probabilities on Snake River steelhead yearling Chinook, subyearling Chinook, and sockeye by colonial waterbirds during 2015–2018. Predation is depicted as the proportion of available fish passing through each river reach that were consumed by all colonies combined foraging within that river reach (*see *7.8 Supplements, Figures 7.S1–7.S6 for colony- and year-specific predation estimates).*

7.4.3 Total Mortality & Mortality Due to Avian Predation

Estimated total mortality (1 – survival) and what proportion of total mortality was due to avian predation varied considerably by salmonid population, river-reach, and year. Estimates of cumulative, total mortality and mortality due to avian predation are presented in *Figure 7.4*. Estimates of reach-specific total mortality and mortality due to colonial waterbird predation are presented in *7.8 Supplements* section, *Figures 7.S1-7.S6*. Results are also summarized below for each salmonid species and population evaluated.

7.4.3.1 Upper Columbia River steelhead trout – Estimates of reach-specific total mortality in UCR steelhead were generally the highest between Rock Island Dam and McNary Dam relative to McNary Dam to John Day Dam and John Day Dam to Bonneville Dam. For instance, total mortality of UCR steelhead ranged from 0.26 (0.11–0.33) to 0.44 (0.36–0.50) between Rock Island Dam and McNary Dam and was less than 0.25 in the two river reaches downstream of McNary Dam (see *7.8 Supplements*, *Figure 7.S1*). Comparisons of total mortality and mortality due to colonial waterbird predation indicated that avian predation was often the dominate source of UCR steelhead mortality within each river reach and year evaluated. For instance, avian predation accounted for more than 50% of all sources of UCR steelhead mortality in the three reaches downstream of Rock Island Dam in most years (see *7.8 Supplements*, *Figure 7.S1*). Between McNary and John Day Dam, predation by Caspian terns and gulls nesting on Blalock Islands accounted for more than 70%, and up to 90%, of all sources of UCR steelhead mortality during 2015–2018 (see *7.8 Supplements*, *Figure 7.S1*). An estimate of total mortality during out-migration between Bonneville Dam and Pacific Ocean could not be calculated because there were no PIT tag detection sites in the lower Columbia River estuary downstream of the bird colonies on East Sand Island.

Estimated cumulative total mortality of UCR steelhead from Rock Island Dam to Bonneville Dam ranged annually from 0.45 (0.36–0.61) to 0.73 (0.68–0.76), indicating that a large proportion, and in many years the majority, of UCR steelhead smolts died prior to reaching Bonneville Dam (*Figure 7.4*). Comparisons of total mortality and mortality due to avian predation indicated avian predation accounted for 42% (30–56%) to 70% (53–87%) of all mortality, suggesting that more UCR steelhead were consumed by avian predators during out-migration to Bonneville Dam than died from all other mortality sources combined in most years (*Figure 7.4*; see also Evans et al. 2019).

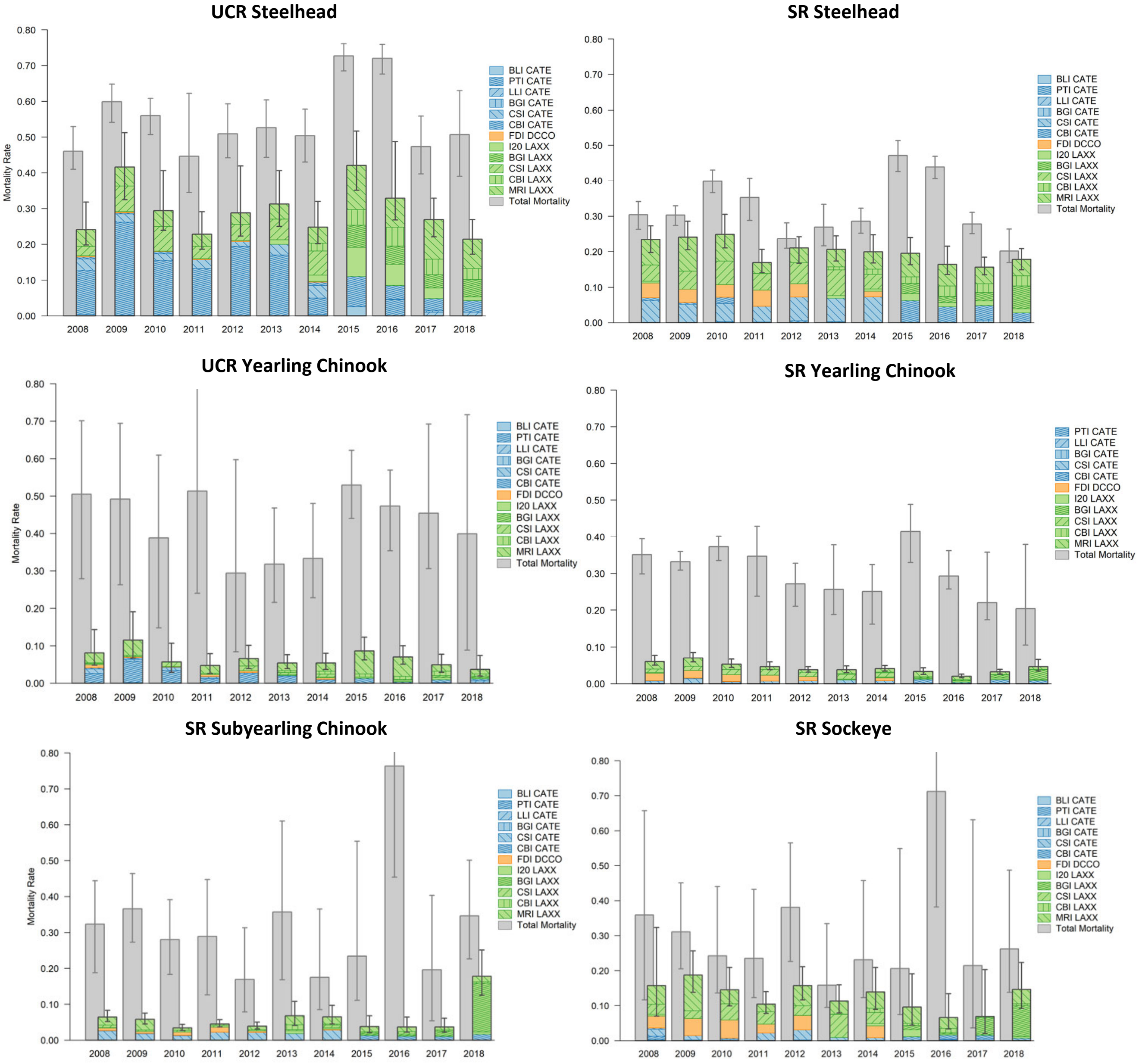


Figure 7.4. Estimated total mortality and mortality attributed to predation by colonial waterbirds on Upper Colombia River (UCR) steelhead and yearling Chinook smolts during passage from Rock Island Dam to Bonneville Dam and Snake River (SR) steelhead, yearling Chinook, subyearling Chinook, and sockeye during passing from Lower Monumental Dam to Bonneville Dam. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), Central Blalock Islands (CBI), and Miller Rocks Island (MRI). Avian species include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and cumulative mortality due to avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011 and the FDI DCCO during 2013 and 2015–2018.

7.4.3.2 Snake River steelhead trout – Estimated total mortality of SR steelhead from Lower Granite Dam to Little Goose Dam and Little Goose Dam to Lower Monumental Dam were low in most years, ranging annually from 0.01 (0.01–0.04) to 0.16 (0.14–0.18) per reach, indicating that the vast majority of SR steelhead survived outmigration to Lower Monumental Dam during 2008–2018. Estimates of total mortality in SR steelhead during passage from Lower Monumental Dam (the upper most foraging range of colonies included in the study) to McNary Dam ranged from 0.06 (0.02–0.08) to 0.19 (0.13–0.19; see *7.8 Supplements*, *Figure 7.S2*). Estimates in SR steelhead between Lower Monumental Dam and McNary Dam were consistently lower than those of UCR steelhead during passage from Rock Island Dam to McNary Dam. Estimates of total mortality in both populations (UCR, SR) were more similar after smolts from reached McNary Dam, suggesting that similar factors influenced the survival of SR and UCR steelhead during passing from McNary Dam to Bonneville Dam (see *7.8 Supplements*, *Figures 7.S1-7.S2*). Comparisons of reach-specific total mortality and mortality due to colonial waterbird predation indicate that avian predation was often the dominate source of SR steelhead mortality within each river reach and year (see *7.8 Supplements*, *Figure 7.S2*). For instance, during passage from Lower Monumental Dam to McNary Dam, avian predation accounted for more than 50% of all sources of SR steelhead mortality in 9 of the 11 years evaluated and more than 80% of all sources of mortality in 6 of the 11 years (see *7.8 Supplements*, *Figure 7.S2*). Similarly, despite low average annual estimates of total mortality, mortality due to avian predation accounted from more than 50% of all mortality sources during smolt passage from McNary Dam to John Day Dam and John Day Dam to Bonneville in most years (see *7.8 Supplements*, *Figure 7.S2*).

Estimates of cumulative total mortality from Lower Monumental Dam to Bonneville Dam ranged annually from 0.20 (0.17–0.26) to 0.47 (0.43–0.51; *Figure 7.4* above). Comparisons of total mortality to mortality due to avian predation indicated avian predation accounted for 37% (33–46%) to 89% (80–86%) of all SR steelhead mortality between Lower Monumental and Bonneville Dams (*Figure 7.4* above). Results for SR steelhead were similar to those for UCR steelhead, indicating that more SR steelhead were consumed by avian predators during out-migration to Bonneville Dam than died from all other mortality sources combined in most years.

7.4.3.3 Upper Columbia River yearling Chinook salmon – Estimates of reach-specific total mortality in UCR yearling Chinook were consistently the highest between Rock Island Dam and McNary Dam relative to McNary Dam to John Day Dam and John Day Dam to Bonneville Dam. For instance, total mortality of UCR yearling Chinook ranged from 0.07 (0.01–0.32) to 0.39 (0.26–0.47) between Rock Island Dam and McNary Dam but was generally less than 0.10 and was as low as 0.03 in the two river reaches downstream of McNary Dam, depending on the reach and year (see *7.8 Supplements*, *Figure 7.S1*). Comparisons of total mortality and mortality due to colonial waterbird predation indicate that avian predation accounted for 3% (1–5%) to 77% (52–83%), depending on river reach and year. Avian predation accounted for larger proportion of all sources of UCR yearling Chinook mortality downstream of McNary Dam staring in 2015, where estimates of both total mortality and mortality due Caspian terns and gulls were elevated (see *7.8 Supplements*, *Figure 7.S3*). Again, an estimate of total mortality during out-migration between Bonneville Dam and Pacific Ocean could not be calculated because there were no PIT tag detection sites in the lower Columbia River estuary downstream of the bird colonies on East Sand Island. It should be noted, however, that of the 14 colonies evaluated, double-crested cormorants breeding on East Sand Island consumed the largest proportion of available UCR yearling Chinook (see *7.8 Supplements*, *Figure 7.S3*).

Estimated cumulative total mortality of UCR yearling Chinook during passage from Rock Island Dam to Bonneville Dam ranged annually from 0.29 (0.08–0.59) to 0.53 (0.44–0.62), indicating that a large proportion, and in some years the majority, of smolts died prior to reaching Bonneville Dam (*Figure 7.4* above). Comparisons of total mortality and mortality due to avian predation indicated that avian predation accounted for 9% (3–33%) to 23% (10–73%) of all UCR yearling Chinook mortality sources (*Figure 7.4* above). Results indicate that relative to UCR and SR steelhead, avian predation accounted for significantly smaller proportion of total mortality and that factors other than avian predation were the dominate source of mortality in UCR yearling Chinook smolts during passage to Bonneville Dam. Due to the high level of predation observed by colonies in the Columbia River estuary, the relative effects of avian predation were likely more substantial below Bonneville Dam, but remain unknown due to lack of mortality estimates from Bonneville Dam to the Pacific Ocean.

Small sample sizes of PIT-tagged UCR yearling Chinook (*Table 7.1* above) resulted in imprecise estimates of total mortality in several river reaches, and years evaluated, as conveyed by the width of 95% CRI. For instance, estimates of total mortality between John Day Dam and Bonneville Dam were essentially non-informative, with 95% CRI ranging from 0.01 to 0.73, for a point estimate of 0.07 in 2011 (see *7.8 Supplements*, *Figure 7.S3*), a year when just 1,705 UCR yearling Chinook smolts were PIT-tagged (*Table 7.1* above). Given such high levels of uncertainty, estimates of total mortality and relative comparisons of total mortality to mortality due to avian predation in UCR yearling Chinook should be interpreted cautiously.

7.4.3.4 Snake River yearling Chinook salmon – Similar to SR steelhead, estimated total mortality of SR yearling Chinook from Lower Granite Dam to Little Goose Dam and Little Goose Dam to Lower Monumental Dam was very low in most, but not all years, ranging annually from just 0.01 (0.01–0.04) to 0.17 (0.14–0.21) per reach. Estimates were less than 0.10 is most years, indicating that the vast majority of SR yearling Chinook survived outmigration to Lower Monumental Dam during 2008–2018. Estimates of total mortality in SR yearling Chinook during passage from Lower Monumental Dam to McNary Dam were also low in most years, ranging annually from 0.03 (0.02–0.04) to 0.16 (0.12–0.19; see *7.8 Supplements*, *Figure 7.S4*). Estimates of total mortality downstream of McNary Dam to Bonneville Dam were highly variable, ranging from just 0.01 (0.01–0.03) to 0.32 (0.23–0.38), depending on reach and year (see *7.8 Supplements*, *Figure 7.S4*). Of the three river reaches evaluated, avian predation accounted for the greatest proportion of total mortality between Lower Monumental Dam and McNary Dam, with birds annually accounted for 7% (4–17%) to upwards of 97% (58–100%) of total mortality (see *7.8 Supplements*, *Figure 7.S4*). Similar to UCR yearling Chinook, avian predation was consistently a minor component of all sources of smolt mortality from McNary Dam to John Day Dam and John Day Dam to Bonneville Dam, with avian predation often accounting for less than

10% of total mortality in most, but not all, river reaches and years (see *7.8 Supplements*, *Figure 7.S4*). Also similar to UCR yearling Chinook, the effects of avian predation on SR yearling Chinook were the greatest in the Columbia River estuary (see *7.8 Supplements*, *Figure 7.S4*), where predation effects could not be compared to total mortality due to a lack of survival estimates to the Pacific Ocean (see *Methods*).

Estimates of cumulative total mortality in SR yearling Chinook during passage from Lower Monumental Dam to Bonneville Dam ranged annually from 0.22 (0.17–0.36) to 0.42 (0.33–0.49; *Figure 7.4* above). Comparisons of total mortality in SR yearling Chinook to mortality due to avian predation indicated avian predation accounted for 7% (4–7%) to 23% (9–17%) of all SR steelhead mortality between Lower Monumental and Bonneville Dams (*Figure 7.4* above). Results for SR yearling Chinook were similar to those for UCR yearling Chinook and suggests that avian predation was relatively minor component of total mortality in most years, especially relative to SR and UCR steelhead, where avian predation was consistently the dominate source of mortality during smolt passage to Bonneville Dam.

7.4.3.5 Snake River sub-yearling Chinook salmon – Analogous to UCR yearling Chinook, small sample sizes of PIT-tagged SR subyearling Chinook resulted in imprecise estimates of total mortality and, consequently, relative comparisons of total mortality to mortality due to avian predation. As such, results should be interpreted cautiously and are only reported below where estimates provide a modicum of certainty.

Estimates of total mortality of SR subyearling Chinook from Lower Granite Dam to Little Goose Dam and Little Goose Dam to Lower Monumental Dam were consistently higher than those observed in SR steelhead and SR yearling Chinook, with estimates ranging annually from 0.05 (0.01–0.10) to 0.23 (0.17–0.30) per reach (see *7.8 Supplements*, *Figure 7.S5*). In most years, estimates were greater than 0.10 per reach, indicate that a relatively large proportion of subyearling Chinook died prior to reaching Lower Monumental Dam, the upper most foraging range of bird colonies included in the study. Estimates of total mortality in SR subyearling Chinook during passage from Lower Monumental Dam to McNary Dam were similar to those upstream of Lower Monumental Dam, ranging annually from 0.04 (0.02–0.08) to 0.21 (0.14–0.30; see *7.8 Supplements*, *Figure 7.S5*). Similar to trends in SR yearling Chinook, estimates of total mortality in SR subyearling Chinook during passage from McNary Dam to John Day Dam and John Day Dam to Bonneville Dam were highly variable, ranging from just 0.03 (0.01–0.13) to 0.62 (0.28–0.81), depending on river reach and year (see *7.8 Supplements*, *Figure 7.S5*). During 2008-2012, larger numbers (> 20,000; *Table 7.1* above) of SR subyearling Chinook were PIT-tagged and comparisons of total mortality to mortality due to avian predation in those years indicated that avian predation accounted for the largest proportion of total mortality between Lower Monumental Dam and McNary Dam, with estimates ranging annually from 28% (11–59%) to 78% (32–98%; see *7.8 Supplements*, *Figure 7.S5*). Downstream of McNary Dam, avian predation was a small component of total mortality, with avian predation accounting for less than 25% of total mortality in most reaches and years during smolt outmigration to Bonneville Dam (see *7.8 Supplements*, *Figure 7.S5*). Again, estimates of total mortality from Bonneville Dam to the Pacific Ocean were not available, so the relative impacts of Caspian terns and double-crested cormorants breeding on East Sand Island on total SR subyearling Chinook mortality were unknown.

Estimates of cumulative total mortality in SR subyearling Chinook from Lower Monumental Dam to Bonneville Dam ranged annually from 0.17 (0.08–0.31) to 0.76 (0.45–0.89; *Figure 7.4* above). Estimates in 2016 were significantly higher than those in other years but were imprecise and thus largely uninformative (*Figure 7.4* above). Comparisons of total mortality to mortality due to avian predation during 2008-2012 indicated avian predation accounted for 12% (7–24%) to 23% (10–63%) of all SR subyearling Chinook mortality between Lower Monumental and Bonneville Dams (*Figure 7.4* above). Results suggest that in most years, avian predation represented a relatively small proportion of total mortality.

7.4.3.6 Snake River sockeye salmon – Similar to UCR Chinook and SR subyearling Chinook, small sample sizes of SR sockeye resulted in imprecise estimates of total mortality and, consequently, relative comparisons of total mortality to mortality due to avian predation. In the case of SR sockeye, the lower 95% CRI bounds associated with estimates of total mortality exceeded the upper 95% CRI bound associated with estimates of cumulative predation in river reaches and years evaluated. Due to such high levels of uncertainty, results should be interpreted cautiously and only general statements regarding relative comparisons of total mortality and mortality due to avian predation are provided for SR sockeye.

Estimates of total mortality of SR sockeye during passage from Lower Granite Dam to Little Goose Dam and Little Goose Dam to Lower Monumental Dam were highly variable, ranging annually from 0.01 (0.1–0.07) to 0.43 (0.39–0.46), depending on the reach and year. Similarly, estimates of total mortality during passage from Lower Monumental Dam to McNary Dam and McNary Dam to Bonneville Dam were also highly variable, but generally lower, than those upstream of Lower Monumental Dam, ranging annually from 0.02 (0.01–0.08) to 0.24 (0.02–0.67), depending on the reach and year (*Figure 7.4* above). Comparisons of total mortality to mortality due to avian predation suggest predation effects were the greatest between Lower Monumental Dam and McNary Dam, with avian predation accounting for roughly 10% to 80% of all sources of SR sockeye mortality annually (see *7.8 Supplements*, *Figure 7.S6*). Estimates were often lower during passage from McNary Dam to John Day Dam and John Day Dam to Bonneville Dam, although point estimates of predation from gulls breeding on Miller Rocks Island resulted in estimates of avian predation accounting for more than 50% of all mortality sources in some years (*Figure 7.4* above). An appreciable proportion of SR sockeye (ca. 0.03 to 0.11 during 2008–2015; see *7.8 Supplements*, *Figure 7.S6*) were consumed by double-crested cormorants breeding on East Sand Island in the Columbia River estuary, but relative comparisons of these impacts to total mortality were not available due to a lack of PIT tag detection sites downstream of East Sand Island.

Estimates of cumulative total mortality from Lower Monumental Dam to Bonneville Dam in SR sockeye ranged annually from 0.16 (0.10–0.33) to 0.71 (0.38–0.88; *Figure 7.4 above*). Gross comparisons of total mortality to mortality due to avian predation indicating that avian predation accounted for more 50% of all SR sockeye mortality sources during passage from

Lower Monumental Dam to Bonneville Dam in 5 of the 11 years evaluated. Results suggest that the cumulative effects of avian predation were generally greater in sockeye salmon than those of other salmon species and populations.

7.5 Discussion

Results of this study address the effects of several avian predator species from multiple breeding colonies on the mortality of six different salmonid populations originating from the UCR and SR basins, providing a comprehensive, system-wide evaluation of avian predation. Results from this and other studies (Evans et al. 2016, Evans et al. 2019, Payton et al. 2019) suggest that avian predation, although not the original cause of salmonid declines in the CRB (Nelson et al. 1991), is now a factor limiting the recovery of some salmonid populations, particularly UCR and SR steelhead. The cumulative effects of avian predation (predation by birds from all breeding colonies combined) on UCR and SR steelhead were between 0.18 and 0.53 (or 18% to 53%) of available smolts annually during 2008–2018. Results suggest that avian predation was the single greatest direct source of smolt mortality during out-migration for UCR steelhead from the middle Columbia River to Bonneville Dam or for SR steelhead from the lower Snake River to Bonneville Dam, with avian predation accounting for more than 50% of all smolt mortality in 9 of the 11 years evaluated. Results indicate that, in some years, UCR and SR steelhead smolt losses due to predation by colonial waterbirds were greater than losses associated with passage through six hydroelectric dams, predation from piscivorous fish, mortality from disease, and mortality from all other remaining sources combined. Even after passage through the impounded sections of the lower Snake River and Columbia River upstream of Bonneville Dam, predation by Caspian terns and double-crested cormorants breeding on East Sand Island in the Columbia River estuary were substantial, ranging annually from 0.10 to 0.38 of available steelhead.

The cumulative effects of avian predation on UCR and SR Chinook smolts (yearling and sub-yearling) and SR sockeye smolts were significantly lower than those on UCR and SR steelhead smolts in most river reaches, and years evaluated. For example, cumulative avian predation probabilities for UCR yearling Chinook and SR yearling and sub-yearling Chinook were often less than 0.10 of available smolts annually. The cumulative effects of avian predation on SR sockeye smolts were intermediate between those of steelhead and Chinook smolts, with estimated predation probabilities generally being between 0.15 and 0.25 of available smolts annually. Several other studies have documented that impacts of avian predation on smolt survival vary significantly based on the species of salmonid, and that steelhead smolts are especially susceptibility to predation from colonial waterbirds (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Osterback et al. 2014, Evans et al. 2016). Potential reasons for the greater susceptibility of juvenile steelhead to avian predation include differences in the size and behavior of steelhead compared with other salmonid species (Evans et al. 2012, Evans et al. 2016). In particular, steelhead smolts are on average longer (fork-length) and tend to be more surface-oriented compared with salmon smolts (Quinn 2005, Beeman and Maule 2006), traits that make steelhead especially susceptible to plunge diving predators like Caspian terns and California gulls (Winkler 1996, Cuthbert and Wires 1999; see also *Chapter 10*).

Mortality due to avian predation compared with total smolt mortality for Chinook indicated that avian predation was generally a small component of total mortality during passage to Bonneville Dam. These results are consistent with those of Evans et al. (2016) and suggest that factors other than avian predation were responsible for the majority of Chinook smolt losses during out-migration. One component of Chinook smolt mortality upstream of Bonneville Dam was likely predation by piscivorous fishes, such as northern pikeminnow (*Ptychocheilus oregonensis*), smallmouth bass (*Micropterus dolomieu*), walleye (*Sander vitreus*), and channel catfish (*Ictalurus punctatus*). Rieman et al. (1991) estimated that approximately 0.14 of juvenile salmonids passing through John Day Reservoir were consumed by piscine predators and that mortality rates were highest for Chinook relative to other salmonid species. Harnish et al. (2014) and Michael et al. (2018) reported increases in the abundance of piscine predators in the Columbia River between Priest Rapids Dam and Bonneville Dam, and hypothesized that piscine predation was the greatest direct source of Chinook smolt mortality in this river reach. Estimates of Chinook smolt mortality associated with the direct effects of dam passage vary by dam and year, with estimates ranging annually from approximately 0.01 to 0.06 of available smolts per dam (Johnson et al. 2011, Hughes et al. 2013, Skalski et al. 2013a, Skalski et al. 2013b). In addition to piscine predators and direct mortality associated with dam passage, other direct sources of mortality for Chinook smolts likely occur (e.g., mortality associated with disease, poor water quality conditions, and other factors), but data to quantify these other mortality sources are generally lacking in the published literature (Dietrich et al. 2011).

Exceptions to the low estimates of predation by colonial waterbirds on Chinook smolts upstream of Bonneville Dam were predation by California and ring-billed gulls breeding on Crescent Island (prior to 2015), Badger Island (after 2015), and Miller Rocks and predation by double-crested cormorants nesting on East Sand Island in the Columbia River estuary. Potential reasons for higher predation probabilities on Chinook and sockeye (see *below*) smolts by these particular gull and cormorant colonies may be related to the large size of these colonies (thousands of breeding pairs; Adkins et al. 2014), behavioral flexibility to exploit temporarily available food sources (Winkler 1996, Collis et al. 2002), or, in the case of gulls, the close proximity of breeding colonies to dams where smolts may be more vulnerable to predation due to delays in passage, injury, mortality associated with turbine passage, or smolts being temporarily stunned or disoriented by hydraulic conditions in the tailrace of dams (Ruggerone 1986, Zorich et al. 2011, Evans et al. 2016; see also *Chapter 10*). In the case of predation by cormorants breeding on East Sand Island, UCR and SR smolt losses may be especially concerning as predation occurs on individuals that have survived freshwater out-migration and are significantly more likely to return as adults compared to those yet to complete out-migration to the estuary (Roby et al. 2003, Evans et al. 2019).

Few studies have documented and quantified cause-specific sources of mortality in SR sockeye smolts during out-migration through the Snake and Columbia rivers (Trushenski et al. 2019), making the results from this study especially important to conservation efforts associated with this critically endangered salmonid population (NOAA 2014). Results from this study indicate that cumulative effects of avian predation on SR sockeye were substantial and represented a large proportion of all sources of sockeye smolt mortality upstream of Bonneville Dam in several, but not all, years. This finding was surprising, as previous research indicated that relative to other juvenile salmonid species, sockeye salmon comprise a small proportion of the diet of colonial waterbirds in the CRB (Collis et al. 2002, Roby et al. 2003, Lyons 2010). Previously published estimates of avian predation rates have indicated similar levels of predation on sockeye and Chinook smolts by double-crested cormorants and Caspian terns, but datasets were limited to just one or two years and there were no estimates of the cumulative effects of avian predators from multiple breeding colonies, including gull colonies, on survival of SR sockeye smolts during out-migration (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Evans et al. 2016). Although small sample sizes of PIT-tagged sockeye smolts resulted in imprecise estimates of avian predation and survival probabilities, sample sizes were sufficient to conclude that the cumulative effects of avian predation on SR sockeye smolts were consistently greater than the cumulative effects of avian predation on SR yearling and sub-yearling Chinook smolts in most river reaches and years evaluated. On average, sockeye smolts are larger than sub-yearling Chinook smolts, but smaller than yearling Chinook smolts (Quinn 2005), so factors other than fish size may be related to the greater susceptibility to avian predation of SR sockeye smolts compared to SR and UCR Chinook smolts. For example, differences in the run-timing of SR sockeye smolts compared to SR yearling and sub-yearling Chinook smolts may increase the susceptibility of SR sockeye to avian predation, with sockeye smolts abundance peaking after most yearling Chinook smolts have out-migrated, but before most sub-yearling Chinook smolts have out-migrated (FPC 2020). Peak abundance of Snake River sockeye smolts in the lower Snake and Columbia rivers also generally occurs during late-May, a period that over-laps with the peak abundance of breeding colonial waterbirds in the CRB (Evans et al. 2012, Adkins et al. 2014, see also *Chapter 10*).

Estimates of avian predation presented herein represent minimum estimates of the total impact of piscivorous waterbirds on salmonid smolts because not all colonies included in the present study were scanned for smolt PIT tags in all years and because not all piscivorous waterbird species were included in this study. For example, PIT tag scanning on the gull colonies at Island 20 and at the Blalock Islands did not commence until 2013 and scanning for smolt PIT tags was conducted intermittently at the double-crested cormorant colony on Foundation Island. Predation impacts by gulls nesting at the Island 20 and Blalock Islands colonies were generally less than 0.02 of available smolts in most years when scanning occurred but estimates of predation by cormorants nesting on Foundation Island were consistently greater than 0.02, particularly predation on SR steelhead and SR sockeye smolts. Predation estimates at the double-crested cormorant colony on East Sand Island were also biased low to an unknown degree due to colony abandonment events, whereby cormorants temporarily dispersed from their nest sites, but remained in the Columbia River estuary in large numbers (several thousand) during the peak of the smolt out-migration period during 2016–2018 (Turecek et al. 2018, Evans et al. 2019). Other breeding colonies of double-crested cormorants also exist on bridges and channel markers in the upper Columbia River estuary, locations were smolt PIT tags could not be recovered following the breeding season. Given this and recent increases in the size of some of these colonies where smolt PIT tags were not collected (e.g., on the Astoria-Megler Bridge), the total impact of all double-crested cormorants on smolt survival in the Columbia River estuary was unknown during the latter half of the study (see also *Chapter 4*). Finally, we did not investigate impacts of predation on smolts by all piscivorous waterbird species in the CRB, such as American white pelicans (*Pelecanus erythrorthynchos)*, Brandt's cormorants (*P. penicillatus*), glaucous-winged/western gulls (*L. glaucescens X L. occidentalis*), common mergansers (*Mergus merganser*), Forster's terns (*Sterna forsteri*), great blue herons (*Ardea herodias*), black-crowned night-herons (*Nycticorax nycticorax*), and western and Clark's grebes (*Aechmophorus* spp.). In the case of American white pelicans, minimum estimates of smolt predation have been reported and indicate that impacts on steelhead and yearling Chinook smolts were generally low (less than 0.01; Evans et al. 2016; see also *Chapter 6*) but predation on sub-yearling Chinook smolts were significantly higher on some stocks (upwards of 0.10; Payton et al. 2020). In case of Brandt's cormorants, which nest in the Columbia River estuary, predation probabilities on smolts were amongst the lowest of any piscivorous waterbird species and colony in the CRB, with estimates consistently less than 0.01 of available smolts (Evans et al. 2016b). Predation estimates on glaucous-winged/western gulls, which also nest in the Columbia River estuary, are currently unknown but maybe relatively small based on the low proportion of salmonids (ca. 4%; Collis et al. 2002) reported in the diet of birds nesting on East Sand Island (see also *Chapter 6*). In the case of non-colonial or semi-colonial piscivorous waterbirds (i.e., Forester's terns, mergansers, herons, and grebes) documented predation impacts on smolt survival were much less than those of colonial piscivorous waterbirds (Wiese et al. 2008; Evans et al. 2016), suggesting that these species pose little risk to smolt survival.

In addition to biotic factors that can influence the susceptibility of smolts to colonial waterbird predation (e.g., predator species, colony sizes and locations, individual fish characteristics, and others; see also *Chapter 10*), abiotic factors can also contribute to the susceptibility of smolts to bird predation and thus smolt survival during outmigration. Petrosky and Schaller (2010) observed a relationship between increasing river flows in the Columbia River and higher rates of steelhead smolt survival during outmigration through the Federal Columbia River Power System (FCRPS), a relationship that has been linked to predation rates by colonial waterbirds, whereby higher river flows decrease fish travel times and consequently lower the exposure of smolts to bird predation. Hostetter et al. (2012) observed that increased river flows were related to a decrease in Caspian tern predation rates on steelhead smolts originating from the Snake River. Payton et al. (2016) observed that faster water transit times (a measure of flow in relation to reservoir levels) were associated with lower predation rates by Caspian terns on steelhead smolts passing through the Wanapum and Priest Rapids reservoirs in the middle Columbia River. Collectively, results from these and other studies indicate that numerous biotic factors and abiotic conditions experienced by smolts during outmigration influence their susceptibility to avian predation (Evans et al. 2019; see also *Chapter 10*).

The modeling approach used in this study utilized both recapture and recovery data to estimate survival and cause-specific mortality simultaneously or jointly. Conversely, several other studies have relied solely on recovery data to estimate predation rates without accounting for survival to within the foraging range of each particular waterbird breeding colony, and without partitioning avian predation for birds from colonies capable of foraging in multiple river reaches (e.g., Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2015, Frechette et al. 2015). The joint mortality and survival (JMS) model used herein was also able to sum predation probabilities across species of avian predators, across breeding colonies, and across river reaches to investigate the cumulative effects of avian predation in the context of smolt survival (Hostetter et al. 2018, Evans et al. 2019, Payton et al. 2019). Finally, in the present study, juvenile salmonids were grouped by species and age-class (for Chinook smolts) to maximize sample sizes of PIT-tagged fish and to provide the most robust dataset possible for investigating the cumulative effects of predation and for modeling the additive effects of predation on fish mortality (see *Chapter 8*). This approach differs from the methods of Evans et al. (2012) and Hostetter et al. (2015), where PIT-tagged fish were grouped strictly by ESA-listed population and availability was based on detections of smolts at fixed locations in space and time.

In conclusion, results of this study suggest that avian predation is now a factor limiting the recovery of some ESA-listed salmonid populations, particularly UCR and SR steelhead. For UCR and SR Chinook, however, avian predation represented a relatively small component of all smolt mortality. Our results suggest that the benefits of managing avian predators to reduce predation will vary greatly by the species of avian predator, by the particular breeding colony, and by the salmonid species and population. Collectively, the results of this study and the analytical framework used to jointly estimate cause-specific mortality and survival provide data to help prioritize where management actions directed at avian predators might be the most beneficial to ESA-listed salmonid populations in the CRB and a method to quantify the effects of avian predation on salmonid smolt survival across large spatial-scales.

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7.8 Supplements

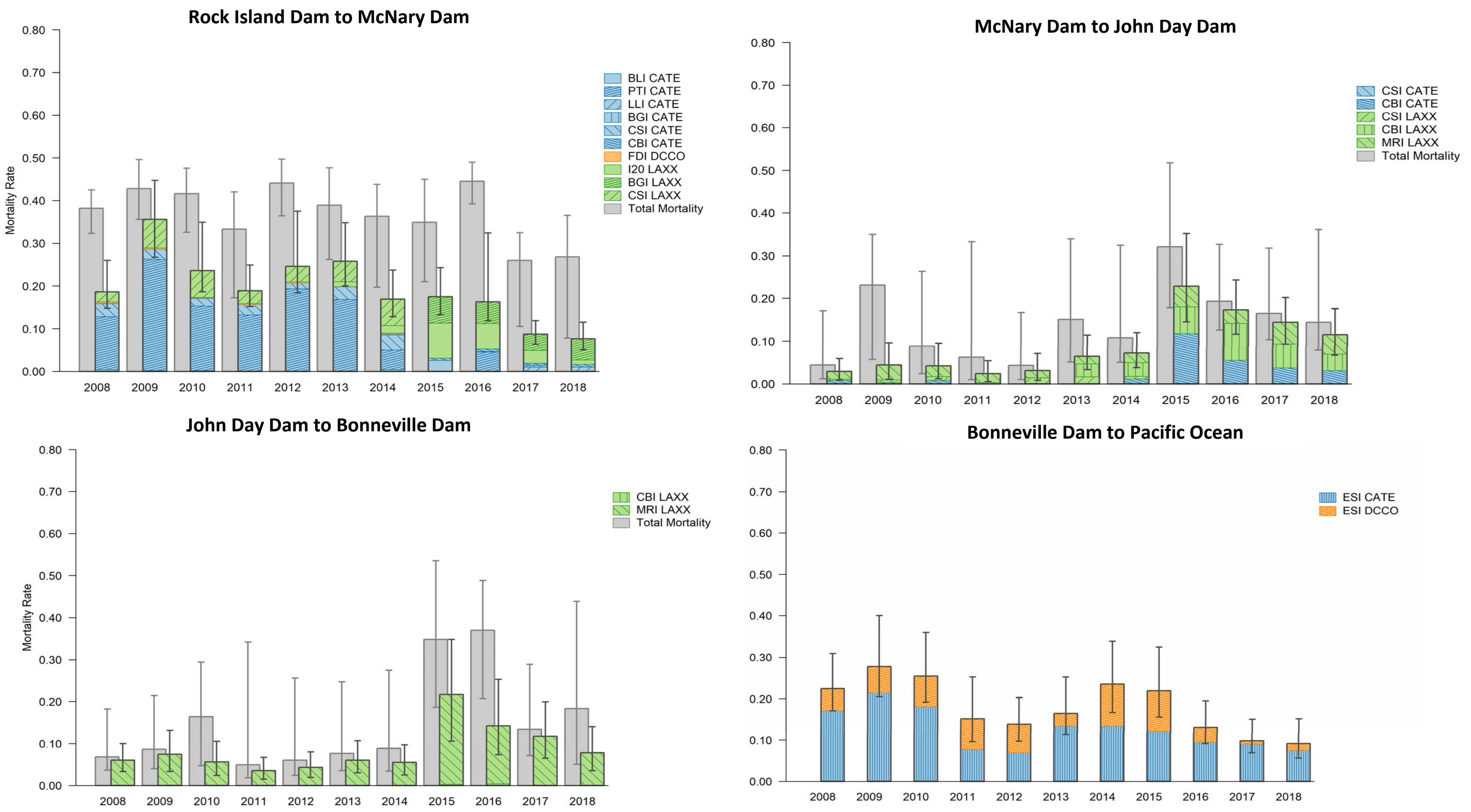

Figure 7.S1. Estimates annual reach-specific predation (colored bars) and total mortality (grey bars) of Upper Columbia River steelhead during passage from Rock Island Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods). Results are those of Evans et al. (2019).

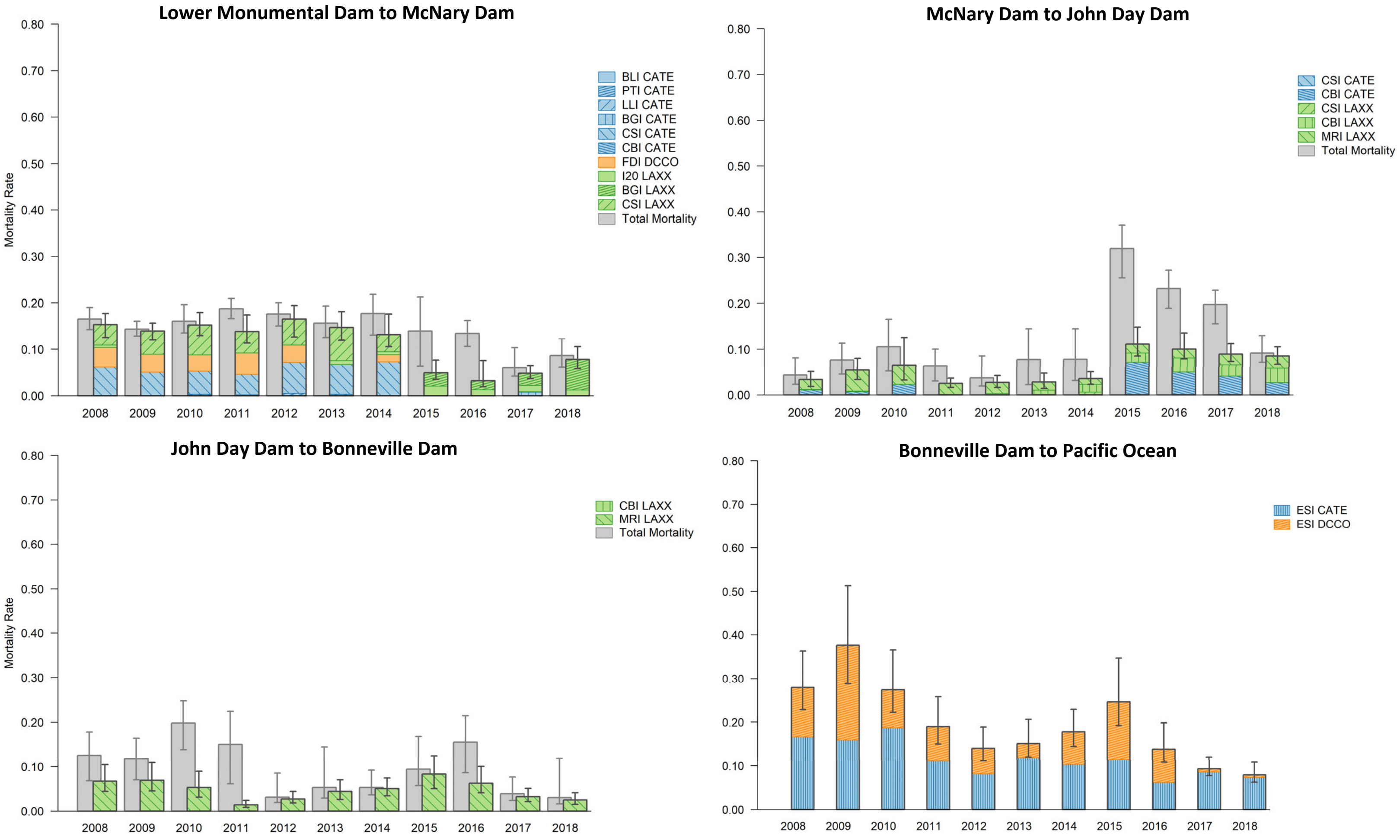

Figure 7.S2. Estimates annual reach-specific predation (colored bars) and total mortality (grey bars) of Snake River steelhead during passage from Rock Island Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, the FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods).

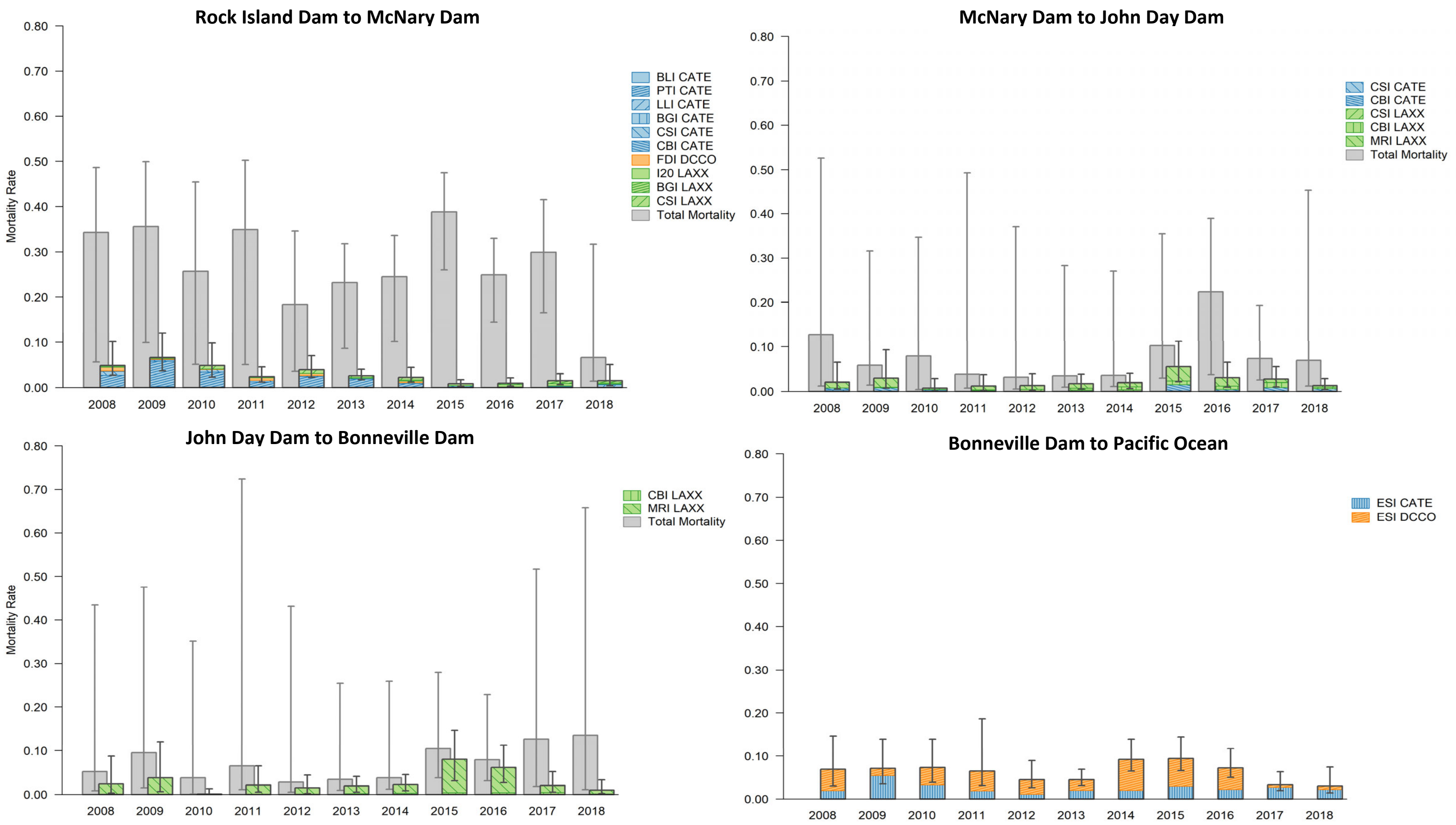

Figure 7.S3. Estimates annual reach-specific predation (colored bars) and total mortality (grey bars) of Upper Columbia River yearling Chinook during passage from Rock Island Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, the FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods).

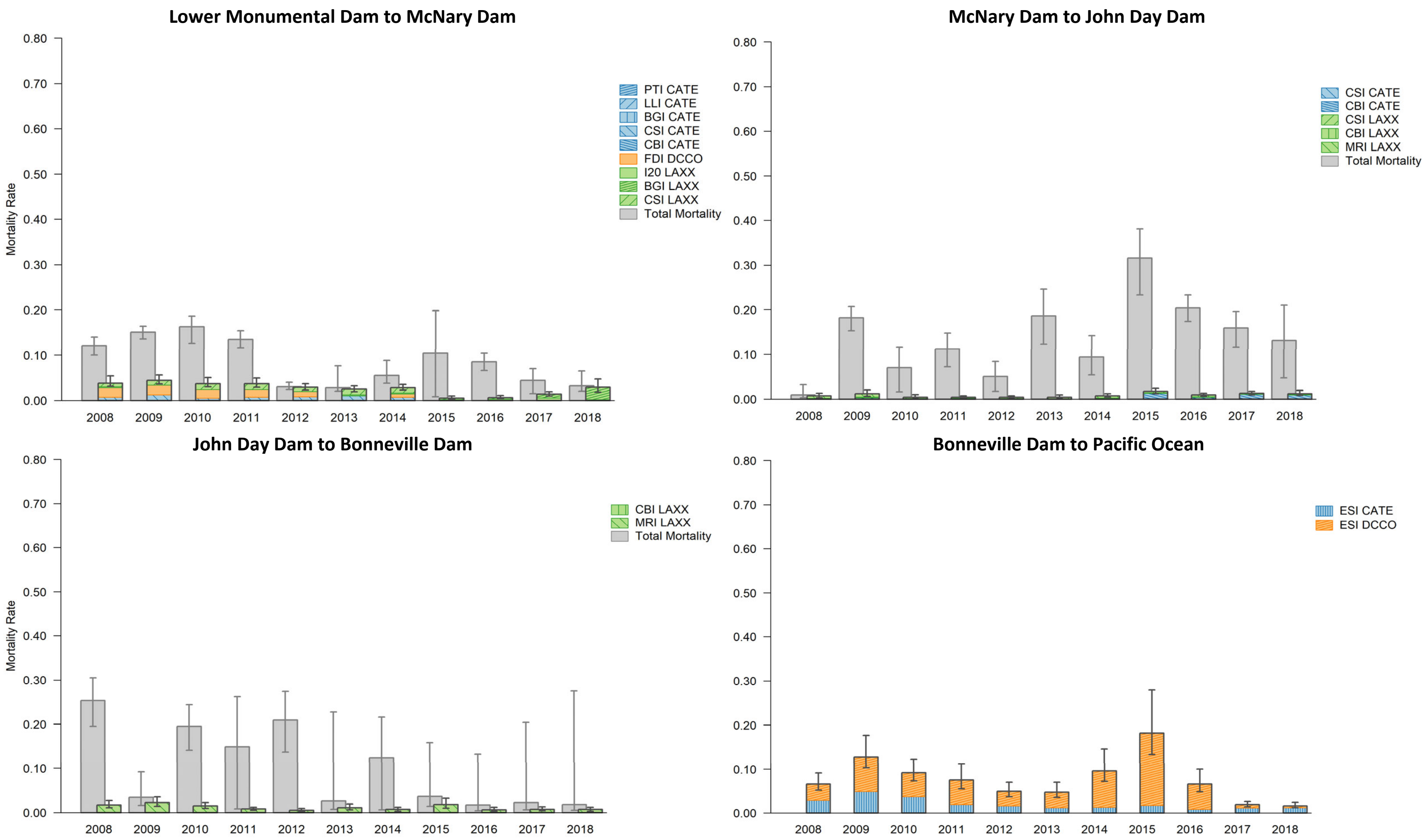

Figure 7.S4. Estimates annual reach-specific predation (colored bars) and total mortality (grey bars) of Snake River yearling Chinook during passage from Lower Monumental Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, the FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods).

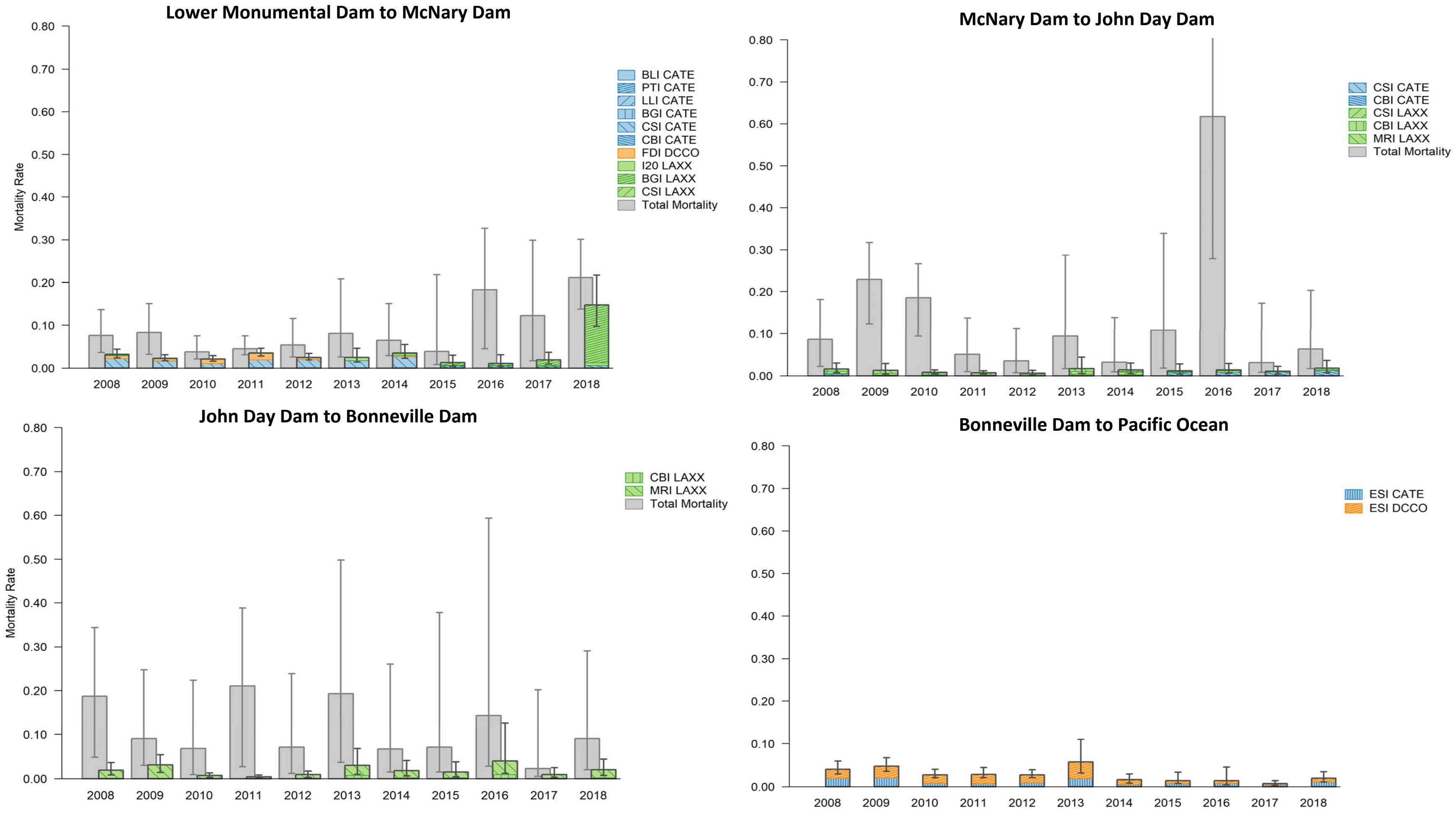

Figure 7.S5. Estimates annual reach-specific predation (colored bars) and total mortality (grey bars) of Snake River subyearling during passage from Lower Monumental Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, the FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods).

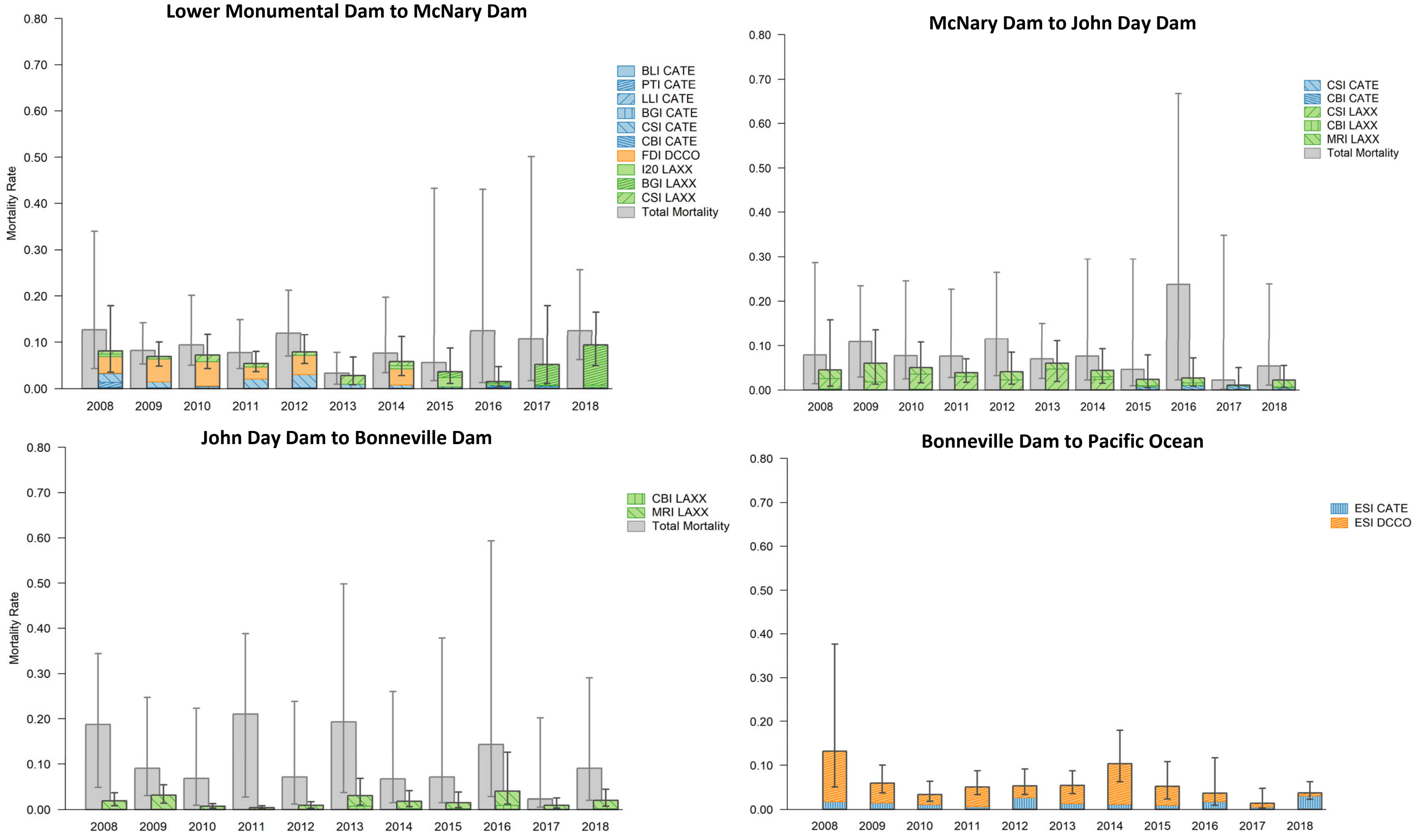

Figure 7.S6. Estimates annual reach-specific predation (colored bars) and total mortality (grey bars) of Snake River sockeye during passage from Lower Monumental Dam to the Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species included in this study include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. No predation estimates were available from I20 and CBI LAXX colonies during 2008–2011, the FDI DCCO during 2013 and 2015–2018. Predation estimates were minimum estimates for the ESI DCCO colony during 2016–2018 (see Methods).

CHAPTER 8: Additive Effects of Avian Predation on the Survival of Juvenile Salmonids in the Columbia River Basin

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8.1 Summary

We investigated the relationship between predation by Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), California gulls (*Larus californicus*), and ring-billed gulls (*L. delawarensis*) and the survival of anadromous salmonids (*Oncorhynchus* spp.) from the Upper Columbia River (UCR) and Snake River (SR) basins over an 11-year study period (2008–2018). The dataset included juvenile salmonids (smolts) from six different Endangered Species Act (ESA) listed populations (SR and UCR steelhead trout [*O. mykiss*], SR yearling and sub-yearling Chinook salmon [*O. tshawytscha*], UCR yearling Chinook, and SR sockeye salmon [*O. nerka*]) that were marked with passive integrated transponder (PIT) tags and exposed to varying levels of avian predation during out-migration to the Pacific Ocean. We used a hierarchical, multinomial state-space model to jointly estimate weekly and annual predation and survival probabilities (proportion of available fish) among time-stratified cohorts to explicitly measure the strength, magnitude, and direction of the relationship between predation and survival. Predation probabilities were estimated in river reaches where smolts were consumed by birds, while survival probabilities were estimated both within these same reaches and through ocean residency following exposure to avian predation as smolts. Spatial-scales included (1) smolt out-migration from Rock Island Dam on the middle Columbia River (for UCR smolts) or Lower Monumental Dam on the lower Snake River (for SR smolts) to Bonneville Dam on the lower Columbia River and (2) smolt out-migration from Bonneville Dam to their return as adults to Bonneville Dam following ocean residency.

Results indicated that increases in Caspian tern predation probabilities on SR and UCR steelhead smolts upstream of Bonneville Dam were associated with statistically significant decreases in survival probabilities to Bonneville Dam. Results suggest that a greater proportion of SR and UCR steelhead smolts would have survived out-migration to Bonneville Dam in the absence of tern predation upstream of Bonneville Dam, with the estimated average annual difference in observed survival versus baseline survival (i.e. survival in the absence of predation) of 0.049 (95% credible interval = 0.027–0.072) for SR steelhead and 0.196 (0.139–0.257) for UCR steelhead. Results also indicated that predation on SR yearling Chinook by terns and on SR steelhead by double-crested cormorants was an additive source of smolt mortality upstream of Bonneville Dam, with increases in predation probabilities associated with statistically significant decreases in survival probabilities. Due to relatively low levels of predation by terns on SR yearling Chinook upstream of Bonneville Dam, however, differences in observed survival versus baseline survival were low at 0.011 (-0.007–0.027), suggesting that only small increases in Chinook smolt survival to Bonneville Dam were possible in the absence of tern predation upstream of Bonneville Dam. In some years, there was evidence of a relationship between consumption probabilities by California and ring-billed gulls and survival probabilities for SR steelhead and SR yearling Chinook smolts to Bonneville Dam, but results were not statistically significant when considered across all years. The statistical power to accurately determine to what degree gull predation may have influenced smolt survival was limited by a truncated time series of available data (2013-2018), coupled with high levels of

uncertainty in estimates of predation probabilities. Gulls are also known to consume dead fish and to kleptoparasitize (steal) fish from other piscivorous waterbirds (such as Caspian terns), so it is likely that gull consumption of smolts was a more compensatory source of mortality compared to predation by terns and cormorants.

There was evidence that higher levels of Caspian tern predation on smolts in the Columbia River estuary were associated with lower returns of adults to Bonneville Dam for both SR and UCR steelhead populations; increases in tern predation probabilities were associated with statistically significant decreases in adult survival probabilities. Results suggested that for every 100 SR and UCR steelhead smolts consumed by terns in the estuary, on average, 6 (3–9) and 14 (6–23) fewer adults, respectively, returned to Bonneville Dam. Results provide evidence that tern predation on SR and UCR steelhead smolts was a partially additive source of mortality to the adult life-stage, despite predation occurring only during the smolt life-stage. There was some evidence that higher levels of double-crested cormorant predation on smolts in the estuary were also associated with lower adult returns to Bonneville Dam for the UCR steelhead population, however, results were not statistically significant when considered across all years. There was no evidence of a statistically significant relationship between predation probabilities by Caspian terns or double-crested cormorants in the estuary and adult returns to Bonneville Dam for salmon populations (evolutionarily significant units or ESUs) from either the SR or UCR basins.

The ability to accurately assess the relationship between predation probabilities and survival probabilities depended on several factors, including the level or magnitude of predation, intra-annual variation in estimates of predation, the salmonid life-stage (smolt or smolt-to-adult) evaluated, and sufficient sample sizes of tagged smolts within a population to generate precise estimates of predation and survival probabilities. Accounting for annual variation in predation and survival probabilities and the use of weekly cohorts within years as replicates were also critical components of the modelling framework. In the case of the four salmon ESUs evaluated (SR yearling Chinook, SR sub-yearling Chinook, UCR yearling Chinook, SR sockeye), the dataset often lacked the requisite components to accurately determine to what degree avian predation was associated with survival. Larger sample sizes of tagged smolts for these ESUs and efforts to scan all active tern, cormorant, and gull colonies for smolt PIT tags in all years, would likely generate a more robust, long-term datasets for future studies. In cases where predation impacts were low and where baseline mortality was high, larger sample sizes of tagged fish were still insufficient to identify the relationship between predation and survival probabilities in several salmon ESUs. However, in these circumstances, where predation impacts were low and baseline mortality was high, reductions in avian predation presumably would not dramatically increase fish survival. Despite these caveats, results of this study provide evidence that predation by Caspian terns and double-crested cormorants limited the survival of several salmonid populations from the Columbia River basin. Our results also indicate that efforts to reduce predation on smolts by Caspian terns and double-crested cormorants would increase the number of smolts from some salmonid populations that survive out-migration to Bonneville Dam. More important from a conservation perspective, our results also indicate that reductions in avian predation on smolts from some salmonid populations would increase the number that survive to adulthood, a key finding for those concerned with the restoration of ESA-listed salmonids in the Columbia River basin.

8.2 Introduction

Predator-prey dynamics are fundamental to ecological processes (Holling 1959), yet the full impact of predation on prey populations is often poorly understood (Serrouya et al. 2015). Predator-prey dynamics are also highly relevant to restoration actions for prey of conservation concern (Sedinger et al. 2010, Sandercock et al. 2011). Assessing the effects of predation on prey populations generally requires information about (1) the number or proportion of available prey consumed, (2) which individuals or life-stages are targeted by which predators, and (3) the levels at which other mortality sources or vital rates may compensate for predation (Caswell 2001, Mills 2012). Quantifying varying levels of predation and its association with prey population dynamics may also provide important information regarding the efficacy of predator management actions to protect endangered species as well as to develop and adapt management strategies for both predator and prey populations.

Previous research has identified predation by Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), California gulls (*Larus californicus*), and ring-billed gulls (*L. delawarensis*) as a substantial mortality factor for some juvenile salmonids (smolts; *Oncorhynchus* spp.) in the Columbia River basin (CRB) that are listed under the U.S. Endangered Species Act (ESA; Collis et al. 2002, Roby et al. 2003, Ryan et al. 2003, Evans et al. 2012, Evans et al. 2019). For instance, Caspian tern predation rates on ESA-listed steelhead trout (*O. mykiss*) during out-migration to the Pacific Ocean in excess of 0.20, or 20% of available smolts, have been documented by numerous studies (Evans et al. 2016, Evans et al. 2019, Payton et al. 2019; see also *Chapter 7*). Identifying high predation rates does not necessarily indicate that predators are limiting prey populations, however, as increases in mortality due to predation may be compensated for by decreases in other mortality factors, or predators may selectively consume individuals with relatively low reproductive values. Therefore, determining whether reductions in avian predation result in higher rates of smolt survival (i.e., predation is an additive source of mortality) or if smolts consumed by birds are destined to die regardless of the level of predation (i.e., predation is compensated for by other mortality sources) may be paramount to understanding the potential efficacy of management aimed at reducing avian predation on salmonid smolts (see also *Chapters 1, 2, and 4*). To date, the extent to which avian predation affects smolt and smolt-to-adult survival in the Columbia River basin (CRB) is largely unknown (Haeseker et al. 2020, Payton et al. 2020).

Recent advances in integrated mark-recapture-recovery models provide novel approaches to investigate predator-prey dynamics, compensation, and the impacts of predation on prey populations (Schaub and Pradel 2004, Sandercock et al. 2011, Peron 2013, Evans et al. 2016, Payton et al. 2020). In a study of the relationship between Caspian tern predation rates on ESA-listed Upper Columbia River (UCR) steelhead smolts and steelhead survival probabilities, Payton et al. (2020) observed that increases in tern predation probabilities were associated with statistically significant decreases in steelhead survival probabilities. Significant relationships between predation and survival were measured across large spatial scales and salmonid life-stages (smolt, smolt-to-adult) and indicated that tern predation was largely an additive source of smolt mortality during the smolt life-stage and a partially additive source of mortality to the adult life-stage. In the case of UCR steelhead, predation by colonial waterbirds was the dominant source of smolt mortality during out-migration; smolt mortality from tern predation was greater than all other mortality sources combined in some river reaches and years (Evans et al. 2019, Payton et at. 2020). Not all salmonid species and populations, however, are subject to such high levels of avian predation during seaward migration. For instance, avian predation rates on Chinook salmon (*O. tshawytscha*) smolts by Caspian terns were less than 0.05, or less than 5% of available smolts (Evans et al. 2016; see also *Chapter 7*). Given highly variable predation rates across salmonid species, it is likely that the degree to which avian predation limits salmonid survival differs by salmonid (prey) and avian (predator) species (Evans et al. 2019).

Due to the uncertainty in estimates of both mortality and survival probabilities, a common method of assessing the relationship between a cause-specific mortality source and survival involves the measurement of correlations (Otis and White 2004, Haeseker et al. 2020). Using correlations to assess whether a mortality source is completely compensatory, however, can be problematic since a lack of data is statistically indistinguishable from a lack of correlation in the data (Wagenmakers et al. 2016). Consequently, the conclusions that can be drawn from the usual correlational analyses of mortality and survival are limited to conclusions of whether there is evidence of a relationship between variables. Given a lack of evidence, we can conclude nothing more as absence of evidence is not evidence of absence or, as noted by Fisher (1925), one should never accept the null hypothesis that variables are unrelated. Furthermore, even with sufficient evidence of a relationship, correlations only allow for relative measurements of the level or strength of evidence that a relationship exists but say nothing of the magnitude of the relationship. Arguably, in terms of wildlife ecology, it is of nominal use to say there is evidence of a relationship without the ability to make further inference, such as to what degree survival has decreased given some specified increase in cause-specific mortality. Methods that directly measure the linear relationship between mortality and survival and that quantify the magnitude of the relationship between mortality and survival are needed to alleviate many, although not all, of the concerns associated with correlational analyses (Payton et al. 2020).
As part this study, we used the recently published methods of Payton et al. (2020) to investigate to what degree variation in predation probabilities by Caspian terns, double-crested cormorants, and California and ring-billed gulls were associated with variation in survival probabilities of different salmonid species and populations. The dataset included six different ESA-listed populations of passive integrated transponder (PIT) tagged juvenile salmonids from the Upper Columbia River (UCR) and Snake River (SR) basins: (1) UCR steelhead, (2) UCR yearling Chinook salmon, (3) SR steelhead, (4) SR yearling Chinook salmon, (5) SR sub-yearling Chinook salmon, and (6) SR sockeye salmon (*O. nerka*). The relationship between predation and survival was evaluated at two spatial scales representing two salmonid life-history stages

(smolt, smolt-to-adult) across multiple years. Results represent the most comprehensive, system-wide assessment of the effects of avian predation on salmonid survival in the CRB conducted to-date.

8.3 Methods

8.3.1 Mark-Recapture-Recovery

The same mark-recapture-recovery field methods described in Chapter 7 were used to model avian predation and fish survival probabilities in this chapter. In brief, we used an 11-year dataset (2008–2018) of uniquely marked (PIT-tagged) juvenile UCR and SR smolts and subsequent recaptures (detections of live fish) and recoveries (detections of dead fish) to evaluate survival, avian predation, and the relationship between these two processes. Each spring (March–July), PIT-tagged smolts were detected passing Rock Island Dam (RIS) on the middle Columbia River (river km [Rkm] 729, as defined by distance to the Pacific Ocean) or Lower Monumental Dam (LMN) on the lower Snake River (Rkm 589) during their out-migration to the Pacific Ocean (*Figure 8.1*). Following passage at RIS or LMN some proportion of tagged smolts were then detected at Bonneville Dam (BON; Rkm 234), at a vessel-towed pair-trawl net detection system downstream of BON (Rkm 75), and/or as returning adults at BON following ocean residency.

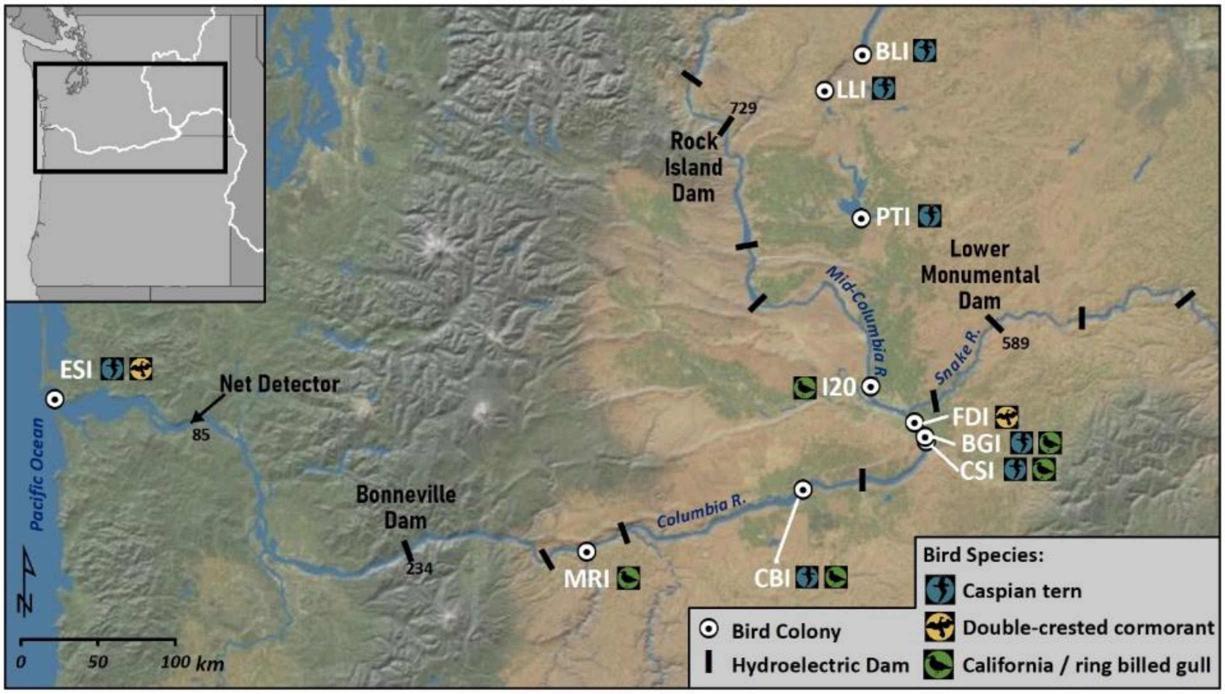

Figure 8.1. Mark-recapture-recovery locations of PIT-tagged smolts initially detected passing Rock Island Dam and Lower Monumental Dam. Downstream recapture locations include Bonneville Dam and a towed net detection system in the Columbia River estuary (Net Detector). Recovery locations include bird colonies on Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks (MRI), and East Sand Island (ESI). Avian species included in this study were Caspian terns, double-crested cormorants, and California and ring-billed gulls. Numbers represent the distance in river km (Rkm) from the Pacific Ocean.

During August-October, after the nesting season of piscivorous colonial waterbirds, PIT tags from depredated smolts were recovered from up to seven different breeding colonies of Caspian terns (hereafter "terns"), five different colonies of California and ring-billed gulls (hereafter "gulls"), and two colonies of double-crested cormorants (hereafter "cormorants"; *Figure 8.1* above). Recoveries of PIT tags of depredated fish on bird colonies are the result of two stochastic processes (Hostetter et al. 2015). First, only a portion of the PIT tags consumed are subsequently deposited on a bird's breeding colony (referred to as deposition probability). Second, only some of the deposited tags are later detected by researchers after the breeding season (referred to as detection probability). Independent annual probabilities of tag deposition for each predator species (tern, cormorant, gull) included in this study were informed by previous research (Hostetter et al. 2015), incorporated here through informative Beta priors, and assumed constant within each year. Probabilities of tag detection following the breeding season varied within each year, with tags deposited earlier in a breeding season less likely to be retrieved than those deposited later. For each bird colony in each year, parameters defining unique logistic detection probability functions were estimated by independent studies based on the intentional sowing of PIT tags identical to those placed in juvenile salmonids on each colony prior to, during (when possible), and after the breeding season (Evans et al. 2012, Evans et al. 2019, Hostetter et al. 2015; see also *Chapter 7* and *Appendix B, Table B17*).

Not all active breeding colony sites were scanned for smolt PIT tags in all years. As such, the time series available for analyses was truncated for some predator species. Specifically, gull colonies on Island 20 and in the Blalock Islands were active but not scanned for smolt PIT tags during 2008-2012, precluding estimates of gull predation in those years. The Foundation Island cormorant colony was also active but not scanned for smolt PIT tags during 2013 and 2015–2018, precluding estimation of predation in those years. Many, if not most, of the cormorants nesting on East Sand Island in the Columbia River estuary dispersed to alternative colony sites higher in the Columbia River estuary that were not accessible for PIT tag recovery (e.g., the Astoria-Megler Bridge) during 2016–2018, preventing accurate estimates of predation by cormorants in the estuary in those years. Overall, tern predation probabilities were available during 2008–2018, gull predation probabilities were available during 2013–2018, and cormorant predation probabilities were available during 2008–2015, depending on the colony.

8.3.2 Relationship Between Predation & Survival

The methods of Payton et al. (2019, 2020) were used to jointly estimate weekly predation and survival probabilities and to measure the strength, direction, and magnitude of the association between them. Estimates of predation were specific to each predator species or species group (in the case of gulls; hereafter "predator species") within each river reach (see also *Chapter 7)*. For each reach j, estimates of weekly smolt survival probabilities (ϕ_j) and weekly predation probabilities (Θ_{jh}) were evaluated, where h indicates the mortality source or hazard. Of the 14 different piscivorous waterbird colonies included in the study, seven were those of Caspian terns ({TERNS}; $\Theta_j^{\{\mathrm{TERNS}\}} = \sum_{h\in\{\mathrm{TERNS}\}} \Theta_{jh}$), five were those of California and ring-billed gulls ({GULLS}, $\Theta_j^{\{GULLS\}} = \sum_{h\in\{GULLS\}} \Theta_{jh}$), and two were those of double-crested cormorants ({CORMS}, $\Theta_j^{\{CORMS\}} = \sum_{h\in\{CORMS\}} \Theta_{jh}$). A final category, referred to as "unexplained mortality," was defined as the unaccounted-for remainder of the unit interval partition,

$$(\Theta_j^{unexplained} = 1 - \phi_j - \textstyle\sum_{h\in\{\{\mathrm{TERNS}\}\cup\{GULLS\}\cup\{CORMS\}\}} \Theta_{jh}).$$

To evaluate the aggregate effects of predation (predation by birds of the same species from multiple colonies), spatial scales included two river reaches were predation occurred and where survival following exposure to predators could be evaluated. The river reaches and salmonid life stages considered were (1) smolt out-migration through the hydrosystem, bounded by Rock Island Dam (RIS) for UCR smolts or Lower Monumental Dam (LMN) for SR smolts to Bonneville Dam (BON) for both UCR and SR smolts and (2) post-hydrosystem survival, bounded by BON for out-migrating SR and UCR smolts back to BON as adults following ocean residency. For the smolt hydrosystem out-migration spatial scale, all recoveries from bird colonies upstream of BON were used to inform predation probabilities and all smolt recaptures at BON and at the Net Detector in the estuary were used to inform smolt survival to BON. For the post-hydrosystem smolt-to-adult survival spatial scale, all recoveries of tags on colonies downstream of BON were used to inform predation probabilities and recaptures of returning adults to BON were used to estimate survival to BON. Recapture probabilities of the returning adults at BON fishways were assumed to be 1.0 (Keefer et al. 2008, Evans et al. 2014).

Simultaneous estimation of survival and predation within the same model allows for the measurement of functional relationships between these processes (e.g., $\phi = f(\boldsymbol{\theta})$). The mathematical expression of the classic compensatory mortality model was described by Anderson and Burnham (1976) as $\phi = \phi^0 - a\theta$, where ϕ denotes the observed probability of survival, ϕ^0 represents baseline survival probability (the expected probability of survival in the absence of predation), θ denotes the predation probability (replacing anthropogenic harvest, K, used by Anderson and Burnham, [1976]), and a defines the constant linear relationship between predation and survival. Direct measurements of the population-level impact on prey survival that can be attributed to the predator are measured with ϕ^Δ. This impact is defined as the difference between estimated baseline survival probabilities (ϕ^0; expected survival probabilities in the absence of predation) and the average annual observed probabilities of survival (measured in the presence of predation, i.e., $\phi^\Delta = \phi^0 - \phi$). For example, when considering predation impacts by Caspian terns that consume steelhead smolts upstream of BON, ϕ^Δ = 0.05 implies that, in the absence of tern predation, steelhead smolt survival to BON may have been 0.05 higher (i.e., represents a proportional increase in fish survival in the absence of predation).

Several levels of temporal correlation were explicitly recognized in our iteration of this model. The yearly number of smolts were apportioned into weekly cohorts for which we developed estimates of year (y)- and week (w)-specific survival probabilities ($\phi_{y,w}$), predation probabilities from the focal predator species group ($\Theta_{y,w}^{\{PRED\}}$), measured probabilities of predation by other predator species ($\Theta_{y,w}^{\{OTHER\}}$), and probabilities of dying due to an unexplained source of mortality ($\Theta_{y,w}^{unexplained}$). Weekly cohorts were not assumed to be mutually independent; instead, the temporal correlation among probabilities of predation, other mortality, and recapture were addressed through logistic random-walk models (Payton et al. 2020). The assumption of a single, constant probability of baseline survival, common to compensatory mortality models, was relaxed in two ways. First, by incorporating weekly variation in predation probabilities and survival probabilities the model was able to account for annual differences in the relationship between predation and survival. To this end, annual probabilities of baseline survival, ϕ_y^0, were assumed to be mutually independent. Second, the assumption of a constant baseline survival probability within each year was relaxed by incorporating a weekly random effect,

$$\operatorname{logit}(\phi_{y,w}^0) = \operatorname{logit}(\phi_y^0) + \epsilon_{y,w}$$

$$\text{where } \epsilon_{y,w} \sim \operatorname{normal}(0, \sigma_\phi^2)\ \forall y, w.$$

In assessing the relationship between smolt survival and avian predation, Payton et al. (2020) found some evidence that the additive rates of predation were different among years. As such, annual values of additivity were assumed to be unique but similar among years,

$$a_y \sim \mathrm{normal}(\mathrm{\mu_a}, \sigma_a^2).$$

To complete the basis of this framework, predation due to other waterbird species ($\Theta_{y,w}^{\{OTHER\}}$) represented a subset of all baseline mortality and thus provided a minimum estimate of "other mortality," analogous to non-harvest mortality or "natural" mortality in other studies of predation (Anderson and Burnham 1976, Sandercock et al. 2011). This compensatory model is expressed by defining weekly survival as a piecewise function,

$$\phi_{y,w} = \begin{cases} \mathrm{\phi_{y,w}^0} - a_y \Theta_{y,w}^{\{\mathrm{PRED}\}}, & \text{when } \Theta_{y,w}^{\{\mathrm{PRED}\}} \le \theta_{y,w}^{saturation} \\ 1 - \Theta_{y,w}^{\{OTHER\}} - \Theta_{y,w}^{\{\mathrm{PRED}\}}, & \text{when } \Theta_{y,w}^{\{\mathrm{PRED}\}} > \theta_{y,w}^{saturation} \text{ and } a < \phi_{y,w}^0 \\ 0, & \text{when } \Theta_{y,w}^{\{\mathrm{PRED}\}} > \theta_{y,w}^{saturation} \text{ and } a \ge \phi_{y,w}^0 \end{cases}.$$

Here, the first sub-function is the classic compensatory mortality model described by Anderson and Burnham (1976), while the second and third sub-functions enforce the boundary constraints whereby survival, avian predation, and the measured subset of other mortality must sum to 1.0. The delineations of these parameter space boundaries are defined by

$$\theta^{saturation} = \begin{cases} \dfrac{1-\phi^0}{1-a}, & \text{when } a < \phi^0 \\ \dfrac{\phi^0}{a}, & \text{when } a \ge \phi^0 \end{cases}.$$

This analysis involved the consideration of multiple predators that simultaneously impacted the same cohorts of fish across overlapping foraging ranges. To develop the most precise and unbiased evaluation of impacts from each predator species or species group, one must also consider the impacts of birds from other colonies foraging in the same river reach, given the requisite data were available. Therefore, we considered all predators simultaneously when evaluating the additive impacts of a "focal" predator. When a complete predation dataset from another predator species group was available, the predation from this group was considered "nuisance" predation and we considered the additive effects of both predator groups simultaneously. Thus, we had three categories of measured predation: $\Theta_{y,w}^{\{\mathrm{focal}\}}$, $\Theta_{y,w}^{\{\mathrm{nuisance}\}}$ and $\Theta_{y,w}^{\{OTHER\}}$. Survival across the spatial scale of reference was modelled as,

$$\phi_{y,w} = \begin{cases} \mathrm{\phi_{y,w}^0} - a_y \Theta_{y,w}^{\{\mathrm{focal}\}} - a_y^{\{n\}} \Theta_{y,w}^{\{\mathrm{nuisance}\}}, & \text{when } \Theta_{y,w}^{\{\mathrm{focal}\}} \le \theta_{y,w}^{saturation} \\ 1 - \Theta_{y,w}^{\{OTHER\}} - \Theta_{y,w}^{\{\mathrm{nuisance}\}} - \Theta_{y,w}^{\mathrm{focal}}, & \text{when } \Theta_{y,w}^{\{\mathrm{focal}\}} > \theta_{y,w}^{saturation} \text{ and } a < \phi_{y,w}^0 \\ 0, & \text{when } \Theta_{y,w}^{\{\mathrm{focal}\}} > \theta_{y,w}^{saturation} \text{ and } a \ge \phi_{y,w}^0 \end{cases}.$$

Prior distributions for all probabilities were defined to be either uniform for univariate parameters (i.e., baseline survival, recapture) or Dirichlet(**1**) for multivariate parameters (i.e., survival and mortality). We assigned $\mu_a \sim \mathrm{normal}(\frac{1}{2}, 3)$ as the enumeration of a vague *a priori* assumption that avian predation was likely equal parts additive and compensatory mortality, with less prior credibility given to hypotheses of over-compensatory or super-additive mortality mechanisms. As with the random errors in the joint mortality and survival model, we assumed a weakly-informative prior of $\mathrm{half\ normal}(0, 5)$ for both σ_ϕ^2 and σ_a^2.

Models were analyzed using the software STAN (Stan Development Team 2020), accessed through R version 3.6.2 (R Development Core Team 2014), and using the rstan package (version 2.19.3; Stan Development Team 2020). To simulate random draws from the joint posterior distribution, we ran four Hamiltonian Monte Carlo (HMC) Markov Chain processes. Each chain contained 4,000 warm-up iterations followed by 4,000 posterior iterations thinned by a factor of 4. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013); only chains with zero reported divergent transitions were accepted. Posterior predictive checks compared simulated and observed annual aggregate raw recapture and recovery numbers to ensure model estimates reflected the observed data. Reported estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% Credible Interval [CRI]) calculated using the HDInterval package (version 0.2.0; Meredith and Kruschke 2016).

8.3.2.1 Model assumptions – In addition to the standard assumptions of the Joint Mortality and Survival model (Payton et al. 2019, see also *Chapter 7*), the accuracy and precision of these analyses depended, in part, on the validity of the following assumptions:

A1. Within each year there was a baseline mortality rate with independent weekly variation and the magnitude of inter-week variation was similar among years.

A2. The additive rate of avian predation was defined as the proportionate reduction in fish survival associated with increasing levels of consumption by a predator species.

A3. Linear relationships between survival rates and predation rates accurately describe the mechanistic relationships between avian predation and fish survival within each year.

To confirm assumption A1, we used a Bayesian equivalent of the Durbin-Watson test to identify serial correlation among weeks and found little evidence of correlation. We additionally found little credibility that the magnitude of weekly variation about the baseline mortality was different among years. A2 relates to the facts that data from the same groups of fish inform both survival and predation rates and that a depredated fish cannot survive, so the hypothesized relationship between predation and survival is necessarily linear. These considerations motivate the construction of the above model rather than a simple linear regression analysis, which would not allow for the direct measure of a strictly linear relationship and is not appropriate when there is uncertainty associated with measuring both the "explanatory" and the "dependent" variables. A3 is a necessary assumption that any additive effect of mortality is similar enough across a year that the direct relationship can be measured across weekly cohorts. A3 also concerns the lack of experimental manipulation and controls required to separate correlation and causation in observational studies, as it is debatable whether causation can be inferred from these types of analyses (Payton et al. 2020).

8.4 Results

8.4.1 Mark-Recapture-Recovery

Annual sample sizes of PIT-tagged smolts varied considerably by location, salmonid species, age-class (yearling, sub-yearling for Chinook), and year and are provided in *Section 8.8 Supplements* (*Table 8.S1* for annual sample sizes of smolts at RIS or LMN and *Table 8.S2* for annual sample sizes of smolts at BON). Average annual numbers of PIT-tagged smolts during the study period were 42,317 SR yearling Chinook, 30,036 SR steelhead, 9,297 SR subyearling Chinook, and 2,037 SR sockeye at LMN and 7,124 UCR steelhead and 3,156 UCR yearling Chinook at RIS. The number of weekly releases ranged from an average of 2 weeks for SR sockeye to 16 weeks for SR yearling Chinook. Average annual numbers of PIT-tagged smolts at BON were 24,991 SR yearling Chinook, 19,992 SR steelhead, 6,789 SR subyearling Chinook, 1,286 SR sockeye, 5,085 UCR steelhead, and 7,180 UCR yearling Chinook. The number of smolts detected at RIS or LMN and subsequently recaptured alive at BON and at the Net Detector downstream of BON also varied by salmonid species, age-class, and year, as did the number recovered dead on bird colonies (see *8.8 Supplements*, *Tables 8.S1* and *8.S2*). In general, the largest average annual number of recovered tags on bird colonies were those of SR steelhead (N = 2,599), SR yearling Chinook (N = 1,028), and UCR steelhead (N = 746; see *8.8 Supplements*, *Table 8.S1* for annual numbers). Only a small number and proportion of PIT-tagged fish last detected as smolts at BON returned to BON as adults, with average annual numbers ranging from 22 SR sockeye adults to 531 SR steelhead adults (see *8.8 Supplements, Table 8.S2* for annual numbers).

8.4.2 Relationship Between Predation & Survival

Colony- and reach-specific predation and survival probabilities for the salmonid species and populations included in this study are presented in detail in *Chapter 7*. Herein we summarize the strength, direction, and magnitude of the relationship between predator-specific predation probabilities and survival probabilities within the two spatial scales and salmonid life-stages considered (smolt passage through the hydrosystem to Bonneville Dam and smolt-to-adult returns to Bonneville Dam following smolt hydrosystem passage). Results are provided by salmonid population, species, and age-class (for Chinook salmon). *Tables 8.1–8.2* provide full output from all models. *Figures 8.2–8.12* are provided to visualize the relationship between weekly estimates of predation and survival for those relationships that were statistically significant.

8.4.2.1 Upper Columbia River steelhead trout – Average annual survival probability for UCR steelhead smolts from RIS to BON was 0.458 (95% credible interval = 0.434–0.482) during 2008–2018 (*Table 8.1*) and ranged annually from 0.308 (0.258–0.366) to 0.567 (0.470–0.665), indicating the majority of UCR steelhead died prior to reaching BON in most years. Average annual estimates of predation by Caspian terns from colonies that foraged upstream of BON was 0.14 (0.13–0.16; *Table 8.1*) and ranged annually from 0.05 (0.03–0.07) to 0.29 (0.21–0.38). Weekly estimates of tern predation probabilities were highly variable and ranged weekly from 0.02 (0.01–0.03) to 0.46 (0.24– 0.72; *Figure 8.2*). There was evidence that increases in tern predation on UCR steelhead smolts above BON were associated with statistically significant decreases in smolt survival to BON (*Table 8.1* and *Figure 8.2*). Results indicated that significantly more steelhead would likely have survived out-migration to BON in the absence of tern predation, with an estimated difference between observed survival and baseline survival (i.e., survival in the absence of tern predation) of 0.196 (0.139–0.257) across all years (*Table 8.1*). Results also suggest that tern predation was a super-additive source (**a** > 1.0) of steelhead smolt mortality (*Table 8.1*). In contrast to predation by Caspian terns, results indicated that predation by double-crested cormorants nesting on Foundation Island, the only cormorant colony upstream of BON that was included in the study, depredated a negligible proportion of available UCR steelhead smolts, with predation probabilities averaging less than 0.01 annually during 2008–2012 and 2014 (*Table 8.1*). There was no evidence of a statistically significant relationship between cormorant predation probabilities on UCR steelhead smolts and UCR steelhead survival probabilities to BON (*Table 8.1*). Similar to terns, gulls consumed a substantial proportion of UCR steelhead smolts that were released from RIS, with an average annual consumption probability of 0.21 (0.18–0.26) and ranging annually from 0.13 (0.10–0.18) to 0.31 (0.24–0.39) of available smolts during 2013–2018 (*Table 8.1*). Despite such high levels of consumption by gulls, however, there was no evidence of a statistically significant relationship between gull consumption of UCR steelhead smolts and smolt survival to BON (*Table 8.1*). Estimates of gull consumption of smolts ranged from super-additive in some years to over-compensatory (**a** < 0) in other years, with no consistent trend identified across all years. Weekly probabilities of smolt consumption by gulls were less variable and less precise (based on the width of 95% credible intervals) compared with those by terns (see also *Chapters 7* and *9*).

Table 8.1. Average annual predation probabilities by Caspian terns, double-crested cormorants, and California and ring-billed gulls and survival probabilities for Upper Columbia River (UCR) and Snake River (SR) steelhead (ST), yearling Chinook salmon (CH1), subyearling Chinook salmon (CH0), and sockeye salmon (SO) smolts during out-migration from Rock Island Dam (for UCR smolts) or Lower Monumental Dam (for SR smolts) to Bonneville Dam. Estimates of the magnitude of the association between predation probabilities on smolt survival ($\boldsymbol{a}$, additivity) and the difference in smolt survival probabilities from estimated baseline smolt survival probabilities ($\boldsymbol{\Phi}^{\Delta}$) are also provided. Statistically significant ($prob[a > 0] > 0.95$) relationships between predation and survival are in bold and denoted in red. Values are reported as medians with 95% credible intervals.

Predator (years)	Prey	Survival	Predation	a	Φ^{Δ}
Caspian terns (2008-2018)	UCR ST	0.458 (0.434, 0.482)	0.14 (0.13, 0.16)	**1.32 (0.86, 1.73)**	**0.196 (0.139, 0.257)**
	SR ST	0.619 (0.597, 0.643)	0.04 (0.04, 0.05)	**1.47 (1.05, 1.95)**	**0.049 (0.027, 0.072)**
	UCR CH1	0.571 (0.497, 0.63)	0.03 (0.02, 0.03)	0.80 (-0.84, 2.43)	0 (-0.038, 0.037)
	SR CH1	0.668 (0.627, 0.709)	0.01 (0.01, 0.01)	**1.48 (0.01, 2.95)**	**0.011 (-0.007, 0.027)**
	SR CH0	0.697 (0.600, 0.786)	0.02 (0.01, 0.02)	1.32 (-0.62, 3.21)	-0.001 (-0.043, 0.038)
	SR SO	0.603 (0.511, 0.677)	0.03 (0.02, 0.03)	0.44 (-1.66, 2.44)	0.009 (-0.046, 0.057)
Double-crested cormorants (2008-2012, 2014)	UCR ST	0.490 (0.458, 0.521)	< 0.01	0.41 (-1.71, 2.34)	0.001 (-0.005, 0.008)
	SR ST	0.620 (0.595, 0.643)	0.04 (0.03, 0.04)	**1.40 (0.07, 3.06)**	**0.047 (-0.005, 0.104)**
	UCR CH1	0.576 (0.492, 0.676)	< 0.01	0.63 (-1.43, 2.78)	0.002 (-0.007, 0.012)
	SR CH1	0.640 (0.608, 0.673)	0.02 (0.01, 0.02)	0.75 (-1.24, 2.89)	0.011 (-0.019, 0.043)
	SR CH0	0.689 (0.594, 0.775)	0.01 (0.01, 0.02)	0.87 (-1.23, 2.87)	0.009 (-0.013, 0.031)
	SR SO	0.698 (0.588, 0.783)	0.03 (0.02, 0.05)	-0.11 (-2.41, 2.03)	-0.003 (-0.067, 0.054)
Ring-billed and California gulls (2013-2018)	UCR ST	0.433 (0.397, 0.468)	0.21 (0.18, 0.26)	0.19 (-0.78, 1.06)	0.040 (-0.168, 0.205)
	SR ST	0.613 (0.581, 0.652)	0.13 (0.12, 0.15)	0.51 (-0.81, 1.57)	0.052 (-0.105, 0.179)
	UCR CH1	0.578 (0.497, 0.653)	0.05 (0.04, 0.05)	0.75 (-0.55, 2.34)	0.038 (-0.041, 0.107)
	SR CH1	0.691 (0.624, 0.760)	0.02 (0.02, 0.02)	0.52 (-1.34, 2.37)	0.010 (-0.025, 0.047)
	SR CH0	0.718 (0.563, 0.859)	0.05 (0.04, 0.07)	0.37 (-1.66, 2.35)	0.020 (-0.110, 0.128)
	SR SO	0.522 (0.409, 0.641)	0.13 (0.09, 0.16)	-0.73 (-2.43, 0.90)	-0.102 (-0.362, 0.121)

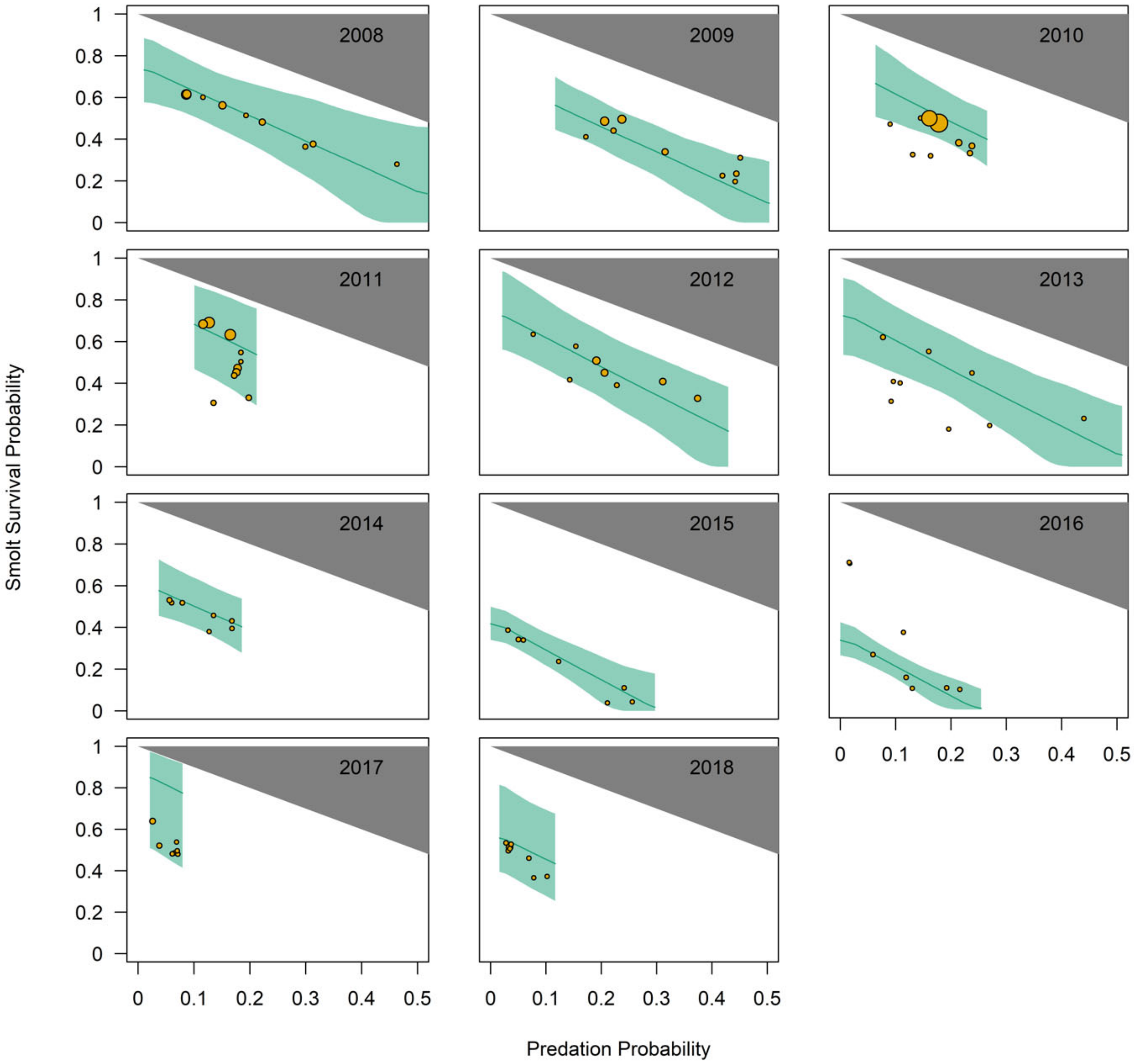

Figure 8.2. Weekly smolt survival probabilities for Upper Columbia River steelhead as a function of Caspian tern predation probabilities during smolt out-migration from Rock Island Dam to Bonneville Dam in each year from 2008 to 2018. The size of circles depicts the relative number of PIT-tagged smolts released at Rock Island Dam each week. Lines represent the best fit estimate of the linear relationship between predation and survival and shading denotes 95% credible intervals around the best fit. Estimates are those previously reported by Payton et al. (2020).

Smolt-to-adult return survival probabilities for UCR steelhead from BON (as smolts) to BON (as adults) averaged 0.028 (0.027–0.030) during 2008-2016 (years with complete adult returns) and ranged annually from 0.002 (0.001–0.004) to 0.084 (0.074–0.095), indicating that the vast majority of UCR steelhead smolts died prior to returning to BON as an adult. The average annual estimate of predation probability by terns nesting in the estuary (i.e., downstream of BON, following hydrosystem passage; *Figure 8.1* above) was 0.11 (0.10–0.13; *Table 8.2*) during 2008-2017 and ranged annually from 0.08 (0.06–0.11) to 0.19 (0.14–0.26). There was evidence that tern predation on UCR steelhead smolts was a partially additive source of mortality to the adult steelhead life-stage, with increases in tern predation probabilities on smolts associated with statistically significant decreases in adult returns to BON in most years evaluated (*Figure 8.3*). Estimated average additivity was 0.14 (0.06–0.23), suggesting that for every 100 UCR steelhead smolts consumed by nesting terns in the estuary, 14 (6–23) fewer returned to BON as adults. The estimated difference between observed survival and baseline survival was 0.019 (0.010–0.031) across all years, suggesting that significantly more SR steelhead smolts would likely have returned to BON as adults in the absence of tern predation in the estuary (*Table 8.2* and *Figure 8.8*). For instance, results suggest that in the absence of tern predation in the estuary, average annual smolt-to-adult return probabilities would likely have been 0.047 (0.037–0.61) instead of 0.028 (0.027–0.030). The average annual predation probability on UCR steelhead smolts by cormorants nesting on East Sand Island was 0.07 (0.06–0.08) of available fish during 2008-2015 (years with estimates of cormorant predation) and ranged annually from 0.04 (0.02–0.06) to 0.12 (0.07–0.18; *Table 8.1* above). There was some evidence that cormorant predation on UCR steelhead smolts was a partially additive source of mortality to the adult life stage, with increasing weekly cormorant predation probabilities on UCR steelhead smolts associated with decreasing adult returns to BON in some of the years evaluated. When data from all years were considered, however, there was not a statistically significant relationship to indicate cormorant predation was an additive source of mortality for UCR steelhead (*Table 8.2*).

All of the gull colonies included in the study were located upstream of BON, so the relationship between gull predation downstream of BON and smolt-to-adult survival to BON could not be evaluated using the spatial scales investigated herein.

Table 8.2. Average annual predation probabilities by Caspian terns and double-crested cormorants on smolts in the Columbia river estuary and smolt-to-adult survival probabilities for Upper Columbia River (UCR) and Snake River (SR) steelhead (ST), yearling Chinook salmon (CH1), subyearling Chinook salmon (CH0), and sockeye salmon (SO) during passage from Bonneville Dam (as smolts) back to Bonneville Dam (as adults). Estimates of the magnitude of the association between predation probabilities on smolts and smolt-to-adult survival ($\boldsymbol{a}$*, additivity) and the difference in smolt-to-adult survival probabilities from estimated baseline smolt-to-adult survival probabilities (*$\boldsymbol{\Phi}^{\Delta}$*) are also provided. Statistically significant (*$prob[a > 0] > 0.95$*) relationships between predation and survival are in bold and denoted in red. Values are reported as medians with 95% credible intervals.*

Predator (years)	Prey	Survival	Predation	a	Φ^{Δ}
Caspian terns (2008-2017)	UCR ST	0.028 (0.027, 0.030)	0.11 (0.10, 0.13)	**0.14 (0.06, 0.23)**	**0.019 (0.010, 0.031)**
	SR ST	0.026 (0.025, 0.027)	0.11 (0.10, 0.12)	**0.06 (0.03, 0.09)**	**0.007 (0.004, 0.010)**
	UCR CH1	0.019 (0.018, 0.021)	0.02 (0.02, 0.02)	0.04 (-0.30, 0.41)	0.001 (-0.006, 0.009)
	SR CH1	0.016 (0.016, 0.017)	0.02 (0.02, 0.02)	-0.14 (-0.42, 0.19)	-0.003 (-0.008, 0.004)
	SR CH0	0.020 (0.018, 0.022)	0.01 (0.01, 0.01)	-0.24 (-0.69, 0.22)	-0.003 (-0.007, 0.002)
	SR SO	0.020 (0.016, 0.024)	0.03 (0.02, 0.03)	0.85 (-0.09, 2.05)	0.017 (-0.001, 0.042)
Double-crested cormorants (2008-2015)	UCR ST	0.030 (0.028, 0.032)	0.07 (0.06, 0.08)	0.05 (-0.03, 0.13)	0.004 (-0.002, 0.010)
	SR ST	0.027 (0.026, 0.028)	0.08 (0.07, 0.09)	-0.03 (-0.07, 0.01)	-0.002 (-0.005, 0.001)
	UCR CH1	0.019 (0.018, 0.021)	0.04 (0.03, 0.04)	-0.04 (-0.17, 0.06)	-0.001 (-0.006, 0.003)
	SR CH1	0.016 (0.016, 0.017)	0.05 (0.05, 0.06)	-0.06 (-0.16, 0.01)	-0.003 (-0.008, 0.002)
	SR CH0	0.020 (0.018, 0.022)	0.02 (0.02, 0.02)	0.01 (-0.41, 0.50)	0 (-0.008, 0.009)
	SR SO	0.023 (0.019, 0.027)	0.04 (0.03, 0.05)	0.17 (-0.26, 0.75)	0.008 (-0.012, 0.034)

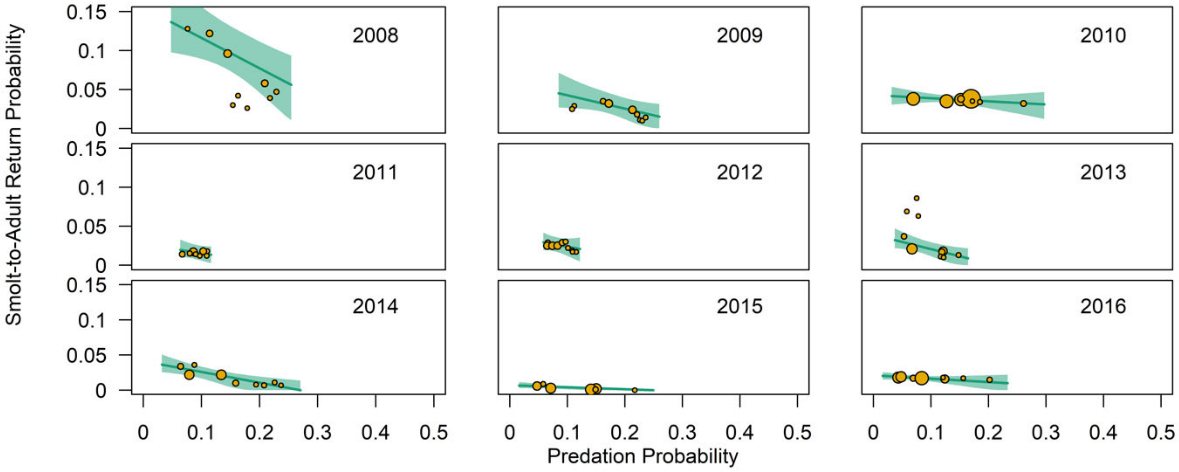

Figure 8.3. Weekly smolt-to-adult survival probabilities for Upper Columbia River steelhead as a function of Caspian tern predation probabilities during out-migration from Bonneville Dam as smolts to return to Bonneville Dam as adults in each year from 2008 to 2016. The size of circles depicts the relative number of PIT-tagged smolts detected at Bonneville Dam each week. Lines represent the estimate of the best linear fit to the data and shading denotes 95% credible intervals around the best fit.

8.4.2.2 Snake River steelhead trout – The average annual survival probability for SR steelhead smolts from LMN to BON was 0.619 (0.597–0.643) during 2008-2018 (*Table 8.1* above) and ranged annually from 0.541 (0.450–0.648) to 0.691 (0.617–0.766). Caspian terns from colonies that foraged upstream of BON consumed, on average, 0.04 (0.05–0.05) of available SR steelhead smolts (*Table 8.1* above). Like tern predation on UCR steelhead, there was a statistically significant relationship between tern predation probabilities on SR steelhead smolts and SR steelhead smolt survival probabilities to BON, with increases in tern predation associated with reductions in smolt survival (*Table 8.1* above and *Figure 8.5*). The estimated difference between observed survival and baseline survival of SR steelhead smolts was 0.049 (0.027–0.072) across all years*,* suggesting that more SR steelhead smolts would likely have survived out-migration to BON in the absence of tern predation upstream of BON (*Table 8.1* above). Cormorant predation probabilities on SR steelhead smolts averaged 0.04 (0.03–0.04) across all years (*Table 8.1* above) and, similar to tern predation, there was a statistically significant relationship between cormorant predation probabilities on SR steelhead and SR steelhead survival probabilities to BON (*Table 8.1* above and *Figure 8.6*). The difference between observed survival and baseline survival in the absence of cormorant predation was 0.047 (-0.005–0.104), suggesting that more SR steelhead smolts would have survived to BON in the absence of cormorant predation upstream of BON (*Table 8.1* above).

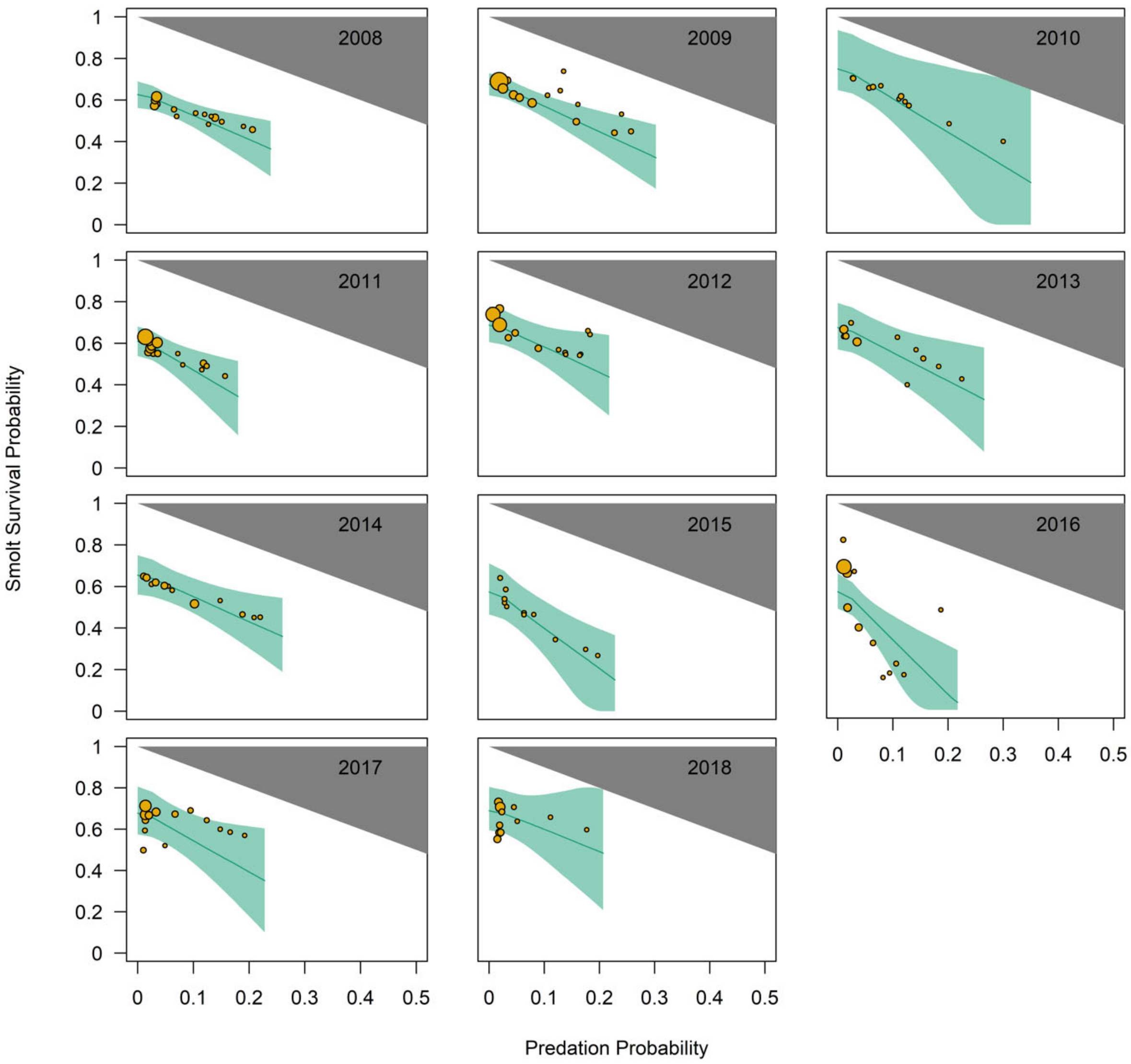

Figure 8.5. Weekly smolt survival probabilities for Snake River steelhead as a function of Caspian tern predation probabilities during smolt out-migration from Lower Monumental Dam to Bonneville Dam in each year from 2008 to 2018. The size of circles depicts the relative number of PIT-tagged smolts released at Lower Monumental Dam each week. Lines represent the best fit estimate of the linear relationship between predation and survival and shading denotes 95% credible intervals around the best fit.

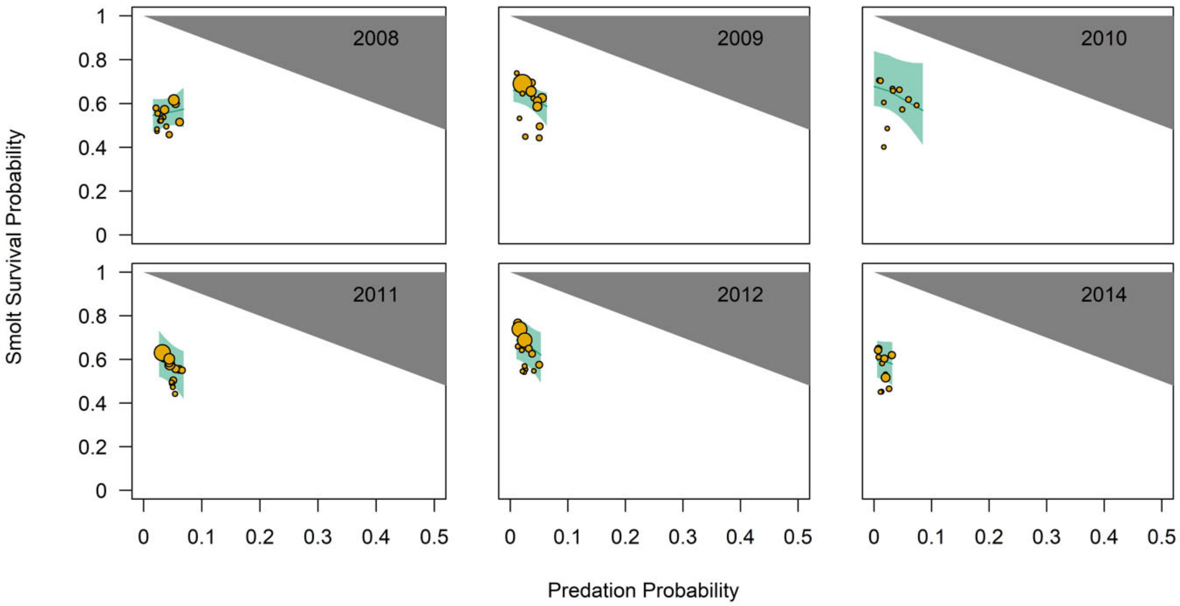

Figure 8.6. Weekly smolt survival probabilities for Snake River steelhead as a function of double-crested cormorant predation probabilities during smolt out-migration from Lower Monumental Dam to Bonneville Dam for each year from 2008 to 2012, plus 2014. The size of circles depicts the relative number of PIT-tagged smolts released at Lower Monumental Dam each week. Lines represent the best fit estimate of the linear relationship between predation and survival and shading denotes 95% credible intervals around the best fit.

Gull consumption probabilities of SR steelhead smolts were consistently the highest of the avian predator species evaluated, with an average annual consumption probability of 0.13 (0.12–0.15) during 2013-2018 (*Table 8.1* above) and ranged annually from 0.09 (0.07–0.12) to 0.19 (0.13–0.26). Despite such high levels of consumption, however, there was only one year (2017) of the six evaluated when there was a statistically significant relationship between increases in gull consumption probabilities of SR steelhead smolts and decreases in SR steelhead smolt survival probabilities to BON (*Figure 8.7*). Trends were suggestive in several other years (2013 and 2015), but when data from all years were considered there was not a statistically significant relationship that indicated gull consumption was an additive source of mortality for SR steelhead. Similar to gull consumption probabilities of UCR steelhead smolts, and unlike tern predation probabilities on both UCR and SR steelhead smolts, gull consumption probabilities of SR steelhead were less precise (based on the width of the 95% credible intervals) and often, but not always, less variable (e.g., in 2014 and 2016, years when there was no evidence of a relationship; *Figure 8.7*).

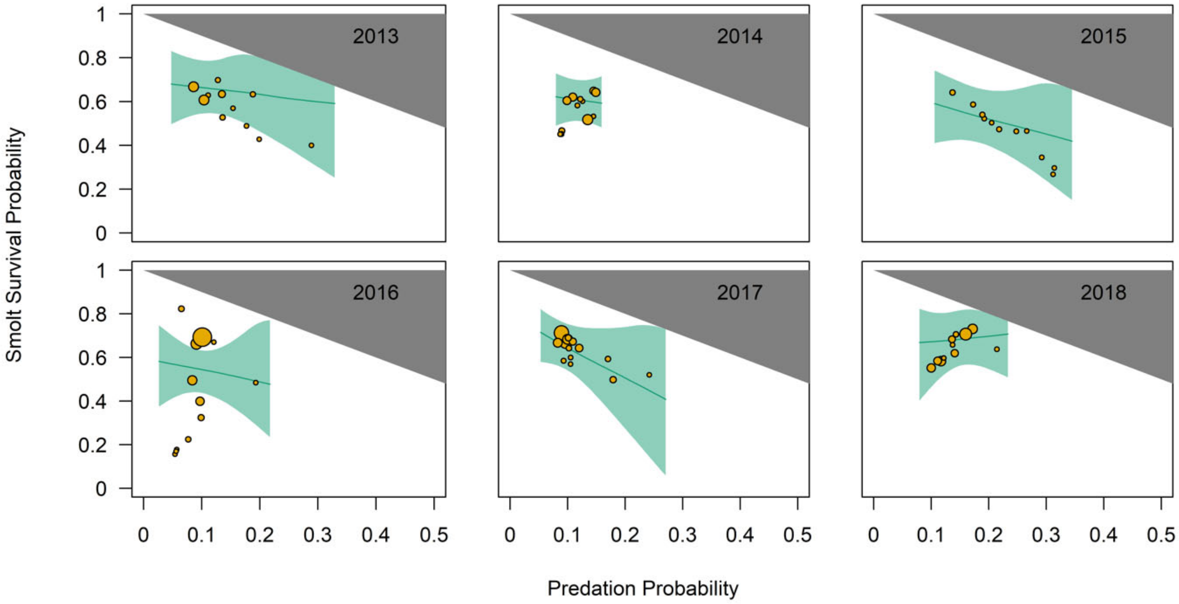

Figure 8.7. Weekly smolt survival probabilities for Snake River steelhead as a function of California and ring-billed gull consumption probabilities during smolt out-migration from Lower Monumental Dam to Bonneville Dam in each year from 2013 to 2018. The size of circles depicts the relative number of PIT- tagged smolts released at Lower Monumental Dam each week. Lines represent the best fit estimate of the linear relationship between predation and survival and shading denotes 95% credible intervals around the best fit.

Smolt-to-adult survival probabilities for SR steelhead from BON (as smolts) to BON (as adults) averaged 0.026 (0.025–0.027; *Table 8.2* above) across all years and ranged annually from 0.007 (0.005–0.008) to 0.061 (0.058–0.064). Estimates of tern predation probabilities on SR steelhead smolts in the estuary were 0.11 (0.10–0.13; *Table 8.2* above) and ranged annually from 0.06 (0.05–0.09) to 0.15 (0.11–0.21). Similar to results for UCR steelhead, there was evidence that tern predation on SR steelhead smolts in the estuary was a partially additive source of mortality to the adult life-stage. Average annual additive was 0.06 (0.03–0.09), suggesting that for every 100 SR steelhead smolts consumed by Caspian terns nesting in the estuary, on average, 6 (3–9) fewer returned to BON as adults. The estimated difference between observed survival and baseline survival was 0.007 (0.004–0.010) across all years, suggesting that significantly more SR steelhead smolts would likely have returned to BON as adults in the absence of tern predation in the estuary (*Table 8.2* above and *Figure 8.8*). Average annual estimates of cormorant predation on SR steelhead smolts in the estuary were significantly lower than those by terns on SR steelhead smolts in the estuary, but were still substantial, at 0.08 (0.07–0.09; *Table 8.1* above) and ranged annually from 0.03 (0.02–0.04) to 0.17 (0.12–0.25). There was no evidence that cormorant predation on SR steelhead smolts in the estuary was an additive source of mortality to the adult life-stage, with estimates ranging from partially additive to over-compensatory, depending on the year.

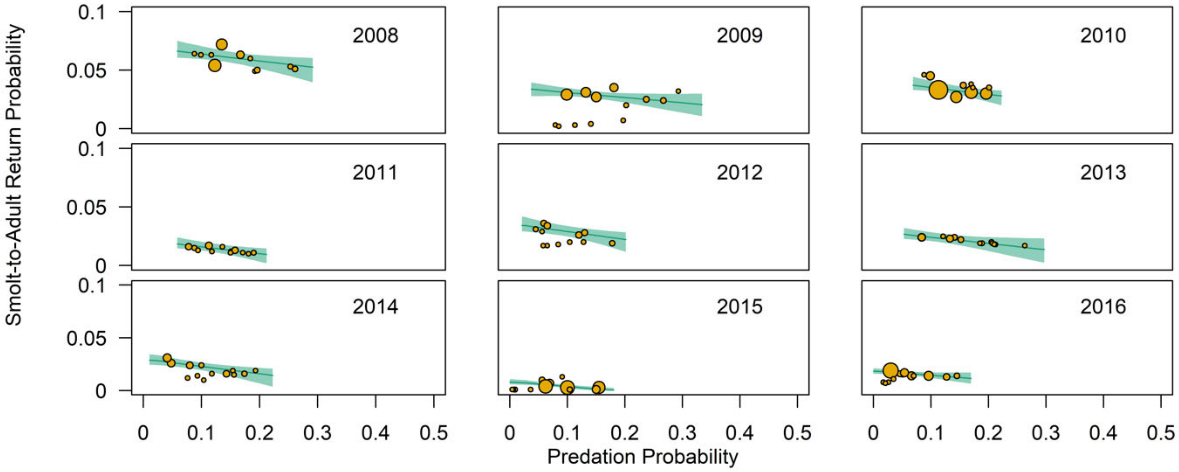

Figure 8.8. Weekly smolt-to-adult survival probabilities for Snake River steelhead as a function of Caspian tern predation probabilities during out-migration from Bonneville Dam as smolts to return to Bonneville Dam as adults in each year from 2008 to 2016. The size of circles depicts the relative number of PIT-tagged smolts detected at Bonneville Dam each week. Lines represent the estimate of the best linear fit to the data and shading denotes 95% credible intervals around the best fit.

8.4.2.3 Upper Columbia River yearling Chinook salmon – Average annual survival probabilities for UCR yearling Chinook smolts from RIS to BON was 0.571 (0.497–0.630*; Table 8.1* above) during 2008-2018 and ranged annually from 0.472 (0.374–0.556) to 0.706 (0.458–0.935). Small samples sizes of PIT-tagged UCR yearling Chinook smolts (see *8.8 Supplements*, *Table 8.S1*) resulted in imprecise estimates of survival in most years (see also *Chapter 7*). Predation probabilities on UCR yearling Chinook smolts averaged 0.03 (0.02–0.03) for terns, < 0.01 for cormorants, and 0.05 (0.04–0.05) for gulls, indicating that only a small proportion of available UCR yearling Chinook smolts were consumed by avian predators upstream of BON (*Table 8.1* above). For years in which terns and gulls consumed more than 0.05 of available UCR yearling Chinook smolts, there was some evidence of an additive relationship between predation and survival probabilities. For instance, in 2015, when gulls consumed 0.07 (0.05–0.11) of available UCR yearling Chinook smolts, higher consumption probabilities were associated with statistically significant lower survival probabilities to BON. When data from all years were considered, however, no statistically significant relationship was observed across all years (*Table 8.1* above). Taken together, imprecise estimates of survival, coupled with low levels of consumption, weakened the model's power to detect the potential additive effects of tern, cormorant, and gull consumption on the survival of UCR yearling Chinook smolts upstream of BON.

Smolt-to-adult survival probabilities for UCR yearling Chinook salmon from BON (as smolts) to BON (as adults) averaged 0.019 (0.018–0.021) and ranged annually from 0.008 (0.006–0.010) to 0.032 (0.030–0.035) during 2008-2015 (years with complete adult returns; *Table 8.2* above). Estimates of tern predation probabilities on UCR yearling Chinook smolts in the estuary were low, averaging 0.02 (0.02–0.02) during 2008-2015 (*Table 8.2* above) and ranged annually from

0.01 (0.01–0.02) to 0.03 (0.02–0.05). Results provided no evidence of a statistically significant relationship between tern predation probabilities on UCR yearling Chinook smolts in the estuary and adult returns to BON (*Table 8.2* above). Estimates of cormorant predation in the estuary on UCR yearling Chinook smolts were slightly higher than those for terns, averaging 0.04 (0.03–0.04) during 2008-2015 (*Table 8.2* above) and ranged annually from 0.02 (0.01–0.03) to 0.08 (0.05–0.12). Similar to tern predation, results provided no evidence of a statistically significant relationship between cormorant predation on UCR yearling Chinook smolts in the estuary and adult returns to BON (*Table 8.2* above).

8.4.2.4 Snake River yearling Chinook salmon – Average annual survival probability of SR yearling Chinook smolts from LMN to BON was 0.668 (0.627–0.709) during 2008-2018 (*Table 8.1* above) and ranged annually from 0.570 (0.487–0.654) to 0.737 (0.658–0.824). Similar to predation probabilities on UCR yearling Chinook smolts, a small proportion of available SR yearling Chinook smolts were consumed by avian predators upstream of BON, with average annual consumption probabilities of 0.01 (0.01–0.01) for terns, 0.02 (0.01–0.02) for cormorants, and 0.02 (0.02–0.02) for gulls (*Table 8.1* above). Unlike estimates of survival in UCR yearling Chinook smolts, however, larger sample sizes of SR yearling Chinook smolts (see *8.8 Supplements*, *Table 8.S1*) resulted in more precise estimates of survival and predation probabilities and thus better resolution regarding the relationships between these probabilities. For instance, despite the low levels of predation and limited variation in predation probabilities among weeks, there was evidence that tern predation was an additive source of mortality for SR yearling Chinook smolts during out-migration to BON, generally similar to the additivity levels measured with respect to tern predation on UCR and SR steelhead upstream of BON (*Table 8.1* above). This relationship is most apparent in years with higher predation probabilities and years with more variation in estimates of predation probability among weeks (e.g., in 2013; *Figure 8.11*). Due to low average annual tern predation probabilities on SR yearling Chinook smolts, however, the estimated difference between observed survival and baseline survival (i.e., survival in the absence of tern predation) was small at 0.011 (-0.007–0.027), suggesting that very few, if any, additional smolts would likely have survived out-migration to BON in the absence of tern predation upstream of BON (*Table 8.1* above). Estimates of consumption probabilities by cormorants and gulls on SR yearling Chinook smolts were also low, less variable than those of terns, and provided no evidence of a statistically significant relationship between predation probabilities and survival probabilities when data from all years were considered (*Table 8.1* above).

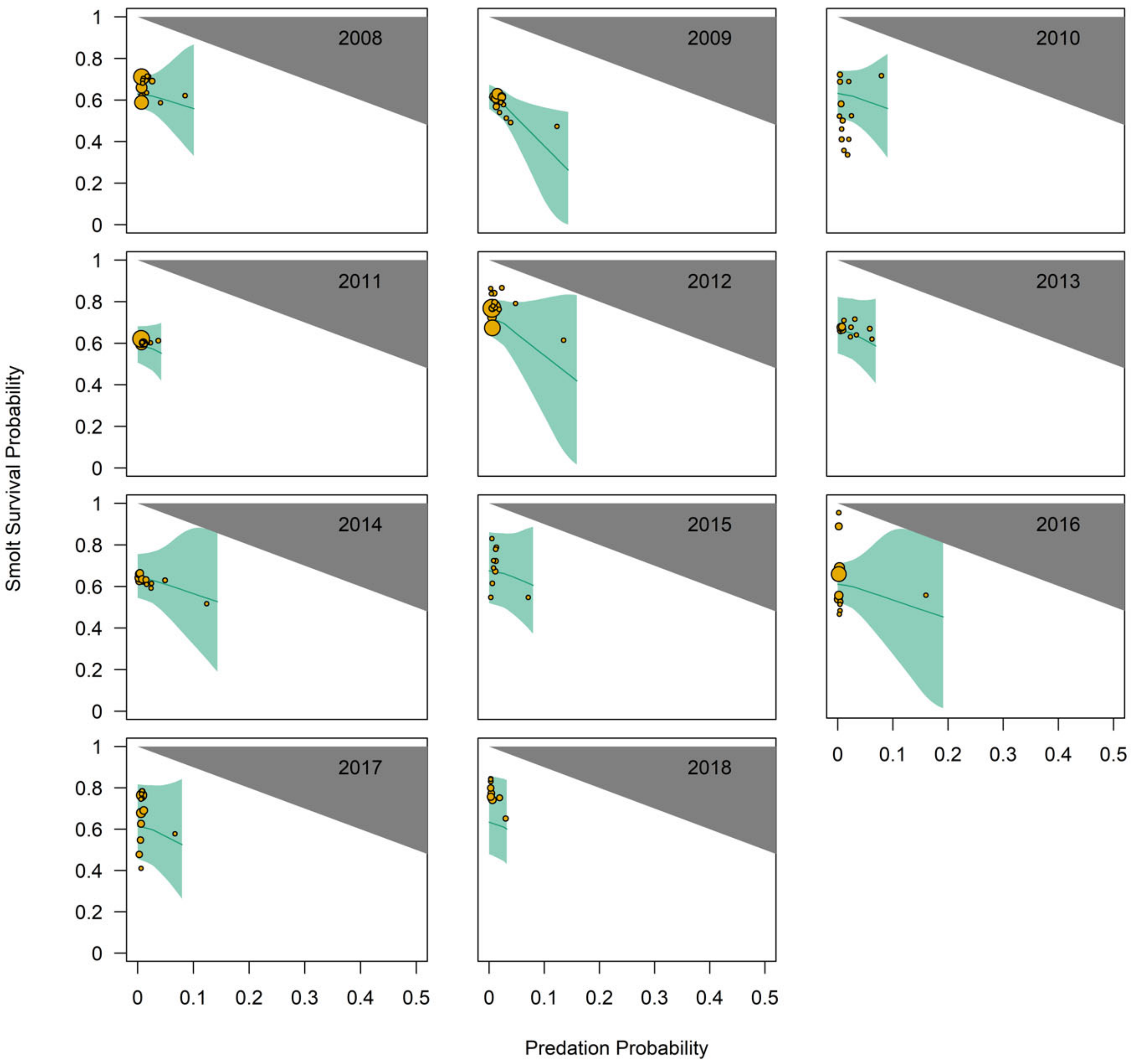

Figure 8.11. Weekly smolt survival probabilities for Snake River yearling Chinook salmon as a function of Caspian tern predation probabilities during smolt out-migration from Lower Monumental Dam to Bonneville Dam in each year from 2008 to 2018. The size of circles depicts the relative number of PIT-tagged smolts released at Lower Monumental Dam each week. Lines represent the best fit estimate of the linear relationship between predation and survival and shading denotes 95% credible intervals around the best fit.

Smolt-to-adult survival probabilities for SR yearling Chinook salmon from BON (as smolts) to BON (as adults) averaged 0.016 (0.016–0.017) during 2008-2015 (*Table 8.2* above) and ranged annually from 0.007 (0.006–0.008) to 0.035 (0.033–0.038). The average annual predation probability on SR yearling Chinook smolts by terns in the estuary was low at 0.02 (0.02–0.02; *Table 8.2* above) and ranged annually from 0.01 (0.01–0.02) to 0.05 (0.04–0.06). Based on the low rates of tern predation on SR yearling Chinook smolts in the estuary and low smolt-to-adult returns to BON, results provided no evidence of a statistically significant relationship between tern predation probabilities in the estuary and adult returns to BON when data across all years were considered (*Table 8.2* above). Average annual predation probability by cormorants on SR yearling Chinook smolts in the estuary was significantly higher than that of terns at 0.05 (0.05–0.06; *Table 8.2* above) and ranged annually from 0.04 (0.02–0.05) to 0.14 (0.10–0.20). Despite higher predation probabilities by cormorants, there was no statistically significant relationship between increasing predation probabilities on SR yearling Chinook smolts by cormorants in the estuary and decreasing smolt-to-adult survival probabilities to BON (*Table 8.2* above).

8.4.2.5 Snake River sub-yearling Chinook salmon – Average annual survival probability of SR sub-yearling Chinook smolts from LMN to BON was 0.697 (0.600–0.786) during 2008-2018 (*Table 8.1* above) and ranged annually from 0.604 (0.161–0.920) to 0.845 (0.520–0.935). Small sample sizes of PIT-tagged SR sub-yearling Chinook smolts (see *8.8 Supplements*, *Table 8.S1*) generated imprecise estimates of survival probabilities in most, but not, all years. The average annual predation probability on SR sub-yearling Chinook smolts by terns was low at 0.02 (0.01–0.02; *Table 8.1* above) and ranged annually from 0.01 (0–0.02) to 0.03 (0.02–0.04). Although the point estimate of average annual additivity was high at 1.32 (*Table 8.1* above), uncertainty in estimates of predation and survival probabilities, coupled with low levels of tern predation and a lack of variation among weeks in estimates of predation probabilities, limited the model's ability to detect a relationship between predation and survival, with no statistically significant trend, either positive or negative, identified when data from all years were considered. Similar to terns, average annual predation probability by cormorants was at 0.01 (0.01–0.02; *Table 8.1* above) and ranged annually from 0.01 (0–0.01) to 0.03 (0.01–0.06); there was no evidence of statistically significant relationship between cormorant predation and survival when data from all years were considered (*Table 8.1* above). Average annual consumption of SR sub-yearling Chinook smolts by gulls was higher than that of terns or cormorants at 0.05 (0.04–0.07) and ranged annually from 0.02 (0.01–0.05) to 0.08 (0.05–0.12) during 2013-2018 (*Table 8.1* above). Analogous to results for tern and cormorant predation, however, small sample sizes of PIT-tagged SR sub-yearling Chinook smolts and uncertainty in estimates of predation and survival, limited the model's power to detect a consistent and statistically significant relationship between gull consumption of SR sub-yearling Chinook smolts and smolt survival to BON.

Smolt-to-adult survival probabilities for SR sub-yearling Chinook salmon from BON (as smolts) to BON (as adults) averaged 0.020 (0.018–0.022) and ranged annually from 0.005 (0.004-0.006) to 0.035 (0.033–0.038; *Table 8.2* above). Average annual predation probability on SR sub-yearling Chinook smolts by terns in the estuary was low at 0.01 (0.01–0.02) and ranged annually from < 0.01 to 0.02 (0.01–0.03; *Table 8.2* above). Average annual predation probability by cormorants in the estuary was slightly higher than that of terns at 0.02 (0.02–0.03) and ranged annually from 0.1 (0–0.4) to 0.03 (0.02–0.05; *Table 8.2* above). As expected, based on the low predation probabilities and low smolt-to-adult survival probabilities, there was no evidence of a statistically significant relationship between tern or cormorant predation on SR sub-yearling Chinook smolts in the estuary and adult returns to BON (*Table 8.2* above).

8.4.2.6 Snake River sockeye salmon – Sockeye smolts were PIT-tagged in limited numbers (see *8.8 Supplements*, *Table 8.S1*) over only a few weeks (2–4) each year. This led to high levels uncertainty in estimates of both survival and predation probabilities during smolt out-migration from LMN to BON (see also *Chapter 7*). Average annual survival probability for SR sockeye smolts was 0.698 (0.588–0.783; *Table 8.1* above) during 2008-2017 (years with complete adult returns) and ranged annually from 0.563 (0.388–0.881) to 0.866 (0.587–0.951). The average annual predation probability for tern predation on SR sockeye smolts was 0.03 (0.02–0.3;) and ranged annually from 0.01 (0.01–0.02) to 0.06 (0.03–0.11). The average annual predation probability for cormorant predation on SR sockeye smolts was 0.03 (0.02–0.05; *Table 8.1* above) and ranged annually from 0.02 (0.01–0.03) to 0.05 (0.02–0.08). Average annual consumption probability for gulls of SR sockeye smolts was substantially higher than those of terns and cormorants at 0.13 (0.09–0.16; *Table 8.1* above) and ranged annually from 0.08 (0.03–0.17) to 0.22 (0.10–0.35). For all three avian predator species (terns, cormorants, gulls), the dataset lacked the precision and weekly replication needed to accurately assess the relationship between predation probabilities and survival probabilities for SR sockeye during smolt out-migration from LMN to BON (*Table 8.1* above).

Smolt-to-adult survival probabilities for SR sockeye salmon from BON (as smolts) to BON (as adults) averaged 0.020 (0.016–0.024) during 2008-2017 (*Table 8.2* above) and ranged annually from 0.006 (0.006-0.008) to 0.037 (0.029–0.046). The average annual predation probability for SR sockeye smolts by terns in the estuary was 0.03 (0.02–0.03) and ranged annually from < 0.01 to 0.08 (0.03–0.13), indicating that, with the notable outlier of 2016, only a small proportion of available SR sockeye smolts in estuary were depredated by terns (*Table 8.2* above and also *Chapter 7*). The average annual predation probability for SR sockeye smolts by cormorants in the estuary was slightly higher than that for terns at 0.04 (0.03–0.05; *Table 8.2* above) and ranged annually from 0.02 (0.01–0.04) to 0.09 (0.02–0.19). Although there was some evidence of a relationship between increased levels of tern and cormorant predation on SR sockeye smolts in the estuary and smolt-to-adult survival to BON in some years, analogous to modelling results for smolt survival upstream of BON, small sample sizes of PIT-tagged SR sockeye smolts (see *8.8 Supplements, Table 8.S2*), coupled with low predation rates and low adult return rates, prohibited an accurate assessment of the relationship between avian predation and survival of SR sockeye salmon, with no consistent trend identified across all study years.

8.5 Discussion

Anadromous salmonid smolts originating from the UCR and SR basins must out-migrate through the foraging ranges of several species of piscivorous colonial waterbirds nesting at multiple sites throughout the CRB. Results of this and other studies indicate that the effects of avian predation on smolts from several ESA-listed salmonid populations can be substantial, particularly predation by Caspian terns on steelhead, where predation probabilities of more than 0.20 or 20% of available smolts have been repeatedly observed over the course of the last two decades (Evans et al. 2012, Hostetter et al. 2015, Evans et al. 2016, 2019, Payton et al. 2019; see also *Chapter 7*). Not all salmonid populations in the CRB, however, are subject to such high levels of avian predation, with predation on UCR and SR yearling Chinook salmon smolts consistently less than 0.05 or 5% of available fish annually (Evans et al. 2016; see also *Chapter 7*). Regardless of the level or magnitude of predation, avian predation does not necessarily indicate a substantial hazard to the short- or long-term survival of salmonids if all fish consumed by birds are destined to die regardless of avian predation. By using the recently published methods of Payton et al. (2020), we were able to directly measure the strength, direction, and magnitude of the relationship between survival and predator-specific predation probabilities on multiple salmonid species and populations from the CRB. Our results provide novel inferences about how increases in fish survival for some salmonid species can be directly attributed to reductions in avian predation.

Results of this study were consistent with those of Payton et al. (2020), indicating that there was a statistically significant relationship between Caspian tern predation probabilities on UCR steelhead smolts and steelhead survival probabilities within and across salmonid life-stages (smolt, smolt-to-adult). In addition, results of this study indicate that increases in Caspian tern predation on SR steelhead smolts were associated with statistically significant decreases in SR steelhead survival within and across salmonid life-stages. In the absence of tern predation upstream of Bonneville Dam, results suggested that significantly more steelhead smolts would likely have survived out-migration to Bonneville Dam. These results support on-going efforts to increase steelhead survival by reducing the number of Caspian terns nesting at certain colonies upstream of Bonneville Dam, in particular those on Goose Island, Crescent Island, and the Blalock Islands (see also *Chapter 2*). Results of this study also provide evidence that predation by Caspian terns nesting on East Sand Island in the Columbia River estuary was a partially additive source of mortality for steelhead in the adult life-stage. Over scales as large as smolt-to-adult returns, representing the majority of an anadromous salmonid's potential lifespan, mortality sources encountered during earlier life-stages will be increasingly compensatory relative to mortality at later life stages. For example, the percentage of UCR and SR steelhead smolts that survive from ocean entry until they return to the Columbia River as adults has been consistently less than 5% in recent years (Evans et al. 2019, Haeseker et al. 2020). Consequently, if piscivorous waterbirds consumed a random sample of available smolts out-migrating through the estuary, then the additive portion of the smolts consumed (i.e. the proportion that would have otherwise survived to adulthood) would similarly be less than 5%. This highlights the importance of recognizing the temporal- and spatial-scales at which the relationship between predation and survival is evaluated, which can have important implications for inferences with respect to the impact of predation on prey populations (Payton et al. 2020).

Our results suggest that Caspian tern predation upstream of Bonneville Dam was a super-additive source of smolt mortality (**a** > 1.0) in some salmonid populations, whereby greater levels of tern predation were associated with additional indirect mortality that further reduced the survival probability of smolts left unconsumed by terns. Mechanisms of super-additivity may be related to the foraging behavior of Caspian terns, where unsuccessful depredation attempts resulted in some proportion of the prey being injured (lethally- or sub-lethally), losses that are akin to latent mortality or crippling losses in harvest management (Williams et al. 2002, Schaub and Lebreton 2004, Payton et al. 2020). An additional mechanism of super-additivity involves instances where smolts captured by terns are stolen by gulls (i.e., kleptoparasitism) prior to the fish being consumed by the foraging tern, its mate, or its young. For example, an unknown, but possibly substantial, proportion of smolts captured by terns and brought back to the breeding colony in their bills to feed mates or young were kleptoparasitized by gulls (Garcia et al. 2010, Patterson 2012, Adkins et al. 2014). These smolt losses, however, were not fully incorporated into tern predation probabilities (Payton et al. 2020). Similarly, the theft of tern-depredated smolts by gulls is a mechanism of compensatory mortality associated with gull consumption of smolts, whereby some proportion of smolts consumed by gulls were already dead or nearly so (see also *below*).

Results presented in this study regarding the impact of Caspian tern predation on survival of Snake River steelhead differ from those of a recently published study by Haeseker et al. (2020). As in the present study, Haeseker et al. (2020) investigated the relationship between Caspian tern predation on Snake River steelhead smolts in the estuary and adult returns to Bonneville Dam. While our model measured a significant inverse relationship between levels of tern predation and the probability of SR steelhead adult returns, their model provided no evidence of a statistically significant correlation between predation and survival probabilities. Haeseker et al. (2020) further interpreted a lack of correlation as evidence that predation was completely compensatory (**a** = 0). As noted previously, interpreting a lack of correlation between two variables as proof that no relationship exists is anathema to statistical inference (Fisher 1925).

Beyond the interpretation of results, the most apparent difference between the modelling efforts of Haeseker et al. (2020) and those in the present study involve the consideration of covariates. Efforts by Haeseker et al. (2020) to incorporate additional factors or covariates that may influence salmonid survival may be crucial in fully understanding the true impact of predation on survival. It is well known that smolt-to-adult survival probabilities vary both within and across years due to a multitude of factors like smolt run-timing, smolt abundance, river conditions, and ocean conditions (Scheuerell et al. 2009, Petrosky and Schaller 2010). However, many of these and other factors almost certainly influence avian predation probabilities as well (e.g., smolt arrival times and abundance in the estuary; see also *Chapters 9* and 10) and, by not accounting for this covariation, the relationship between predation and survival was almost certainly obfuscated in the Haeseker et al. (2020) analysis. In contrast, the models herein omitted consideration of specific environmental covariates. However, in the present study it was assumed that average survival and predation probabilities were independent from year-to-year, whereas Haeseker et al. (2020) assumed that predation and survival probabilities from any given week were as similar to those in the next consecutive week as they were to those from any other week of any year, even those weeks that were decades apart in time. While some covariates considered by Haeseker et al. (2020) were annual variables (i.e., their values only varied by year), there was no explicit term to recognize all unexplained variation among years and, again, none of these covariates were used to explain variations in tern predation probabilities. Further inherent to this modelling choice by Haeseker et al. (2020) is the assumption that any relationship between survival and predation is constant across all years. As demonstrated by the results herein, this is inconsistent with the empirical data. The inverse linear relationship between survival and predation is apparent when plotting weekly predation probabilities versus weekly survival probabilities for each year separately, and the differences among years in the magnitude of this relationship also become evident (*Figure 8.13* above). If annual differences in the relationship between survival and predation probabilities are ignored and all years combined, the relationship is not only obscured but seemingly reversed (see *8.8 Supplements*, *Figure 8.S3*).

We believe that this difference in underlying assumptions about the role of year as a covariate is the principal explanation for the discrepancy in results of Payton et al. (2020) and those of Haeseker et al. (2020). Nevertheless, as discussed above, even after accounting for differences in survival and predation probabilities among years, there was no apparent relationship between cormorant predation probabilities on SR steelhead smolts in the estuary and smolt-to-adult survival probabilities. As such, further exploration of covariates that may affect survival and/or predation probabilities, including those investigated by Haeseker et al. (2020), are warranted and should be conducted before concluding that double-crested cormorants nesting in the Columbia River estuary feed exclusively on smolts destined to die from other causes.

In addition to accounting for year-effects, the ability to accurately assess the relationship between predation and survival depends on several other factors, including (1) the magnitude or level of cause-specific mortality (Sandercock et al. 2011, Bischof et al. 2009), (2) variation in cause-specific mortality (Schaub and Lebreton 2004, Wolfe et al. 2015), and (3) the precision of survival and cause-specific mortality estimates (Cooch et al. 2014, Payton et al. 2020). In the present study, the magnitude of Caspian tern predation on SR and UCR steelhead smolts was high and predation was variable across weeks within the same year. In the case of predation on Chinook or sockeye salmon smolts, however, the datasets often lacked these requisite components and were thus less suitable for determining the degree to which survival was related to avian predation. For instance, even in cases where large sample sizes of tagged fish were available, as with SR yearling Chinook salmon, low levels of predation (generally < 0.05 of the available smolts) and a lack of variation in estimates of predation, inhibited the model's ability to detect statistically significant relationships between predation and survival. Furthermore, because predation probabilities on salmon smolts were generally low and represented a small fraction of all smolt mortality (1 – survival), estimated differences in observed survival versus baseline survival ($\mathbf{\Phi}^{\Delta}$) were also often small (i.e., near zero) and would likely remain so regardless of the level of additivity. Consequently, in the absence of tern, cormorant, and gull predation, only small increases in salmon smolt survival were possible, particularly survival to adulthood, where the vast majority of fish die from causes other than avian predation before returning to Bonneville Dam.

Results of this study also provided evidence of a relationship between double-crested cormorant predation and smolt survival for SR steelhead upstream of Bonneville Dam. Similar to tern predation on SR steelhead smolts, the point estimate of additivity suggests that cormorant predation was a super-additive source of smolt mortality (**a** > 1.0), but unlike tern predation, the bounds of the credible interval about the point estimate of additivity ranged from 0.07 to 3.06, indicating that precise estimates of additivity due to cormorant predation were lacking. Following passage through the hydrosystem to below Bonneville Dam, there was no evidence that cormorant predation on SR steelhead smolts in the estuary were associated with reduced smolt-to-adult survival. Cormorants nesting in the estuary have a diverse and highly varied diet (Roby et al. 2003) and, compared with Caspian terns, double-crested cormorants consumed a smaller proportion of available steelhead and consumed steelhead more in proportion to their relative availability or abundance (see also *Chapter 9*). Measuring the impact of double-crested cormorant predation on salmonid smolt survival in the estuary may necessitate the examination of other covariates to understand to what degree cormorant predation limits steelhead survival. It is also possible that predation rates by terns, cormorants, and gulls were correlated with other ecological processes that influenced smolt survival both upstream and downstream of Bonneville Dam, like changes in river flows, changes in prey availability, prey-switching, ocean conditions, and other factors (Petrosky and Schaller 2010, Haeseker et al. 2012, Payton et al. 2016; see also *Chapters 9* and *10*). As such, additional research is needed to better understand mechanisms of additivity and how biotic and abiotic conditions experienced by smolts during out-migration influence predation probabilities, relationships that could either magnify or mask the effects of avian predation on fish survival in the CRB.

Although sufficient data to understand to what degree consumption by gulls limited smolt survival were generally lacking in the present study, some proportion of California and ring-billed gull consumption of juvenile salmonids is clearly compensatory. Unlike Caspian terns and double-crested cormorants, gulls acquire much of their food energy by scavenging, and are known to consume dead fish and to kleptoparasitize dead fish from other piscivorous waterbirds (Antolos et al. 2005, Winkler 2020). Relatively large smolts, such as steelhead, are disproportionately kleptoparasitized by gulls compared with smaller-sized juvenile salmonids, such as Chinook salmon and sockeye salmon (Adkins et al. 2011). Selective kleptoparasitism could enhance gull consumption probabilities for steelhead smolts relative to other salmon smolts. Previous research also indicates that smolts are especially susceptible to gull predation in the tailrace of hydroelectric dams, areas where smolts may be injured or stunned because of dam passage or where turbulent hydraulic conditions may disorient fish, thereby increasing their risk of being depredated by gulls (Ruggerone 1986, Zorich et al. 2011, Evans et al. 2016). The proportion of smolts consumed by gulls in the current study that were healthy fish capable of surviving out-migration versus dead or moribund fish, however, was unknown and warrants additional research (see also *Chapter 6*).

In conclusion, this study investigated the direct relationship between predation probabilities by multiple avian predator species and survival probabilities of multiple species and populations of salmonids. Our results provide novel evidence with respect to the full impact of avian predation on salmonid populations across short- and long-term timescales. Results suggest efforts to reduce tern and cormorant predation on smolts may increase the number of steelhead that survive out-migration to Bonneville Dam and, more importantly from a conservation perspective, the number that survive to adulthood. These are important findings for those concerned with the restoration of Upper Columbia River and Snake River steelhead in the Columbia River basin, populations that are listed under the U.S. Endangered Species Act. Additional research, however, is needed to better understand the relationship between avian predation and survival probabilities for various salmon populations, particularly smolt-to-adult survival, where predation probabilities are generally lower and less variable than those for steelhead, and where avian predation constitutes a relatively minor component of all sources of fish mortality.

8.6 Acknowledgments

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8.8 Supplements

Table 8.S1. Numbers of PIT-tagged salmonid smolts interrogated/released (Rel) at Lower Monumental Dam (for Snake River [SR]) or Rock Island Dam (for Upper Columbia River [UCR]) and subsequently recaptured alive (Live) at Bonneville Dam or at a Net Detector downstream of Bonneville Dam or recovered (Dead) on Caspian tern, double-crested cormorant, or California and ring-billed gull colonies. Species include steelhead, Chinook salmon, and sockeye salmon. Chinook salmon age-classes include yearling (1) and subyearling (0).

Year	SR Steelhead (Rel–Live–Dead)	SR Chinook 1 (Rel–Live–Dead)	SR Chinook 0 (Rel–Live–Dead)	SR Sockeye (Rel–Live–Dead)	UCR Steelhead (Rel–Live–Dead)	UCR Chinook 1 (Rel–Live–Dead)
2008	28485-3350-4241	58586-5001-1859	15822-1642-434	738-62-42	7271-1107-1034	907-246-27
2009	51818-7938-7829	51368-5983-2473	12495-1106-398	2629-368-120	7114-1205-1098	756-224-29
2010	7693-1427-1090	10866-1689-360	27229-2115-550	563-73-15	7365-1447-969	1076-269-30
2011	56787-3093-4712	95195-4416-2368	25082-1029-463	10946-354-322	7757-583-794	1705-364-30
2012	42984-3000-2645	65609-5650-1299	10196-592-181	2697-244-113	6711-845-570	1885-537-38
2013	15626-1487-1015	21850-1594-374	1360-64-36	807-46-16	5893-846-680	5826-1532-118
2014	22154-1890-2118	35900-2721-1027	2024-148-48	1383-151-56	7663-1017-775	5551-1375-159
2015	3632-652-341	5674-625-227	942-13-18	470-68-15	7069-1189-792	5908-1050-197
2016	32335-5602-1558	50218-5395-794	901-29-5	517-47-6	6764-1577-533	5781-1742-130
2017	38582-3035-1927	40477-2567-307	2875-231-21	334-25-7	7436-797-514	2962-693-35
2018	30297-3631-1113	29746-1844-217	3339-365-41	1326-150-33	7366-912-338	2368-385-19

Table 8.S2. Numbers of PIT-tagged salmonid smolts interrogated/released (Rel) at Bonneville Dam and recovered (Dead) on Caspian tern or double-crested cormorant colonies downstream of Bonneville Dam or detected returning as an adult (SAR) to Bonneville Dam. Species include steelhead, Chinook salmon, and sockeye salmon. Chinook age-classes include yearling (1) and subyearling (0). Years omitted were those without complete adult returns to-date.

Year	SR Steelhead (Rel–Dead–SAR)	SR Chinook 1 (Rel–Dead–SAR)	SR Chinook 0 (Rel–Dead–SAR)	SR Sockeye (Rel–Dead–SAR)	UCR Steelhead (Rel–Dead–SAR)	UCR Chinook 1 (Rel–Dead–SAR)
2008	20072-2754-1221	25350-717-897	18292-348-648	168-6-3	2688-328-228	1854-38-49
2009	23593-3429-677	36142-1891-580	11308-268-57	1828-46-67	2684-380-72	3507-101-82
2010	40265-4309-1307	68640-2630-1049	12279-107-350	1373-23-39	12634-1298-469	17192-451-557
2011	8362-672-123	14066-342-145	9208-79-297	816-14-8	2485-215-40	2449-75-33
2012	9296-638-268	22744-495-380	9981-131-159	1440-33-34	3538-214-93	4229-56-76
2013	11949-609-278	14714-228-334	1443-23-25	602-6-20	4017-187-88	6912-78-138
2014	13684-869-326	18131-604-142	1784-20-15	1726-39-23	4183-366-90	7188-216-56
2015	34664-3549-138	23285-1500-159	632-6-9	3300-59-18	6679-682-16	11104-464-131
2016	27119-1341-441			263-11-3	8139-443-141	
2017				250-4-0		

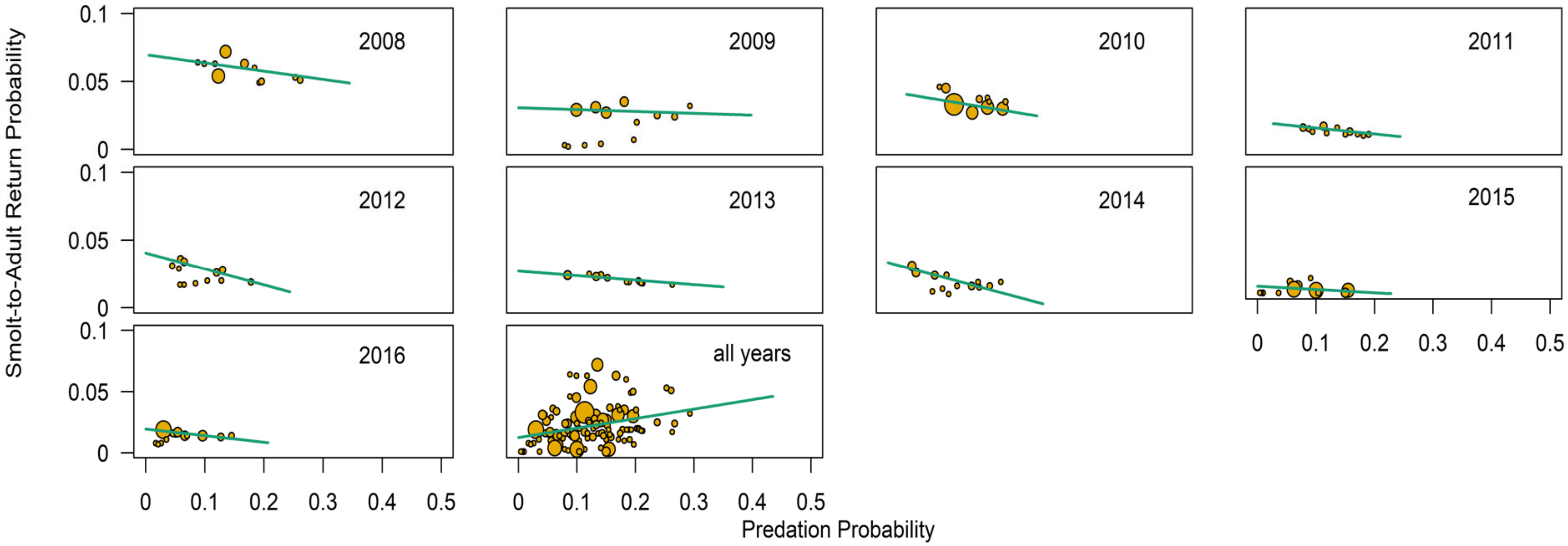

Figure 8.S3. Weekly smolt-to-adult survival probabilities to Bonneville Dam for Snake River steelhead as a function of Caspian tern predation probabilities in the Columbia River estuary in each year from 2008 to 2016 and all years combined (all years). The size of the circles depicts the relative number of PIT-tagged steelhead smolts available passing Bonneville Dam. Blue lines represent simple weighted least-squares regression lines among the estimates (weights relative to sample size). By ignoring year-effects, as was done by Haeseker et al. (2020), the relationship between predation and survival probabilities is obfuscated (see Discussion).

CHAPTER 9: Functional Responses Across Predator Species, Space, and Time: How Piscivorous Waterbirds Respond to Changes in Juvenile Steelhead Abundance

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9.1 Summary

How predators respond to changes in prey availability (i.e. functional responses) is foundational to predator-prey dynamics, foraging theory, and the stability of predator-prey systems. In the Columbia River basin, USA, predation by Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), and California and ring-billed gulls (*Larus californicus* and *L. delawarensis*) on out-migrating steelhead trout (*Oncorhynchus mykiss*) smolts is considered a factor limiting the recovery of multiple steelhead populations. We used an extensive, 13-year dataset on bird abundance across seven waterbird breeding colonies (three tern colonies, two cormorant colonies, and two gull colonies) and a juvenile steelhead tag-recovery dataset with >645,000 detections and >32,000 recoveries of tagged steelhead on those seven waterbird colonies to quantify predation probabilities and functional responses across predator species, colony locations, and years. Weekly predation probabilities varied by bird species, location (estuary, inland), and week, and generally ranged between 0.01–0.30 at three tern colonies, 0.01–0.20 at two cormorant colonies, and 0.03–0.13 at two gull colonies. Per capita predation probabilities (i.e. per bird) were an order of magnitude higher at inland tern and cormorant colonies relative to estuary colonies of the same species. Terns displayed consistent Type II functional responses across all colonies and years, with per capita predation probabilities peaking at minimal prey availability and declining as steelhead availability increased. Cormorants nesting at the large (several thousand breeding pairs) colony in the estuary displayed a Type III functional response, but cormorants nesting at the small (several hundred breeding birds) inland colony displayed a Type II response. Functional responses for two gull colonies were not explicitly quantified due to a lack of weekly gull abundance data; however, gull predation remained consistent across large ranges of steelhead availability, suggestive of a Type I or III functional response. For terns, the magnitude of predation combined with a Type II functional response suggest possible population-level impacts that could destabilize small or declining prey populations. For cormorants in the estuary, a Type III functional response suggests prey switching behaviors and predation targeted at specific locations and periods of high prey availability. Awareness of predator-specific foraging strategies, location-specific functional responses, and per capita predation probabilities may lead to better predator management strategies aimed at increasing steelhead survival, while also improving our understanding of density-dependent predation effects on prey populations and the complexity of predator-prey interactions in natural systems.

9.2 Introduction

Pulses in resource availability produce abundant yet ephemeral foraging opportunities for consumers (Yang et al. 2008). Prey resource synchrony (e.g., timing of prey emergence, migrations, births) can constrain the ability of consumers to exploit temporary resource abundance, but predator-prey relationships are often mediated by consumer foraging strategies (Armstrong et al. 2016). At short timescales, behavioral aggregation, prey switching, and functional responses of consumers determine how an individual, population, or species maximizes energy intake during pulses of high resource availability (Solomon 1949, Ims 1990, Yang et al. 2008). It is often hypothesized that generalist and mobile consumers exploit spatiotemporal variation in resource availability through functional and behavioral responses, while specialist and spatially restricted predators experience constraints in processing high prey densities due to restrictions on handling or consumption time (Holling 1959a, Ims 1990, O'Donoghue et al. 1998b, Yang et al. 2008, Armstrong et al. 2016). Understanding how functional responses vary among generalist and specialist consumers and across spatial and temporal scales requires large-scale, multi-year, multi-species field studies that are rare and difficult to sustain (O'Donoghue et al. 1998a, 1998b, Armstrong et al. 2016, Ellis et al. 2020), and thus limits our understanding of these processes in natural environments (Chan et al. 2017).

Functional responses describe how per capita kill rates are related to changes in prey abundance (Solomon 1949, Holling 1959a, 1959b). Identifying predator functional responses is foundational to behavioral and population ecology (Abrams 1990), foraging theory (Stephens and Krebs 1986), resource limitation (Hassell 1978), and the stability of predator-prey systems (Oaten and Murdoch 1975). Functional responses are commonly categorized into three types (*Figure 9.1*; Holling 1959a):

Type I. The number of prey killed per predator per unit time is directly proportional to changes in prey density.

Type II. The number of prey killed per predator per unit time increases with increasing prey density, but levels off at an upper threshold due to handling time (i.e. the time required to capture and consume prey).

Type III. The number of prey killed per predator per unit time responds slowly when prey are scarce, rises rapidly as prey density increases, then levels off at high densities.

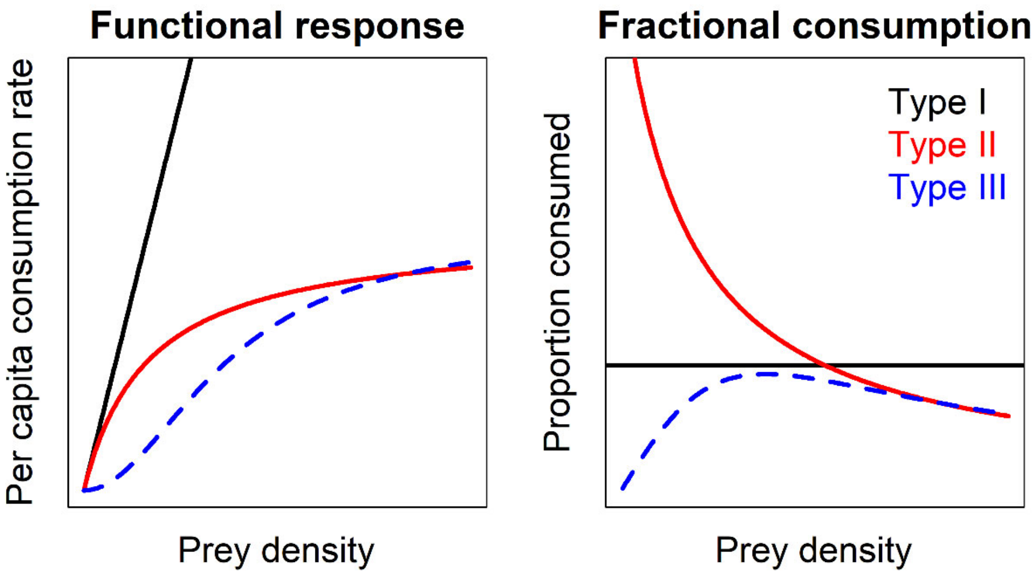

Figure 9.1. Type I, II, and III functional responses expressed as per capita consumption rates (i.e., number of prey consumed per predator per unit time; left) and corresponding fractional consumption rates (i.e., per capita predation probability; right). Mark-recovery studies generally quantify some form of fractional consumption (right).

Both Type II and III functional responses suggest declining per capita consumption rates in response to increasing prey availability. Type III differs from type II at low prey densities where an accelerating phase precedes the deceleration of per capita consumption (*Figure 9.1* above). While functional responses were originally formulated in terms of number of prey consumed per predator per unit time (Holling 1959a), many studies now examine functional responses using predation rates or probabilities (e.g., changes in per capita predation probability as a function of prey abundance; *Figure 9.1* above; Juliano 2001). Evaluating functional responses using predation probabilities is particularly advantageous as mark-recovery methods provide a diversity of opportunities to quantify cause-specific mortality probabilities and are widely used across ecosystems and taxa (e.g., Schaub and Pradel 2004, Hostetter et al. 2015, Payton et al. 2019).

Anadromous Pacific salmonids (*Oncorhynchus* spp.) are key components linking aquatic and terrestrial ecosystems throughout western North America (Gende et al. 2002, Quinn 2018). Predation on returning adult salmonids by bears and other consumers are well-studied phenomena, whereby multiple predator species respond to pulses in adults returning to freshwater ecosystems (e.g., Gende et al. 2002, Quinn et al. 2003, Scheuerell et al. 2007, Schindler et al. 2013, Deacy et al. 2018). Similarly, but perhaps less visually captivating, are responses of avian, fish, and mammalian predators to annual pulses of hundreds of thousands of out-migrating juvenile salmonids (e.g., Rieman et al. 1991, Petersen 1994, Yurk and Trites 2000, Evans et al. 2012). Pulses of juvenile salmonids may last for weeks to months, dominating the prey base and providing an important ephemeral food source for a diversity of predators (Collis et al. 2002, Weitkamp et al. 2012, Erhardt et al. 2018). While knowledge of density–

dependent predation in response to pulses of out-migrating juvenile salmonids (i.e. functional responses) is critical to improving our understanding of predator-prey dynamics in these systems, predator-specific relationships remain poorly understood (e.g., Hostetter et al. 2012, Furey et al. 2016).

In the Columbia River basin, USA, avian predation on out-migrating juvenile salmonids is considered a factor limiting the recovery of some salmonid populations listed under the U.S. Endangered Species Act (ESA; NOAA 2008). Combined across multiple breeding colonies of piscivorous waterbirds, avian predation on out-migrating juvenile steelhead (*O. mykiss*) can exceed 30% and is a primary source of mortality for out-migrating steelhead (Evans et al. 2016, 2019, Payton et al. 2019). Predation rates by three taxa of avian predators are particularly notable: Caspian terns (*Hydroprogne caspia*; hereafter terns), double-crested cormorants (*Phalacrocorax auritus*; hereafter cormorants), and mixed colonies of California and ring-billed gulls (*Larus californicus* and *L. delawarensis*; hereafter gulls; Hostetter et al. 2015, Evans et al. 2016, Payton et al. 2019). Foraging ecology across these three taxa is highly variable. Gulls are generalists that forage on a variety of aquatic and terrestrial food resources (Winkler 2020). Cormorants are pursuit-divers of aquatic prey, but their diets include a diverse array of fish species, sizes, and habitats (Dorr et al. 2020). Terns, conversely, have a relatively restricted range of prey types and sizes, and forage in the top meter of the water column (plunge-divers; Cuthbert and Wires 2020). Foraging ecology variation among waterbird species is also evident from previous diet studies, where juvenile salmonids comprised 30–80% of tern diets (by mass) compared to 15–45% of cormorant diets and 0–15% of California and ring-billed gull diets in the Columbia River basin (Collis et al. 2002; Roby et al. 2002). Steelhead are especially susceptible to predation by waterbirds, with predation probabilities sometimes two to five times higher than those of other juvenile salmonid species (Collis et al. 2001; Evans et al. 2012; Evans et al. 2016). Comparisons of total predation probabilities, per capita predation probabilities, and functional responses across three predator species located in freshwater and estuarine ecosystems provides a tremendous opportunity to explore how predator foraging strategies, ranging from diets with high proportions of salmonids (terns) to less specialized (cormorants) to generalists (gulls), influence predator-specific functional responses and variation in these processes across locations and years.

Herein, we investigate species-specific predation probabilities and functional responses using mark-recovery data collected across three avian predator species nesting on four different islands over a 13-year study in the Columbia River basin. Our study evaluates the form of and support for different functional responses among different predator species in a natural system. We hypothesized that functional responses vary by the foraging behavior of each predator species, where pulses of out-migrating steelhead smolts quickly swamp more specialized predators (terns; Type II response), while more generalist predators (cormorants and gulls) exploit resource pulses through changes in per capita predation rates (Type I or III). We also explored how functional responses vary by colony location where we expected increased evidence of predator swamping at colonies located in freshwater regions relative to estuary locations due to decreased prey diversity and faster transit times of steelhead smolts in freshwater portions of the basin. Overall, this study provides several novel insights into predator prey dynamics by (i) exploring how predators respond to changes in abundance of out-migrating steelhead smolts, (ii) directly quantify support for predator functional responses using mark-recovery data across multiple predator species, and (iii) comparing functional responses across a spectrum of specialist to generalist predators replicated amongst multiple locations and years.

9.3 Methods

9.3.1 Study Area

This study was conducted at three Caspian tern colonies, two double-crested cormorant colonies, and two colonies of mixed California and ring-billed gulls in the Columbia River basin (*Figure 9.2*). Colonies were located in the inland portion of the basin (i.e., Columbia Plateau; Adkins et al. 2014) and the Columbia River estuary (*Figure 9.2*). Following methods of Evans et al. (2012), we used detection and recovery data from Passive Integrated Transponder (PIT) tagged out-migrating juvenile steelhead detected while passing Rock Island Dam (RIS; river km [Rkm] 730), Lower Monumental Dam (LMN; Rkm 589), McNary Dam (MCN; Rkm 470), and Bonneville Dam (BON; Rkm 225; *Figure 9.2*) and recoveries of tags from the aforementioned bird colonies (*Figure 9.2*).

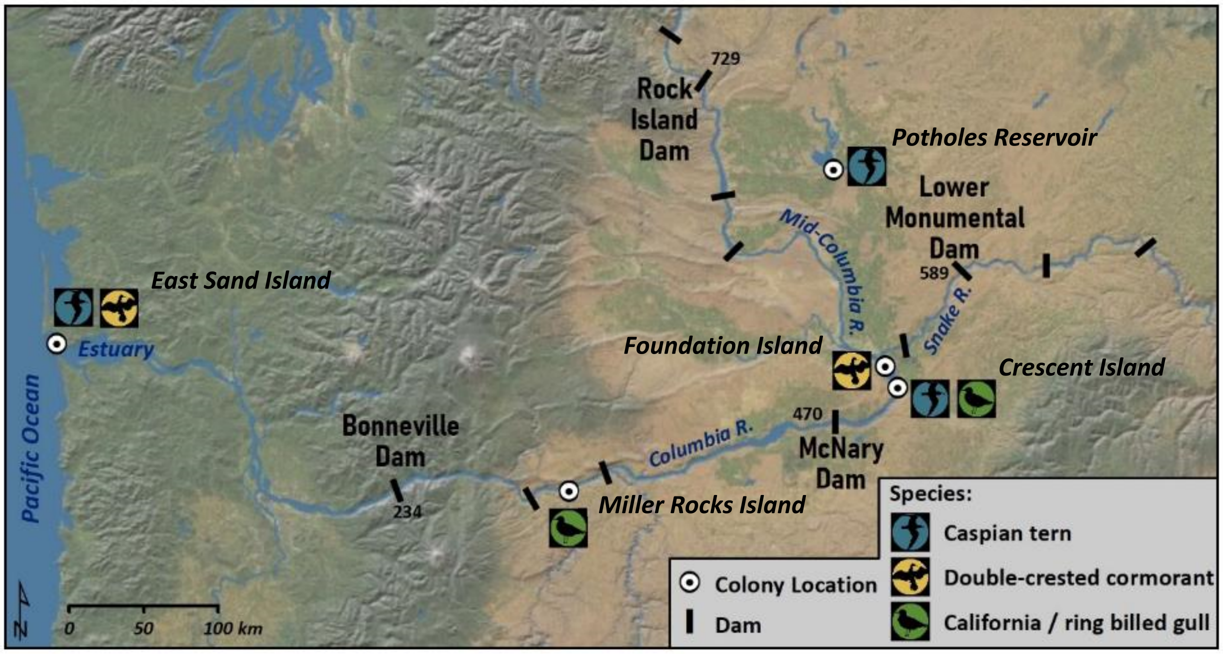

Figure 9.2. Locations of bird nesting colonies and hydroelectric dams within the Columbia River basin, USA, where data were collected for analyses.

9.3.2 Colony Monitoring

Boat, aerial, and blind-based surveys were used to count the number of birds nesting at each colony. Our study period includes 2007–2019, however, we restricted colony-specific analyses to years in which both counts of birds and recovery of PIT tags occurred at a given colony. For tern colonies, weekly abundance was the average of multiple counts of adult birds on the nesting colony during each week across the breeding season. The same approach was used for cormorants nesting at Foundation Island in the Columbia Plateau region. Counts of cormorants nesting at East Sand Island in the Columbia River estuary were collected via aerial photography once per week or every other week and recorded as the number of active nests. For simplicity of comparisons, the number of adults was calculated as 2 × number of active nests. In weeks without East Sand Island cormorant count data, we interpolated abundance as the average of the previous and subsequent week. Data were more limited at gull colonies, which were only counted on one occasion per breeding season. We therefore estimate weekly predation probabilities associated with gull colonies but do not quantify per capita predation probabilities and functional responses. Additional details on colony survey protocols are provided in Collis et al. (2002) and Adkins et al. (2014).

9.3.3 Fish Tagging & Recovery

Methods defining weekly juvenile steelhead availability to avian predators followed those of Evans et al. (2012). Each year, tens of thousands of juvenile steelhead are PIT tagged and released throughout the Columbia River basin (PSMFC 2020). A portion of PIT-tagged juveniles are subsequently detected passing downstream hydroelectric dams during outmigration to the Pacific Ocean (*Figure 9.2* above). Following Evans et al. (2012), we considered PIT-tagged steelhead detected at the nearest dam upstream of an avian colony with adequate PIT tag detection capabilities as available to avian predators at the subsequent downstream colony (*Figure 9.2* above). We restricted analyses to the middle 95% of the annual steelhead outmigration period to prevent issues associated with small weekly sample sizes of available fish which may occur during the leading and trailing weeks of each run. Steelhead availability corresponded to approximately 5–10 weeks (April–June) at inland locations and 5–8 weeks (April–June) at estuary locations. Detection data were retrieved from the PIT tag Information System (PTAGIS), a regional salmonid tag database maintained by the Pacific States Marine Fisheries Commission in January 2020 (PSMFC 2020).

Recovery of PIT tags on bird colonies followed the methods of Ryan et al. (2001) and Evans et al. (2012). Colonies were scanned for PIT tags each year after nesting birds dispersed (August–November) using flat-plate or pole-mounted PIT tag antennas. Not all tags consumed by birds are subsequently detected by researchers as (i) some proportion of tags are destroyed during the digestion process or deposited off-colony (hereafter deposition probability; ψ) and (ii) tags deposited on colony may be destroyed or lost prior to scanning efforts or be present but missed by researchers (hereafter detection probability; p; Hostetter et al. 2015). To address these processes, coinciding studies quantified annual colony-specific detection probabilities where

PIT tags with known codes were distributed across each bird colony during the breeding season and detections of these tags after the breeding season used to estimate weekly colony-specific detection probabilities (Evans et al. 2012, 2019, Hostetter et al. 2015). Previous studies also quantified tern, cormorant, and gull deposition probabilities in the Columbia River basin, which were found to vary among predator species but not spatial location (Hostetter et al. 2015). To address these processes in our study, we incorporated informative priors describing annual weekly colony-specific detection probabilities and predator-specific deposition probabilities derived from these previous results (see section *9.3.4 Predation Probabilities* for details). While more complex processes are likely associated with detection and deposition probabilities, this approach addresses the three primary concerns in this study, particularly (i) imperfect detection of deposited tags, (ii) tags deposited earlier in the season are less likely to be recovered during post-season scanning efforts, and (iii) imperfect and predator-specific differences in deposition probabilities (Evans et al. 2012, Hostetter et al. 2015).

9.3.4 Predation Probabilities

The probability of recovering an available PIT-tagged steelhead is the product of three processes: the probability (i) a steelhead was consumed (hereafter “predation probability”; ϑ), (ii) the tag from a consumed steelhead was deposited on a breeding colony (deposition probability; ψ), and (iii) a deposited tag was detected by researchers during scanning efforts (detection probability; *p;* Hostetter et al. 2015). For simplicity, parameters are indexed by week (*j*) and year (*k*), but subscripts for predator species and colony are dropped. We assume the number of PIT-tagged steelhead recovered in week *j* of year *k* (n_{jk}) is a binomial random variable. Specifically,

(1) $n_{jk} \sim \text{binomial}(N_{jk}, \theta_{jk}\psi p_{jk})$

where $N_{jk}$is the number of PIT-tagged steelhead available in week *j* of year *k*, θ_{jk} is the predation probability in week *j* of year *k*, ψ is the predator-specific deposition probability, and p_{jk} is the probability of detecting a tag deposited in week *j* of year *k*. As previously described, informative priors were used to inform detection probabilities (p_{jk}) and predator-specific deposition probability (ψ). Predation probability (θ_{jk}) was modeled as a weekly random walk on the logit-scale reflecting the tendency of predation probabilities to vary across weeks. Here,

(2) $\theta_{1k} \sim \text{beta}(1, 1)$ for week 1
(3) $\text{logit}(\theta_{jk}) \sim \text{normal}(\text{logit}(\theta_{j-1k}), \sigma_k^{\theta})$ for week 2, 3, …, *J*

where σ_k^{θ} describes the year-specific standard deviation of the random walk.

9.3.5 Functional Responses

Functional responses describe how per capita (i.e., per bird) predation probabilities are affected by prey abundance (Juliano 2001; *Figure 9.1* above). Only slight modifications to the previously described predation probability submodel (equations 2 and 3) are required to model weekly per capita predation probabilities. Specifically, for functional responses analyses we define predation probabilities (θ_{jk}) as a function of per capita predation probabilities (λ_{jk}) and predator abundance (C_{jk}),

(4) $$\theta_{jk} = \lambda_{jk} C_{jk}.$$

Which is equivalent to deriving per capita predation probabilities as the total predation probability divided by the number of predators ($\lambda_{jk} = \theta_{jk}/C_{jk}$). The formulation in eq (4), however, allows us to directly model per capita predation probabilities as a function of prey abundance.

We fit three models reflecting Type I, II, and III functional responses (Juliano 2001),

(5) $$\text{logit}(\lambda_{jk}) = \beta_{0k}^{\lambda}$$ Type I

(6) $$\text{logit}(\lambda_{jk}) = \beta_{0k}^{\lambda} + \beta_{1k}^{\lambda} N_{jk}$$ Type II

(7) $$\text{logit}(\lambda_{jk}) = \beta_{0k}^{\lambda} + \beta_{1k}^{\lambda} N_{jk} + \beta_{2k}^{\lambda} N_{jk}^{2}$$ Type III

Equation 5 assumes no relationship between per capita predation probabilities and prey abundance (N_{jk}; Type I). Equation 6 allows a monotonic relationship between per capita predation probabilities and prey abundance (Type II if $\beta_{1k}^{\lambda} < 0$), while equation 7 allows a quadratic relationship between per capita predation probabilities and prey abundance typical of a Type III response (Juliano 2001, Ellis et al. 2020; *Figure 9.1* above). Support for each functional response was evaluated by summarizing the direction and support of $\boldsymbol{\beta}^{\lambda}$ parameters (Juliano 2001, Ellis et al. 2020). Type II models require $\beta_{1k}^{\lambda} < 0$ (monotonic decline), while Type III responses require $\beta_{1k}^{\lambda} > 0$ and $\beta_{2k}^{\lambda} < 0$ (i.e. convex quadratic relationship; *Figure 9.1* above; Juliano 2001, Ellis et al. 2020).

We modelled functional responses independently for each colony, but shared information across years through a hyperparameter. Specifically,

(8) $$\beta_{0k}^{\lambda} \sim \text{normal}(\mu_0, \sigma_0)$$

(9) $$\beta_{1k}^{\lambda} \sim \text{normal}(\mu_1, \sigma_1)$$

(10) $$\log(\beta_{2k}^{\lambda}) \sim \text{normal}(\mu_2, \sigma_2)$$

where $\boldsymbol{\mu}$ and $\boldsymbol{\sigma}$ describe colony-specific means and standard deviations, respectively. We used a log-normal distribution for β_{2k}^{λ} to enforce $\beta_{2k}^{\lambda} > 0$, the n simply subtracted rather than added β_{2k}^{λ} in equation 7. We found this greatly improved convergence while also preventing ecologically implausible concave relationships that result in per capita predation probabilities indefinitely increasing as prey abundance increased (Juliano 2001).

For analysis purposes we used the number of PIT-tagged steelhead (N_{jk}) as an index for the total number of steelhead available (tagged + untagged steelhead). Weekly estimates of total steelhead abundance are generally unavailable or incomparable across river reaches (e.g., estuary vs upriver). During our study, tens of thousands of steelhead were annually tagged upriver of our study sites and volitionally released into the rivers and tributaries to continue outmigration. Thus, we expected the general trend in run-timing for tagged steelhead to be similar to the total population (Marsh et al. 1999). Changes in the numbers of PIT-tagged steelhead available across years, however, may reflect changes in tagging studies more than changes in total steelhead abundance. We therefore standardized prey abundance as weekly proportions of the total annual run to account for the possibility that intra- but not inter-annual changes in tagged steelhead abundance reflected changes in the total population. Weekly proportions sum to 1.0 within a year and functional responses describe relationships between per capita predation probabilities and changes in relative prey abundance.

9.3.6 Implementation

We implemented all models in a Bayesian framework using the software package stan (Stan Development Team 2018) accessed through R version 3.6.2 (R Development Core Team 2018) using the package rstan version 2.19.3 (Stan Development Team 2020). We ran 4 parallel Hamiltonian Monte Carlo simulations containing 2000 adaptation iterations, followed by 2000 posterior iterations. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013). Chains were not considered valid and accepted unless Gelman-Rubin statistics for all parameters were < 1.01 and zero divergent transitions were reported. Posterior predictive checks were used to evaluate model fit with respect to colony-specific weekly recovery counts (Gelman et al. 2013) and found no evidence for a lack of fit. We present estimated results as posterior medians along with 95% highest posterior density intervals (95% CrI). For brevity, in-text figures display results from 2012, a year with data available at all colonies, while results from all years are provided as supplements (see section *9.8 Supplements*).

Unless previously noted, vague priors were used for all parameters. Specifically, we used $\mathrm{inv.logit}(\mu_0) \sim \mathrm{beta}(1,1)$ for the intercept hyperparameter, $\mathrm{normal}(0, 5)$ for the regression slope hyperparameters (μ_1 and μ_2), and $\mathrm{half-normal}(0, 5)$ for standard deviations. We used informative priors to describe predator-specific deposition probabilities where

$\psi^{Tern} \sim \mathrm{beta}(16.20, 6.55)$, $\psi^{Cormorant} \sim \mathrm{beta}(15.98, 15.29)$, and $\psi^{Gull} \sim \mathrm{beta}(33.71, 183.61)$,

which reflects prior knowledge of the expected mean and variance of deposition probabilities ($\mathrm{E}[\psi]$ = 0.71, 0.51, and 0.16 for terns, cormorants, and gulls, respectively; Hostetter et al. 2015).

9.4 Results

9.4.1 Colony Monitoring

Analyses included 6–13 years of colony-specific count and tag recovery data (*Table 9.1*). During the steelhead outmigration period, bird abundance varied dramatically across weeks, colonies, and years (*Table 9.1*; *Figure 9.3*). Weekly counts of terns at ESI in the estuary ranged from 2,373–18,922 with a multi-year mean of 9358 individuals (*Table 9.1*), while weekly counts of terns nesting at CSI in Columbia Plateau region ranged from 115–729 with a multi-year mean of 496 individuals (*Table 9.1*). Weekly counts of cormorants at ESI ranged from 16,020–32,568 with a multi-year mean of 24,600 individuals, while weekly counts of cormorants at FDI in Columbia Plateau region ranged from 72–683 with a multi-year mean of 363 individuals (*Table 9.1*). At gull colonies, count data included one survey per year during peak colony attendance with average multi-year counts of 4,446 and 7,157 for gulls nesting on MRI and CSI, respectively (*Table 9.1*). Across numerous colonies, changes in weekly bird counts generally tracked changes in out-migrating steelhead abundances (*Figure 9.3*; see also *9.8. Supplements, Figures 9.S1a-g*).

Table 9.1. Data used to investigate predation probabilities and functional responses. Columns denote predator species, nesting island, years, annual number of weeks with adequate predator and prey data, weekly bird counts, and annual numbers of tagged steelhead available and recovered on a bird colony (mean and range for all columns). Avian predators included Caspian terns (Tern), double-crested cormorants (Cormorant), and California and ring-billed gulls (Gull) nesting on islands in the Columbia River basin (Crescent [CSI], Foundation [FDI], East Sand [ESI], Potholes [PTI], Miller Rocks [MRI]). See Figure 9.2 for locations of breeding colonies and hydroelectric dams.

					Steelhead	
Predator	Island	Years	Wks/yr	Num. Birds	Available	Recovered
Tern	PTI	6 (2008–13)	6 (6–7)	416 (169–589)	6,751 (5,653–7,904)	333 (281–419)
Tern	CSI	8 (2007–14)	7 (5–9)	496 (115–729)	24,866 (6,925–50,511)	454 (97–1042)
Tern	ESI	12 (2007–18)	7 (5–8)	9,358 (2,373–18,922)	19,995 (8,698–56,380)	1,289 (292–4,336)
Cormorant	FDI	6 (2007–12)	7 (5–9)	363 (72–683)	31,857 (6,925–50,511)	353 (76–653)
Cormorant	ESI	8 (2007–15)	5 (3–6)	24,600 (16,020–32,568)	20,425 (6,899–56,380)	733 (111–1,545)
Gull	CSI	8 (2007–14)	7 (5–9) †	7,157 (5,601–7,187) †	24,866 (6,925–50,511)	117 (41–249)
Gull	MRI	13 (2007–19)	7 (5–10) †	4,446 (3,223–6,016) †	14,949 (5,453–31,817)	136 (31–341)

† Counts of gull colonies occurred once per year during peak breeding season in late-May

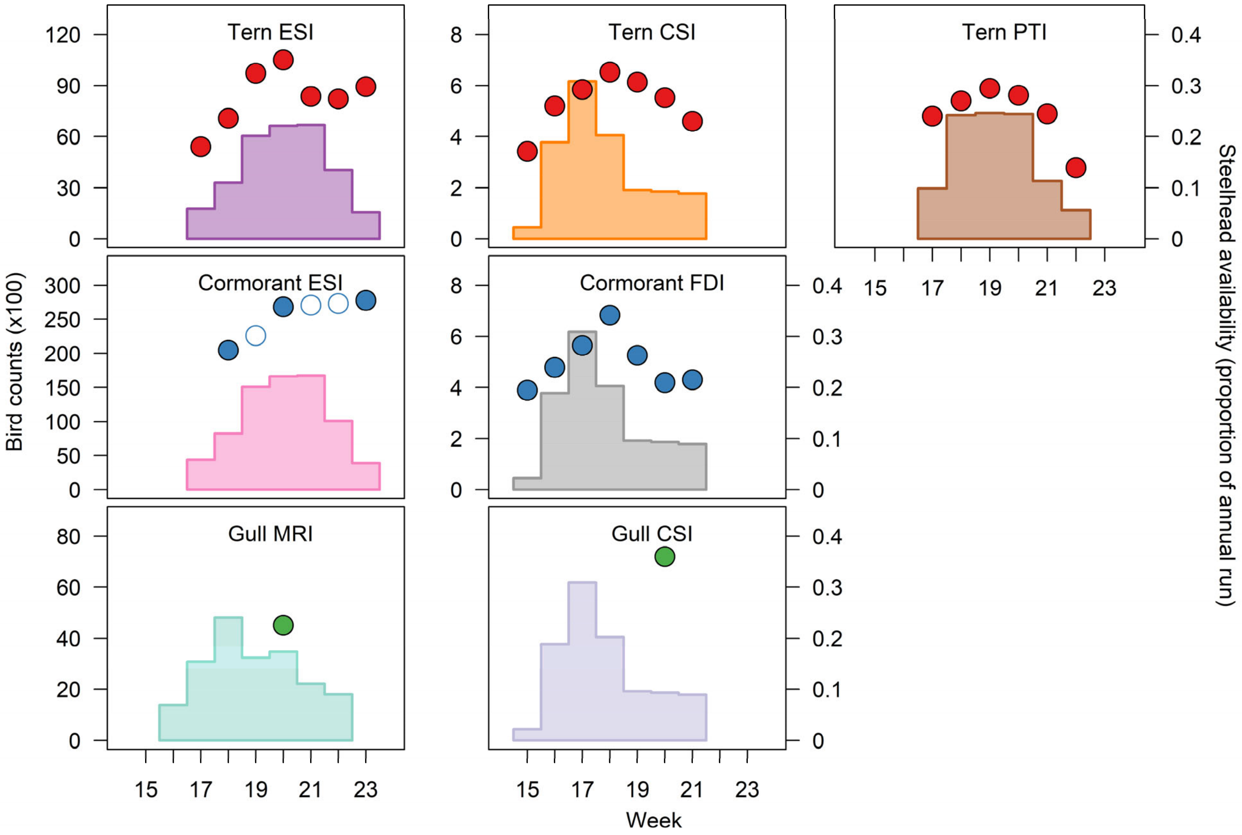

Figure 9.3. Weekly predator abundance (circles) and prey availability (histograms) by colony and bird species in 2012. Avian predators include Caspian terns (top row), double-crested cormorants (middle row), and California and ring-billed gulls (bottom row) nesting on islands in the Columbia River basin (Crescent [CSI], Foundation [FDI], East Sand [ESI], Potholes [PTI], Miller Rocks [MRI]). Y-axis scales vary by panel as noted. See 9.8. Supplements, Figures 9.S1a-g for complete set of colony × year data. For ESI cormorants, filled and open circles denote observed and interpolated counts, respectively.

9.4.2 Steelhead Trout Availability & Recovery

In total, > 645,000 PIT-tagged steelhead detections were used as a measure of prey availability in this study. From these detections, > 32,000 PIT tags were recovered on avian colonies (*Table 9.1* above). Weekly numbers of available steelhead varied from < 200 to > 18,000 individuals per week providing a large range of prey abundances to evaluate functional responses (*Table 9.1* above; *Figure 9.3* above; see also *9.8. Supplements, Figures 9.S1a-g*). The numbers of tags annually recovered on a given colony ranged from 31 (MRI gulls 2018) to 4,336 (ESI terns 2010), with the highest number of tags recovered on tern and cormorant colonies (*Table 9.1* above).

9.4.3 Predation Probabilities

Weekly predation probabilities varied by predator species, location, year, and week (*Figure 9.4*; see also *9.8 Supplements, Figures 9.S2a-g*). Weekly predation probabilities were generally greatest at tern colonies, where 0.01–0.30 of available steelhead were often consumed (*Figure 9.4*; see also *9.8 Supplements, Figures 9.S2a-g*). For cormorants, predation probabilities differed between colony locations, with weekly predation probabilities generally higher (0.01–0.20) at the large ESI cormorant colony in the estuary (thousands of breeding birds) and lower (0.01–0.06) at the much smaller colony on FDI in the Columbia Plateau region (several hundred breeding birds; *Table 9.1* above; *Figure 9.4*; see also *9.8 Supplements, Figures 9.S2a-g*). At gull colonies, weekly predation probabilities generally ranged from 0.03–0.13 (*Figure 9.4*; see also *9.8 Supplements, Figures 9.S2a-g*).

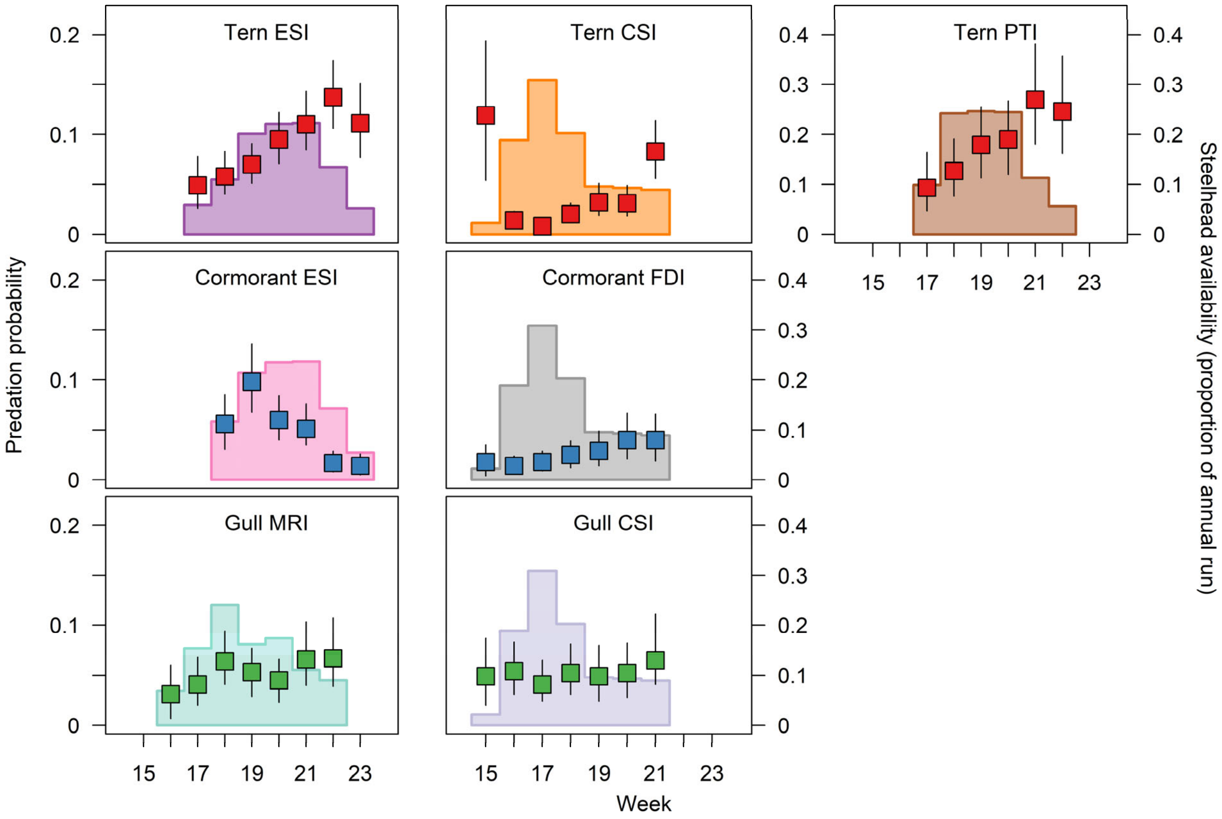

Figure 9.4. Weekly predation probabilities (squares indicate medians, vertical bars indicate 95% credible intervals) and prey availability (histograms) by colony and bird species in 2012. Avian predators include Caspian terns (top row), double-crested cormorants (middle row), and California and ring-billed gulls (bottom row) nesting on islands in the Columbia River basin (Crescent [CSI], Foundation [FDI], East Sand [ESI], Potholes [PTI], Miller Rocks [MRI]). Y-axis scales for predation probability vary for tern PTI panel all other y-axes remain consistent across panels. See 9.8 Supplements, Figures 9.S2a-g for complete set of colony × year results.

Per capita predation probabilities were an order of magnitude higher at inland tern and cormorant colonies relative to estuary colonies (*Figure 9.5*). When standardizing to 1,000 birds, tern per capita predation probabilities were generally near 0.01–0.02 in the estuary, but 0.10–0.50 at inland colonies (*Figure 9.5*; see also *9.8 Supplements, Figures 9.S3a-e*). Similarly, cormorant per capita predation probabilities standardized to 1000 birds were often near 0.005–

0.01 in the estuary, but 0.10–0.20 at the inland colony (*Figure 9.5*; see also *9.8 Supplements, Figures 9.S3a-e*). Higher per capita predation probabilities at inland relative to estuary colonies were evident across all years (see *9.8 Supplements, Figures 9.S3a-e*).

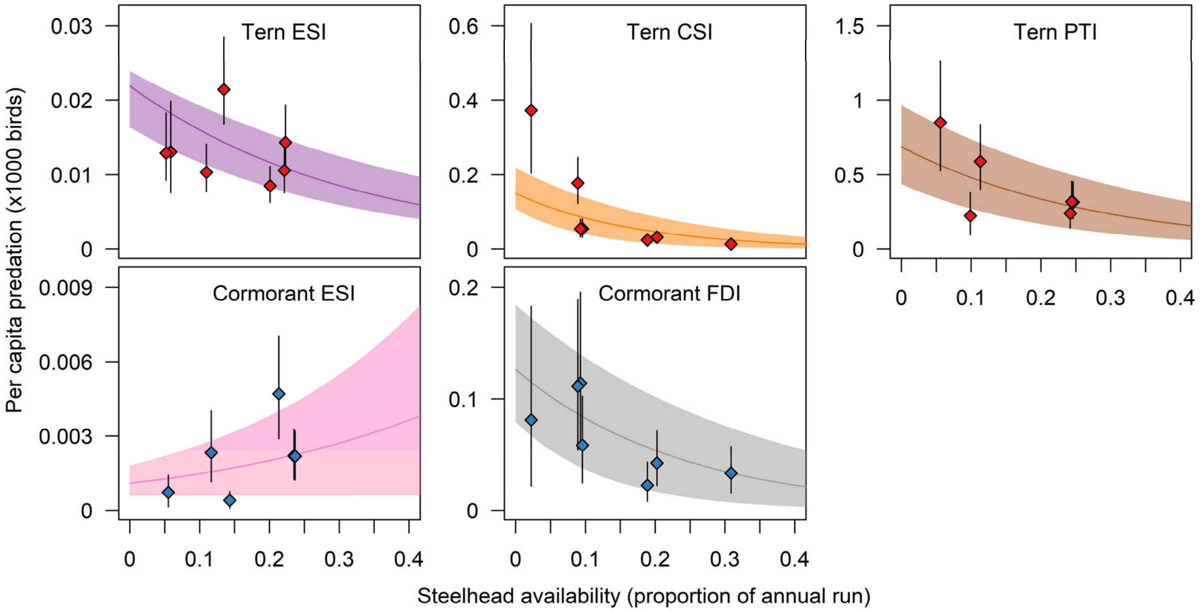

Figure 9.5. Weekly per capita predation probabilities (medians, 95% credible intervals) as a function of relative steelhead availability in 2012. Colony-specific functional response curve most supported by the data is shown (median and 95% credible interval). Avian predators include Caspian terns (top row) and double-crested cormorants (bottom row) nesting on islands in the Columbia River basin (Crescent [CSI], Foundation [FDI], East Sand [ESI], Potholes [PTI]). Y-axis scales vary by panel as noted. See 9.8 Supplements, Figures 9.S3a-e for complete set of colony × year results.

9.4.4 Functional Responses

We found overwhelming support for a Type II functional response across all three tern colonies (*Figure 9.5* above; *Table 9.2*; see also *9.8 Supplements, Figures 9.S3a-e*). At all tern colonies, per capita predation probabilities declined as prey availability increased (*Figure 9.5* above; *Table 9.2*; see also *9.8 Supplements, Figures 9.S3a-e*). Predictions from both the linear (Type II) and quadratic (Type III) models supported trends similar to those observed in empirical data, where tern per capita predation probabilities were highest at minimal prey availability and declined as prey availability increased (i.e. shape of a Type II response), with no credible support for declining predation probabilities at low prey availabilities (i.e. Type III response).

Table 9.2. Support for Type II and Type III functional responses. Type II responses require a negative relationship between per capita predation probabilities and prey abundance ($\mu_1 < 0$), while Type III responses necessitate a polynomial relationship where $\mu_1 > 0$ (i.e. per capita predation probabilities increase at low prey availability). Values are posterior medians (95% credible intervals) of the hyperparameter (μ_1) fit using the Type II and Type III models. Avian predators include Caspian terns (Tern) and double-crested cormorants (Cormorant) nesting on islands in the Columbia River basin (Crescent [CSI], Foundation [FDI], East Sand [ESI], Potholes [PTI]), and prey are PIT-tagged steelhead smolts out-migrating from the Columbia River basin.

		Type II		**Type III**	
Predator	Island	μ_1	Pr(μ_1<0)	μ_1	Pr(μ_1>0)
Tern	PTI	-3.32 (-6.52 – -0.47)	0.98	-3.02 (-6.35 – 1.05)	0.07
Tern	CSI	-5.32 (-8.40 – -2.25)	1.00	-5.11 (-8.39 – -1.92)	0.00
Tern	ESI	-2.97 (-3.50 – -2.47)	1.00	-3.16 (-3.97 – -2.15)	0.00
Cormorant	FDI	-3.03 (-6.10 – 0.53)	0.96	-2.82 (-6.22 – 0.71)	0.06
Cormorant	ESI	3.01 (0.20 – 5.83)	0.02	3.27 (0.26 – 6.26)	0.99

For terns nesting in the estuary (ESI), there was a 1.00 probability μ_1< 0 in the Type II model indicating per capita predation probabilities declined with increasing steelhead abundance (*Table 9.2* above). Median estimates of annual relationships were negative in all 12 years analysed for ESI terns (i.e. β_{1k}^{λ}< 0 for all *k*; see *9.8 Supplements, Figures 9.S3a-e*). Conversely, the Type III model provided no evidence of a decline in predation at low prey availability (Pr(μ_1>0) = 0.00 in Type III model; *Table 9.2* above). At the CSI (inland) tern colony, there was also a 1.00 probability μ_1< 0 in the Type II model indicating per capita predation probabilities declined with increasing steelhead abundance (*Table 9.2* above). We found no credible support for a Type III response at the CSI tern colony (Pr(μ_1>0) = 0.00 in Type III model; *Table 9.2* above). At the PTI tern colony, probability μ_1< 0 in the Type II model was 0.98 (*Table 9.2* above). Support for a Type III response was low, with a 0.07 probability that per capita predation probabilities declined at low prey availability (Pr(μ_1>0) = 0.07 in Type III model; *Table 9.2* above).

Cormorant functional responses varied by colony location (*Table 9.2* above). For cormorants nesting in the estuary (ESI), there was strong support for a Type III response where per capita predation probabilities initially increased at low prey availability (*Table 9.2* above; *Figure 9.5* above; see also *9.8 Supplements, Figures 9.S3a-e*). Here, the Type III hyperparameter describing how per capita predation probabilities initially increased at low prey availability (μ_1) was 3.27 (95% CrI = 0.26–6.26) with a 0.99 probability μ_1> 0 (*Table 9.2* above). Both the Type II and Type III models indicated that ESI cormorant per capita predation probabilities increased as prey availability increased (*Table 9.2* above), but the inflection point at which per capita predation probabilities must decline (e.g., due to handling or digestion time; Type III) was not observed.

At the inland cormorant colony (FDI), per capita predation probabilities declined as prey availability increased (Type II) with a 0.96 probability μ_1< 0 (*Table 9.2* above; *Figure 9.5* above). Support for a Type III response was minimal, with only a 0.06 probability that per capita predation probabilities declined at low prey availability (*Table 9.2* above). Type II responses were supported in all five years at the FDI cormorant colony (see *9.8 Supplements, Figures 9.S3a-e*).

We were unable to quantify functional responses for the CSI and MRI gull colonies due to a lack of data on weekly gull abundance (*Table 9.1* above). Gull predation probabilities often remained relatively consistent across large ranges of prey availability (*Figure 9.4* above; see also *9.8 Supplements, Figures 9.S2.a-g*). CSI and MRI gull colonies consisted of > 3,000 individuals when counted (*Table 9.1* above) with gull nesting activities typically encompassing the entire steelhead outmigration period of April to June. Although not explicitly quantifiable, results suggest a Type I or III response for gulls due to consistent predation probabilities across wide variation in prey availability and the presence of large numbers of gulls during the outmigration period (*Figure 9.3* and *Figure 9.4* above; see also *9.8 Supplements, Figures 9.S3a-e*). As such, gull per capita predation probabilities are likely low and relatively flat across wide variation in prey availability (Type I), while reductions in per capita predation probabilities may occur at prey densities greater than those observed in this study (Type III).

9.5 Discussion

Terns, cormorants, and gulls consumed juvenile steelhead throughout the entire outmigration period. Per capita tern predation probabilities were highest when steelhead availability was low and generally declined as prey availability increased, reflecting declining predator efficiency as prey availability increases (Type II response). Tern Type II responses were consistent across colonies located in both freshwater and estuarine ecosystems. Cormorant functional responses varied by location, where a Type II response was supported at a colony located in a strictly freshwater area and Type III response supported at a colony located in an estuarine area. Functional responses for gulls could not be explicitly quantified but weekly predation probabilities remained relative consistent across wide ranges in prey availability suggestive of a Type I or III response. Our results support hypotheses of prey switching for generalist predators exposed to diverse prey bases (e.g., gulls, cormorants in an estuary; Type III response) and predator swamping theories for specialist predators exposed to pulses in prey availability (e.g., terns; Type II response). Additionally, findings from this study provide novel insights into high and variable avian predation probabilities on out-migrating juvenile steelhead across multiple predator species (Evans et al. 2012, 2019, Hostetter et al. 2015, Payton et al. 2019), predator swamping by synchronous pulses of migratory prey (Ims 1990, Tucker et al. 2008, Hostetter et al. 2012, Furey et al. 2016), and variation in predator functional responses related to predator foraging strategies and spatial processes (O'Donoghue et al. 1998a, 1998b, Roby et al. 2002 p. 200, Gilg et al. 2006, Vucetich et al. 2011, Armstrong et al. 2016).

Consistent with our predictions, terns displayed persistent Type II functional responses across years and colonies. Type II functional responses were stronger and more consistent for terns (specialists) than cormorants (generalists), which displayed Type II and Type III responses at an inland and estuary colony, respectively. Generalist predators often follow Type III functional responses due to their ability to better exploit resource pulses relative to specialist predators (O'Donoghue et al. 1998b, Yang et al. 2008, Armstrong et al. 2016). For example, cormorant and gull diets are more diverse than tern diets and reflect a shifting mosaic of foraging opportunities in the Columbia River basin (Collis et al. 2002). Relative to more specialist tern diets, cormorants and gulls may switch prey in response to dynamic pulses in resources integrating across a wider range of prey sizes (Hostetter et al. 2012, Osterback et al. 2014), species (Collis et al. 2002, DeBruyne et al. 2013), and in the case of gulls, terrestrial food sources (Collis et al. 2002, Winkler 2020).

Predator-swamping is more likely to occur in central place foraging and movement limited species relative to highly mobile species that can track or respond to heterogenous resources (Yang et al. 2008, Armstrong et al. 2016 and citations therein). For example, based on median travel times during outmigration, individual steelhead generally remain within foraging range of an individual tern colony for < 4 days even though terns may forage > 80 km from their colony (Smith et al. 2002, Harnish et al. 2012, Evans et al. 2016). As such, time becomes a limiting constraint in the ability of birds to exploit pulses in juvenile out-migrating salmonids. Prey switching and behavioral aggregation processes, however, allows consumers to track resource waves, extend foraging opportunities, and increase resource-specific consumption during periods of resource pulses (Holling 1959a, Armstrong et al. 2016). For example, the relatively consistent weekly predation probabilities by gulls across wide ranges of steelhead availability may be a function of prey switching or behavioral aggregation, where gulls dramatically increase steelhead consumption during pulses in abundance and forage elsewhere when steelhead abundance is low.

In the estuary, cormorant per capita predation probabilities increased as steelhead availability increased implying that the *number* of prey consumed per predator increases as availability increases. An increasing functional response does not reflect limitless predation, but instead may be attributable to the low contribution of specific prey to the diet of a predator (Chan et al. 2017, Ellis et al. 2020). Herein, we likely observed the lower end of a Type III response for ESI cormorants, which is difficult to distinguish from a Type I response (*Figure 9.1* above; Jeschke et al. 2004, Chan et al. 2017). Consumption rates that are strictly increasing (Type I response) are unlikely for most species but have been observed in field studies where a prey species is a low proportion of a predator's diet and a satiation point remains undetected (Jeschke et al. 2004, Chan et al. 2017, Ellis et al. 2020). Collis et al. (2002) observed that juvenile salmonids accounted for 16% of the diet of cormorants nesting on East Sand Island and just 3% of diet of gulls nesting on Miller Rocks Island. By comparison, salmonids accounted for > 30% and 70% of the diet of terns nesting on East Sand Island and Crescent Island, respectively (Roby et al. 2002, Antolos et al. 2005). Due to this lower reliance on salmonids as a primary prey source, cormorant and gull per capita consumption can substantially increase during pulses of high steelhead availability, resulting in relatively stable or even increasing per capita predation probabilities across large ranges of steelhead availability.

Functional response analyses often involve three related objectives, (1) identifying the shape of a functional response, (2) comparing the support for Type I, II, or III functional responses, and (3) exploring the mechanistic relationships affecting functional responses (e.g., Holling's disc equation [Holling 1959b]; Juliano 2001). Our study addressed objectives #1 and #2, with empirical information comparing the consistency of functional responses across multiple predators, locations, and years. Our use of proportional changes in prey availability allowed detailed insights into predator responses to changes in relative prey density (objectives 1 and 2), however, more detailed information on prey abundance is required to identify the prey density at which predator satiation occurs or the mechanisms causing these relationships (e.g., handling time, exposure time; Holling 1959a, 1959b, Juliano 2001, Tully et al. 2005). Technological and sampling advances may provide new opportunities to directly estimate spatiotemporal juvenile salmonid abundance in the Columbia River basin (e.g., PIT tag detectors that span the width of the river and randomized sampling schemes) that will greatly assist future functional response studies. Similarly, integrating data on predator diets, abundance, and mark-recovery studies will be crucial for identifying the mechanistic processes leading to prey saturation in terns and the possible lack thereof in ESI cormorants.

Quantifying the shape of predator functional responses is critical to determining how predation affects prey populations. Rapid responses to resource pulses emphasize relationships with local surroundings including behavioral aggregation and prey switching (Yang et al. 2008). Changes in weekly steelhead availability are predominantly driven by non-predation factors such as hatchery releases, water flows, and temperatures (Zabel 2002, Smith et al. 2002). As such, our study addresses how multiple predator species respond to changes in seasonal prey availability and the consistency of these relationships across locations and years. Avian predation is linked to changes in steelhead outmigration survival and adult return rates (Payton et al. 2020), however, demographic responses related to predator foraging strategies likely occur at different time scales than measured herein. Many fish species aggregate in high densities during critical life-history events (e.g., juvenile migration, adult spawning) even when overall abundance may be low. Understanding if prey aggregations continue to attract predators at low abundances (e.g., Gende et al. 2002, Quinn et al. 2003, Middleton et al. 2013, Osterback et al. 2013) and how this influences individual prey susceptibility to predation (i.e., predator swamping; Ims 1990, Tucker et al. 2008, Hostetter et al. 2012, Furey et al. 2016) can be crucial for understanding the efficacy of predator management strategies to improve prey population persistence. For example, predator attraction (i.e., behavioral response) may result in high levels of predation that reduce prey abundance, drive density dependent Allee effects (Allee 1931), and destabilize prey populations at low abundances. In these situations, predation may affect the recovery of prey populations even if it was not a cause of the original decline (Neuenhoff et al. 2018). Conversely, declines in predation at low prey abundances (Type III functional response) are stabilizing and, while they can regulate prey populations, they are less likely to drive populations toward extinction (Solomon 1949).

For ESA-listed salmonid populations, predator functional responses can play a critical role in evaluating the impacts of predation and predator management strategies. For example, impacts of tern predation may be highest for early- or late-season migrants due to predator swamping during periods of high prey abundance. Conversely, high concentrations of prey may attract cormorants and gulls that target periods of peak prey availability, thus leading to increased salmonid susceptibility at specific locations or times (e.g., en masse releases of hatchery reared steelhead smolts, at dams where smolts congregate, or during periods of highly synchronized out-migration). Understanding predator-specific responses can guide system-specific management actions focused on local- (e.g., areas of prey concentrations) and landscape-level (e.g., inland vs estuary) factors. Quantifying how predators respond to pulses in prey availability has direct implications to behavioral ecology and predator-prey dynamics, while also providing critical information to identify predator- and location-specific predation impacts (Roby et al. 2002; Osterback et al. 2013; Payton et al. 2020), which can in turn guide targeted- and system-specific actions aimed at conserving prey populations of management concern.

Our approach to evaluating functional responses across predators, space, and time provides insights into how different predators respond to changes in resource availability, and why some predators may not respond at all. Foraging strategy and spatial mechanisms (e.g., prey switching, behavioral responses, predator swamping) contributed to functional responses that were consistent across years and locations for one predator species (terns), but differed among locations for a second predator species (cormorants). Our results illustrate the complexity of predator–prey interactions in natural systems and highlight the need to jointly investigate predator and prey abundance, predation probabilities, per capita predation probabilities, and predator foraging strategies to understand processes affecting complex communities of predators and their prey.

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9.8 Supplements

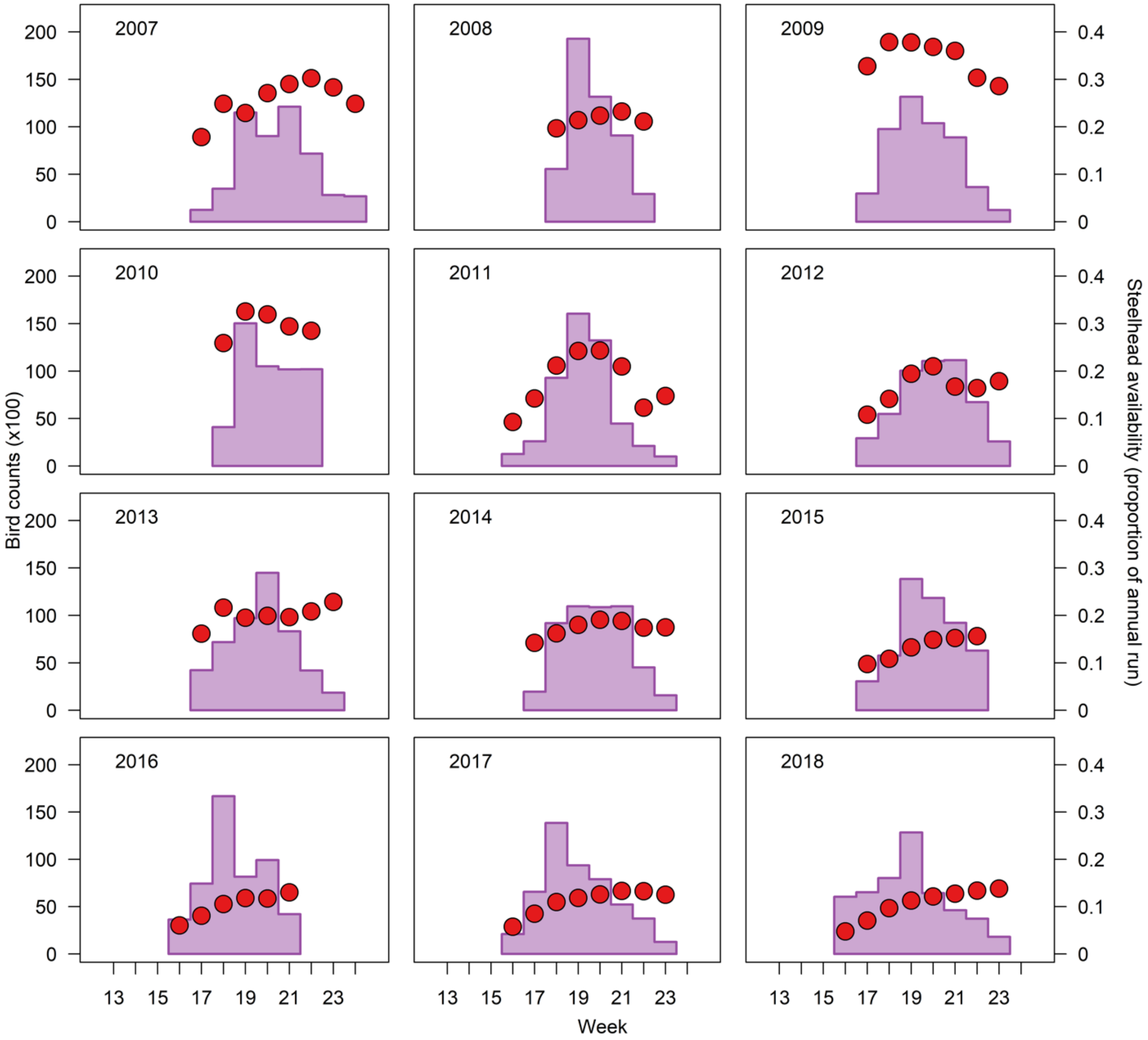

Figure 9.S1a. Weekly abundance of Caspian terns on the East Sand Island colony (circles) and relative availability of tagged steelhead smolts in the Columbia River estuary (histograms) by year during 2007–2018.

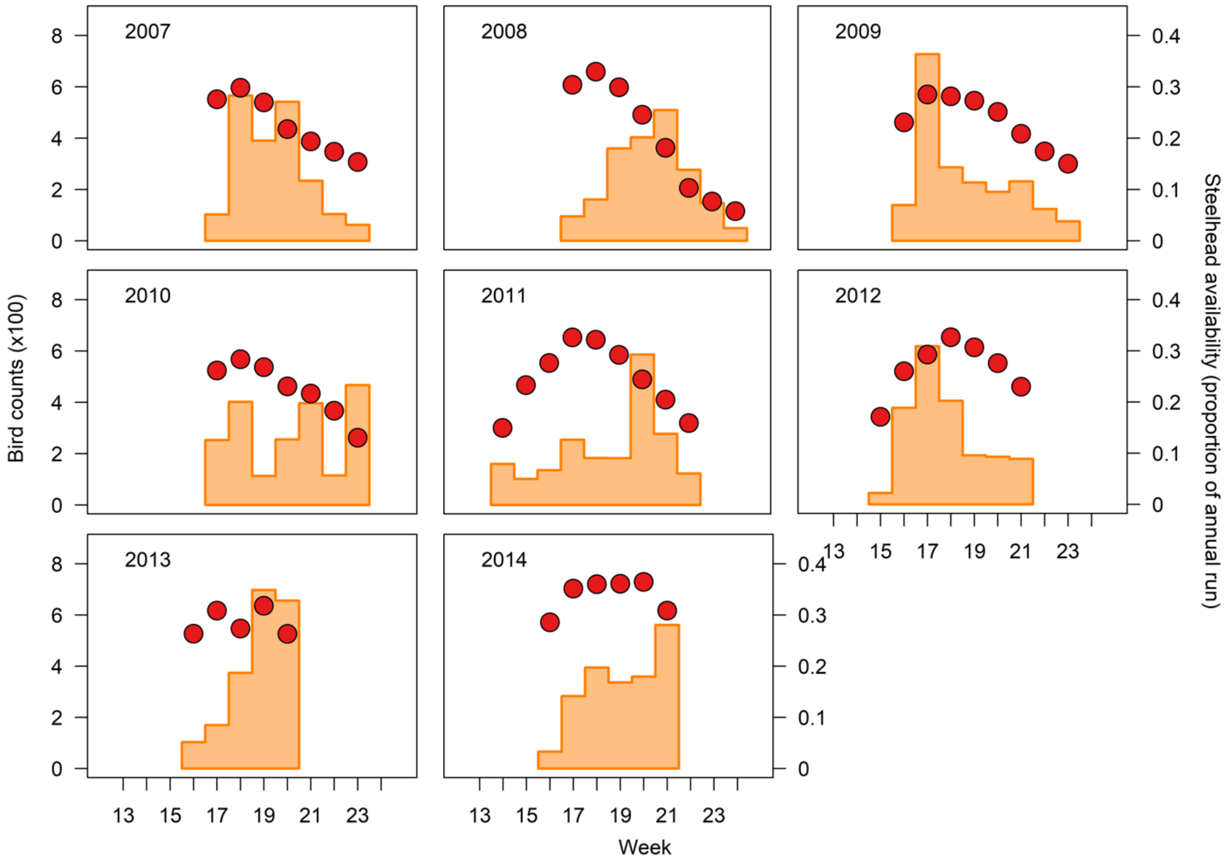

Figure 9.S1b. Weekly abundance of Caspian terns on the Crescent Island colony (circles) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2007–2014.

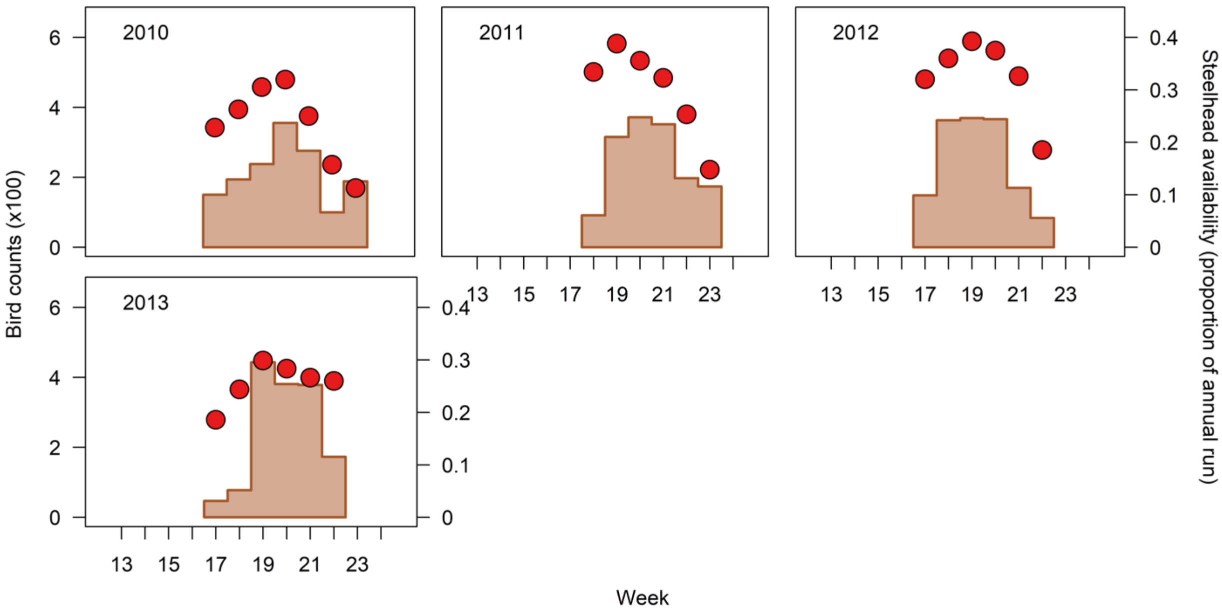

Figure 9.S1c. Weekly abundance of Caspian terns on the Potholes Island colony (circles) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2010–2013.

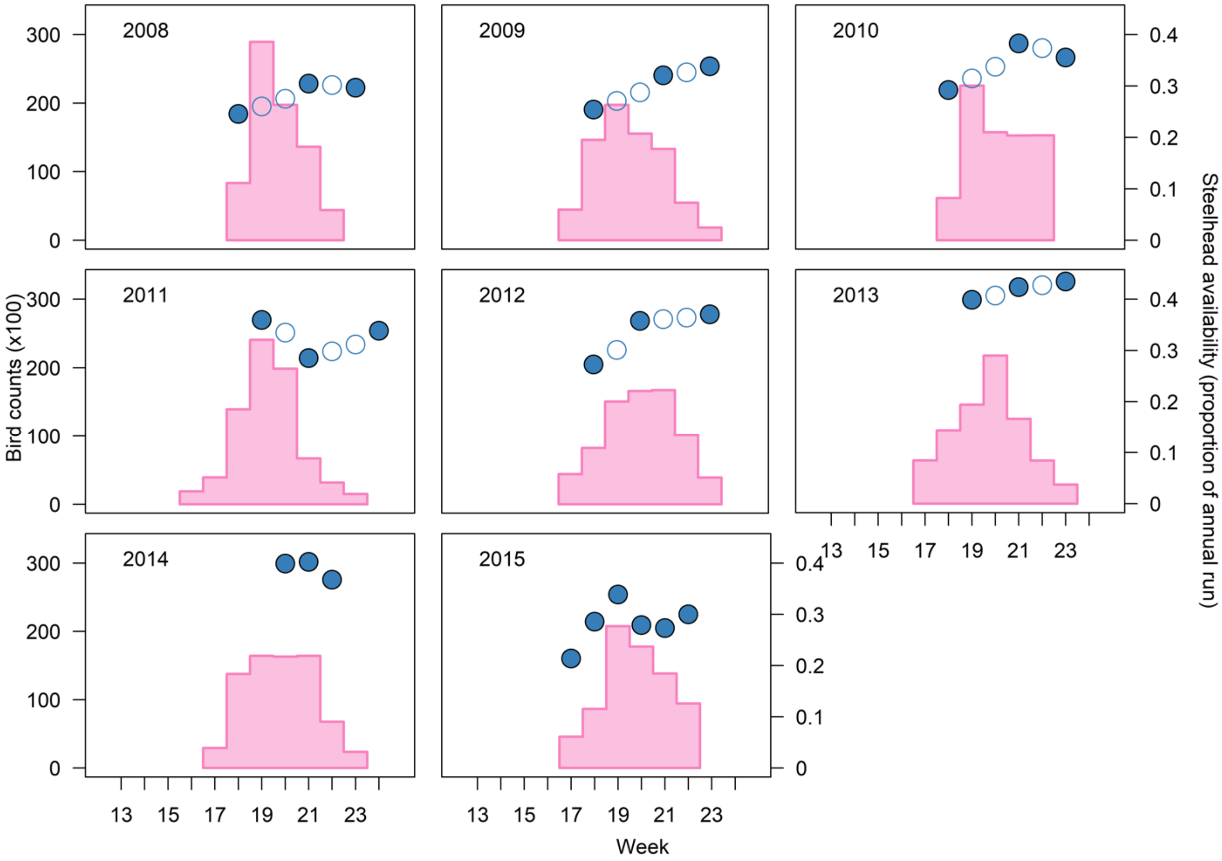

Figure 9.S1d. Weekly abundance of double-crested cormorants on the East Sand Island colony (circles) and relative availability of tagged steelhead smolts in the Columbia River estuary (histograms) by year during 2008–2015. Filled and hollow circles denote observed and interpolated counts, respectively. See Methods in main text for additional details.

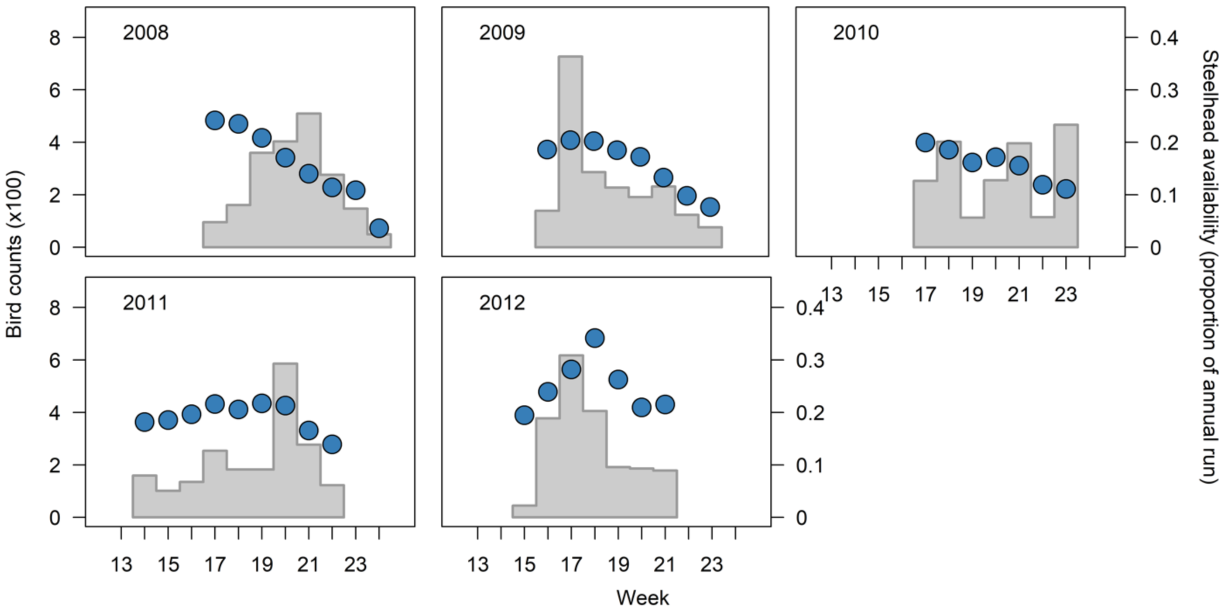

Figure 9.S1e. Weekly abundance of double-crested cormorants on the Foundation Island colony (circles) and relative availability of tagged steelhead in the mid-Columbia River (histograms) by year during 2008–2012.

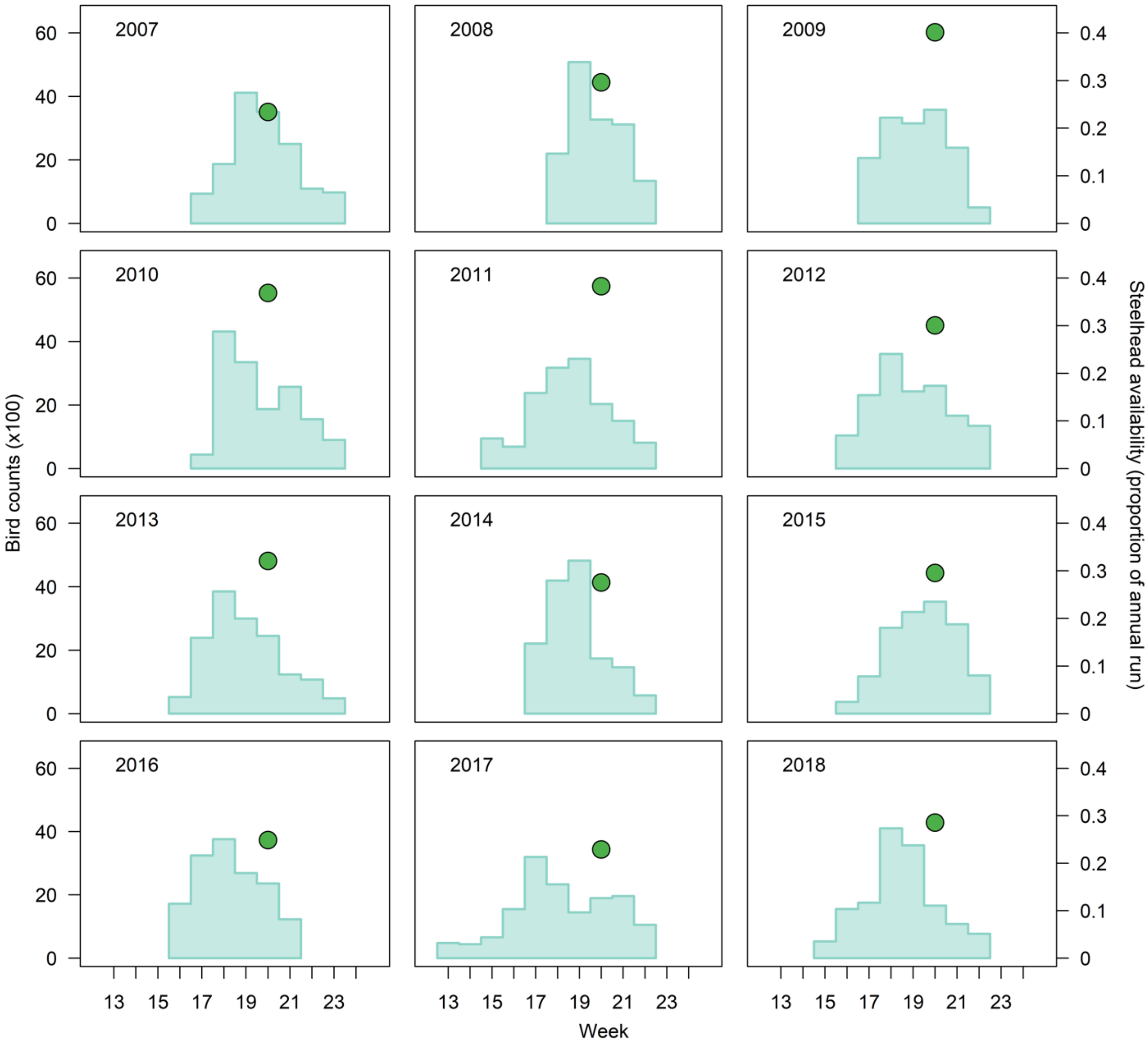

Figure 9.S1f. Weekly abundance of California and ring-billed gulls on the Miller Rocks colony (circles) and relative abundance of tagged steelhead smolts in the mid-Columbia river (histograms) by year during 2007–2018. Gull colony counts occurred once per year during the peak of the breeding season in late-May.

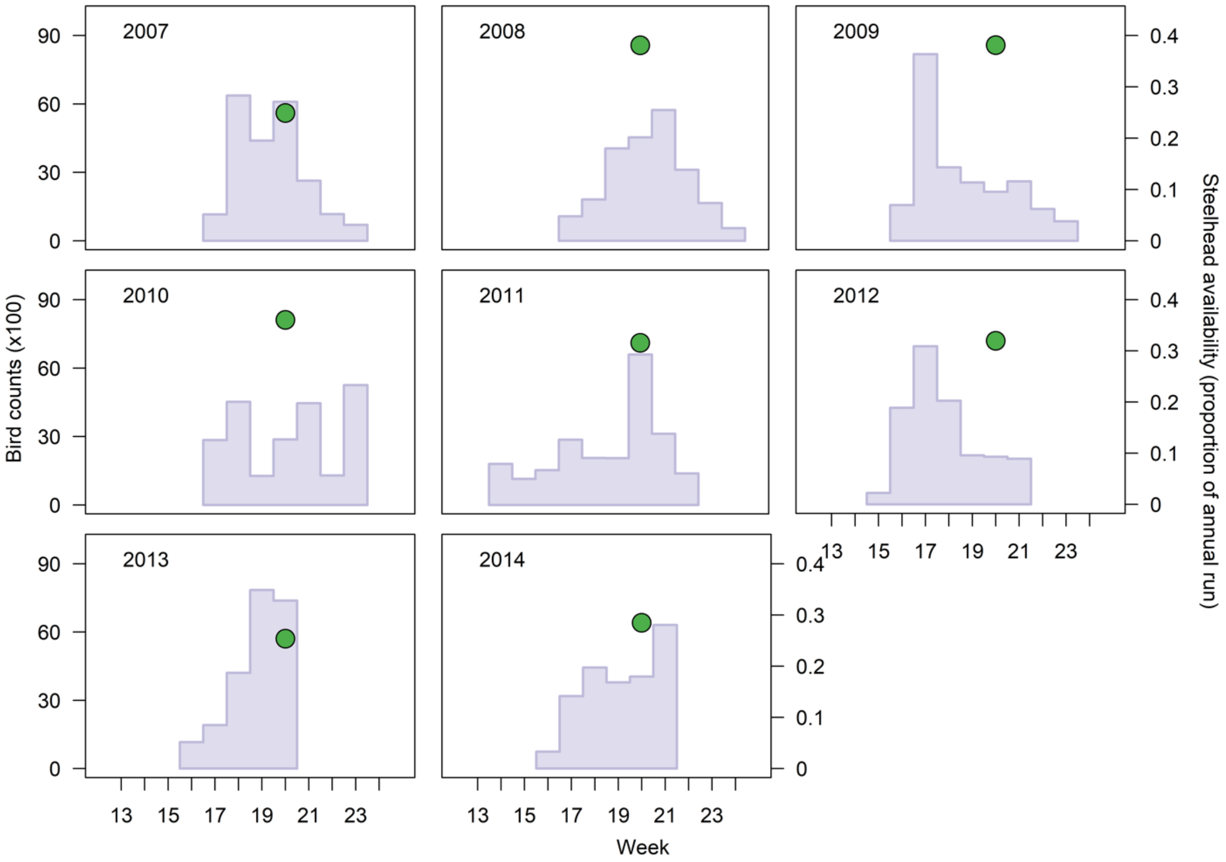

Figure 9.S1g. Weekly abundance of California and ring-billed gulls on the Crescent Island colony (circles) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2007–2014. Gull colony counts occurred once per year during the peak of the breeding season in late-May.

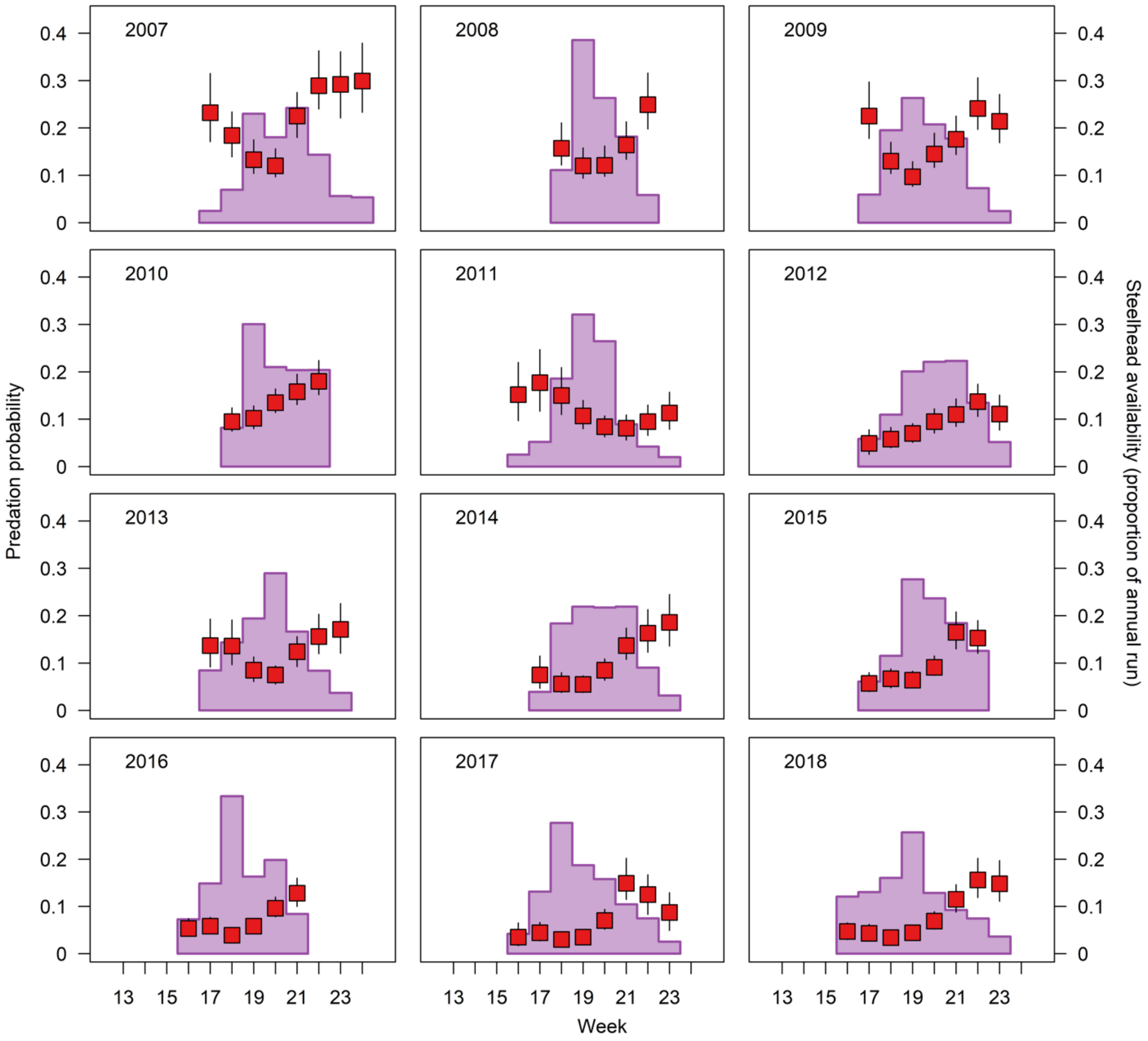

Figure 9.S2a. Weekly predation probabilities for Caspian terns nesting on the East Sand Island colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the Columbia River estuary (histograms) by year during 2007–2018.

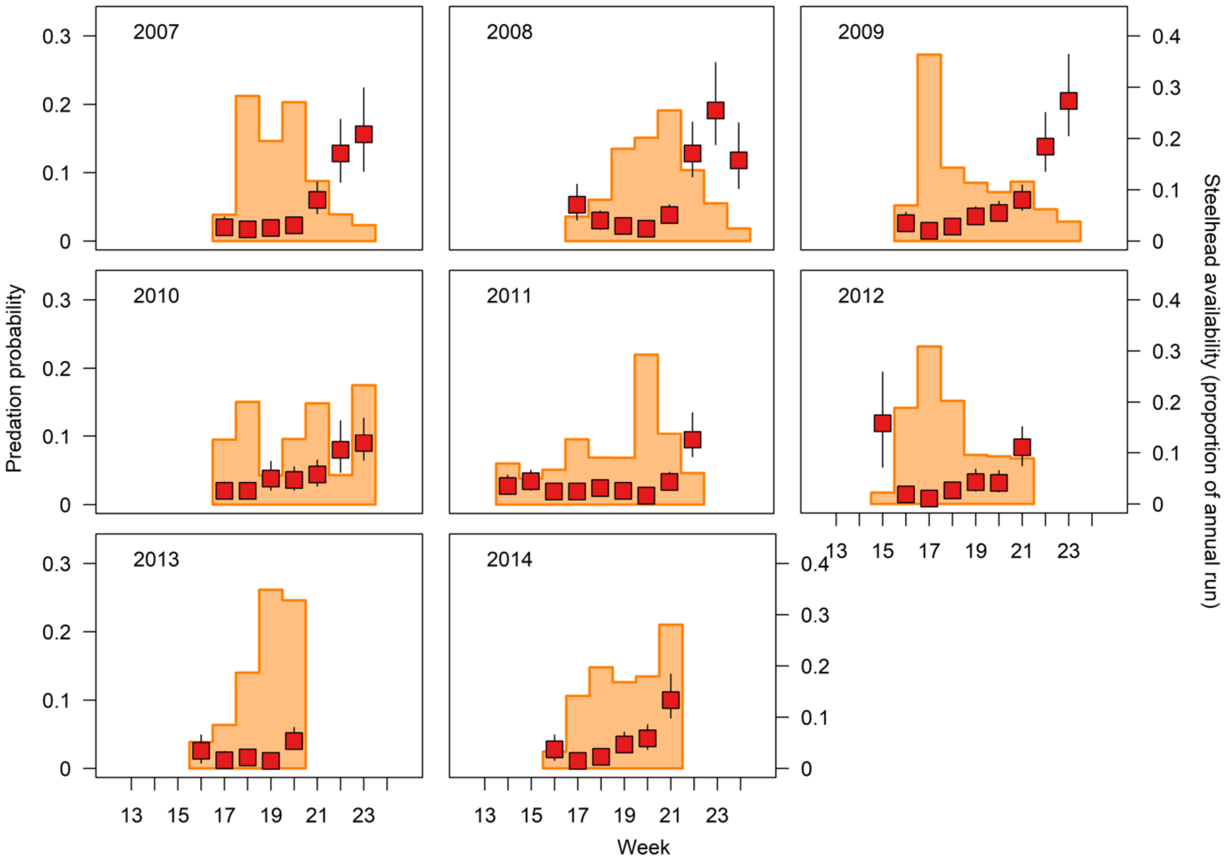

Figure 9.S2b. Weekly predation probabilities for Caspian terns nesting on the Crescent Island colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2007–2014.

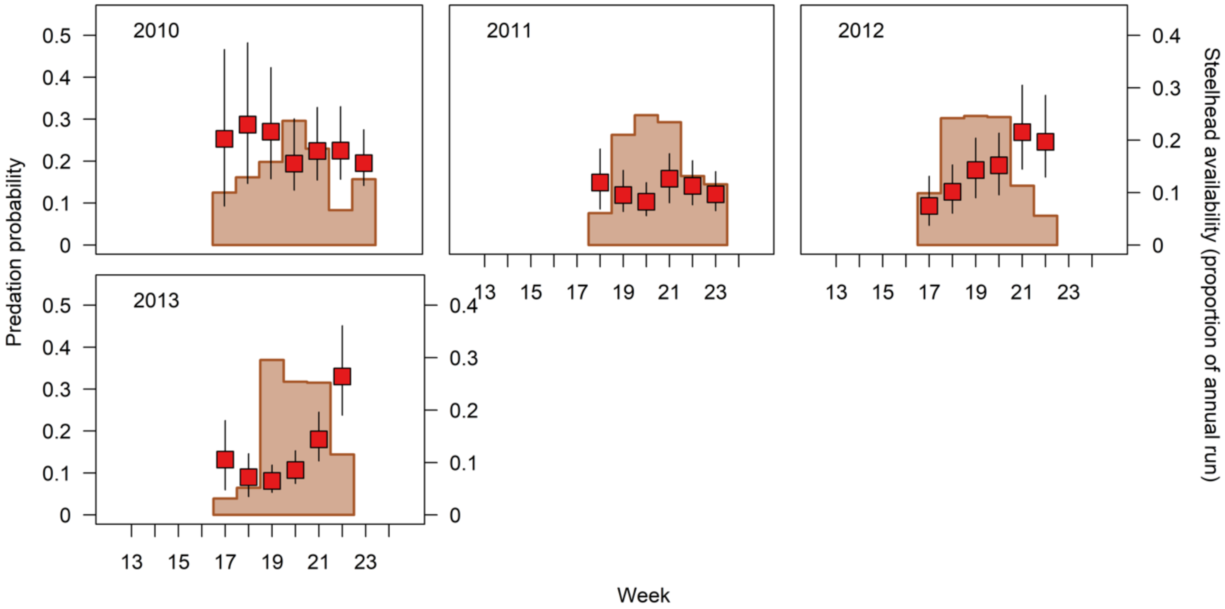

Figure 9.S2c. Weekly predation probabilities for Caspian terns nesting on the Potholes Island colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2010–2013.

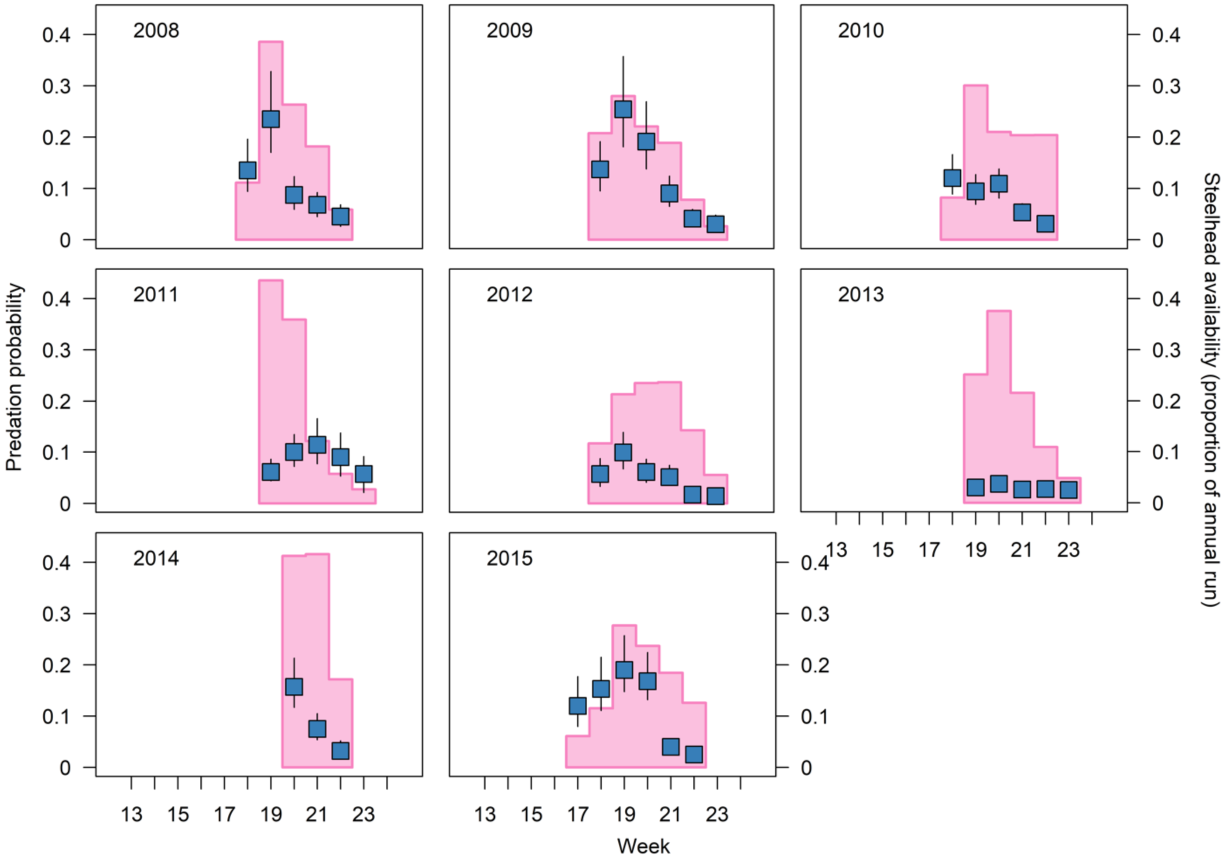

Figure 9.S2d. Weekly predation probabilities for double-crested cormorants nesting on the East Sand Island colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the Columbia River estuary (histograms) by year during 2008–2015.

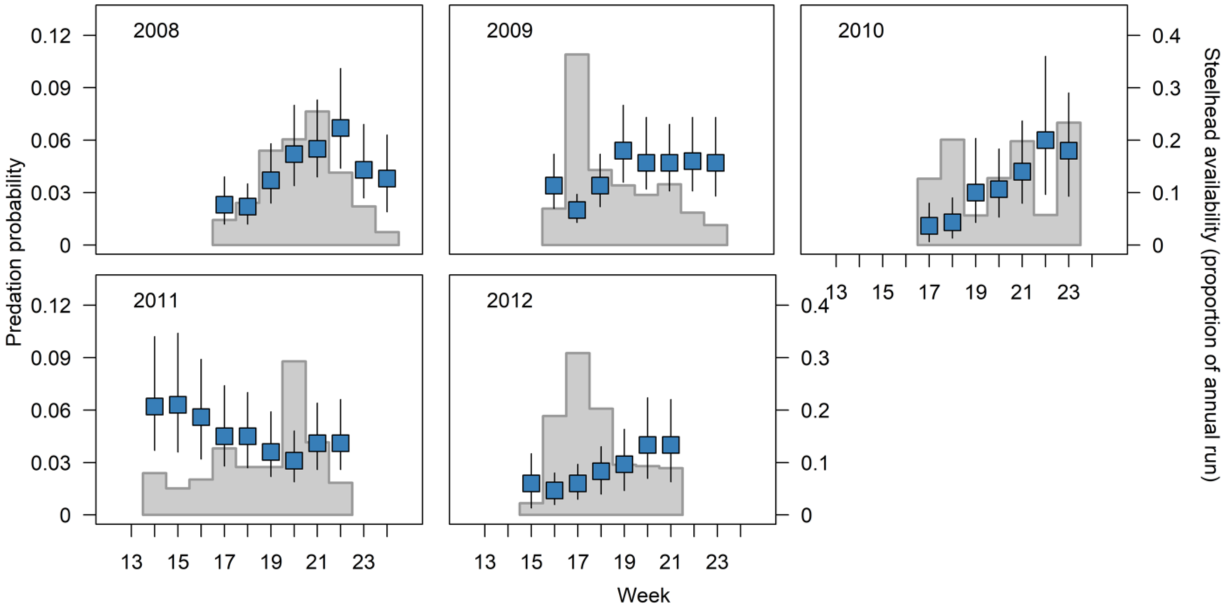

Figure 9.S2e. Weekly predation probabilities for double-crested cormorants nesting on the Foundation Island colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2008–2012.

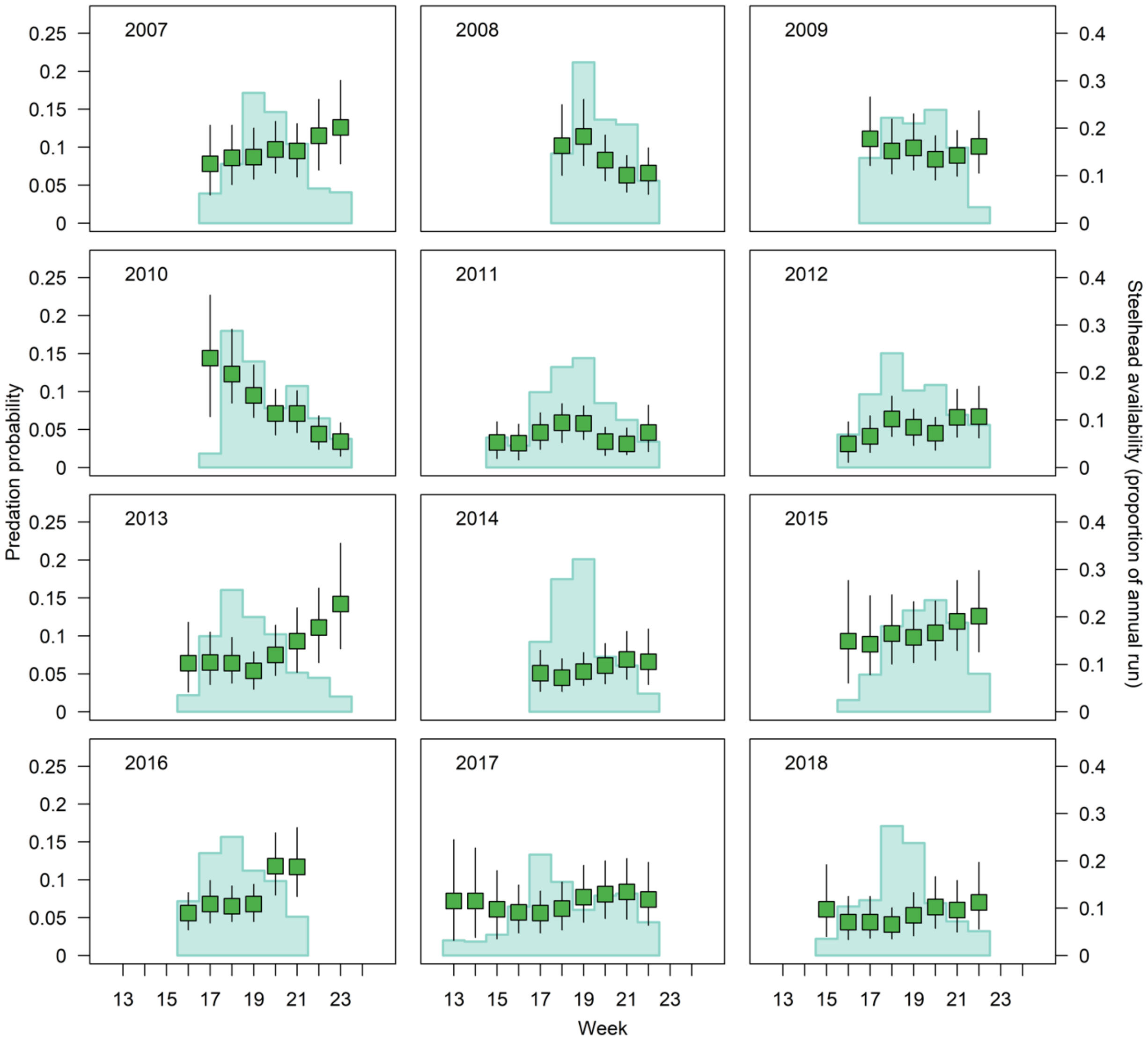

Figure 9.S2f. Weekly predation probabilities for California and ring-billed gulls nesting on the Miller Rocks colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2007–2018.

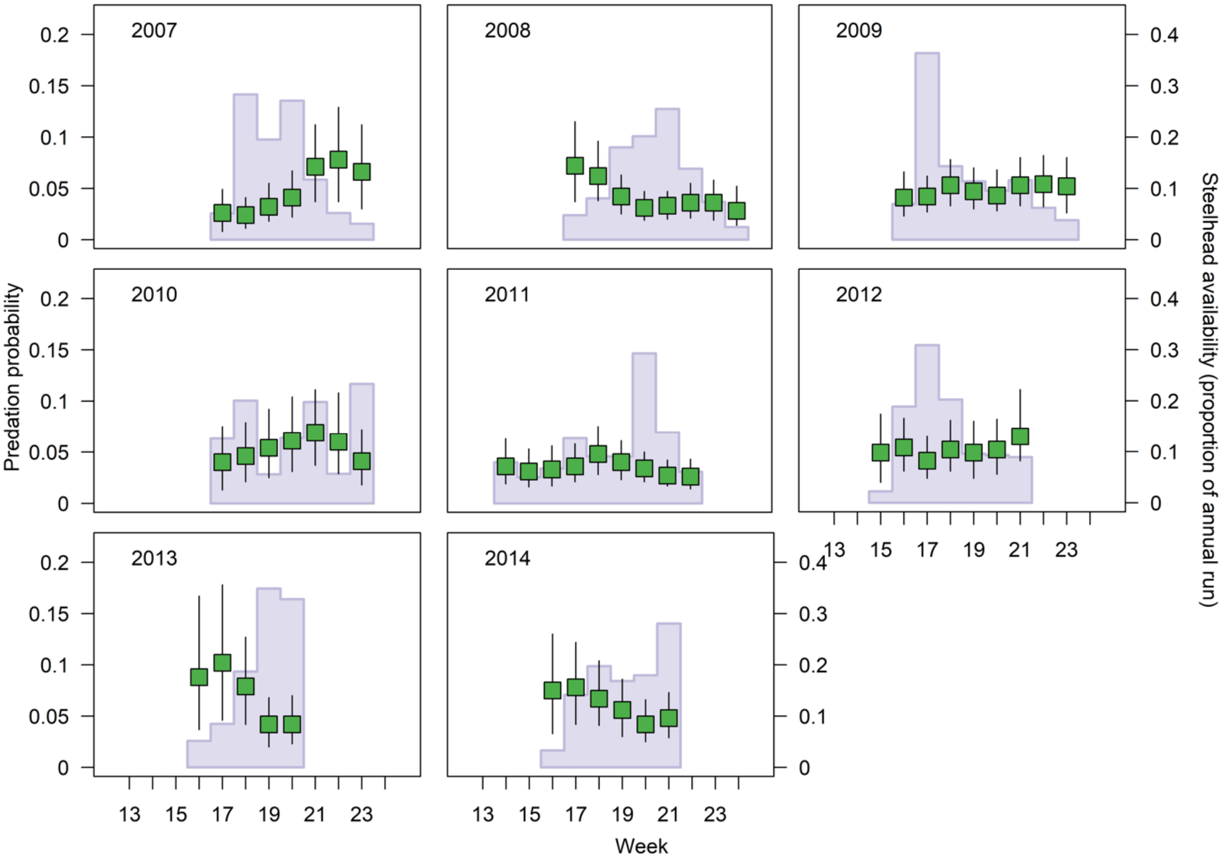

Figure 9.S2g. Weekly predation probabilities for California and ring-billed gulls nesting on the Crescent Island colony (squares; median and 95% credible interval) and relative availability of tagged steelhead smolts in the mid-Columbia River (histograms) by year during 2007–2014.

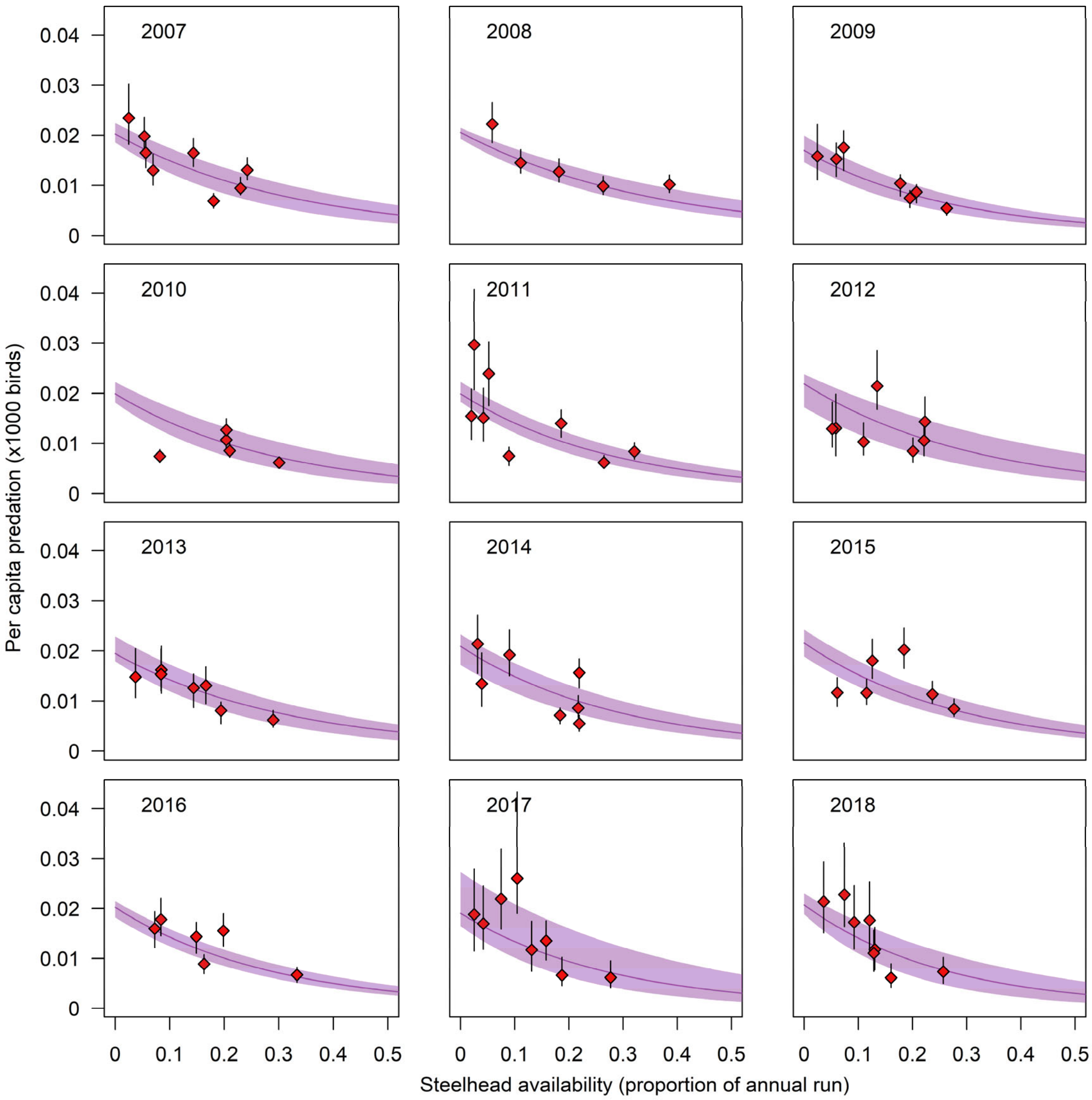

Figure 9.S3a. Weekly per capita predation probabilities for Caspian terns nesting on the East Sand Island colony (medians, 95% credible intervals) as a function of relative availability of tagged steelhead smolts in the Columbia River estuary by year during 2007–2018. The functional response curve most supported by the data is shown for each year (Type II; median and 95% credible interval).

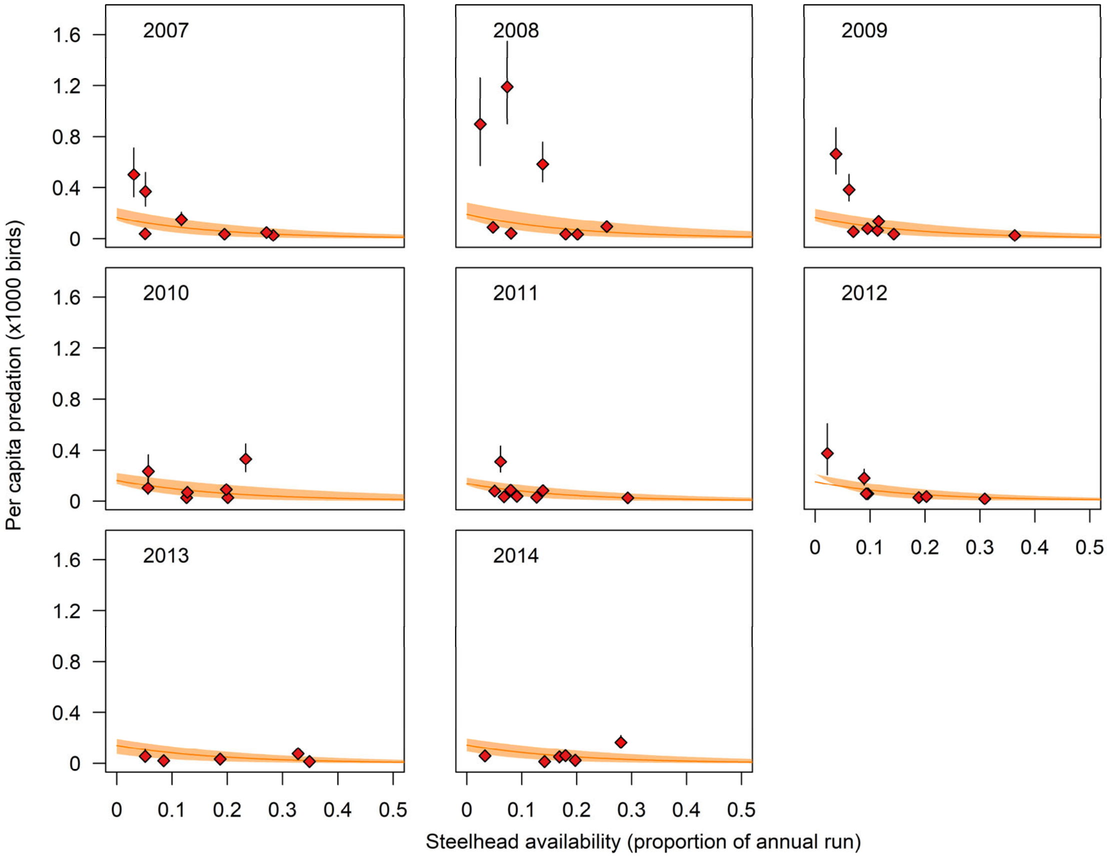

Figure 9.S3b. Weekly per capita predation probabilities for Caspian terns nesting on the Crescent Island colony (medians, 95% credible intervals) as a function of relative availability of tagged steelhead smolts in the mid-Columbia River by year during 2007–2014. The functional response curve most supported by the data is shown for each year (Type II; median and 95% credible interval).

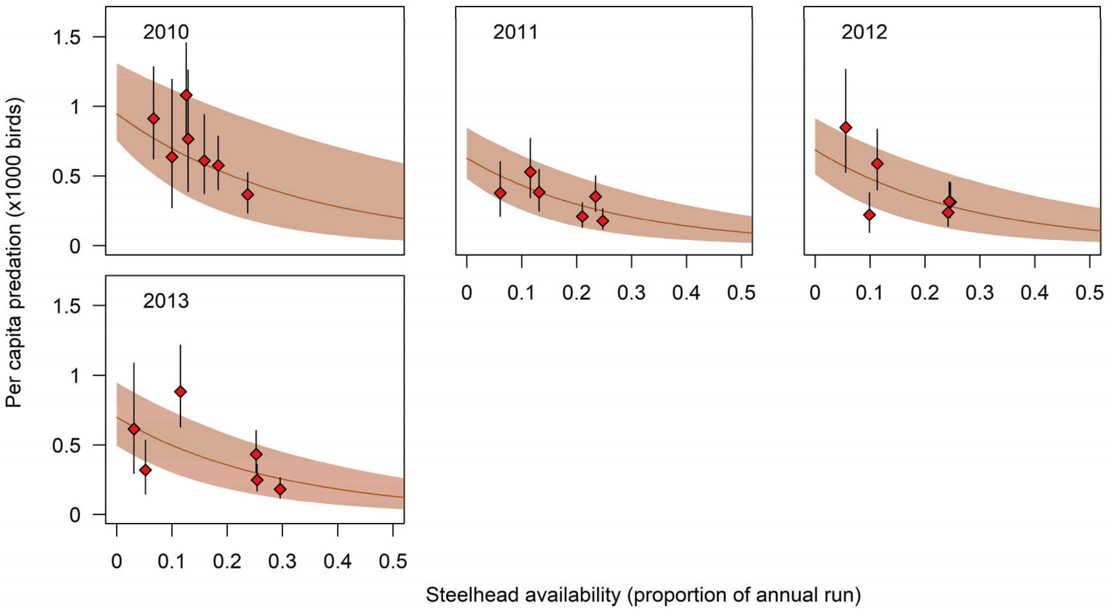

Figure 9.S3c. Weekly per capita predation probabilities for Caspian terns nesting on the Potholes Island colony (medians, 95% credible intervals) as a function of relative availability of tagged steelhead smolts in the mid-Columbia River by year during 2010–2013. The functional response curve most supported by the data is shown for each year (Type II; median and 95% credible interval).

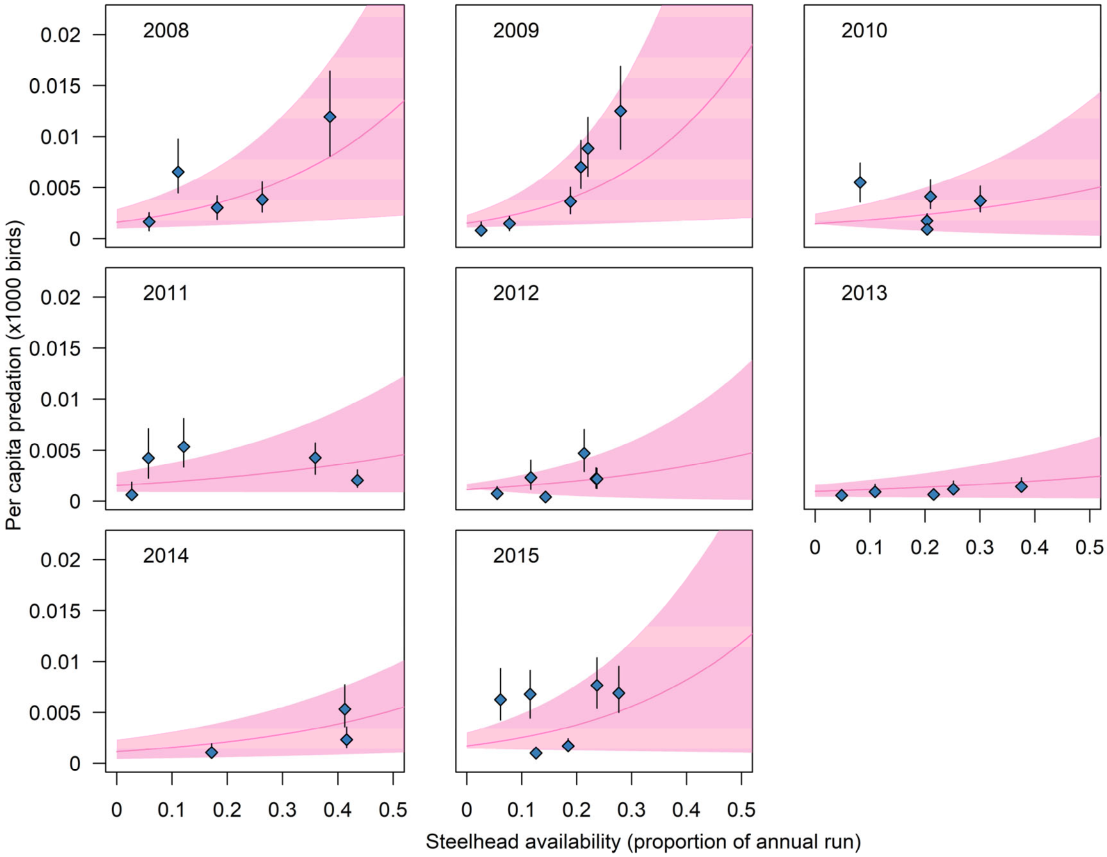

Figure 9.S3d. Weekly per capita predation probabilities for double-crested cormorants nesting on the East Sand Island colony (medians, 95% credible intervals) as a function of relative availability of tagged steelhead smolts in the Columbia River estuary by year during 2008–2015. The functional response curve most supported by the data is shown for each year (Type III; median and 95% credible interval).

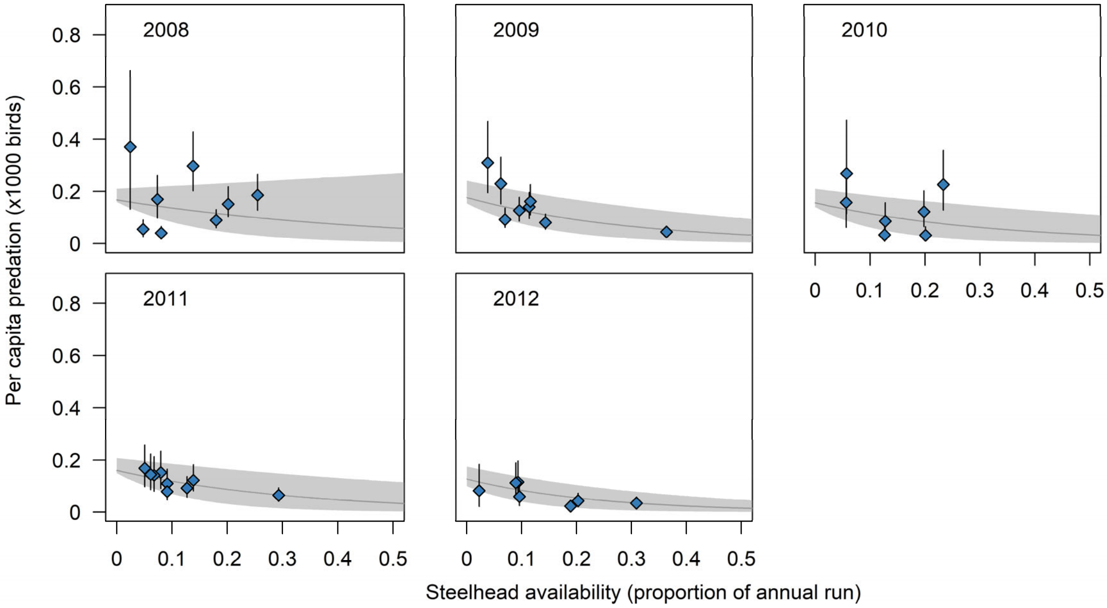

Figure 9.S3e. Weekly per capita predation probabilities for double-crested cormorants nesting on the Foundation Island colony (medians, 95% credible intervals) as a function of relative availability of tagged steelhead smolts in the mid-Columbia River by year during 2008–2012. The functional response curve most supported by the data is shown for each year (Type II; median and 95% credible interval).

CHAPTER 10: Factors Affecting the Susceptibility of Juvenile Salmonids to Avian Predation

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10.1 Summary

This chapter synthesizes mark-recovery studies of avian predation on fish populations to highlight our current knowledge of factors affecting fish susceptibility to predation by multiple bird species. Specifically, we examine peer-reviewed publications, and reports from academic, governmental, and non-governmental agencies to identify commonalities and differences in susceptibility of juvenile salmonids (Salmonidae) to predation by colonial waterbirds across North America and Europe. Our primary focus relates to mark-recovery studies of anadromous juvenile (smolt) Pacific salmonids (*Oncorhynchus* spp.) and piscivorous colonial waterbirds in the Columbia River basin (CRB), as these studies provide long-term data sources to evaluate predator-prey interactions. We first summarize relative susceptibility across salmonid species and populations as these are often the units of conservation interest. We then review the factors hypothesized to influence salmonid susceptibility to avian predation, which are grouped into four major categories: (i) environmental factors, (ii) prey density, predator density, and migration timing, (iii) prey characteristics, and (iv) predator characteristics. Our synthesis focuses on predation by Caspian terns (*Hydroprogne caspia*; terns), double-crested cormorants (*Phalacrocorax auritus*; cormorants), and a variety of gull species (*Larus* spp.; gulls) as these are the most well studied avian predators of salmonid populations. Where data are available, relationships with American white pelicans *Pelecanus erythrorhynchos*, heron species (Ardeidae), and other avian predators are also described. Predator-prey interactions were highly dynamic and varied across salmonid populations and species of avian predator. Inferences across studies supported multiple hypotheses regarding predator-prey dynamics, including the influence of prey characteristics (e.g., fish size, condition), predator characteristics (e.g., foraging ecology, location of breeding colonies), environmental factors that influence prey exposure to predators (e.g., river flows, turbidity), availability of alternative prey, and variation in predator and prey abundances. Mark-recapture-recovery studies of avian predation on fish populations have greatly improved our understanding of the factors affecting fish susceptibility to avian predation, the relative contributions of abiotic and biotic factors to predation susceptibility, and the extent to which avian predation affects fish survival and the viability of prey populations.

10.2 Introduction

Questions of how fish survival and population viability are affected by avian predation have been the subject of debate for decades (Steinmetz et al. 2003, Cowx 2008). Mark-recovery studies are increasingly used to investigate avian predation on a variety of fish species (e.g., Pacific salmon and trout [*Oncorhynchus* spp.], Atlantic salmon and trout [*Salmo spp.*], suckers [Catostomidae], roach [*Rutilus rutilus*], common bream [*Abramis brama*], perch [*Perca fluviatilis*]) in coastal and inland ecosystems across North America and Europe (Dieperink et al.

2001, Jepsen et al. 2010, Evans et al. 2012, 2016a, Osterback et al. 2013, Scoppettone et al. 2014, Skov et al. 2014, Weitkamp et al. 2014). Applications of fish mark-recovery studies to investigate avian predation is expanding rapidly due to recognition of avian predation as a substantial source of fish mortality in some systems (Dieperink et al. 2001, Steinmetz et al. 2003, Teuscher et al. 2015, Payton et al. 2019) and the generality of these approaches across common fish tag types and survey methods (e.g., Passive Integrated Transponder [PIT] tags [Ryan et al. 2001], coded wire tags [CWT; Evans et al. 2011], radio telemetry tags [Schreck et al. 2006], acoustic tags [Evans et al. 2016b], Carlin tags [Feltham and MacLean 1996]). Although diverse in application, mark-recovery studies share multiple commonalities in their ability to identify individual- and cohort-level factors affecting predator-prey interactions, predation mortality, and the relative susceptibility of prey populations. Although mark-recovery studies have been widely used for more than 20 years to investigate the relative susceptibility of fish populations to avian predation across North America and Europe, no comprehensive synthesis summarizing the knowledge gained from these studies is currently available.

Herein, we synthesize mark-recovery studies of avian predation on fish populations to highlight our current knowledge on the factors affecting fish susceptibility to bird predation. This review provides crucial information on how predation risk varies among individuals within a prey population and elucidates possible mechanisms responsible for variation in risk of avian predation among populations. The >10-year long history of research on avian predation of salmonids in the Columbia River basin (CRB) provides a substantial component of this synthesis. We also include the increasing number of mark-recovery studies investigating avian predation on anadromous Pacific and Atlantic salmon and trout populations across North America and Europe. In the CRB, as in many Northern Hemisphere systems, the breeding season for colonial waterbirds (March–August) generally coincides with the peak outmigration period of anadromous juvenile salmonids (April–August) (Evans et al. 2012, Evans et al. 2016c, Adkins et al. 2014) resulting in considerable, yet dynamic overlap among avian predator species and prey populations (*Figure 10.1*, *Figure 10.2*). As part of this work we also focus on differences among predator species, particularly Caspian terns (*Hydroprogne caspia*; hereafter terns), double-crested cormorants (*Phalacrocorax auritus*; hereafter cormorants), and a variety of gull species (*Larus* spp.; hereafter gulls) as these are the most well-studied avian predators of salmonid populations. Where data are available, relationships with American white pelicans (*Pelecanus erythrorhynchos*), herons (Ardeidae), or other avian predators are also described.

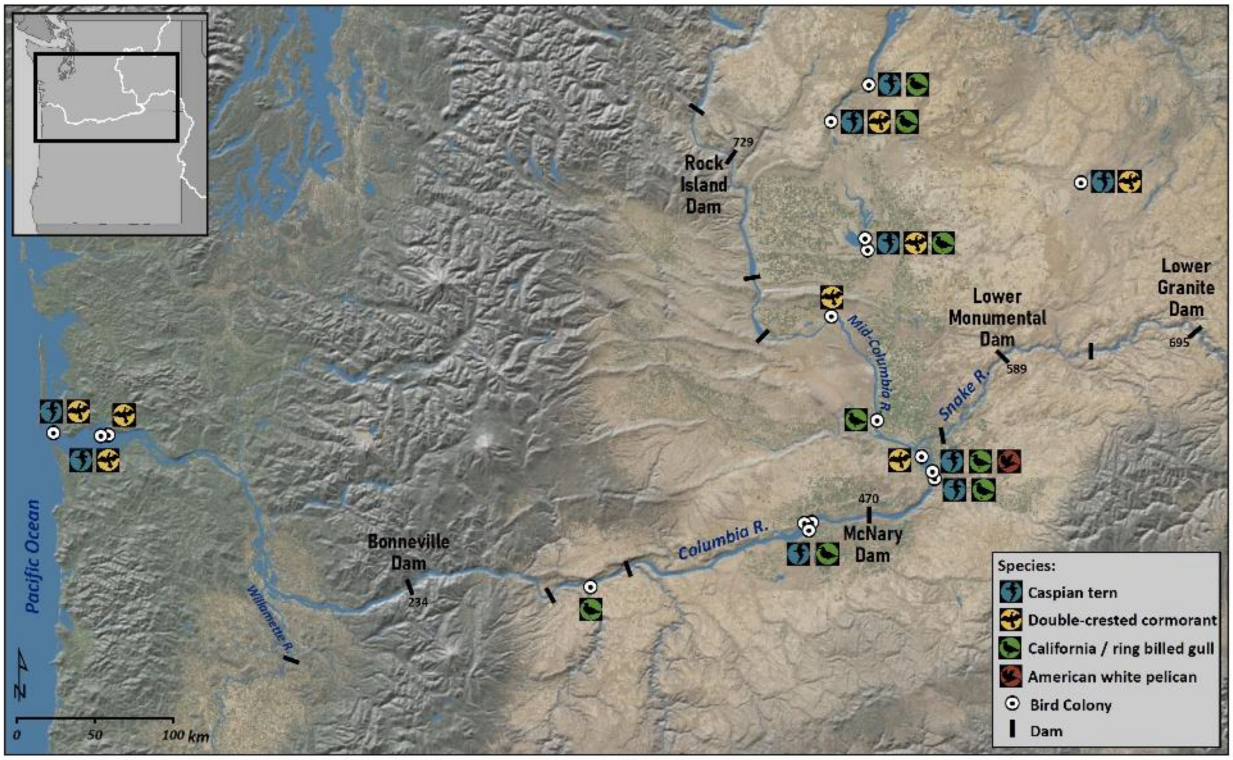
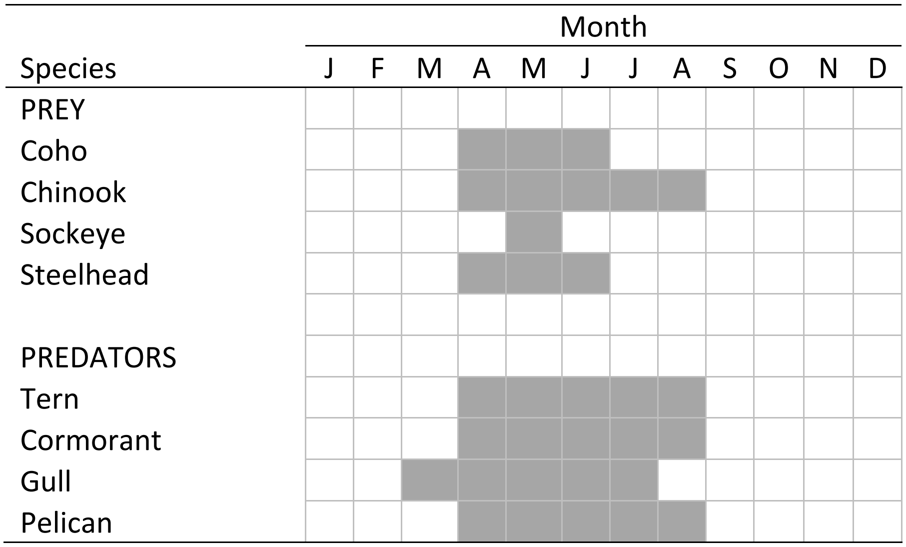


Figure 10.1. Locations of recently active breeding colonies of piscivorous waterbirds in the Columbia River basin. Numbers represent the distance in river km (Rkm) to the mouth of the Columbia River.



Species	J	F	M	A	M	J	J	A	S	O	N	D
PREY												
Coho				■	■	■						
Chinook				■	■	■	■	■				
Sockeye					■							
Steelhead				■	■	■						
PREDATORS												
Tern				■	■	■	■	■				
Cormorant				■	■	■	■	■				
Gull			■	■	■	■	■					
Pelican				■	■	■	■	■				

Figure 10.2. General periods of peak abundance for out-migrating juvenile salmonid species (prey) and avian predator species (predators) nesting in the Columbia River basin by month (January [J] to December [D]). These designations reflect the largest proportion of individuals; however, smaller numbers of both prey and predators often exist in the system during most months. Predator species include Caspian terns (Tern), double-crested cormorants (Cormorant), various gull species (Gull), and American white pelicans (Pelican).

Our review concentrates on publications and publicly available reports of original studies investigating avian predation on salmonid populations. We attempt to identify common trends across studies, highlight important differences, and note inconsistencies when they are observed. References were identified by searching journals and databases. Full citations are provided in the Literature Cited and we continue to update our database as new references become available. The conclusions of this paper are generalizations based on our collective interpretation and understanding of these studies.

10.3 Methods

We focus this synthesis on studies of avian predation on anadromous salmonids (e.g., sockeye salmon [*O. nerka*], Chinook salmon [*O. tshawytscha*], coho salmon [*O. kisutch*], steelhead trout [*O. mykiss*], and Atlantic salmon and trout [*Salmo* spp.] when available) that applied some form of mark-recovery methods to estimate predation rates or investigate fish susceptibility to avian predation. We emphasized mark-recovery studies due to their ability to investigate population-specific (e.g., evolutionarily significant unit [ESU] or distinct population segment [DPS], hereafter simply ESU), cohort-specific (e.g., migration-timing, age class), and individual-specific (e.g., fish length) factors affecting susceptibility to predation. While our primary review was limited to the above metrics, we also include information from relevant observational or diet analysis studies investigating biomass or number of salmonids consumed when findings are especially supportive or contradictory of those from mark-recovery studies. Although important, studies describing opportunistic tag recoveries associated with avian predation or studies that do not directly analyze tag recovery data are generally excluded from this review (e.g., Montevecchi et al. 1998, Aarestrup et al. 2000, Karppinen et al. 2014, Schwinn et al. 2017).

We first describe susceptibility to avian predation of salmonid species, ESUs, and cohorts, as these are often the units of primary conservation interest. We next explore factors hypothesized to influence salmonid susceptibility to avian predation. Hypotheses are grouped into four major categories: (1) environment factors, (2) prey density, predator density, and migration-timing, (3) prey characteristics, and (4) predator characteristics. We selected these categories based on available information and to remain relatively consistent with the foundational work of Leopold (1933), which suggested broadly classifying factors affecting predation into prey density, predator density, prey characteristics, predator characteristics, and alternative food quality and availability.

Factors within each of these four categories may exert considerable influence on the susceptibility of salmonid smolts to avian predation (*Table 10.1*). When complex interactions are involved, it is difficult to identify the ultimate and proximate factors affecting predation susceptibility from any single study. Synthesis of numerous localized studies that focused on a subset of prey species, predator species, locations, or years provides a framework to identify and generalize conclusions consistent across broader spatial and temporal scales. This is the aim of our review paper.

To accomplish this goal the *Results* section is divided into four sub-sections describing relative predation susceptibility as a function of: (1) salmonid species and population, (2) environmental factors, (3) prey density, predator density, and migration-timing, and (4) prey characteristics. Given the differences observed among avian predator species, each subsection discusses predator-specific relationships for terns, cormorants, and gulls, when data are available. Finally, we conclude the paper with a section on general conclusions and critical uncertainties that have yet to be resolved.

Table 10.1. Examples of common hypotheses used to explain variation in susceptibility of salmonid smolts to avian predation.

ENVIRONMENTAL FACTORS

Exposure: environmental factors that increase prey exposure to predation (e.g., travel time, river discharge rate, turbidity) and thereby increase predation susceptibility.

Alternative prey: environmental factors that enhance availability of alternative prey (e.g., marine forage fish) and thereby decrease predation susceptibility, especially in estuarine environments.

Spatial heterogeneity: spatial factors that influence predation susceptibility (e.g., increased risk near dams vs. in open water of the reservoir, increased risk at the freshwater-marine water interface).

PREY DENSITY, PREDATOR DENSITY, AND MIGRATION-TIMING

Predator swamping: increased numbers of prey decreases individual susceptibility to predation.

Prey switching: increased numbers of prey increases susceptibility to predation due to predator functional response (increased per capita consumption of salmonids) and/or numerical response (increased number of individual predators).

Proportional predation: avian predation on prey populations in proportion to relative availability of prey types.

Predator density: predation susceptibility increases as the number of predators (e.g., colony size) increases.

PREY CHARACTERISTICS

Bigger is better: larger prey are less susceptible to predation (size-efficiency hypothesis).

Disruptive size-selectivity: smaller and larger prey are relatively less susceptible to predation (e.g., difficult to capture [small], exceed predator gape width [large]).

Individual quality: prey in degraded condition (e.g., physical injury, disease, poor osmoregulation) are more susceptible to predation.

Rearing environment: Rearing in hatchery environments promotes behavioral characteristics that increase predation susceptibility (e.g., boldness, predator naiveté, surface orientation).

PREDATOR-SPECIFIC RELATIONSHIPS

Central place foraging: predation susceptibility decreases with increasing distance from breeding colony locations for central-place foraging predators.

Predator characteristics: predator-specific foraging strategies (e.g., pursuit vs. ambush) and/or predator physical characteristics (e.g., size) determine which factors influence smolt susceptibility to predation.

10.4 Results

10.4.1 Salmonid Species & Populations

Salmonids exist as a variety of semi-isolated populations across an extensive geographical range (Quinn 2018). Anadromous salmonid classification is generally based on differences in phenotypic traits (e.g., body size, migration timing [spring, summer, fall, winter]), age class at out-migration (subyearling, yearling), spatial distribution, and genetic isolation (Waples 1991, Quinn 2018). Groups of interest may consist of species, evolutionarily significant units (ESUs), distinct population segments (DPSs), and populations within these classifications (e.g., watershed within an ESU; Waples 1991, Quinn 2018). In the United States, protective status is often directed at the ESU/DPS level (hereafter simply ESU), while conservation, management, and enhancement may focus on species, ESUs, and local populations.

It is highly probable that susceptibility to avian predation varies among salmonid species and populations given their inherent differences (Evans et al. 2012). Many studies of avian predation on salmonid populations have concentrated on estimating avian predation rates for a single salmonid population of interest (e.g., Feltham and MacLean 1996, Dieperink et al. 2001, Schreck et al. 2006, Osterback et al. 2014, Hostetter et al. 2015a) or multiple co-occurring populations exposed to similar communities of avian predators (e.g., Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Sebring et al. 2013, Frechette et al. 2015, Evans et al. 2016b). In this section, we summarize these studies to describe the relative predation susceptibility across salmonid species and populations.

The majority of mark-recovery studies evaluating salmonid population-specific susceptibility to avian predation were conducted in the CRB, and these studies found that predation susceptibility varied among salmonid species and the avian predator species that each salmonid species was exposed to (*Table 10.2*).

Table 10.2. Relative susceptibility of salmonid species or populations to predation by Caspian terns (TERN), double-crested cormorants (CORMORANT), mixed gull species (GULL), and similar mark-recovery studies that combined multiple predator species or occurred outside the Columbia River basin (OTHER).

TERN
Predation susceptibility: Steelhead > sockeye ≈ coho ≈ yearling Chinook > subyearling Chinook. Variation among populations and across years but relatively consistent across colony locations (Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Evans et al. 2012, Zamon et al. 2013; Roby et al. 2015, Evans et al. 2016c, Evans et al. 2016b, Evans et al. 2019c, Roby et al. 2017b, Roby et al. 2017a).
CORMORANT
Predation susceptibility: subyearling Chinook ≈ sockeye ≈ coho ≈ yearling Chinook ≈ steelhead. Variation among populations and across years but relatively consistent across colony locations (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Zamon et al. 2013, Roby et al. 2015, Evans et al. 2016c, Evans et al. 2016b, Evans et al. 2019c).
Exceptions may be Lower Columbia River Chinook and coho populations that suffer high predation rates in the estuary, which may be attributable to increased estuary residency times of juveniles, difference in out-migration run-timing, and releases of hatchery fish in close proximity to colony sites (Zamon et al. 2013; Sebring et al. 2013).
GULL
Predation susceptibility: Steelhead > subyearling Chinook ≈ sockeye ≈ yearling Chinook. Variation among populations, across years, and colony locations (Evans et al. 2012, Roby et al. 2015, Evans et al. 2016b).
OTHER
In San Francisco Bay, fall-run subyearling Chinook were more susceptible to tern predation relative to spring-run yearling Chinook, which may be attributable to differences in out-migration run-timing and releases of hatchery fish in close proximity to colony sites (Evans et al. 2011).
In two Oregon estuaries, cormorants disproportionately depredated coho salmon relative to steelhead, which may be attributable to increased estuary residency time of juvenile coho relative to steelhead (Clements et al. 2012).
In the River Gudenå (Denmark), Atlantic salmon (*S. salar*) and sea trout (*S. trutta*) displayed similar susceptibility to avian predation (combined grebes and herons; Jepsen et al. 1998).
In the River Skjern estuary (Denmark), Atlantic salmon were more susceptible to great cormorant predation relative to sea trout (Dieperink et al. 2002, Koed et al. 2006).

For terns, studies consistently found higher predation susceptibility for steelhead trout relative to salmon species (e.g., Chinook, coho, sockeye) in both freshwater and estuary systems (*Table 10.2* above; *Figure 10.3*; Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Good et al. 2007, Evans et al. 2012, Zamon et al. 2013, Roby et al. 2015, Evans et al. 2016c, Evans et al. 2016b, Evans et al. 2019c, Roby et al. 2017b, Roby et al. 2017a). Within salmon populations, susceptibility to tern predation was generally lowest for subyearling Chinook, followed by similar levels of risk for sockeye, coho, and yearling Chinook (*Table 10.2* above and citations therein). Lower Columbia River Chinook and coho salmon populations may experience higher susceptibility to predation from terns nesting in the CR estuary relative to Chinook and coho salmon populations originating farther upriver, providing exceptions to these generalities (Zamon et al. 2013, Sebring et al. 2013). In San Francisco Bay, fall-run subyearling Chinook were more susceptible to tern predation compared to spring-run Chinook, which may have been driven by differences in run-timing, releases of hatchery fish in close proximity to colony sites, and subsequent exposure to tern predation between these ESUs (Evans et al. 2011).

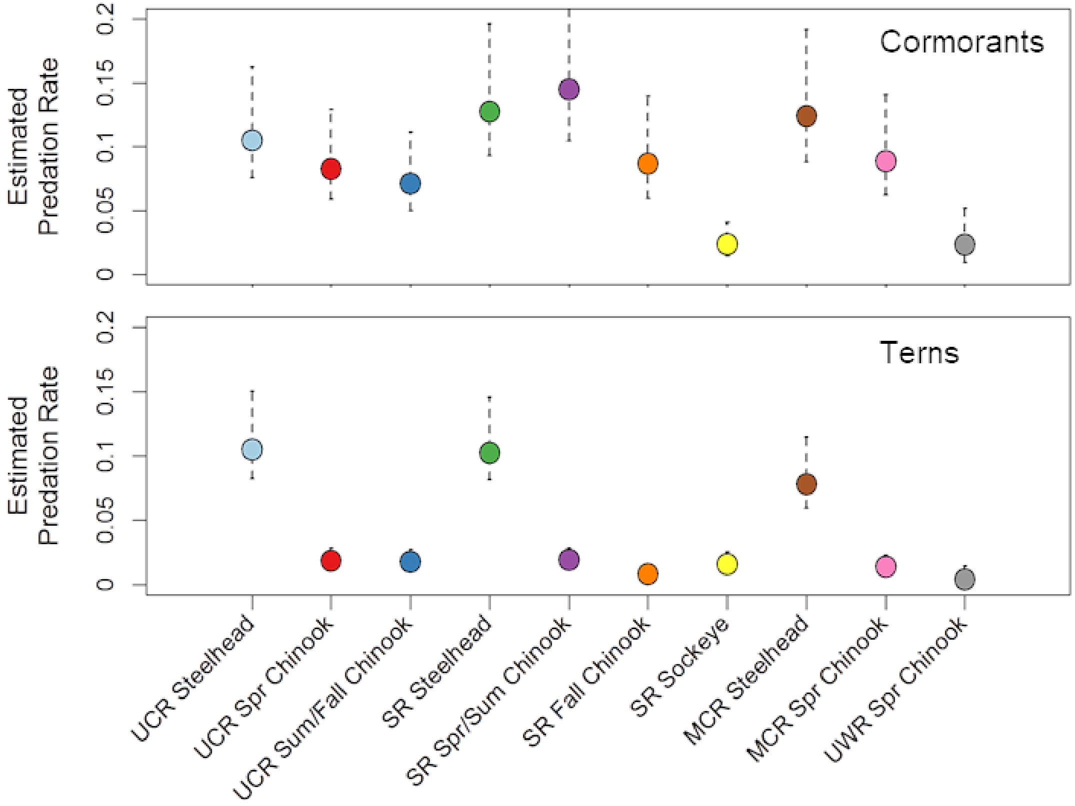


Figure 10.3. Estimated annual predation rates (proportion of available fish consumed) on PIT-tagged populations of juvenile salmonids (Upper Columbia River [UCR], Snake River [SR], Middle Columbia River [MCR], Upper Willamette River [UWR]) by double-crested cormorants (top) and Caspian terns (bottom) nesting on East Sand Island in 2015. Error bars represent 95% credible intervals for predation rates. Figure adapted from Evans et al. (2016c).

For cormorants, most studies found little to no support for differences in predation susceptibility between steelhead and salmon species, or between salmon ESUs in freshwater and estuary systems (*Table 10.2* above; *Figure 10.3* above; Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Zamon et al. 2013, Roby et al. 2015, Evans et al. 2016c, Evans et al. 2016b, Evans et al. 2019c). Lower Columbia River Chinook and coho populations experienced higher predation susceptibility to cormorants nesting in the Columbia River estuary, differences that may be due to the greater abundance and late run-timing of these populations relative to other populations of juvenile salmonids in the CRB (Zamon et al. 2013, Sebring et al. 2013). In two other estuaries in Oregon, USA, cormorants disproportionately depredated coho salmon relative to steelhead, which may be attributable to increased estuary residency time for coho salmon smolts in those systems (Clements et al. 2012).

For gulls, information was generally less available, but several studies found some evidence of higher predation susceptibility for steelhead trout relative to salmon species, and minor variation among salmon ESUs in susceptibility (*Table 10.2* above; Frechette et al. 2012, Evans et al. 2012, Roby et al. 2015, Evans et al. 2016b). Studies evaluating population-specific susceptibility to gull predation were generally restricted to freshwater systems in the CRB (Evans et al. 2012, Roby et al. 2015, Evans et al. 2016b); however, studies in nearshore and estuary systems have also occurred in coastal California (Frechette et al. 2012).

Salmonid populations that undergo long-distance out-migration may be particularly susceptible to avian predation due to the cumulative effects of predation by birds nesting at multiple colony locations (Evans et al. 2019a). In the CRB, several salmonid populations must pass through the foraging ranges of multiple tern, cormorant, and gull colonies during out-migration. For example, juvenile Upper Columbia River steelhead were depredated by birds nesting at up to 14 different colonies, with cumulative (all colonies combined) predation rates in excess of 40% of available fish in some years (Evans et al. 2019a). Similarly, Snake River steelhead often experience high cumulative avian predation rates due to predation by > 10 separate colonies during out-migration (Payton et al. 2019).

Species-specific susceptibly to avian predation was also observed across Atlantic salmon (*S. salar*) and sea trout (*S. trutta*). For example, in the River Skjern estuary (Denmark), Atlantic salmon were more susceptible to great cormorant (*Phalacrocorax carbo sinensis*) predation relative to sea trout (*Table 10.2* above; Dieperink et al. 2002, Koed et al. 2006). However, Atlantic salmon and sea trout displayed similar susceptibility when recoveries were combined across grebes and herons in the River Gudenå, Denmark (*Table 10.2* above; Jepsen et al. 1998).

10.4.2 Environmental Factors

In this section, we summarize studies focused on linking large-scale and local environmental factors to the relative susceptibility of juvenile salmonids to avian predation. Environmental factors are known to influence salmonid survival across multiple life stages. For example, studies have linked variation in salmonid survival to large-scale climate indices such as the

Pacific Decadal Oscillation (Mantua et al. 1997) and the North Pacific Gyre Oscillation (Di Lorenzo et al. 2008), presumably through the regulation of predators and/or food resources (Scheuerell and Williams 2005, Zabel et al. 2006, Emmett et al. 2006, Scheuerell et al. 2009). Juvenile salmonid out-migration is also characterized by extreme fluctuations in local environmental conditions (e.g., river discharge, temperature, salinity, turbidity) that may directly impact survival and predation susceptibility (e.g., Connor et al. 2003, Smith et al. 2003, Hostetter et al. 2012).

Similar to survival, susceptibility to avian predation is likely influenced by numerous environment factors affecting prey exposure to predators, the availability of alternative prey, and spatial heterogeneity in habitats. Further, environmental factors that influence both within- and among-year variation in predation susceptibility may vary between freshwater and estuarine systems (Hostetter et al. 2012, Lyons et al. 2014, Evans et al. 2016c). Understanding variation in predation rates, relative susceptibility among prey populations, and how various management actions and future conditions may alter predator-prey dynamics requires identifying the environmental factors that influence susceptibility of juvenile salmonids to avian predation (Lyons et al. 2014).

Multiple studies have identified associations between environmental factors and smolt susceptibility to avian predation (*Table 10.3*). Per capita (i.e. per bird) predation rates on salmonids were higher at breeding colonies located in freshwater systems relative to colonies in estuaries (*Table 10.3*, *Figure 10.4*), a result supported by both mark-recovery (Evans et al. 2012; see also *Chapter 5*) and diet analysis studies (Collis et al. 2002, Roby et al. 2002, Lyons et al. 2010). Similarly, susceptibility of salmonid smolts to both tern and cormorant predation in the Columbia River estuary was related to large scale climatic indices (North Pacific Gyre Oscillation and spring upwelling), likely due to associations with availability of marine forage fish (alternative prey) in the estuary (*Table 10.3*; Lyons et al. 2014, Evans et al. 2016c).

Table 10.3. Environmental factors associated with the relative susceptibility of juvenile salmonids to predation by Caspian terns (TERN), double-crested cormorants (CORMORANT), mixed gull species (GULL), and similar mark-recovery studies that combined multiple predator species or occurred outside the Columbia River basin (OTHER).

TERN
Ocean indices (North Pacific Gyre Oscillation and spring upwelling) associated with predation susceptibility in the Columbia River estuary (Evans et al. 2016c).
Faster transit times associated with reduced predation within a year (Evans et al. 2013).
Increased water discharge associated with reduced predation susceptibility at an inland (freshwater) colony (Hostetter et al. 2012).
Increased water clarity associated with decreased predation susceptibility at an inland colony (Hostetter et al. 2012).
Increased depredation in open waters of a reservoir relative to near dams (Evans et al. 2016b).

Table 10.3 (Cont.)

Per capita (per bird) consumption of salmonids was higher for birds nesting in the freshwater areas relative to the marine or brackish areas (Evans et al. 2012; see also *Appendix C*).

Predation probability generally decreased with distance from colony (Schreck et al. 2006, Evans et al. 2016b, Hostetter et al. 2018, Evans et al. 2019c, Payton et al. 2019).

Table 10.3 (Cont.)

Terns commuted long distances from their colony location, including over land to forage on juvenile salmonids (regularly >30 km, occasionally >90 km; Evans et al. 2012, Evans et al. 2016b, Hostetter et al. 2018, Evans et al. 2019a, Roby et al. 2017b, Roby et al. 2017a).

CORMORANT

North Pacific Gyre Oscillation and Columbia River discharge were associated with predation susceptibility in the Columbia River estuary (Lyons et al. 2014, Evans et al. 2016c)

Increased turbidity associated with decreased predation susceptibility at an inland colony (Hostetter et al. 2012).

Foraged across habitats including open waters of a reservoir and near dams (Evans et al. 2016b).

Per capita consumption of salmonids was higher for birds nesting in the freshwater areas relative to the marine or brackish areas (Evans et al. 2012; see also *Appendix C*).

Predation probability generally decreased with distance from colony (Schreck et al. 2006, Evans et al. 2016b, Evans et al. 2019c).

GULL

Depredation concentrated near dams relative to open waters of a reservoir (Evans et al. 2016b, Roby et al. 2017b).

Per capita predation was highly variable based on colony location, even within the same reservoir (Evans et al. 2016b, Roby et al. 2017b).

Data deficient for other environmental factors.

OTHER

Pelicans foraged in open waters of a reservoir and near dams (Evans et al. 2016b) and can travel long distances from their colony location to forage (> 80 km; Evans et al. 2016a).

Atlantic salmon and sea trout susceptibility to predation by great cormorants and grey herons was highest near region of ocean entry (i.e. increasing salinity; Dieperink et al. 2001, 2002, Koed et al. 2006).

In the river Gudenaa (Denmark), grey heron depredation on Atlantic salmon was concentrated near a dam (Koed et al. 2002)

Masu salmon (*O. masou*) susceptibility to grey heron predation was higher in shallow relative to deeper waters in both stream and tank experiments (Miyamoto et al. 2018).

Predation rates on stocked rainbow trout decreased with increasing distance to cormorant and American white pelican colonies in Idaho (Meyer et al. 2016, Chiaramonte et al. 2019).

Coho and steelhead susceptibility to gull predation decreased with increasing distance to gull colony in coastal California (Frechette et al. 2012, 2015).

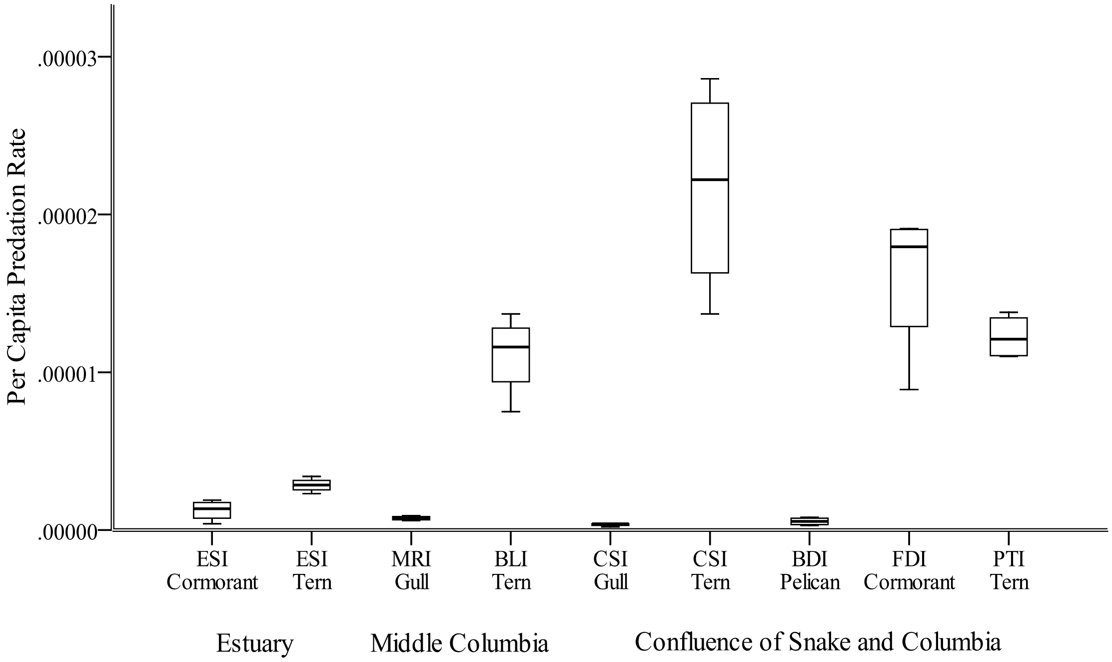


Figure 10.4. Boxplots of minimum per capita (per bird) predation rates on juvenile salmonids (Chinook, sockeye, and steelhead combined) by Caspian terns (Tern), double-crested cormorants (Cormorant), California gulls and ring-billed gulls (Gull), and American white pelicans (Pelican) nesting on islands in the Columbia River estuary (ESI = East Sand Island), or inland on the mid-Columbia River (MRI = Miller Rocks islands; BLI = Blalock islands) and near the Snake River–Columbia River confluence (CSI = Crescent Island; BDI = Badger Island; FDI = Foundation Island; PTI = Potholes Reservoir islands) during 2007–2010. Per capita predation rates were not adjusted for on-colony PIT deposition probabilities, resulting in minimum estimates of per capita predation. Figure from Evans et al. (2012).

Terns, cormorants, and gulls are central-place foragers (Orians and Pearson 1979), and the susceptibility of an individual fish to predation generally decreased with increasing distance from breeding colonies (*Table 10.3* above; *Figure 10.5*; Schreck et al. 2006, Frechette et al. 2012, Frechette et al. 2015, Meyer et al. 2016; Evans et al. 2016b, Roby et al. 2017b, Hostetter et al. 2018, Evans et al. 2019a, Chiaramonte et al. 2019). It is important to note, however, that multiple studies have documented long-distance foraging trips from breeding colonies (e.g., >20 km; Frechette et al. 2012, Evans et al. 2012, Evans et al. 2016b, Meyer et al. 2016; Roby et al. 2017a, Roby et al. 2017b, Hostetter et al. 2018, Chiaramonte et al. 2019), with Caspian terns regularly commuting >30 km and occasionally >90 km from inland nesting colonies to forage in the Columbia River (*Table 10.3* above; *Figure 10.5*; Evans et al. 2012, Evans et al. 2016b, Roby et al. 2017a, Roby et al. 2017b, Hostetter et al. 2018).

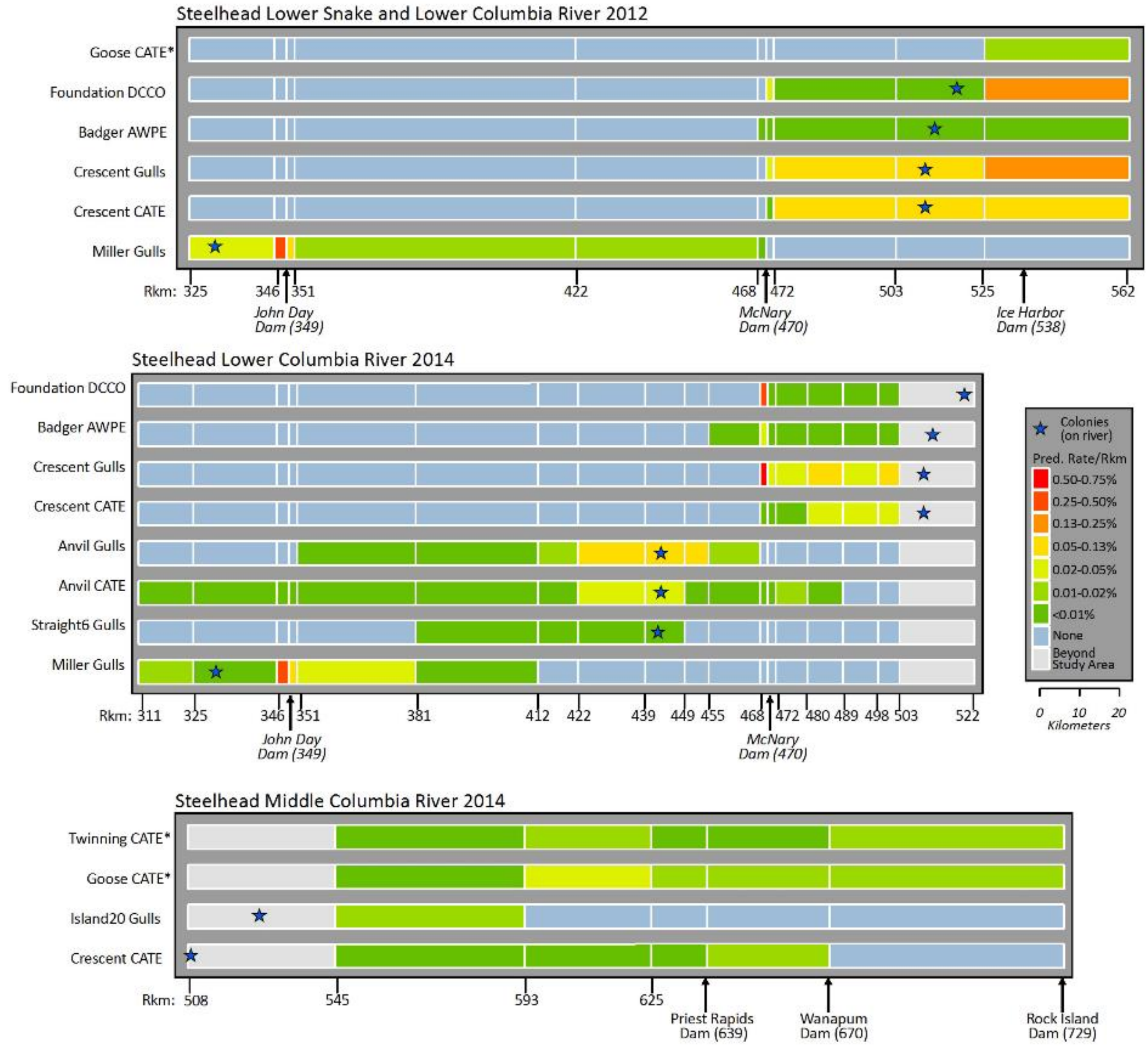


Figure 10.5. Colony-specific locations of predation on tagged steelhead smolts in sections of the lower Snake River, lower Columbia River, and middle Columbia River during 2012 and 2014. Results are depicted as predation rates per river km (Rkm). Predator species include Caspian terns (CATE), double-crested cormorants (DCCO), California and ring-billed gulls (Gulls), and American white pelicans (AWPE). An asterisk next to a colony site denotes that the colony is located off-river. Figure adapted from Evans et al. (2016b).

Additional factors associated with increased fish susceptibility to tern predation included longer out-migration travel times (Hostetter et al. 2012, Evans et al. 2013) and increased turbidity (Hostetter et al. 2012). Predation rates were occasionally higher in the open waters of a reservoir relative to near dams *(Table 10.3* above; *Figure 10.5* above; Evans et al. 2016b). Hypotheses for these increases in susceptibly included increased exposure to predation due to longer out-migration travel times (Hostetter et al. 2012, Evans et al. 2013), decreased prey reaction times in more turbid waters (Hostetter et al. 2012 and citations therein), and broadly searching for prey across a variety of habitats within foraging range of the breeding colony (Cuthbert and Wires 1999).

Additional factors associated with increased fish susceptibility to cormorant predation included decreased turbidity, likely due to increased prey encounter rates (Hostetter et al. 2012 and citations therein), and relatively consistent predation susceptibility across habitats, including the open waters of reservoirs and near dams (*Table 10.3* above; *Figure 10.5* above; Evans et al. 2016b). The volume of water spilled at dams was positively associated with smolt susceptibility to cormorant predation in the CR estuary, possibly due to connections between spill, river discharge, smolt abundance, and the availability of alternative prey during high flow events in the estuary (Evans et al. 2016c).

For gulls, predation was often concentrated near dams relative to the open waters of reservoirs (*Figure 10.5* above; Evans et al. 2016b, Roby et al. 2017b). Increased susceptibility of salmonid smolts to gull predation near dams was potentially linked to the close proximity of many gull colonies to dams in CRB (*Figure 10.5* above), prolonged or delayed smolt out-migration times near dams, and/or turbulent hydrologic conditions that may injure or temporary disorient smolts near dams (Ruggerone 1986, Evans et al. 2016). Information directly linking environmental factors to susceptibility of salmonid smolts to gull predation, however, are generally lacking (*Table 10.3* above).

For pelicans, there was no evidence of concentrated foraging in open reservoirs versus dams on the Columbia River (Evans et al. 2016b). Similar to terns, pelicans are capable of commuting long distances (> 80 km) to forage on salmonids (Evans et al. 2016b) and other fish (Evans et al. 2016a).

Atlantic salmon and sea trout susceptibility to predation by great cormorants and grey herons was highest in estuaries and near regions of ocean entry (*Table 10.3* above; Dieperink et al. 2001, Dieperink et al. 2002, Koed et al. 2006). Overall, results across both Pacific and Atlantic salmonids indicate predation risk is highly variable across estuaries and driven by multiple factors (*Table 10.3* above). For example, susceptibility to avian predation in estuaries may increase due to salinity gradients (Dieperink et al. 2001, Dieperink et al. 2002, Koed et al. 2006, Kennedy et al. 2007) or decrease due to increased availability of alternative prey (Lyons et al. 2014, Evans et al. 2016c).

10.4.3 Prey Density, Predator Density, & Migration Timing

This section characterizes our current knowledge of juvenile salmonid susceptibility to avian predation as a function of prey density, predator density, and prey migration-timing, with a particular emphasis on linking results from multiple field studies to functional responses commonly used to explain and evaluate predator-prey dynamics (Solomon 1949, Holling 1959). A recently completed detailed analysis of functional responses in Caspian tern, double-crested cormorant, and California and ring-billed predation to changes in steelhead smolt availability in the Columbia River basin are also presented in *Chapter 9*.

The timing and synchrony of life-history events (e.g., birth, migration) can dramatically affect the survival and fitness of prey species (Solomon 1949, Ims 1990). For example, large numbers of synchronized prey may 'swamp' the short-term capacity of predators to consume them, thus increasing an individual's survival probability (Ims 1990). Alternatively, initial increases in prey abundance may increase predation rates if predators switch to prey items as they become more common (i.e. prey-switching) or develop an improved search image that increases capture efficiency (Solomon 1949, Holling 1959, Denno and Lewis 2009). How predators respond to changes in prey abundance are generally described by a predator's functional response – the rate at which a predator kills prey at different prey densities (Holling 1959). Despite widespread acceptance of these functional forms, most applications have focused on theoretical approaches with few studies directly linking these concepts to field data from wild populations (Miller et al. 2006; see also *Chapter 9*).

Mark-recovery studies investigating avian predation on fish populations provide opportunities to explore relationships among survival, consumption rates (e.g., biomass or number consumed), predation rates (e.g., proportion consumed), and their connections with predator functional responses in complex multi-predator systems. In the CRB, out-migration timing plays a major role in determining salmonid smolt survival (Hostetter et al. 2011, Haeseker et al. 2012, Evans et al. 2014) and smolt-to-adult survival (Scheuerell et al. 2009, Evans et al. 2014, Haeseker et al. 2020). Multiple studies now suggest that earlier migrating individuals experience higher survival compared to later migrating individuals (Scheuerell et al. 2009, Hostetter et al. 2011, Haeseker et al. 2012, Evans et al. 2014). Similarly, differences in the timing and synchrony of life-history stages (e.g., migration-timing) may directly influence juvenile salmonid susceptibility to avian predation (*Table 10.4*).

Table 10.4. Prey density and run-timing factors associated with the relative susceptibility of juvenile salmonids to predation by Caspian terns (TERN), double-crested cormorants (CORMORANT), mixed gull species (GULL), and similar mark-recovery studies that combined multiple predator species or occurred outside the Columbia River basin (OTHER).

TERN
Predation susceptibility decreased as smolt abundance increased (i.e. predator swamping; Hostetter et al. 2012, Roby et al. 2015, Roby et al. 2017a, Roby et al. 2017b; see also *Chapter 9*).
Predation susceptibility increased for late-season (June) migrants (Roby et al. 2015, Evans et al. 2016b).
Predation susceptibility increased as colony size increased (Good et al. 2007, Hostetter et al. 2012, Evans et al. 2013, Evans et al. 2019c).
CORMORANT
Predation susceptibility increased as smolt abundance increased (Roby et al. 2015, Roby et al. 2017b, Evans et al. 2016c), but this trend was not detected at all colonies (Hostetter et al. 2012; see also *Chapter 9*).

Table 10.4 (Cont.)

Predation susceptibility was highest in May–June when birds were nesting and smolt abundances were highest (Evans et al. 2016c).
Predation susceptibility was highly variable within a season across salmonid populations (Zamon et al. 2013, Sebring et al. 2013, Roby et al. 2015, Evans et al. 2016c).
Predation susceptibility increased as colony size increased (Hostetter et al. 2012).
GULL
Predation susceptibility remained similar across a wide range of smolt abundances (Roby et al. 2017b; see also *Chapter 9*).
OTHER
Susceptibility to combined tern and cormorant predation in the Columbia River estuary decreased for late-season (June) migrants (Kennedy et al. 2007).
Great cormorant predation on sea trout was highest early in the season and decreased thereafter (Källo et al. 2020).
Increased predation by grey herons associated with increased Masu salmon density in both stream and tank experiments (Miyamoto et al. 2018).
Mergansers (*Mergus* spp) disproportionately depredated Atlantic salmon on the first several days after hatchery releases in River North Esk, Scotland (Feltham and MacLean 1996)

Multiple studies identified relationships between juvenile salmonid predation susceptibility and prey density, predator density, or prey migration-timing (*Table 10.4* above). Susceptibility to both tern and cormorant predation increased as colony size increased (*Table 10.4* above). For example, colony-specific tern predation increased as the number of terns counted increased both within (weekly) and across (annual) breeding seasons (*Figure 10.6*). Tern and cormorant colony size, however, is not the sole factor influencing predation impacts. Predation can remain fairly consistent across colony sizes, or conversely, predation may vary substantially even though colony size remained similar, with both situations often attributed to variation in other environmental factors (*Table 10.4* above; *Figure 10.6*; Good et al. 2007, Hostetter et al. 2012, Evans et al. 2013, Lyons et al. 2014, Evans et al. 2019c; see section *10.4.2 Environmental Factors* above). Information to support or disprove relationships between juvenile salmonid susceptibility and gull colony size were lacking (*Table 10.4* above).

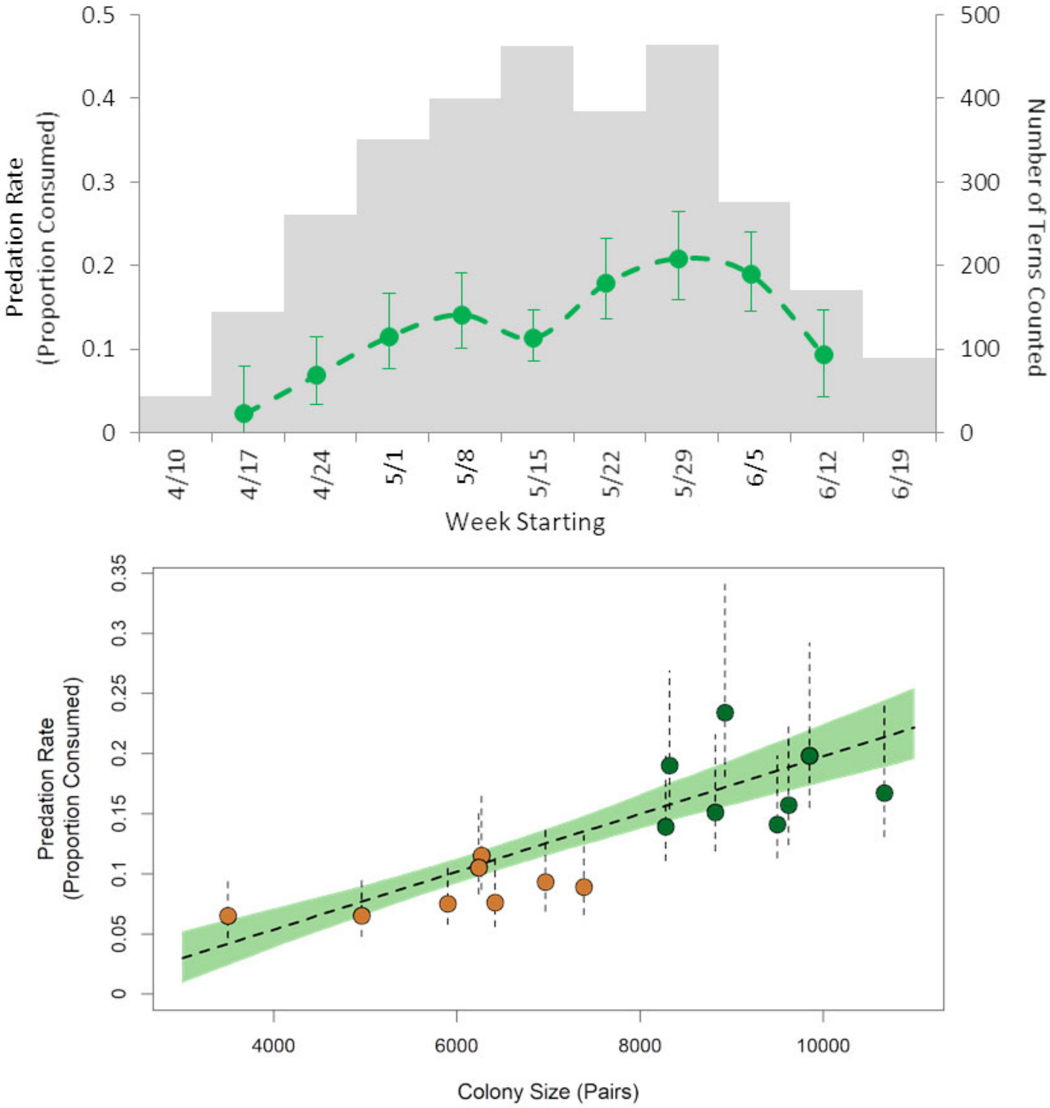


Figure 10.6. Weekly predation rates on Upper Columbia River (UCR) steelhead by terns nesting on Goose Island (top) and annual predation rates on UCR steelhead by terns nesting on East Sand Island prior to (green dots) or following (orange dots) management actions to reduce colony size (bottom). Error bars represent 95% confidence intervals (top) or 95% credible intervals (bottom). Dotted line represents the best fit, linear association between colony size and predation rates, with shading (light green) denoting 95% credible intervals of the best fit. Figures adopted from Evans et al. (2013) and Evans et al. (2019c).

Predator-specific functional responses may best explain the relationships between prey abundance and smolt susceptibility to avian predation (*Table 10.4* above; *Figure 10.7*; see also *Chapter 9*). Tern predation on juvenile steelhead followed a Type II functional response, whereby individual smolt susceptibility to tern predation declined as smolt abundance increased (*Figure 10.7*; Hostetter et al. 2012, Roby et al. 2015, Roby et al. 2017a, Roby et al. 2017b, Evans et al. 2016b; see also *Chapter 9*).

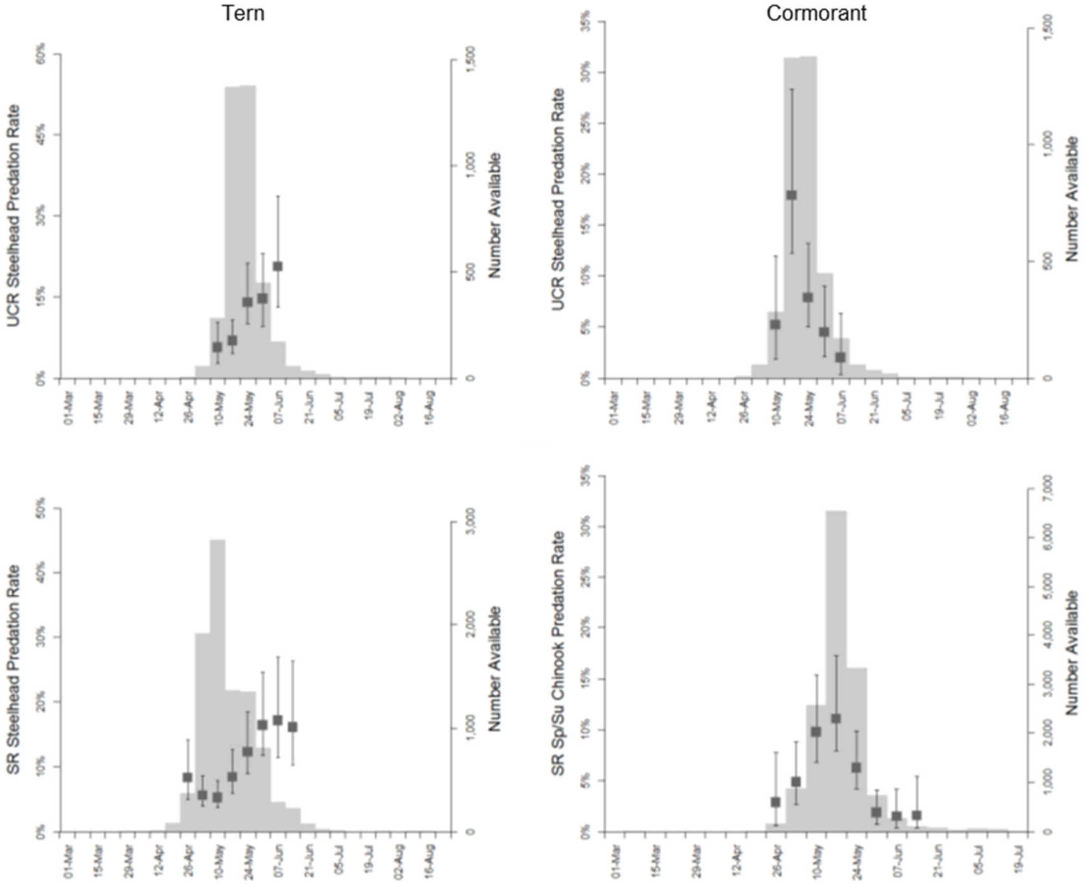


Figure 10.7. Weekly population-specific salmonid (Upper Columbia River [UCR], Snake River [SR]) predation rates East Sand Island by Caspian terns (tern; left) and double-crested cormorants (cormorant; right) compared to the number of available PIT-tagged fish (histogram). Error bars denote 95% credible intervals. Figure adapted from Roby et al. (2015). See also Chapter 9.

Conversely, functional responses associated with cormorant predation on juvenile steelhead varied by colony (see *Chapter 9*). At East Sand Island, cormorants displayed a Type III functional response, whereby susceptibility to cormorant predation was lower during periods of low smolt abundance and increased as smolt abundance increased (*Table 10.4* above; *Figure 10.7* above; see also *Chapter 9*). At Foundation Island, however, cormorants displayed a Type II functional response more similar to that of terns (see *Chapter 9)*. Across these studies, increases in predation rates as prey abundance increased may be driven by cormorant prey-switching, while the tapering of predation rates at high abundances may arise due to eventual swamping (Hostetter et al. 2012, Roby et al. 2015, Evans et al. 2016b, Roby et al. 2017a, Roby et al. 2017b, Chapter 9). Together, these trends imply that juvenile salmonid susceptibility to cormorant predation may vary by colony location, is driven by a host of factors, and may be greatest during May–June in the CRB (*Table 10.4* above; *Figure 10.7* above; Evans et al. 2016c; see also *Chapter 9*).

In our review, only one study presented relationships between prey abundance and juvenile salmonid susceptibility to gull predation (Roby et al. 2017b). Here, steelhead susceptibility to gull predation remained similar across a wide range of smolt abundances, most reflective of a Type I functional response or the increasing period of a Type III functional response (*Table 10.4* above; Roby et al. 2017b; see also *Chapter 9*).

10.4.4 Prey Characteristics

Mark-recovery studies based on uniquely marked individuals allow insights into the dynamics of predator-prey interactions and their related ecological processes (Lebreton et al. 1992). Identifying the influence of individual fish characteristics on predation susceptibility provides crucial information on how predation risk varies among individuals within a population (e.g., variation in fish length among individuals) and elucidates possible mechanisms responsible for variation in predation risk among prey populations. In this section, we synthesize studies describing relationships between individual fish characteristics and avian predation susceptibility. Based on the available literature and hypotheses investigated therein, we broadly assigned prey characteristics into length, condition, and rearing-type categories (*Table 10.5*).

Table 10.5. Prey characteristics (e.g., fish length, condition, rearing-type) associated with the relative susceptibility of juvenile salmonids to predation by Caspian terns (TERN), double-crested cormorants (CORMORANT), mixed gull species (GULL), and similar mark-recovery studies that combined multiple predator species or occurred outside the Columbia River basin (OTHER).

TERN
Quadratic relationship with smolt length, whereby susceptibility to predation was highest for fish 175–225 mm, and lower for shorter (<175 mm) and longer (>225 mm) individuals (Hostetter et al. 2012, Evans et al. 2019b).
Rearing-type (wild, hatchery) was not consistently associated with differences in susceptibility to predation (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2012, Evans et al. 2016c). Higher predation rates on hatchery-reared individuals were occasionally observed but may be attributable to differences in smolt size or run-timing (Roby et al. 2015, Evans et al. 2016c, Evans et al. 2019c).
Smolts in degraded condition (descaling, body injuries, disease symptoms) were more susceptible to predation; however, differences were often small and often only a small proportion of fish were classified as in degraded condition (Schreck et al. 2006, Hostetter et al. 2012, Roby et al. 2015, Evans et al. 2019b).
Lower gill Na^+, K^+ - ATPase activity associated with increased predation susceptibility in the Columbia River estuary (Schreck et al. 2006).

Table 10.5 (Cont.)

CORMORANT
Rearing-type was not consistently associated with differences in predation susceptibility (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Evans et al. 2016c, Evans et al. 2019c). Increased predation rates on hatchery-reared individuals were occasionally observed but may be attributable to differences in run-timing (Hostetter et al. 2012, Roby et al. 2015).
Smolts in degraded condition (descaling, body injuries, or disease symptoms) were more susceptible to predation; however, differences were small and often only a small proportion of fish were classified as in degraded condition (Hostetter et al. 2012, Roby et al. 2015).
No relationship between smolt length and susceptibility (Hostetter et al. 2012, Evans et al. 2019b).
GULL
Quadratic relationship with fish length, whereby susceptibility was highest for fish 175–240 mm, and lower for shorter (<175 mm) and longer (>240 mm) individuals (Evans et al. 2019b).
Hatchery-reared susceptibility occasionally > naturally reared susceptibility, but differences were often small (Evans et al. 2019b).
No evidence that smolts in degraded condition (descaling, body injuries, or disease symptoms) were more susceptible to predation (Evans et al. 2019b).
OTHER
No consistent relationship detected between subyearling Chinook length (60–120 mm) and susceptibility to combined predation from Caspian terns and double-crested cormorants in the Columbia River estuary (Sebring et al. 2013).
No consistent relationship detected between steelhead length (mean = 154 mm) and predation susceptibility to combined Caspian tern and cormorant predation in the Columbia River estuary (Kennedy et al. 2007).
Smolt length displayed a quadratic relationship with susceptibility to western gull predation in coastal California (Osterback et al. 2014).
No relationship detected between trout length (mostly 200–350 mm) and susceptibility to predation by cormorants or American white pelicans (Teuscher et al. 2015, Chiaramonte et al. 2019).
Longer Masu salmon displayed increased susceptibility to grey heron predation (75–160 mm; Miyamoto et al. 2018).
Shorter Atlantic salmon and sea trout displayed increased susceptibility to great cormorant and grey heron predation (165–230 mm; Dieperink et al. 2001, Dieperink et al. 2002, Källo et al. 2020).
Atlantic salmon depredated by mergansers (*Mergus* spp) were on average shorter (mean = 117 mm) than the average length of the tagged population (mean = 122) in River North Esk, Scotland (Feltham and MacLean 1996)

Table 10.5 (Cont.)

Hatchery-reared steelhead susceptibility > naturally reared steelhead susceptibility to combined Caspian tern and cormorant predation in the Columbia River estuary, but differences may be partially attributable to differences in smolt length, condition, or run-timing (Kennedy et al. 2007).

Naturally reared steelhead susceptibility > hatchery-reared steelhead susceptibility to western gull predation in coastal California (Osterback et al. 2014).

Lower gill Na^+, K^+ - ATPase activity associated with increased susceptibility to combined predation from Caspian terns and double-crested cormorants in the Columbia River estuary (Kennedy et al. 2007).

Within populations of juvenile salmonids, higher susceptibility to avian predation was attributed to differences in rearing-type, condition, behavior, and size (*Table 10.5* above; Collis et al. 2001, Schreck et al. 2006, Kennedy et al. 2007, Hostetter et al. 2012, Hostetter et al. 2012, Evans et al. 2019b, Evans et al. 2019c). Overall, individual fish characteristics substantially affected smolt susceptibility to avian predation, however, relationships varied by predator species (*Table 10.5* above).

Size (e.g., fork length) was the most common individual fish characteristic recorded and regularly associated with variation in susceptibility to tern and gull predation (*Table 10.5* above and citations therein). A quadratic function of length best described susceptibility to tern and gull predation, where susceptibility was lower for shorter (<175 mm) and longer (>225) individuals (*Figure 10.8*, *Figure 10.9*; Hostetter et al. 2012, Osterback et al. 2014, Evans et al. 2019b). Conversely, most studies found no support for an effect of smolt length on susceptibility to cormorant predation across a variety of fish lengths (130–375 mm; *Table 10.5* above; Hostetter et al. 2012, Chiaramonte et al. 2019). Kennedy et al. (2007) found no consistent relationship between steelhead length (mean = 154 mm) and avian predation susceptibility in the Columbia River estuary, however, Kennedy et al. (2007) combined recoveries across predator species (terns and cormorants), which may have masked predator-specific relationships observed in other studies (*Table 10.5* above).

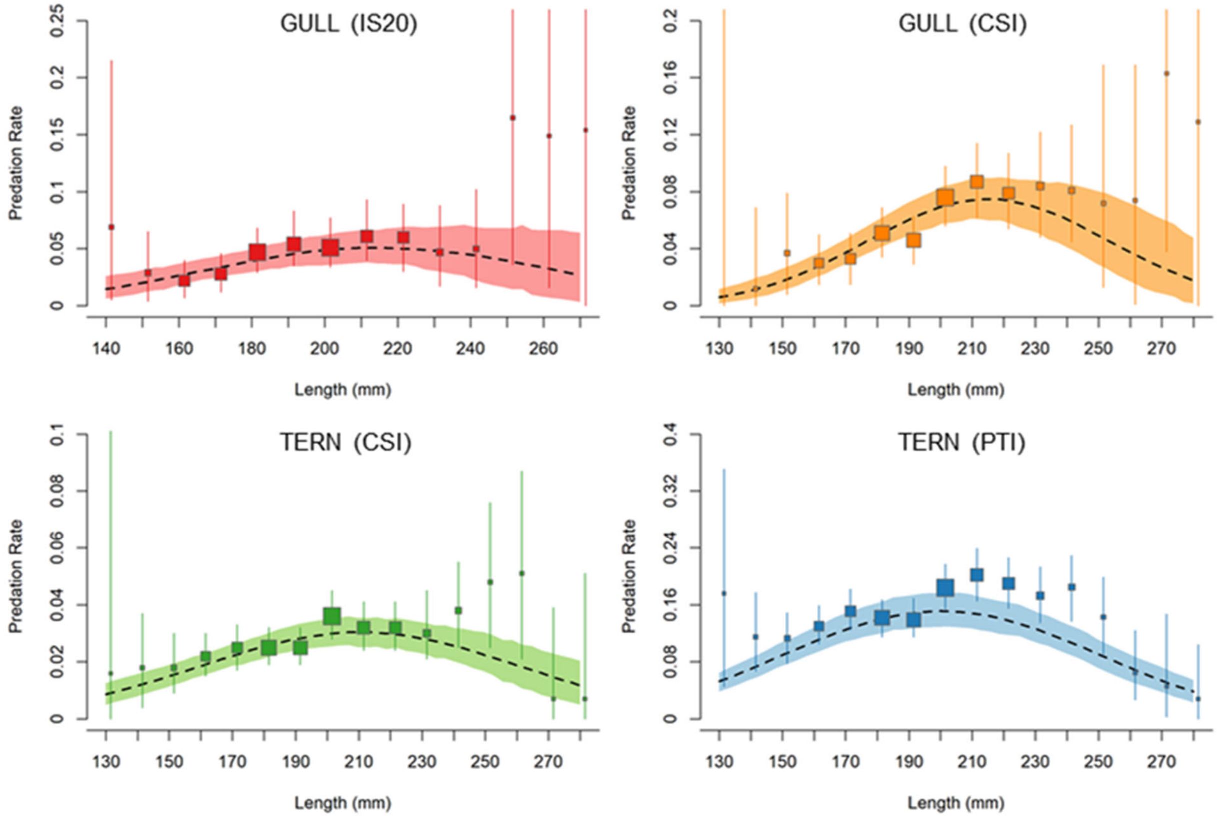


Figure 10.8. Weighted estimates of fork-length specific (rounded to nearest cm) annual steelhead predation rates (boxes, error bars represent 95% credible intervals) by Caspian terns (TERN) and gull species (GULL) nesting at Island 20 (IS20), Crescent Island (CSI), or Potholes Reservoir (PTI). Data are from PIT-tagged upper Columbia River steelhead smolts released at Rock Island Dam, Washington during 2008–2017. Curve shaded region represents the best fit from a logistic regression model with the shaded region representing the 95% credible intervals. Adapted from Evans et al. (2019b).

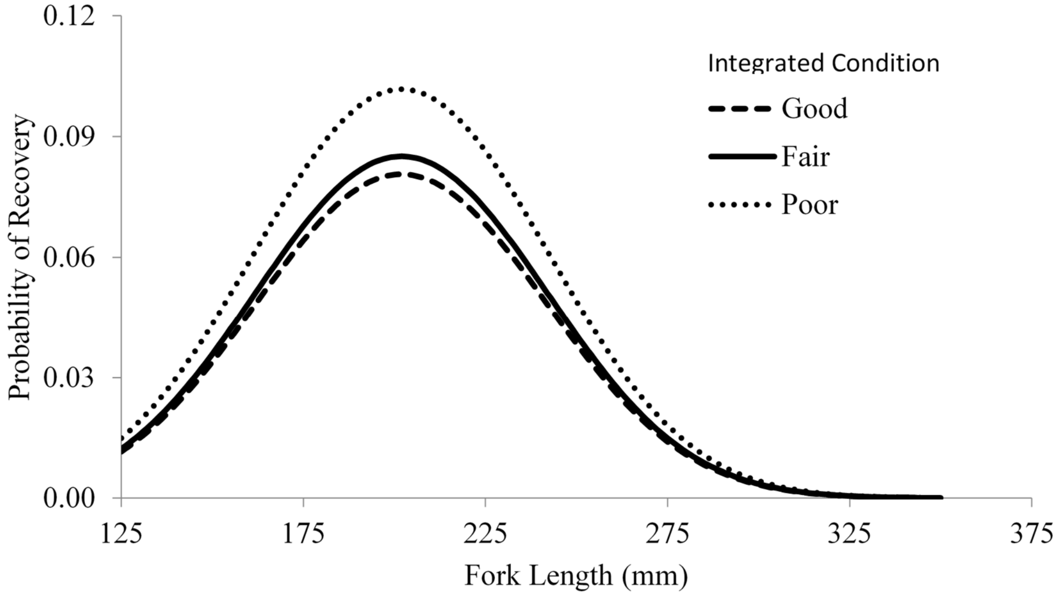


Figure 10.9. Recovery probability of PIT-tagged juvenile steelhead on the Crescent Island Caspian tern colony as a function of fork length and integrated condition value. Figure from Hostetter et al. (2012).

Smolts in degraded condition (descaling, body injuries, disease presence) were more susceptible to tern and cormorant predation, although differences were often small (*Table 10.5* above; *Figure 10.9* above; Schreck et al. 2006, Hostetter et al. 2012, Roby et al. 2015, Evans et al. 2019b). There was no support for condition-dependent susceptibility to gull predation where results were reported (Evans et al. 2019b). Lower gill Na^+ K^+ - ATPase activity (osmoregulatory ability) was consistently linked to increased predation susceptibility in estuaries (*Table 10.5* above; Schreck et al. 2006, Kennedy et al. 2007).

Rearing-type often displayed no appreciable or consistent differences in predation susceptibility for terns, cormorants, or gulls (*Table 10.5* above; *Figure 10.10*; Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2012, Evans et al. 2016c). Increased predation of hatchery-reared individuals was occasionally observed, but concurrent differences in smolt size or run-timing may also explain these results (*Figure 10.10*; Ryan et al. 2003, Kennedy et al. 2007, Hostetter et al. 2012, Roby et al. 2015, Evans et al. 2016c, Evans et al. 2019c). Increased susceptibility of naturally reared steelhead relative to hatchery-reared steelhead by gulls in coastal California was reported by a single study (Osterback et al. 2014).

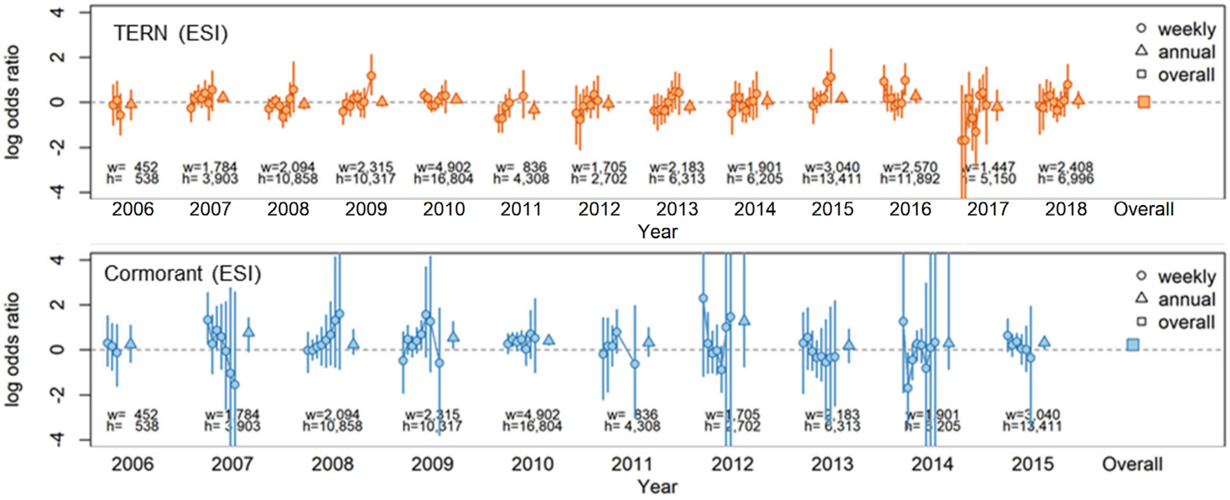


Figure 10.10. Relative susceptibility of Snake River steelhead by rearing-type to predation by Caspian terns (top) or double-crested cormorants (bottom) nesting on East Sand Island (ESI). Values represent the log odds ratio of predation, with values < 0 indicating greater odds of predation for hatchery-reared fish and values > 0 indicating greater odds of predation for naturally reared fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Figure from Evans et al. (2019c).

10.5 Discussion

This synthesis supports the hypothesis that variation in juvenile salmonid susceptibility to avian predation is associated with a complex and dynamic suite of individual fish characteristics, migration-timing, environmental variables, and predator characteristics. We demonstrate how synthesizing results across numerous localized studies revealed commonalities across prey and predator species, while also identifying important differences across space (e.g., marine vs freshwater systems) and predator characteristics. Across salmonids, the juvenile life stage is important to population viability and is often subject to high mortality (Kareiva et al. 2000, McClure et al. 2003, Good et al. 2007, Quinn 2018), with bird predation increasingly recognized as a substantial source of total juvenile mortality in multiple systems (Dieperink et al. 2001, Schreck et al. 2006, Clements et al. 2012, Evans et al. 2019b, Payton et al. 2019). A detailed understanding of the mechanisms influencing fish susceptibility to avian predation, however, is only recently coming to light (Hostetter et al. 2012, Osterback et al. 2014, Tucker et al. 2016). Below, we summarize our key findings, expand upon possible cumulative effects in multi-predator systems, and discuss critical data gaps or uncertainties in our understanding of factors that influence fish susceptibility to avian predation.

10.5.1 Variability Among Salmonid Species & ESUs

Estimation of population-specific predation rates is the most common objective of mark-recovery studies of avian predation. While such direct estimation topics should find a place in any predator-prey study, the relative susceptibility among prey populations also provides crucial information on prey risk of predation, predator-selectivity, mechanisms driving predator-prey dynamics, and study design considerations. For terns, studies consistently found higher predation susceptibility for steelhead trout relative to salmon species (i.e. Chinook, coho, sockeye), while susceptibility among salmon populations was generally similar for sockeye, coho, and yearling Chinook, but lower for subyearling Chinook (Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Evans et al. 2012, Zamon et al. 2013; Roby et al. 2015, Evans et al. 2016b, Evans et al. 2016c, Roby et al. 2017a, Roby et al. 2017b, Evans et al. 2019c). For cormorants, most studies found little to no support for differences in predation susceptibility between steelhead trout and salmon species, or among salmon ESUs in freshwater and estuary systems (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Zamon et al. 2013; Roby et al. 2015, Evans et al. 2016b, Evans et al. 2016c, Evans et al. 2019c). One exception was the higher susceptibility of Lower Columbia River Chinook and coho salmon to cormorant predation in the estuary, which was attributable to greater residency times, differences in out-migration run-timing, and the release of large numbers of hatchery fish in close proximity to cormorant colonies (Zamon et al. 2013, Sebring et al. 2013).

Identifying the relative susceptibility of prey populations is also a first step in understanding the possible mechanisms underlying predator-prey interactions. For example, higher susceptibility of juvenile steelhead to tern predation is often attributed to differences in species-specific length (e.g., steelhead > salmon), migration-timing, and behavioral characteristics (Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Evans et al. 2012). Studies expanding on these hypotheses provided strong support for size-selective tern predation (Hostetter et al. 2012, Evans et al. 2019c) and variability due to migration-timing (Hostetter et al. 2012, Roby et al. 2015, 2017a). Conversely, cormorant predation displayed little to no size-selectivity, whereby smolts of all sizes were equally susceptible to predation (Hostetter et al. 2012, Chiaramonte et al. 2019). Species- and ESU-level variation in susceptibility to tern and cormorant predation may therefore be largely explained by variation in individual fish characteristics (e.g., length), migration-timing, and the subsequent biotic (e.g., predator and prey densities) and abiotic (e.g., river flow, temperature) factors associated with migration-timing.

10.5.2 Environmental Factors

Avian predation of juvenile salmonids occurs in spatially and temporally complex systems. Local- and large-scale environmental factors influence juvenile salmonid survival and susceptibility to avian predation. Relationships with large-scale climate indices in the Columbia River estuary and lower per capita predation rates in estuarine relative to freshwater systems were common across predator species and likely arise from complex factors influencing the availability and abundance of forage fishes in estuary systems (Collis et al. 2002, Roby et al. 2002, Lyons et al. 2014, Evans et al. 2019c; see also *Appendix C*).

Increased turbidity can affect fish susceptibility to predation due to a potential reduction in predator-prey encounter rates, reduced use of cover, and decreased reaction time (Gregory 1993, De Robertis et al. 2003, Strod et al. 2008). Increased turbidity was associated with decreased fish susceptibility to cormorant predation in both experimental (Strod et al. 2008) and field settings (Hostetter et al. 2012), likely due to reductions in predator-prey encounter rates. Conversely, increased turbidity was associated with increased fish susceptibility to tern predation in an observation study, but the mechanism for this relationship remains unknown (Hostetter et al. 2012).

Estuaries may act as survival bottlenecks for migrating juvenile salmonids. Local conditions, however, greatly influence the suite of predators (e.g., birds, mammals, fish), environmental factors (e.g., salinity, flow), and individual fish characteristics (e.g., osmoregulatory ability) influencing predation susceptibility in estuaries (Dieperink et al. 2001, Schreck et al. 2006, McMichael et al. 2010, Clements et al. 2012). Smolts less physiologically prepared to enter marine water were disproportionately consumed by birds in the Columbia River estuary, an area of high predation risk (Schreck et al. 2006, Kennedy et al. 2007). Outside the CRB, avian predation of sea trout also concentrated on the first two days trout were exposed to brackish waters, possibly due to osmoregulatory stress experienced during this transition (Dieperink et al. 2001). In two other estuaries in Oregon, USA, cormorants disproportionately depredated coho relative to steelhead, which may be attributable to increased juvenile coho estuary residency times (Clements et al. 2012).

Overall, studies have attempted to identify the influence of environmental factors on salmonid survival for decades (Kareiva et al. 2000, McClure et al. 2003). Identifying these relationships is notoriously difficult due to the complexity of salmonid life-histories, whereby different life-stages utilize various habitats and experiences in one life-stage may affect mortality in subsequent life-stages (Budy et al. 2002, Muir et al. 2006, Schreck et al. 2006, Ferguson et al. 2006). Similar to relationships with salmonid survival, identifying the effects of environmental factors on within- and among-year variation in predation susceptibility remains challenging and is only partially understood.

10.5.3 Prey Density, Predator Density, & Migration Timing

In the CRB, multiple studies suggest that earlier migrating individuals often experience higher survival compared to later migrating individuals of the same ESU (Scheuerell et al. 2009, Hostetter et al. 2011, Haeseker et al. 2012, Evans et al. 2014, Evans et al. 2016b) and that avian predation may be a primary mortality source during juvenile outmigration for some populations (Evans et al. 2016a, Evans et al. 2019b, Payton et al. 2019). While no single study has fully linked the dynamics of migration timing, survival, and avian predation, our synthesis provides several unique perspectives on these shared processes.

First, a number of studies now suggest predator swamping mechanisms may decrease predation susceptibility as the number of prey available increases (Ims 1990), including studies of juvenile salmonid survival and predation (Hostetter et al. 2012, Roby et al. 2015, Roby et al. 2017a, Roby et al. 2017b, Furey et al. 2016; see also *Chapter 9*). Predator swamping often occurs quickly under a Type II functional response, where rates of consumption (i.e. prey consumed per predator per unit time) level off at an upper limit due to predator satiation or handling time, resulting in predation rates that decrease as prey abundance increases (Denno and Lewis 2009; see also *Chapter 9*). Tern diet studies in the CRB consistently find high proportions of salmonids, particularly for terns nesting at inland, freshwater colony locations (Collis et al. 2002, Roby et al. 2002, Roby et al. 2003, Antolos et al. 2005). As such, tern consumption of salmonids by biomass may quickly reach a maximum as the number juvenile salmonids increases. This results in terns consuming more juvenile salmonids (by number or biomass) as availability increases up to a threshold, while predation rates and individual smolt susceptibility to tern predation declines due to the effects of predator swamping (*Figure 10.7* above; Hostetter et al. 2012, Roby et al. 2015, Roby et al. 2017a, Roby et al. 2017b, Evans et al. 2016b; see also *Chapter 9*).

Second, functional responses likely vary among predator species due to diet specialization and differences in foraging strategies (Solomon 1949, Holling 1959). For example, the diets of cormorants are generally more diverse and lower in percent salmonids relative to terns nesting at nearby colonies (Collis et al. 2002; see also *Chapter 9*). While a Type II functional response may best describe tern predation, there was some evidence for a Type III response to describe cormorant predation in the CR estuary (see *Chapter 9*). For cormorants, it appeared that consumption increased as smolt availability increased (e.g., prey-switching to more common prey), but will eventually approach a threshold where consumption is maximized (*Figure 10.7* above; Hostetter et al. 2012, Roby et al. 2015, Evans et al. 2016b, Roby et al. 2017a; see also *Chapter 9*). The threshold of prey abundance that eventually swamps the predator (e.g., abundance at which consumption rates flatten) may be higher for cormorants relative to terns due to their increased capacity for prey-switching and exploitation of temporarily available prey. Differences in functional responses, however, do not imply cormorants are consuming more or less salmonids than terns, but instead describe how each of these predator species respond to changes in prey abundance (Solomon 1949, Holling 1959; see also *Chapter 9*).

Third, the functional responses of gulls are poorly understood (see *Chapter 9*). Gulls have substantial capacity to exploit temporary food sources (i.e. prey switching) due to their omnivorous and diverse diets (Collis et al. 2002). The single study describing relationships between prey availability and gull predation (Roby et al. 2017b) found that steelhead susceptibility to gull predation remained similar across a range of smolt abundances in a single year, most reflective of a Type I or Type III functional response. Type I functional responses are generally unrealistic in natural settings as (i) predators will eventually become satiated as prey abundance continues to increase and (ii) even if not satiated, predators are limited by handling time (Denno and Lewis 2009). Instead, prey-switching or local numerical responses through spatial redistribution (e.g., gulls forage on smolts when they are available) may assist in describing these relationships and understanding these relationships should be the focus of future studies (see also *Chapter 9*).

Clearly, investigations of predator-specific functional responses and their influences on prey population viability are topics worthy of additional study beyond this review paper. Understanding functional responses, predator swamping, prey-switching, migration-timing, and prey survival in multi-predator systems requires information on predator abundance, prey abundance, and per capita consumption or predation rates. Additionally, while most current analyses of per capita predation rates focus on annual colony size and annual predation rates (Evans et al. 2012, Evans et al. 2019c), there is also important insights to be gained from within season variation in these processes (e.g., Evans et al. 2013, Roby et al. 2017b). Accomplishing these objectives requires simultaneous studies to monitor abundance at different bird colonies, mark-recovery studies evaluating population-specific predation rates, and regional databases on prey availability (see *Chapter 9*). Some of these topics may be preliminarily explored using new analytical procedures and existing data (e.g., comparisons of predation rates, consumption rates, colony sizes, and how they relate to survival), while others will require new field studies and data collection approaches (e.g., concurrent monitoring of colonies, diets, and mark-recovery). We recommend that future studies directly investigate and attempt to quantify predator-specific functional responses across multiple predators, years, and colony locations to describe functional relationships and their implications to juvenile salmonid survival under different levels of predation pressure (see also *Chapter 9*).

10.5.4 Individual Fish Characteristics

Size-selective susceptibility to tern and gull predation occurred within the size range of most juvenile salmonids (60–300 mm). For terns and gulls, results support a hypothesis of disruptive size-selectivity where the smallest and largest individuals are less susceptible to predation (Hostetter et al. 2012, Evans et al. 2019b). Reduced susceptibility of larger and smaller fish may arise from multiple selection pressures including reduced encounter rates or capture efficiency for smaller prey and increased swimming ability or growth beyond predator gape width for larger prey (Hostetter et al. 2012, Evans et al. 2019b). ESU-specific predation rates provide further support for size-selective predation, whereby tern predation is often highest on steelhead, lower for yearling Chinook, coho, and sockeye, and lowest for subyearling Chinook, which generally follows the average length distributions for these species (Evans et al. 2012, Roby et al. 2015). Similarly, cormorant predation displayed little to no size-selectivity across a variety of length ranges, corresponding to the lack of trends in ESU-specific cormorant predation rates (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2012, Zamon et al. 2013, Roby et al. 2015, Evans et al. 2016c, Evans et al. 2016b, Evans et al. 2019c).

Disruptive size-selective predation by terns was most frequently documented in steelhead populations, with predation susceptibility highest for steelhead near 175–225 mm in length (Hostetter et al. 2012, Evans et al. 2019b). Most juvenile salmon (e.g., sockeye, subyearling Chinook, yearling Chinook) are < 175 mm and studies of predation on salmon generally display a linear, positive relationship between length and predation susceptibility (Hostetter et al. 2015b, Roby et al. 2017a). Linear size-selective tern predation, whereby larger fish suffer disproportional predation pressure, directly contrasts with multiple studies supporting 'bigger is better' hypotheses for salmon survival in the CRB where survival increased for larger individuals (Zabel et al. 2005, Hostetter et al. 2011, Evans et al. 2014, Hostetter et al. 2015, Faulkner et al. 2019). Most studies have thus far separately investigated survival and predation, leaving the direct effect of size-selective predation of larger individuals on outmigration survival and prey demographics (e.g., disproportionate consumption of larger individuals with higher survival probabilities) poorly understood (Evans et al. 2019c). While both size-selective predation and size-selective survival are now well described, the direct link between these processes and effects on prey populations remain unexplored.

Predation rates can be high without affecting population growth rates due to a host of factors including mortality compensation (via survival, reproduction, or immigration), disproportionate predation of individuals with low reproductive values (e.g., doomed surplus; Errington 1946), and the relative importance of the affected demographic rates to population growth (e.g., juvenile vs adult survival; Caswell 2001). In a relatively unique approach to jointly examine these factors, a multi-year study of Snake River steelhead and Upper Columbia River steelhead investigated the effects of individual fish characteristics on outmigration survival, susceptibility to avian predation, and adult returns (Hostetter et al. 2011, Hostetter et al. 2012, Connon et al. 2012, Evans et al. 2014, Evans et al. 2019a). Together, these studies found that juvenile steelhead in degraded condition (e.g., body injuries, descaling, or external symptoms of disease) were less likely to survive juvenile outmigration, less likely to return as adults, and more likely to be consumed by avian predators. Certainly, all depredated individuals had their fitness reduced to zero, however, condition-dependent avian predation suggests that birds disproportionately depredated fish with lower overall fitness (i.e. less likely to survive to adulthood). Levels of condition-dependent predation, however, were generally small, only a small proportion of juvenile salmonids appeared physically degraded, and predation and survival occasionally displayed stronger relationships with length- and migration-timing factors than condition-dependent influences (Hostetter et al. 2011, Evans et al. 2014). Together, these results suggest that condition-dependent predation exists, but may play a relatively minor role on population-level dynamics. Similar studies directly linking individual fish characteristics to consistent trends in predation, life-stage specific survival, and survival to adulthood are generally lacking for other salmonid populations.

Hatchery-rearing systems may select for bolder individuals that are more active, more surface oriented, less able to endure sustained swimming, more aggressive, and less experienced with predators relative to their naturally-reared counterparts (reviewed by Maynard et al. 1995, Sundström et al. 2004). Additionally, hatchery-reared fish are often larger than their naturally reared equivalents, making it difficult to disentangle competing hypotheses describing the influences of inherent behavioral tendencies and size-selectivity on survival and predation of hatchery- vs naturally-reared smolts (Miyamoto et al. 2018). Hypotheses for behavioral tendencies increasing avian predation susceptibility of hatchery-reared salmonids are logical, but thus far are unconvincingly supported. For example, multiple studies from the CRB observed little support for differences in the predation susceptibility of hatchery-reared vs naturally-reared individuals and when observed, differences were often small and inconsistent (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2012, Evans et al 2016c, Evans et al. 2019c). Instead, factors such as length, migration-timing, and predator characteristics appeared to better explain differences in smolt susceptibility to avian predation relative to inherent differences between hatchery-reared and naturally-reared salmonids.

Pacific salmonids rely on both endogenous (e.g., hormone concentrations) and environmental cues (e.g., photoperiod, water temperatures, stream flows) for initiating outmigration (Quinn 2004, Beckman et al. 2000). Various studies have indicated a general trend where earlier migrating juveniles experience higher survival (Scheuerell et al. 2009, Hostetter et al. 2011, Haeseker et al. 2012, Evans et al. 2014), however, there remains a tradeoff between earlier migration and attaining a larger size as a means to improve outmigration survival (Zabel and Achord 2004, Zabel et al. 2005, Scheuerell et al. 2009, Hostetter et al. 2015b). Ultimately, migration timings that maximize fitness should be favored (e.g., survival to adulthood), which may include migrating when predation risk is lower, waiting until a critical size is attained (Beamish and Mahnken 2001), increasing migration synchrony to swamp predators (Ims 1990), or exploiting optimal conditions in the estuary or nearshore ocean (Pearcy 1992). Further studies aimed at teasing apart these mechanisms and their complex interactions with predation susceptibility and total survival will lead to improved models to predict population-level responses of management actions and future conditions for salmonid populations (Sogard 1997, Steinmetz et al. 2003).

10.5.5 Conclusions

Predation is often a key mortality factor in natural systems, although the impacts of predators on prey population dynamics can be highly variable (Estes et al. 2011). The importance of predation as a key factor in population fluctuations may range from nearly negligible to substantial and direct impacts of population growth rates are usually even more difficult to detect (Caswell 2001). Prey density, predator density, and the spatiotemporal overlap of predators and prey populations are often key factors in describing predator-prey interactions (Solomon 1949, Holling 1959, Skalski and Gilliam 2001, Miller et al. 2006). While there is a growing interest to investigate theories of predator functional responses using empirical field data from wild populations, few studies have directly quantified these relationships (Gross et al. 1993, Skalski and Gilliam 2001, Miller et al. 2006, Rosenbaum and Rall 2018). Fish mark-recovery studies focused on avian predation of juvenile salmonids provide tremendous opportunities to bridge this gap and jointly investigate survival, predation, and the ecological processes determining prey-population dynamics (Rosenbaum and Rall 2018, Evans et al. 2019a, Payton et al. 2019).

Discerning the level of detail required to make adequate predictions about population-level impacts is an important challenge in predator-prey studies. Deposition probability (i.e. probability a consumed tag is deposited on a bird colony) and detection probability (i.e.

probability a deposited tag is detected by researchers) are required for accurate comparisons of predation rates among predator species, colonies, and years in mark-recovery studies of avian predation on fish populations (Hostetter et al. 2015a; see also *Appendix A.2*). To understand the influences of predation on prey population dynamics (e.g., survival, population growth rates), however, researchers should consider several additional candidate details. First, individual fish characteristics can dramatically influence predation susceptibility within and across populations (see *Individual Fish Characteristics*). Selective predation on individuals with low reproductive value (e.g., sick, injured, doomed to die from other causes) or characteristics associated with reduced fitness can result in predation having little effect on the viability of prey populations (Errington 1946). Conversely, selective predation on individuals with high reproductive value (e.g., larger fish) can result in larger effects on prey population viability than expected from simple evaluation of total predation rates. Second, evaluation frameworks should aim to identify consistent population- and community-level factors affecting predation and survival by incorporating multiple prey species (e.g., steelhead vs Chinook) while also acknowledging differences across populations within a species (e.g., Upper Columbia River steelhead, Snake River steelhead). Failure to do so will likely lead to poor understanding of multi-species factors affecting both survival and predation and their influences on ecosystem-based management strategies. Third, factors influencing predation susceptibility can be driven by predator-specific characteristics (e.g., foraging strategies, functional responses). Here, fish susceptibility to predation by terns (ambush plunge-divers), cormorants (pursuit divers), and gulls (opportunistic surface feeders) displayed several unique relationships (e.g., different functional responses, size-selectivity) that were often masked when analyses reduced the community of predators to simply ‘avian predation’. Fourth, the apparent simplicity among predation, survival, and population viability can often be misleading. Methods that jointly model predation and survival and the factors affecting those processes now provide new opportunities to explore predator-prey dynamics and directly evaluate the impacts of predation on the long-term viability of prey populations (Schaub and Pradel 2004, Hostetter et al. 2018, Payton et al. 2019, Haeseker et al. 2020, Payton et al. 2020).

Finally, many studies of avian predation on fish populations excellently described the interactions between a single predator and prey species and how individual prey characteristics may result in increased predation risk (Hostetter et al. 2012, Osterback et al. 2014, Tucker et al. 2016, Evans et al. 2019c). However, predators and prey do not occur in isolation, but are nested inside complex multi-predator, multi-prey systems that include a variety of ecological processes (Sih et al. 1998). In particular, the spatial and temporal overlap of avian predators and fish populations create a host of unique challenges regarding the spatial distribution of predators and prey, interactions among predators (e.g., competition, interference, synergistic), and how management actions to change predator abundance at one location affects predators, prey, and cumulative predation over larger spatial scales. Holistic studies focused on these interacting factors will have tremendous application to both basic and applied ecology and should be emphasized in future study designs.

10.6 Acknowledgments

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APPENDIX A: Standardized Methods for Data Collection & Analysis

We began investigating the nesting and foraging ecology of piscivorous colonial waterbirds (i.e., Caspian terns [*Hydroprogne caspia*], double-crested cormorants [*Phalacrocorax auratus*], American white pelicans [*Pelecanus erythrorhynchos*], and various gull species [Larus spp.]) in the Columbia River Basin in 1997. The impetus for this work was to assess the impacts of predation by colonial waterbirds on the survival Endangered Species Act (ESA)-listed juvenile salmonids (*Oncorhynchus spp.)*; information that was needed to help develop management initiatives to improve smolt survival in the basin (USFWS 2005; USACE 2014, 2015). The geographic region within the Columbia River Basin which was the focus for these studies included the Columbia River estuary, the lower Columbia and Snake rivers, and other waterbodies where colonial waterbirds might nest that were within known or suspected foraging range of anadromous juvenile salmonids in the Columbia and Snake rivers (*Figure A.1*). We also investigated nesting and foraging ecology of Caspian terns at nest sites outside the Columbia River Basin at islands constructed by the U.S. Army Corps of Engineers as alternative nesting habitat for terns displaced from managed colonies within the basin (USFWS 2005; USACE 2014). Field work was conducted primarily during the breeding season for piscivorous colonial waterbirds and the outmigration period for juvenile salmonids, which largely overlap (April–July). The data metrics that have been of primary concern to resource managers have been colony size (i.e., number of breeding pairs) and predation rates on juvenile salmonids using recoveries of Passive Integrated Transponder (PIT) tags on bird colonies. The detailed methods used to collect and analyze these data are provided herein.

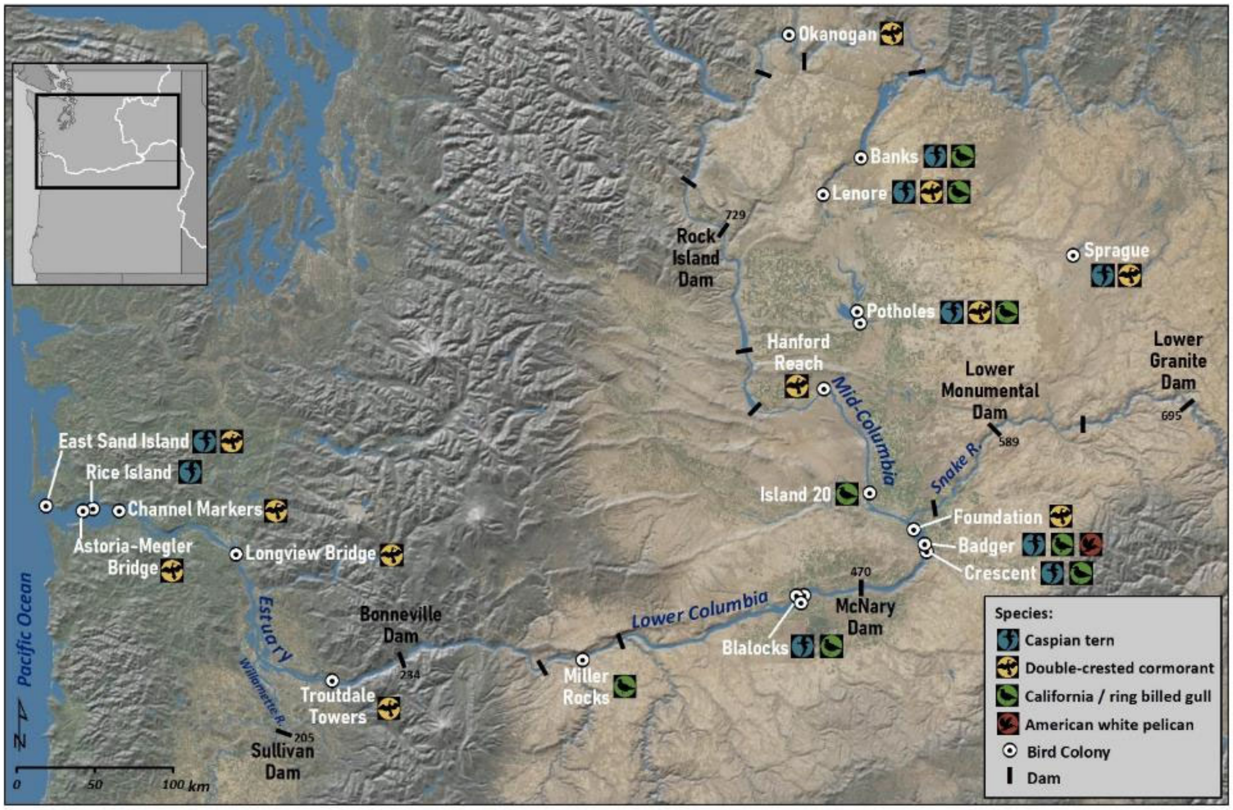

Figure A.1. Locations of piscivorous waterbird colonies and mainstem hydroelectric dams in the Columbia River basin. Numbers represent the distance in river km (Rkm) to the mouth of the Columbia River.

A.1 Colony Size

We completed aerial-, boat-, and ground-based surveys early in the breeding season to identify all active breeding colonies of piscivorous waterbirds within the study area (*Figure A.1* above). Colonies were visited daily or periodically (weekly or monthly) throughout the breeding season to assess nesting chronology (pre-laying, incubation, chick rearing, and fledging periods) and colony attendance (number of adults on colony). Estimates of colony size (number of breeding pairs) for piscivorous waterbirds were obtained late in incubation, when the greatest numbers of nesting adults are aggregated on colony (Bullock and Gomersal 1981; Gaston and Smith 1984). Although it is possible that small colonies (i.e. < 10 breeding pairs) may have been missed during these surveys, we are confident that all colonies of consequence were identified within the study area.

A.1.1 Caspian Terns

Caspian terns prefer to nest on the ground on predator-free islands (Cuthbert and Wires 1999), typically on unvegetated substrate (Quinn and Sirdevan 1998). We monitored Caspian tern colonies within and outside (i.e., at Corps-constructed islands, see *above*) the Columbia River basin during the 1997–2018 and 2008–2018 breeding seasons, respectively. All colonies were monitored from blinds or other nearby vantages to avoid disturbance to the colony that might affect colony size and productivity estimates, as well as to prevent unintended impacts (e.g., nesting failure) at the colony.

At large Caspian tern colonies in the Columbia River estuary (i.e., Rice Island and East Sand Island), we estimated colony size using high-resolution, vertical aerial photography of the colony taken at the peak in colony attendance (late May, corresponding to the late incubation period). We then used a custom application developed in ArcGIS to count target birds in the resulting aerial imagery. To estimate the number of breeding pairs on the colony, we used the average of three independent counts of all adult terns (incubating and non-incubating) on the colony in the aerial imagery multiplied by the proportion of all terns counted on plots that that were incubating at the time of the image capture. Confidence intervals for colony size were calculated using a Monte Carlo simulation procedure to incorporate the variance in the multiple counts from the aerial photography and the ground counts of incubating terns. These estimates were calculated one thousand times using random draws from the sample distributions. Standard errors and confidence intervals for colony size were derived from the resulting distributions.

At smaller colonies in the Columbia Plateau region and those outside the Columbia River basin, we estimated colony size using direct ground counts of incubating adults. We conducted our ground counts to estimate colony size at the peak in colony attendance (early May to mid-June, depending on colony). We based our Caspian tern colony size estimates on the maximum number of incubating terns counted on the colony late in the incubation period. Due to the limitations of this method, no measure of variance for our estimates of colony size at small tern colonies are available but were not expected to be great. When possible, estimates of the number of breeding pairs from ground counts were verified using counts of nesting adults in oblique aerial photography taken near the peak in incubation period. During the 2016–2018 breeding seasons, we used counts of nesting adults in oblique aerial photography taken near the peak in incubation period to estimate the number of breeding Caspian terns at the USACE-constructed islands in southern Oregon and northeastern California.

A.1.2 Double-crested Cormorants

Unlike Caspian terns, which generally nest on islands with unvegetated substrate, double-crested cormorants nest in a variety of habitats including in trees, on islands with rocky or sandy substrate, on emergent vegetation in marshes, and on artificial structures such as bridges, navigational markers, and transmission towers (Dorr et al. 2014). As with Caspian terns, we observed double-crested cormorants at colonies within the Columbia River Basin from blinds or other nearby vantages to avoid disturbing the nesting birds, with visits to colonies occurring on a daily or weekly basis to assess nesting ecology (see *below*).

At the large terrestrial double-crested cormorant colony on East Sand Island, we estimated the peak number of attended nests using high-resolution, vertical aerial photography of the colony taken at the peak in colony attendance (late May to early June). We used the average of three independent counts of the number of attended nests visible in aerial imagery as our estimate the colony size (number of breeding pairs) and used the standard errors from these counts to estimate a confidence interval for the estimate. Beginning in 2008, we expanded the use of aerial photography to estimate colony attendance (number of adults on colony) across the entire breeding season. High-resolution aerial imagery of the cormorant colony was taken approximately every 2 weeks from early May to early September. Beginning in 2006, Brandt's cormorants began nesting within the much larger double-crested cormorant colony on East Sand Island. In each year from 2006–2018, we delineated the boundaries of nesting Brandt's cormorants within the entire colony area from nearby observation blinds so that the two species of cormorants could be counted separately from aerial images. As was the case with Caspian terns, we developed and used a custom application in ArcGIS to count attended nests or individual birds in the aerial imagery of the East Sand Island cormorant colony.

At all other double-crested cormorant colonies, which were smaller than the East Sand Island colony and/or were in trees or on man-made structures (i.e., bridges, transmission towers, or channel markers), colony size was estimated using aerial-, ground-, or boat-based counts of the peak number of active nests during the late incubation and early chick rearing periods. Any well-maintained nest structure attended by a cormorant adult and/or chick was considered active. Estimates of colony size at arboreal colonies (i.e., the Foundation Island, North Potholes Reserve, and Okanagan colonies; *Figure A.1* above) should be considered minimums, as vegetation partially obscured some nests over the course of the breeding season.

A.1.3 Gull Species

Like terns, gulls in the Columbia River Basin (i.e. California gulls [*L. californicus*], ring-billed gulls (*L. delawarensis*], and glaucous-winged/western gulls [Larus spp.]) nest on the ground on predator-free islands (Pierotti and Annett 1995; Winkler 1996; Hayward and Verbeek 2008; Pollet et al. 2012). However, unlike terns, gull nest in and amongst low growing vegetation. For the select number of years when colony size estimates of gulls are available, we used counts of either vertical or oblique aerial imagery of gull colonies during late incubation to derive our estimates. Most, but probably not all, of the gulls present on a colony were visible in the aerial imagery due to some nests being obscured by vegetation, as such these counts should be considered minimums. Furthermore, since we did not have ground counts of incubating gulls at the time of the aerial photo census, counts from aerial imagery are an index to the number of breeding pairs utilizing the colony.

A.1.4 American White Pelicans

Like gulls, American white pelicans nest on the ground amongst vegetation on predator-free islands (Knopf and Evans 2004). We used aerial imagery taken of the colony during the late incubation period to estimate colony size (i.e. number of breeding pairs) at American white pelican colonies in the Columbia River estuary (Miller Sands Spit and Rice Island) and counted the number of individuals at colonies on the lower Columbia River (Badger island and Crescent Island). Although most pelicans present on these colonies were visible in the aerial imagery, some nesting individuals were likely obscured by overhanging trees or shrubs, especially at the Badger Island colony where the pelican's nest under dense sagebrush. As such, counts of American white pelican breeding pairs at colonies in the Columbia River estuary should be considered minimum estimates, whereas counts of adult pelicans at Badger Island should be considered an index to the number of breeding pairs using that island.

A.2 Predation Rates

Since 1987, passive integrated transponder (PIT) tags have been placed in juvenile salmonids (from the Columbia River basin to study their behavior and survival following tagging and release (Prentice et al. 1990). PIT tags can provide specific information on individual fish, including species (Chinook salmon [*O. tshawytscha*], coho salmon [*O. keta*], sockeye salmon [*O. nerka*], steelhead trout [*O. mykiss*], and others), population (Evolutionary Significant Unit or Distinct Population Segment [ESU/DPS]), run-type (spring, summer, fall, winter), rear-type (hatchery, wild), release location, release time, and various fish attributes (e.g., size, condition, and others). Post-release recaptures of PIT-tagged fish can also provide information on migration timing, dam passage routes, smolt survival, and smolt-to-adult return rates based on detections (passive interrogations) of live fish passing hydroelectric dams or other sites with in-river PIT tag detection capabilities. PIT tag detections can also be used to assess mortality of individual fish based on recoveries of tags from fish that succumbed to harvest, predation, or other known sources of mortality.

Beginning in 1996, recoveries of smolt PIT tags on piscivorous waterbird colonies have been used to calculate avian predation rates or probabilities (percentage or proportion of available fish consumed by birds [Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Evans et al. 2012, Sebring et al. 2013, Hostetter et al. 2015, Evans et al. 2016a, Payton et al. 2019, Evans et al. 2019, Payton et al. 2020). Studies have been conducted at colonies located throughout the Columbia River basin, with the principal predator species on salmonid smolts identified as Caspian terns, double-crested cormorants, California gulls, ring-billed gulls, and American white pelicans. Since studies began in 1996, over 1.5 million PIT tags (detections of individual fish) from 11 different fish species have been recovered on over 25 different piscivorous colonial waterbird breeding sites in the Columbia River basin. These detections provide a large, multiyear dataset in which to evaluate predation. The vast majority (ca. 99%) of tags were from anadromous juvenile salmonids, but tags from other fish species (Pacific lamprey [*Entosphenus tridentatus*], white sturgeon [*Acipenser transmontanus*], bull trout [*Salvelinus confluentus*], northern pikeminnow [*Ptychocheilus oregonensis*], smallmouth bass [*Micropterus dolomieu*], and American shad [*Alosa sapidissima*]) and tags implanted in adult salmonids have also been recovered on bird colonies.

Prior to 2014, most predation rate estimates were reported as minimum estimates of predation (e.g., Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Evans et al. 2012, Sebring et al 2013) because an unknown and unaccounted for proportion of ingested PIT tags by colonial waterbirds were deposited at off-colony locations (e.g., at loafing/roosting/foraging locations) and because a proportion of deposited tags were lost or damaged (e.g., blow off the colony during wind storms, washed away by flooding events, or otherwise damaged) or because tags were simply not detected by researchers after the breeding season (Hostetter et al. 2015). Here, we report predation rate estimates that have been corrected for PIT tag deposition and detection probabilities for all colonies and years where adequate data were available for analyses. This retrospective analysis of historic PIT tag datasets provides accurate, unbiased, and standardized estimates of avian predation rates across years, piscivorous waterbird species, and colonies; information needed to assess the relative effects of bird predation on smolt mortality and to assess the efficacy of on-going avian predation management initiatives aimed at increasing smolt survival (see *Chapters 1–4).*

A.2.1 PIT Tag Recovery

The methods of Evans et al. (2012) were used to recover (electronic detection) smolt PIT tags *in situ* on colonies of piscivorous waterbirds after the birds dispersed from each colony following the breeding season (August–October, depending on the colony and year). The area occupied by nesting birds on each colony was determined using surveys (aerial or ground-based) conducted during the peak of the breeding season (typically May–June, depending on the colony and year). In brief, PIT tag antennas were used to systematically scan the entire colony area (referred to as a “pass”) occupied by nesting birds, with a minimum of two complete passes conducted at each colony (*Figure A.2*). The orientation or directionality of antennas relative to nesting substrate was changed following each pass, a technique used to achieve higher rates of PIT tag detection (Ryan et al. 2003, Evans et al. 2016a). In addition to electronic detection, PIT tags were also physically removed from some colonies using a tow behind sweeper magnet (*Figure A.2*). The physical removal of PIT tags reduces tag collision, a phenomenon that renders PIT tags near each other undetectable using electronics. The physical removal of PIT tags (and subsequent hand scanning of each tag to acquire its unique code) increases PIT tag detections at sites where tag densities were high, like on the East Sand Island and Crescent Island Caspian tern colonies (Evans et al. 2012, Evans et al. 2016a). Both physical and electronic PIT tag recovery were conducted concurrently when conditions permitted.

Figure A.2. Examples of equipment used to recover PIT tags from bird colonies, which include a flat-plate antenna (upper left panel), a hand-held antenna (upper middle panel), and a magnet (upper right panel). A flat-plate antenna and tow-beyond magnet attached to an ATV were used to scan the Caspian tern colony on East Sand Island (lower left panel) and a hand-held antenna system was used to scan all other colonies, including the double-crested cormorant colony on East Sand Island (lower right panel).

Tag codes stored locally on PIT tag receivers were uploaded to a central web-based storage drive at the completion of each scanning session, along with metadata regarding the recovery date, bird species, colony location, and pass number. Following validation and removal of duplicate records, newly detected tag codes were reported to the PIT Tag Information System (PTAGIS) using guidelines and protocols established by the PIT-tag Steering Committee (PSFMC 2019).

A.2.2 Predation Rate Calculations

The hierarchical Bayesian model of Hostetter et al. (2015) was used to estimate avian predation rates based on recoveries of smolt PIT tags from each bird colony. Predation rate estimates were calculated using the proportion of smolt PIT tags found on each bird colony from the available population of PIT-tagged fish found in-river (i.e., smolt availability), and then adjusting that rate by the probability that a consumed PIT tag was subsequently deposited on that colony (i.e., deposition probability) and then detected by researchers following the nesting season (i.e., detection probability; *Figure A.3*).

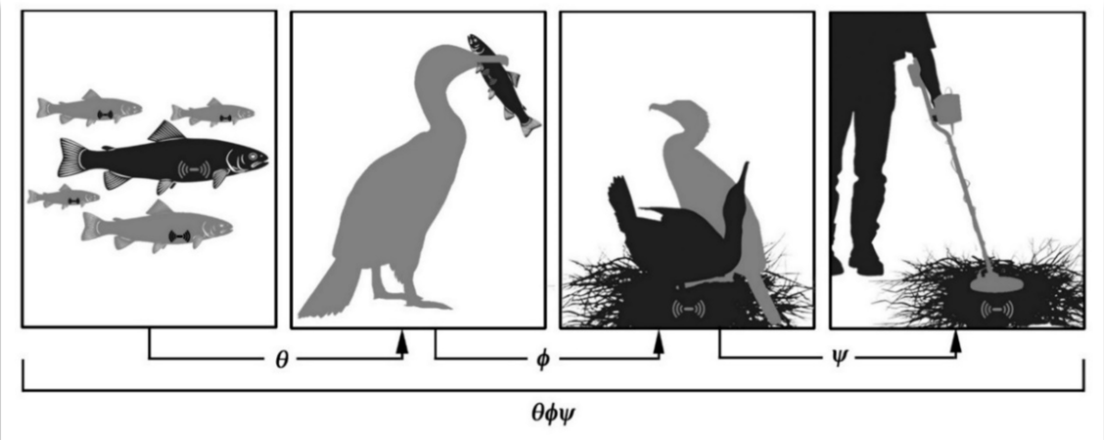

Figure A.3. Conceptual model of the tag-recovery process in studies of avian predation. The probability of recovering a fish tag on a bird colony is the product of three probabilities: a fish was consumed (predation probability, ϑ), deposited on the nesting colony (deposition probability, ϕ), and detected by researchers (detection probability, ψ). Figure from Hostetter et al. (2015).

A.2.2.1 Smolt availability – Smolt availability was based on fish last released or recapture (passive interrogations) in-river from 1 March to 31 August, which reflects the period of overlap between active smolt out-migration and the breeding season for piscivorous colonial waterbirds in the Columbia River basin (Evans et al. 2012; Adkins et al. 2014). PIT-tagged fish were grouped by ESA-listed ESU/DPS, representing a unique combination of the species (steelhead trout, Chinook salmon, sockeye salmon, coho salmon), run-type (spring, summer, fall, or winter), and river-of-origin (Snake River, Upper Columbia River, Middle Columbia River, Lower Columbia River, or Willamette River). The classification of fish from each ESU/DPS followed that of NOAA (2014) and was based on the release location and rear-type (hatchery, wild) of tagged fish relative to the geographic boundary of each ESU/DPS. Fish within each ESU/DPS were further grouped by week and year. Tagging and in-river detection (recapture) data for each PIT-tagged fish were retrieved from PTAGIS, a regional mark-recapture-recovery database (PSFMC 2019).

Smolt availability to birds nesting at each colony was further limited to those fish last interrogated alive upstream and within the maximum foraging range of each bird colony included in the study (see also Evans et al. 2012). For most ESUs/DPSs, this approach eliminated the need to correct or adjust for survival to the foraging vicinity of each colony. For Snake River and Upper Columbia River ESUs/DPSs that originated wholly upstream of McNary Dam, detections were based on PIT-tagged smolts last released/recaptured alive passing Lower Granite Dam (river km [Rkm] 695) and Lower Monumental Dam (Rkm 589) on the lower Snake River, Rock Island Dam on the middle Columbia River (Rkm 729), McNary Dam (Rkm 470) on the lower Columbia River, Bonneville Dam (Rkm 234) on the lower Columbia River, or Sullivan Dam (Rkm 203) on the lower Willamette River.

Maps of release, recapture, and recovery sites for each ESA-listed PIT-tagged ESU/DPS included in the study are provided in *Figures A.4–A.8*.

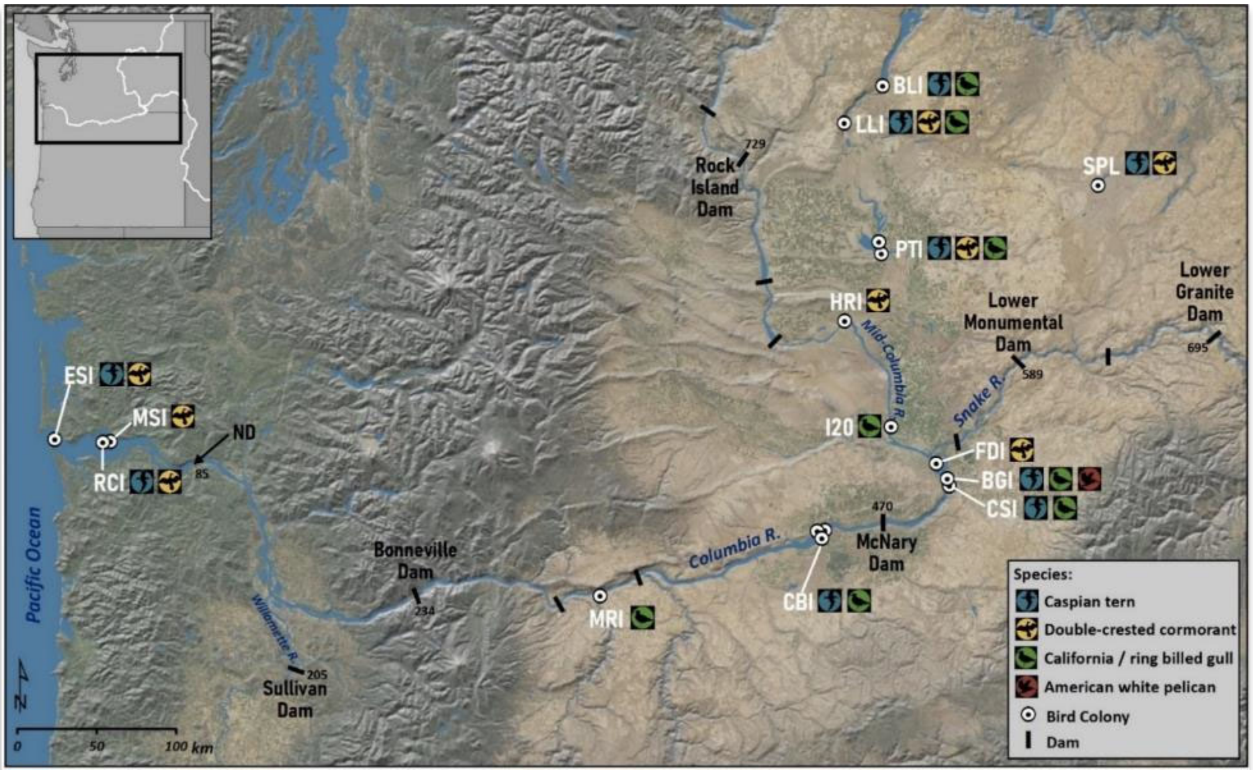

Figure A.4. Recapture-recovery locations of PIT-tagged juvenile salmonids used to estimate avian predation rates. Recapture locations were used to determine fish availability to birds at each colony (see Figures A.5–A.8 for release locations associated with each downstream recapture site). Recovery locations were used to determine the number of fish consumed by birds at each colony. Recapture locations include Lower Granite and Lower Monumental dams on the Snake River, Rock Island Dam on the middle Columbia River, McNary Dam, Bonneville Dam, and a net detector (ND) in the lower Columbia River, and Sullivan Dam on the Willamette River. Avian species are denoted and include birds nesting on Sprague Lake Island (SPL), Banks Lake Island (BLI), Lenore Lake Island (LLI), Potholes Reservoir islands (PTI), Hanford Reach Island (HRI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), Central Blalock islands (CBI), Miller Rocks Island (MRI), Miller Sands Spit Island (MSI), Rice Island (RCI), and East Sand Island (ESI). Distances are reported as river km (Rkm) to the Pacific Ocean.

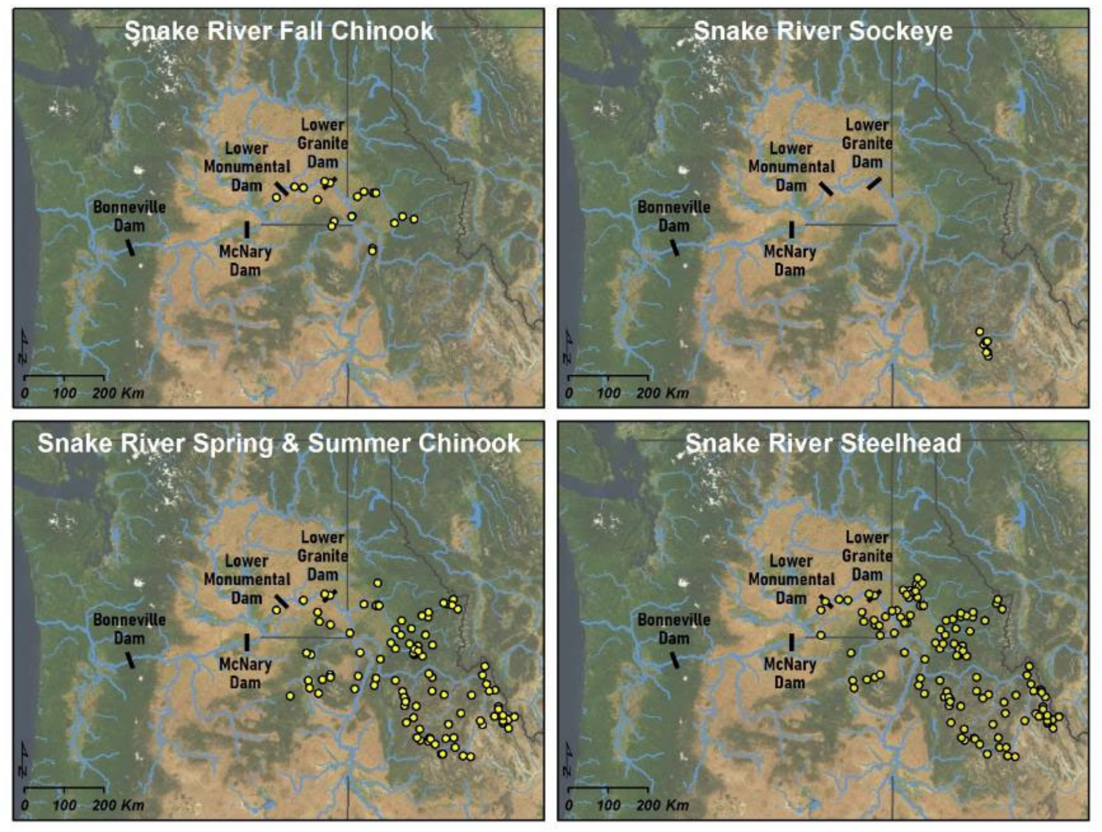

Figure A.5. Release locations for PIT-tagged Snake River steelhead, spring/summer Chinook, fall Chinook, and sockeye used in avian predation rate analyses. The number of released fish subsequently recaptured alive at Lower Granite and Lower Monumental dams on the lower Snake River and McNary and Bonneville dams on the lower Columbia River were used to determine fish availability to each bird colony based on the maximum foraging range of each colony (see Figure A.4 above for colony locations).

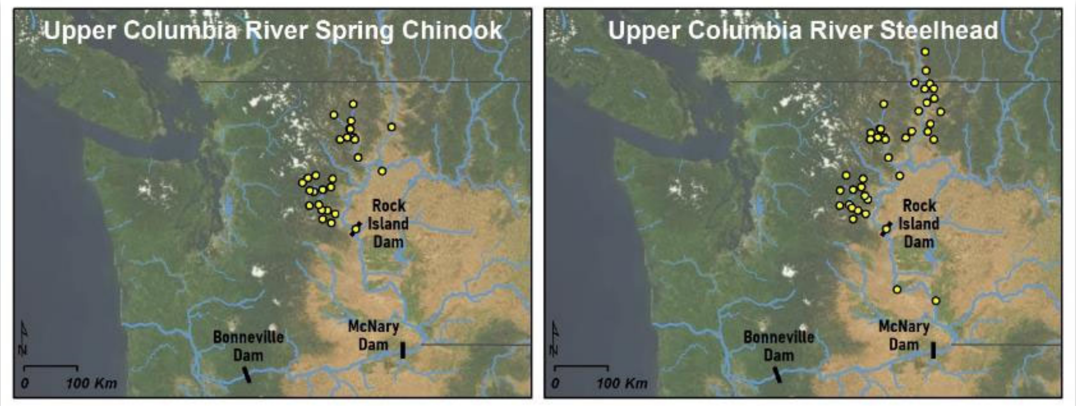

Figure A.6. Tagging and release locations for PIT-tagged Upper Columbia River steelhead and spring Chinook used in avian predation rate analyses. The number of released fish subsequently recaptured alive at Rock Island Dam on the middle Columbia River and McNary and Bonneville dams on the lower Columbia River were used to determine fish availability to each bird colony based on the maximum foraging range of each colony (see Figure A.4 above for colony locations).

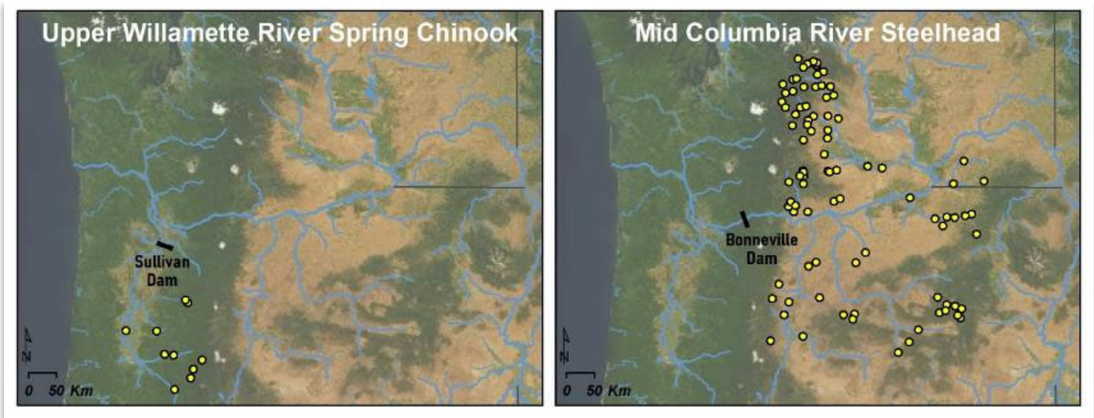

Figure A.7. Tagging and release locations for PIT-tagged Willamette River spring Chinook and Middle Columbia River steelhead used in avian predation rate analyses. The number of released fish subsequently recaptured alive at Sullivan Dam (for Willamette River spring Chinook) or Bonneville Dam (for middle Columbia River steelhead) were used to determine fish availability to each bird colony based on the maximum foraging range of each colony (see Figure A.4 above for colony locations).

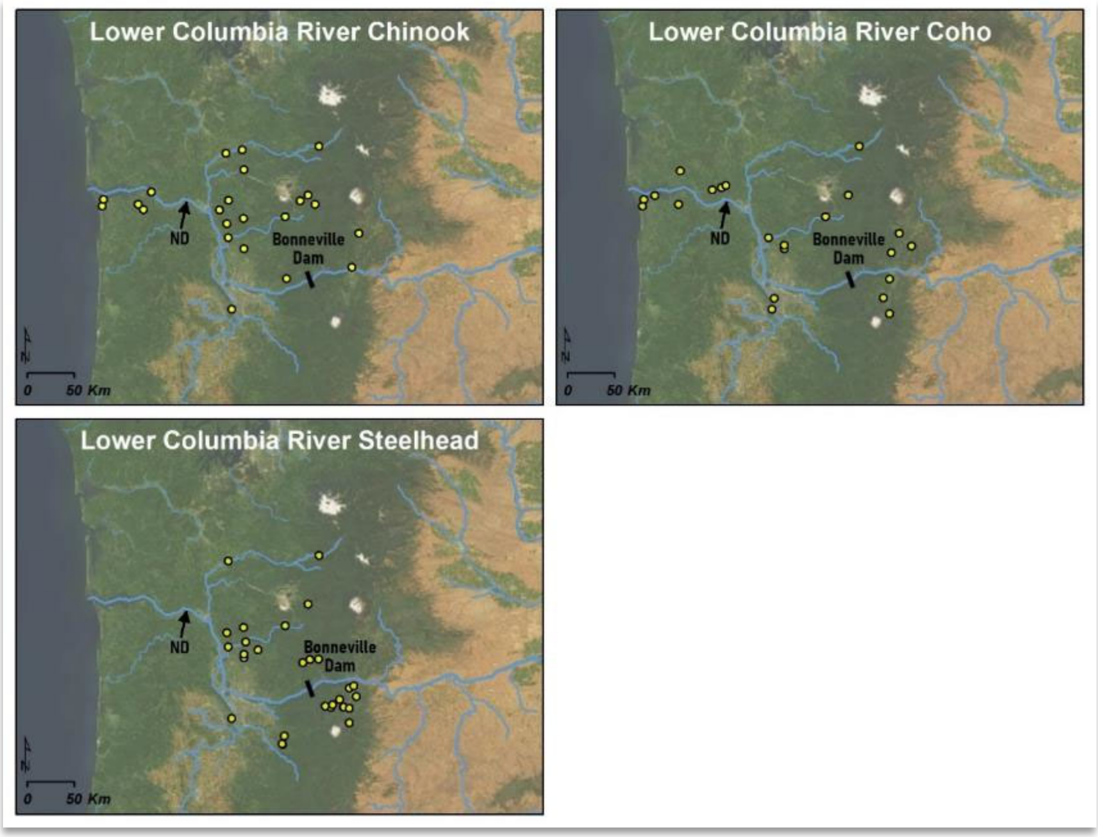

Figure A.8. Tagging and release locations for PIT-tagged Lower Columbia River steelhead, chinook, and coho used in avian predation rate analyses. The number of released fish subsequently recaptured alive at Bonneville Dam or released downstream of Bonneville Dam were used to determine fish availability to each bird colony based on the maximum foraging range of each colony (see Figure A.4 above for colony locations).

For PIT-tagged smolts from Middle Columbia River (MCR) ESUs/DPSs, predation rates were limited to those colonies that foraged exclusively downstream of all MCR smolt release and recapture sites. This was necessary because MCR smolts from several tributaries (e.g., John Day River, Deschutes River, and others) of the Columbia River enter the Columbia River downstream of a given colony's foraging range, preventing inclusion of these fish in predation analyses for some, but not all, colonies in the Columbia River basin.

For PIT-tagged smolts from Lower Columbia River (LCR) ESUs/DPSs, fish that originated both upstream and downstream of Bonneville Dam, availability was based on combination of recaptures at Bonneville Dam and releases of tagged smolts downstream of Bonneville Dam within the geographic boundary of each ESU/DPS. This approach was necessary because a large number and proportion of available LCR PIT-tagged smolts were released downstream of Bonneville Dam but still upstream of the bird colony of interest (e.g., East Sand Island at Rkm 8). In total, up to 24 release locations were included in LCR predation analyses, spanning from the Skipanon (Rkm 13) to Hood River (Rkm 273), depending on the ESU/DPS (*Figure A.8* above). For those LCR fish released upstream of Bonneville Dam, an estimate of survival to Bonneville Dam was necessary and was accomplished by incorporating interrogation records of live fish passing Bonneville Dam and the pair-trawl net detection system in the Columbia River estuary (near Rkm 85; *Figure A.5* above). Annual cohorts within each ESU/DPS were determined by release week and the primary tributary associated with each release site. Once adjusted for survival to Bonneville Dam, releases above Bonneville Dam were combined with those below Bonneville Dam, where no measure of survival could be determined and thus all fish were assumed to be available to birds nesting in the Columbia River estuary (see *LCR Predation Rate Calculations below* for additional details).

In addition to fish volitionally interrogated passing PIT tag arrays or released into tributaries downstream of Bonneville Dam (collectively referred to as "in-river migrants"), PIT-tagged smolts that were loaded into barges at dams on the lower Snake River and transported and release below Bonneville Dam near Skamania Landing (Rkm 225) were also included in predation rate analyses (referred to as "transported fish"). Availability of transported fish was based on fish collected at the Lower Granite Dam, Little Goose Dam (Rkm 635), or Lower Monumental Dam Juvenile Bypass Systems (JBS) and subsequently loaded into a fish barge. Fish were classified as being collected for transportation based on a unique combination of the antenna site (e.g., detected entering a raceway) and date at each JBS. Downstream interrogation histories, JBS facility collection reports, and other sources (e.g., NOAA, USACE, and Fish Passage Center Technical Reports) were used to validate and otherwise proof classifications to ensure accurate assignment of each fish's outmigration history (in-river, transported). Due to small numbers of PIT-tagged fish (generally < 500), smolts collected at JBS facilities and transported using trucks, as opposed to barges, were not included in predation rate analyses (see also Evans et al. 2016a).

Not all ESA-listed ESUs/DPSs in the Columbia River basin were included in predation rate analyses, as some species are either not PIT-tagged or insufficient numbers of tagged fish were available for predation rate calculations. ESA-listed ESUs/DPSs that were not included in the study were Columbia River chum salmon (*O. keta),* Willamette River steelhead trout*,* bull trout, and Eulachon (*Thaleichthys pacificus*). Several other fishes of conservation concern (e.g., Pacific lamprey, White sturgeon, coastal cutthroat trout [*O. clarki clarki*], and others) are PIT-tagged and have been confirmed in the diet of piscivorous colonial waterbirds in the Columbia River basin (Lyons et al. 2014; PSMFC 2019), however, inclusion of these non-listed fish species was beyond the scope of this particular study.

A.2.2.2 Deposition & detection probabilities – Not all PIT tags that are ingested by colonial waterbirds are subsequently deposited on their nesting colony (Hostetter et al. 2015). For instance, a portion of PIT tags consumed by birds are damaged and rendered unreadable following digestion or are regurgitated off-colony at loafing, staging, or other areas used by birds during the nesting season. Deposition probability (i.e. probability that a tag consumed by a nesting bird is deposited on its breeding colony) was previously estimated by intentionally feeding PIT-tagged fish to Caspian terns, double-crested cormorants, and California and ring-billed gulls, with the proportion of known ingested tags subsequently recovered on those colonies used to estimate deposition probabilities (see also Hostetter et al. 2015). The distribution of the median deposition probability derived from those studies were 0.71 (95% credible interval [CRI] = 0.51–0.89) for terns, 0.51 (95% CRI = 0.34–0.70) for cormorants, and 0.15 (95% CRI = 0.11–0.21) for gulls. Results from deposition studies indicated that deposition probabilities did not vary significantly within or between years for each predator species (terns, cormorants, gulls) evaluated, so deposition probabilities were used to correct data in all study years. For Brandt's cormorants breeding East Sand Island, we assumed deposition probabilities for double-crested cormorants nesting on East Sand Island due to similarities in the nesting habitat, forage strategy, and relative size of double-crested and Brandt's cormorants (see also *Chapter 5* and Evans et al. 2016b). No suitable deposition probabilities were available for American white pelicans, however, so estimated impacts of pelican predation on PIT-tagged smolts presented herein are minimum values, values that were corrected for detection probabilities (see *below*) but not deposition probabilities. As such, estimates of America white pelican predation rates should not be directly compared to those of the other piscivorous colonial waterbird species included in the study.

Not all PIT tags deposited by birds on their nesting colony are subsequently detected by researchers after the nesting season (Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2015). For example, tags can be blown off the colony during windstorms, washed away during flooding events, or otherwise damaged or lost during the nesting season. Furthermore, the detection methods used to find PIT tags on bird colonies are not 100% efficient, with some proportion of detectable tags missed by researchers during the scanning process. Unlike deposition probabilities, detection probabilities (i.e., probability that a tag deposited by a bird is detected by researchers after the nesting season) often vary significantly within and between breeding seasons, variation that necessitates a direct measure of detection probabilities for each bird species (terns, cormorants, gulls, and pelicans) and colony, in each study year (Evans et al. 2012, Hostetter et al. 2015, Payton et al. 2019). To address this, PIT tags with known tag codes were intentionally sown on each bird colony (hereafter referred to as "control tags") prior to, during (where possible), and immediately following the nesting season to quantity PIT tag detection probability during the nesting season. Control tags sown on bird colonies were the same size and type as those used to mark most juvenile salmonids from the Columbia River basin (12-mm [length] × 2-mm [width], full duplex). During each discrete sowing period, control tags were haphazardly sown throughout the area occupied by nesting birds during the breeding season. Detections (i.e. recoveries) of control tags during scanning efforts after the breeding season were then used to model the probability of detecting tags that are deposited at different times during the breeding season via logistic regression. Equal numbers of control tags were sown during each discrete time period and sample sizes were selected by considering historic sample sizes. This approach allows direct comparisons of independent detection probabilities, with similar precision between years (see also Payton et al. 2019).

Detection and deposition probabilities (collected referred to as recovery probabilities), many of which have been previously reported in published studies (i.e., Evans et al. 2012, Sebring et al. 2013, Evans et al. 2016a, Hostetter et al. 2015, Hostetter et al. 2018, Evans et al. 2019, Payton et al. 2019), are provided in *Appendix B, Table B.17*.

A.2.2.3 Predation rate model – Following the methodology of Hostetter et al. (2015), predation rates were modeled independently for each salmonid ESU/DPS, bird colony, and year. The probability of recovering a PIT tag from a smolt on each colony was modelled as the product of the three probabilities described above, the probability that (1) the fish was consumed (θ), (2) the PIT tag was deposited on-colony (ϕ), and (3) the PIT tag was detected on-colony after the breeding season (ψ_i; *Figure A.3* above):

$$k_i \sim \text{Binomial}(n_i, \theta_i * \phi * \psi_i)$$

where k_i is the number of smolt PIT tags recovered from the number available (n_i) in week *i*. The detection probabilities (ψ_i) and predation probabilities (θ_i) were each modeled as a function of time. The probability, ψ_i, that a tag, consumed in week *i* and deposited on the colony is detected, is assumed to be a logistic function of week. That is:

$$\text{logit}(\psi_i) = \beta_0 + \beta_1 * i$$

where β_0 and β_1 are both derived from non-informative priors (normal [0, 1000]).

Predation rates nearer together in time are more similar than those further apart in time (Evans et al. 2016a; Payton et al. 2019). To reflect this, variation in weekly predation probabilities, θ_i, was modeled as a random walk process with mean μ_θ and variance σ_θ^2, where:

$$\text{logit}(\theta_i) = \mu_\theta + \sum_{w \le i} \varepsilon_w$$

and $\varepsilon_w \sim normal(0,\ \sigma_\theta^2)\ \forall\, w$. We placed non-informative priors on these two hyperparameters: logit^{-1} $(\mu_\theta) \sim uniform(0,1)$ and $\sigma_\theta^2 \sim uniform(0,20)$. This allows each week (*i*) to have a unique predation probability (θ_i), while still sharing information among weeks improving precision.

Informative Beta priors were used to model deposition probability(ϕ). The shape parameters (α, β) are dependent on the predator species (cormorant, tern, gull) and are assumed to be mutually independent from colony to colony. For terns, we assumed α = 16.20 and β = 6.55, for cormorants, we assumed α = 15.98 and β = 15.29, for gulls we assumed α = 33.71 and β = 183.61 (see Hostetter et al. 2015 for details). No measure of deposition probability was available for white pelicans.

Weekly predation estimates were defined as the estimated number of PIT-tagged smolts consumed divided by the total number available each week. Annual predation rates were derived as the sum of the estimated number of PIT-tagged smolts consumed each week divided by the total number of PIT-tagged smolts available:

$$\sum\nolimits_{i \in breeding\ season} (\theta_i * n_i) \Big/ \sum\nolimits_{i \in breeding\ season} (n_i)$$

Summation of weekly consumption estimates is necessary to accurately reflect weekly variation and autocorrelation of predation rates and thus to create unbiased annual rates with accurate assessments of precision (Hamilton 1994).

The geographic extent of release locations for Lower Columbia River ESUs/DPSs required additional consideration, as a common weekly predation rate, $\theta_{c,i}$, in week i for colony c, cannot be assumed for fish released downstream of Bonneville Dam. Rather, the proximity to East Sand Island respective to the release location of these fish varied widely, depending on the ESU/DPS and year. In recognizing this, we partitioned release locations into three categories, those released in tributaries whose confluence with the Columbia River was (1) upstream of Bonneville Dam (BON), (2) downstream of BON, but upstream of the estuary net detector (ND), or (3) downstream of ND. Recognizing the constraints of the data available, we assumed the week specific predation rate of releases downstream of ND was approximately equal to the predation rate of those fish who successfully migrated to the ND in the same week. Since predation rates produced for other ESUs/DPSs were calculated from availability at BON, we therefore used likelihood methods to construct comparable aggregate predation rates “from BON” for those released above BON and “from release” otherwise.

The number of fish released from a tributary r downstream of ND in week i, $n_{r,i}^{below\ ND}$, were all assumed to be available to predation immediately following release. So, the number of tags from these fish recovered from colony c, denoted $k_{r,i,c}^{below\ ND}$ was modelled

$$k_{r,i,c}^{below\ ND} \sim Binomial(n_{r,i}^{below\ ND}, \theta_{c,i}^{LCR} * \phi * \psi_{c,i}).$$

Fish released upstream of ND but downstream of BON were released in closer geographic proximity to ESI than BON. Therefore, the number of fish released from one of these tributaries r in week j, $n_{r,j}^{between\ BON\ and\ ND}$, contributed in full to the number of LCR fish available to predation in the estuary. However, due to variable tributary lengths and flows, we could not assume all these fish survived to ND, nor could we assume the week in which they were available to predation i to be equal to their week of release j.

Arrivals at ND were assumed to follow a Poisson process with rates estimated for each species released within a tributary and, where applicable, adjusted for week of release. That is, in many tributaries, fish of a species released earlier in the year tend to have greater travel times than those released later in the year. The average travel time to ND, $\lambda_{r,j}$, for fish released in week j from tributary r, was modelled as

$$\lambda_{r,j} = \exp(\beta_{r,0} + \beta_{r,1} * j).$$

where $\beta_{r,1}$ was only included given sufficient statistical credibility of being smaller than zero ($p[\lambda_{r,1} < 0] > 0.9$).

Mark-recapture-recovery methods informed estimates of survival to and recapture at EST. For each species released from tributary r, survival to ND, $S_{r,j}$, was assumed equal for all fish released within a week j and was assumed to follow a logit random walk. Recapture rates at EST in week i, p_i^{ND}, for each species was assumed to be equal across all release groups and follow a logit random walk.

The likelihood contribution for a fish released upstream of ND but downstream of BON in week j from tributary r, recaptured at ND in week i, and later recovered from colony c was proportional to

$$S_{r,j}^{release\ to\ ND} * \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} * p_i^{ND} * \theta_{c,i}^{LCR} * \phi * \psi_{c,i}.$$

The likelihood contribution for a fish released upstream of ND but downstream of BON in week j from tributary r, not recaptured at ND, but later recovered from colony c was proportional to

$$S_{r,j}^{release\ to\ ND} \sum_{j \geq i} \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} * (1 - p_i^{ND}) * \theta_{c,i}^{LCR} * \phi * \psi_{c,i}.$$

The number of fish released from one of these tributaries r upstream of BON in week j considered available (i.e., alive at BON) had to be estimated. Similar to fish released downstream of BON but upstream of ND, fish released upstream of BON could not be assumed to survive to BON nor to EST, nor could we assume the week in which they were available to predation i to be equal to their week of release j. Accordingly, the likelihood contributions of these fish were calculated in a similar fashion, modelling travel time and survival to BON with added parameters for recapture rates in week i at BON, p_i^{BON}, and survival in week i from BON to ND, $S_i^{BON\ to\ ND}$, assumed to be equal across release groups. For example, the likelihood contribution for a fish released upstream of ND but downstream of BON in week j from tributary r, recaptured at BON in week i, recaptured at EST in week i, and later recovered from colony c was proportional to

$$S_{r,j}^{release\ to\ BON} \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} p_i^{BON} S_i^{BON\ to\ ND} p_i^{ND} \theta_{c,i}^{LCR} * \phi * \psi_{c,i}.$$

Annual predation rates were derived as the sum of the estimated number of PIT-tagged smolts consumed each week divided by the estimated total number of PIT-tagged smolts available. Let

$$\begin{aligned} \hat{n}_i^{available} = & \sum_r n_{r,i}^{below\ ND} + \sum_r n_{r,j}^{between\ BON\ and\ ND} * \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} \\ & + \sum_r n_{r,j}^{between\ BON\ and\ ND} * \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} * S_{r,j}^{release\ to\ BON} \end{aligned}$$

and

$$\begin{aligned} \hat{n}_i^{consumed} = & \ \theta_{c,i}^{LCR} \\ & * \left[\sum_r n_{r,i}^{below\ ND} + \sum_r \sum_{j \geq i} n_{r,j}^{between\ BON\ and\ ND} * \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} * S_{r,j}^{release\ to\ ND} \right. \\ & \left. + \sum_r \sum_{j \geq i} n_{r,j}^{between\ BON\ and\ ND} * \frac{\lambda_{r,j}^{j-i} \exp(-\lambda_{r,j})}{(j-i)!} * S_{r,j}^{release\ to\ BON} * S_i^{BON\ to\ ND} \right] \end{aligned}$$

then

$$\hat{\theta}^{annual} = \sum_i \hat{n}_i^{consumed} \Big/ \sum_i \hat{n}_i^{available}.$$

A.2.2.4 Per capita predation rates – To monitor the effectiveness of avian predation management plans to reduce predation rates by reducing colony size, we compared estimates of ESU/DPS-specific predation rates and estimates of peak colony sizes (number of breeding pairs) across colonies and study years, to the extent possible (e.g., where sufficient data existed for the analysis). Per capita (per breeding pair) predation rates were calculated by dividing the annual ESU/DPS-specific predation rate by the peak measure of colony size each year:

$$Annual\ Per\ Capita\ Predation\ Rate_y = \sum_w (\theta_{wy} * n_{wy}) / \sum_w (n_{wy}) \Big/ C_y$$

where C_y is the colony size in year, *y.* The relationship between colony size and predation rates across years was further evaluated by generating a posterior distribution of least squares regression lines, calculated from random samples of per capita predation rates taken from the posterior distributions associated with the above defined estimates. The strength and direction of the relationship can be inferred from the median value of the resulting posterior distribution associated with the slope of the resulting posterior distribution with the 95% credible interval (CRI) defined as the 2.5th and 97.5th quantiles. Statistical credibility was identified by the credibility interval of the slope parameter not overlapping with zero.

A.2.2.5 Statistical derivation – Models were analyzed using the software STAN (Stan Development Team 2020), accessed through R version 3.6.2 (R Development Core Team 2014), and using the rstan package (version 2.19.3; Stan Development Team 2020). To simulate random draws from the joint posterior distribution, we ran four Hamiltonian Monte Carlo (HMC) Markov Chain processes. Each chain contained 4,000 warm-up iterations followed by 4,000 posterior iterations thinned by a factor of 4. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013); only chains with zero reported divergent transitions were accepted. Posterior predictive checks compared simulated and observed annual aggregate raw recapture and recovery numbers to ensure model estimates reflected the observed data. Reported estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% Credible Interval [CRI]) calculated using the HDInterval package (version 0.2.0; Meredith and Kruschke 2016). Annual predation rates were calculated for salmonid ESUs/DPSs where ≥ 500 PIT-tagged individuals were available to birds to avoid imprecise results that may occur from small sample sizes of available PIT-tagged smolts (Evans et al. 2012).

A.2.2.6 Modelling assumptions – Results of predation rate analyses were based on the following assumptions (see also Hostetter at al. 2015 and Evans et al. 2016a for further description):

A1. Information from PTAGIS on releases and in-river interrogations (recapture histories) of PIT-tagged fish used in the study were complete and accurate.
A2. PIT-tagged fish detected in-river were available to birds nesting downstream (i.e., fish were actively migrating and thus available to avian predators downstream).
A3. Predation, deposition, and detection probabilities for PIT-tagged fish were independent.
A4. Detection probabilities for control PIT tags sown on-colony were equal to those of PIT tags deposited by birds on-colony.
A5. Deposition probabilities for PIT tags measured in previous years were equal to those of PIT tags consumed by birds during all study years.
A6. PIT tags from consumed fish were egested by avian predators within a relatively short time period (one week) from when the PIT-tagged fish was last interrogated alive.
A7. PIT-tagged fish are representative of non-tagged fish belonging to the same ESU/DPS.

To help meet the first assumption (A1), irregular entries in PTAGIS were either verified by the respective coordinator of the PIT-tagging effort or were censored from the analysis. Detections of PIT-tagged salmonids at recapture sites upstream of bird colonies is deemed the most appropriate measure of fish availability given the downstream movement of smolts, the ability to standardize data across sites, and the ability to define unique groups of salmonids by a known location and passage date (Assumption A2). Assumption A2 assumes all PIT-tagged fish last interrogated passing a dam or released via barge were alive and available to predators downstream. If large numbers of fish halt their out-migration or died immediately following release/recapture and prior to reaching the foraging range of the colony, predation rates would be biased low. The fate of each PIT tag implanted in a smolt is assumed to be independent (A3). Lack of independence among PIT-tagged fish could potentially bias predation probabilities and overinflate measures of precision (i.e., credibility intervals). Detection probability estimates (A4) were generally high (ca. 70%, depending on the colony and year) suggesting that any possible violations of assumption A4 would have little effect on estimates of predation rates at most colonies and years. Deposition rate data collected in year's past (when multiple estimates of deposition rates were measured for each species of avian predator over the course several time periods both within and across years) showed no evidence of inter- or intra-annual trends in deposition probabilities (Assumption A5). Due to high species-specific (tern, cormorant, gull) differences in deposition probabilities, recovery probabilities (detection x deposition = recovery) were variable and highly influential in predation rate estimates (see *Appendix B, Table B.17*). Assumption A6 relates to the use of the last date of live fish recapture as a proxy for the date a PIT tag was deposited on a bird colony. This assumption needs to be only roughly true because detection efficiency did not change dramatically on a weekly basis. Assumption A7 relates to inference regarding the susceptibility of a PIT-tagged fish to consumption as it relates to all fish (tagged, untagged) from the same cohort. There are few empirical data to support or refute assumption A7, except that the run-timing and abundance of PIT-tagged fish is often in agreement with the run-timing and abundance of untagged fish passing dams in the Columbia River basin. For some salmonid populations, fish were randomly selected for tagging and were tagged in concert with the run passing a given dam to better ensure that a representative sample of fish were available for analyses (see Evans et al. 2014 and Evans et al. 2019). For other groups of fish, however, individuals may have been culled for tagging based on their rear-type, condition, size or only a small number or proportion of available fish were PIT-tagged of a given ESU/DPS due to a lack of funding and/or due to shifting regional research, monitoring, and evaluation priorities each year.

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APPENDIX B: Summary Data Tables

B.1 Colony Size

Table B.1. Name and location of Caspian tern breeding colonies within the Columbia River Basin.

State	Geographic Location	General Location	Colony Name	Species	Latitude	Longitude
Oregon	Columbia River Basin	Lower Columbia River Estuary	East Sand Island	Caspian tern	46.263	-123.968
Oregon	Columbia River Basin	Upper Columbia River Estuary	Rice Island	Caspian tern	46.250	-123.716
Oregon	Columbia River Basin	Mid-Columbia River	Three Mile Canyon Island	Caspian tern	45.816	-119.966
Washington	Columbia River Basin	Mid-Columbia River	Badger Island	Caspian tern	46.111	-118.938
Washington	Columbia River Basin	Mid-Columbia River	Crescent Island	Caspian tern	46.093	-118.931
Washington	Columbia River Basin	Mid-Columbia River	Blalock Island Complex- Long Island	Caspian tern	45.896	-119.646
Washington	Columbia River Basin	Mid-Columbia River	Blalock Island Complex - Middle Island	Caspian tern	45.895	-119.647
Washington	Columbia River Basin	Mid-Columbia River	Blalock Island Complex - Southern Island	Caspian tern	45.895	-119.651
Washington	Columbia River Basin	Mid-Columbia River	Blalock Island Complex - Rock Island	Caspian tern	45.909	-119.629
Washington	Columbia River Basin	Mid-Columbia River	Blalock Island Complex - Anvil Island	Caspian tern	45.898	-119.647
Washington	Columbia River Basin	Mid-Columbia River	Miller Rocks Island	Caspian tern	45.649	-120.908
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Goose Island	Caspian tern	46.986	-119.311
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Northeast Potholes	Caspian tern	47.034	-119.327
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Solstice Island	Caspian tern	47.026	-119.367
Washington	Columbia River Basin	Columbia Plateau	Lenore Lake - North Rock	Caspian tern	47.483	-119.521
Washington	Columbia River Basin	Columbia Plateau	Lenore Lake - Shoal Island	Caspian tern	47.480	-119.524
Washington	Columbia River Basin	Columbia Plateau	Banks Lake - Goose Island	Caspian tern	47.649	-119.291
Washington	Columbia River Basin	Columbia Plateau	Banks Lake - Twining Island	Caspian tern	47.625	-119.303
Washington	Columbia River Basin	Columbia Plateau	Sprague Lake - Harper Island	Caspian tern	47.248	-118.085

Table B.2. Name and location of Caspian tern breeding colonies outside of the Columbia River Basin and USACE-constructed tern islands.

State	Geographic Location	General Location	Colony Name	Species	Latitude	Longitude
Oregon	Interior Oregon	Willamette Valley	Fern Ridge Reservoir	Caspian tern	44.071	-123.281
Oregon	Interior Oregon	Warner Valley	Crump Lake Island	Caspian tern	42.284	-119.844
Oregon	Interior Oregon	Summer Lake WMA	Dutchy Lake Island	Caspian tern	42.967	-120.769
Oregon	Interior Oregon	Summer Lake WMA	East Link Island	Caspian tern	42.956	-120.715
Oregon	Interior Oregon	Summer Lake WMA	Gold Dike Island	Caspian tern	42.912	-120.779
Oregon	Interior Oregon	Malheur NWR	Malheur Lake New Tern Island	Caspian tern	43.293	-118.817
Oregon	Interior Oregon	Malheur NWR	Singhus Ranch Island	Caspian tern	43.379	-118.826
California	Northeastern California	Lower Klamath NWR	Sheepy Lake Island	Caspian tern	41.968	-121.792
California	Northeastern California	Lower Klamath NWR	Orems Unit Island	Caspian tern	41.962	-121.634
California	Northeastern California	Tule Lake NWR	Tule Lake Sump 1B Island	Caspian tern	41.843	-121.478
California	San Francisco Bay	Don Edwards San Francisco Bay NWR	Pond A16	Caspian tern	37.447	-121.971
California	San Francisco Bay	Don Edwards San Francisco Bay NWR	Pond SF2	Caspian tern	37.491	-122.133

Table B.3. Name and location of double-crested cormorant breeding colonies within the Columbia River Basin.

State	Geographic Location	General Location	Colony Name	Nesting Species	Latitude	Longitude
Washington	Columbia River Basin	Lower Columbia River Estuary	Jetty A Channel Marker	Double-crested cormorant	46.266	-124.038
Oregon	Columbia River Basin	Lower Columbia River Estuary	East Sand Island	Double-crested cormorant	46.262	-123.991
Oregon	Columbia River Basin	Lower Columbia River Estuary	Sand Island Channel Marker #1	Double-crested cormorant	46.266	-123.993
Oregon	Columbia River Basin	Lower Columbia River Estuary	Sand Island Channel Marker #2	Double-crested cormorant	46.267	-123.981
Oregon	Columbia River Basin	Lower Columbia River Estuary	Desdemona Sands	Double-crested cormorant	46.210	-123.880
Oregon	Columbia River Basin	Upper Columbia River Estuary	Astoria-Megler Bridge	Double-crested cormorant	46.216	-123.862
Oregon	Columbia River Basin	Upper Columbia River Estuary	Rice Island	Double-crested cormorant	46.250	-123.716
Oregon	Columbia River Basin	Upper Columbia River Estuary	Miller Sands Spit	Double-crested cormorant	46.247	-123.675
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #1	Double-crested cormorant	46.234	-123.714
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #2	Double-crested cormorant	46.233	-123.714
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #3	Double-crested cormorant	46.235	-123.706
Washington	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #4	Double-crested cormorant	46.256	-123.677
Washington	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #5	Double-crested cormorant	46.257	-123.669
Washington	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #6	Double-crested cormorant	46.262	-123.666
Washington	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #7	Double-crested cormorant	46.261	-123.642
Washington	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #8	Double-crested cormorant	46.262	-123.637
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #9	Double-crested cormorant	46.253	-123.544
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #10	Double-crested cormorant	46.252	-123.534
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #11	Double-crested cormorant	46.261	-123.516
Oregon	Columbia River Basin	Upper Columbia River Estuary	Estuary Channel Marker #12	Double-crested cormorant	46.262	-123.503
Washington	Columbia River Basin	Upper Columbia River Estuary	Lewis and Clark Bridge	Double-crested cormorant	46.105	-122.962
Washington	Columbia River Basin	Lower Columbia River	Troutdale Transmission Towers	Double-crested cormorant	45.568	-122.411
Washington	Columbia River Basin	Mid-Columbia River	Okanogan River Delta	Double-crested cormorant	48.099	-119.716
Washington	Columbia River Basin	Mid-Columbia River	Miller Rocks	Double-crested cormorant	45.649	-120.908
Oregon	Columbia River Basin	Mid-Columbia River	Hanford Reach - Locke Island Complex	Double-crested cormorant	46.704	-119.467
Washington	Columbia River Basin	Mid-Columbia River	Foundation Island	Double-crested cormorant	46.167	-118.999
Washington	Columbia River Basin	Mid-Columbia River	Crescent Island	Double-crested cormorant	46.093	-118.931
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Northeast Arm 1	Double-crested cormorant	47.078	-119.348
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Northeast Arm 2	Double-crested cormorant	47.076	-119.350
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Middle North Arm	Double-crested cormorant	47.066	-119.412
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Northwest Arm	Double-crested cormorant	47.041	-119.439
Washington	Columbia River Basin	Columbia Plateau	Sprague Lake - Harper Island	Double-crested cormorant	47.248	-118.085
Washington	Columbia River Basin	Columbia Plateau	Lenore Lake - North Rock	Double-crested cormorant	47.483	-119.521
Washington	Columbia River Basin	Columbia Plateau	Lenore Lake - Shoal Island	Double-crested cormorant	47.480	-119.524
Washington	Columbia River Basin	Columbia Plateau	Crab Creek Colony Complex	Double-crested cormorant	47.407	-119.096
Washington	Columbia River Basin	Lower Snake River	Lyons Ferry RR Bridge	Double-crested cormorant	46.593	-118.222

Table B.4. Name and location of Brandt's cormorant breeding colonies within the Columbia River Basin.

State	Geographic Location	General Location	Colony Name	Nesting Species	Latitude	Longitude
Oregon	Columbia River Basin	Lower Columbia River Estuary	East Sand Island	Brandt's cormorant	46.262	-123.991
Oregon	Columbia River Basin	Lower Columbia River Estuary	East Sand Island - Pile Dike	Brandt's cormorant	46.260	-123.995
Oregon	Columbia River Basin	Upper Columbia River Estuary	Astoria-Megler Bridge	Brandt's cormorant	46.216	-123.862

Table B.5. Name and location of gull (Larus sp.) breeding colonies within the Columbia River Basin.

State	Geographic Location	General Location	Colony Name	Nesting Species	Latitude	Longitude
Oregon	Columbia River Basin	Lower Columbia River Estuary	East Sand Island	Western/Glaucous-winged & Ring-billed Gulls	46.263	-123.968
Oregon	Columbia River Basin	Upper Columbia River Estuary	Rice Island	Western/Glaucous-winged & Ring-billed Gulls	46.250	-123.716
Oregon	Columbia River Basin	Upper Columbia River Estuary	Miller Sands Spit	Western/Glaucous-winged & Ring-billed Gulls	46.247	-123.675
Oregon	Columbia River Basin	Upper Columbia River Estuary	Pillar Rock Island	Western/Glaucous-winged & Ring-billed Gulls	46.251	-123.584
Oregon	Columbia River Basin	Mid-Columbia River	Three Mile Canyon Island	California and/or Ring-billed Gulls	45.816	-119.966
Washington	Columbia River Basin	Mid-Columbia River	Little Memaloose Island	California and/or Ring-billed Gulls	45.653	-121.081
Washington	Columbia River Basin	Mid-Columbia River	Miller Rocks	California and/or Ring-billed Gulls	45.649	-120.908
Washington	Columbia River Basin	Mid-Columbia River	Island 20	California and/or Ring-billed Gulls	46.313	-119.253
Washington	Columbia River Basin	Mid-Columbia River	Crescent Island	California and/or Ring-billed Gulls	46.093	-118.931
Washington	Columbia River Basin	Mid-Columbia River	Badger Island	California and/or Ring-billed Gulls	46.111	-118.938
Washington	Columbia River Basin	Mid-Columbia River	Blalock Islands - Big Blalock Island	California and/or Ring-billed Gulls	45.914	-119.621
Washington	Columbia River Basin	Mid-Columbia River	Blalock Islands - Rock Island	California and/or Ring-billed Gulls	45.909	-119.629
Washington	Columbia River Basin	Mid-Columbia River	Blalock Islands - Anvil Island	California and/or Ring-billed Gulls	45.898	-119.647
Washington	Columbia River Basin	Mid-Columbia River	Blalock Islands - Straight Six Island	California and/or Ring-billed Gulls	45.898	-119.661
Washington	Columbia River Basin	Columbia Plateau	Banks Lake - Goose Island	California and/or Ring-billed Gulls	47.649	-119.291
Washington	Columbia River Basin	Columbia Plateau	Banks Lake - Twinning Island	California and/or Ring-billed Gulls	47.625	-119.303
Washington	Columbia River Basin	Columbia Plateau	Potholes Reservoir - Goose Island	California and/or Ring-billed Gulls	46.986	-119.311
Washington	Columbia River Basin	Columbia Plateau	Sprague Lake - Harper Island	California and/or Ring-billed Gulls	47.248	-118.085
Washington	Columbia River Basin	Columbia Plateau	Lenore Lake - North Rock	California and/or Ring-billed Gulls	47.483	-119.521
Washington	Columbia River Basin	Columbia Plateau	Lenore Lake - Shoal Island	California and/or Ring-billed Gulls	47.480	-119.524

Table B.6. Name and location of American white pelican breeding colonies within the Columbia River Basin.

State	Geographic Location	General Location	Colony Name	Nesting Species	Latitude	Longitude
Oregon	Columbia River Basin	Upper Columbia River Estuary	Miller Sands Spit	American White Pelicans	46.244	-123.675
Oregon	Columbia River Basin	Upper Columbia River Estuary	Rice Island	American White Pelicans	46.250	-123.716
Washington	Columbia River Basin	Mid-Columbia River	Badger Island	American White Pelicans	46.111	-118.938
Washington	Columbia River Basin	Mid-Columbia River	Crescent Island	American White Pelicans	46.093	-118.931

Table B.7. Estimated peak number of breeding pairs of Caspian terns at colony sites in the Columbia River estuary from 1997 to 2019.

	Columbia River Estuary - Caspian Tern Breeding Pairs	
	Upper Estuary	Lower Estuary
Year	Rice Island	East Sand Island
1997	7,134	0
1998	8,766	0
1999	8,328[a]	547
2000	588[a]	8,513
2001	0[a]	8,982
2002	0[a]	9,933
2003	0[a]	8,325
2004	0[a]	9,502
2005	0[a]	8,822
2006	0[a]	8,929
2007	0[a]	9,623
2008	0[a]	10,668[a]
2009	0[a]	9,854[a]
2010	0[a]	8,283[a]
2011	3[a]	6,969[a]
2012	0[a]	6,416[a]
2013	0[a]	7,387[a]
2014	0[a]	6,269[a]
2015	0[a]	6,240[a]
2016	0[a]	5,915[a]
2017	0[a]	3,500[a]
2018	0[a]	4,959[a]
2019	0[a]	3,861[a]

[a] *= Active management used to dissuade or limit the number of Caspian terns breeding at the site*

Table B.8. Estimated peak number of breeding pairs of Caspian terns at colony sites in the interior of Oregon and in northeastern California from 2008 to 2018. Unless noted, the US Army Corps of Engineers (USACE) constructed the site as part of the Caspian Tern Management Plan for the Columbia River Estuary.

	Interior Oregon and Northeastern California - Caspian Tern Breeding Pairs									
	Willamette Valley	Warner Valley	Summer Lake WMA			Lower Klamath NWR		Tule Lake NWR	Malheur NWR	Goose Lake Valley
Year	Fern Ridge	Crump Lake	Dutchy Lake	East Link	Gold Dike	Sheepy Lake	Orems Unit	Tule Lake Sump 1B	Malheur Lake /Singhus Ranch[a]	Goose Lake[a]
2008	0	430	NA	NA	NA	NA	NA	NA	NA	246
2009	0	690	8	7	NA	NA	NA	NA	NA	0
2010	0	71	0	29	NA	258	NA	NA	NA	0
2011	0	35	0	2	0	188	2	34	150	0
2012	0	115	0	10	4	212	NA	207	232	0
2013	0	223	NA	21	0	316	NA	79	530	0
2014	0	1	NA	22	0	520	NA	109	134	0
2015	0	3	NA	27	0	336	NA	545	148	0
2016	--	27	NA	22	0	95	NA	100	14	0
2017	--	26	NA	27	0	245	NA	274	21	0
2018	--	86	NA	70	0	238	NA	627	420	0

[a] = Naturally occurring island and not constructed by USACE; NA = Site not available as nesting habitat; -- = Data not available

Table B.9. Estimated peak number of breeding pairs of Caspian terns at colony sites in the Columbia Plateau region from 1997 to 2019.

	Columbia Plateau Region - Caspian Tern Breeding Pairs								
	Lower Columbia River					Potholes Reservoir	Banks Lake	Sprague Lake	Lenore Lake
Year	Crescent Is.	Blalock Is. Complex	Badger Is. /Foundation Is.	Three Mile Canyon Is.	Miller Rocks	Solstice Is./Goose Is. /Northeast Is.	Twinning Is /Goose Is.	Harper Island	North Rock/ Shoal Is.
1997	614	--	0	354	0	259	--	--	0
1998	357	--	0	210	0	--	--	--	0
1999	552	--	0	238	0	--	--	50	0
2000	548	--	0	260	0	129	10	20	0
2001	657	--	0	2	17	248	23	20	0
2002	578	--	0	0	0	250	--	--	0
2003	509	--	0	0	0	205	21	--	0
2004	530	0	0	0	0	191	41	--	0
2005	476	6	0	0	0	329	26	10	0
2006	448	110	0	0	0	273	20	7	0
2007	355	26	0	0	0	282	48	0	0
2008	388	104	0	0	0	293	27	11	0
2009	349	79	0	0	0	487	61	4	0
2010	375	136	0	0	0	416	34	4	0
2011	419	20	36	0	0	422	19	4	0
2012	422	6	60	0	0	463	22	30	0
2013	393	26	0	0	0	340	13	1	0
2014	474	45	0	0	0	159[a]	67	8	2
2015	0[a]	677	0	0	0	2[a]	64	10	16
2016	0[a]	483	0	0	0	144[a]	6	3	39
2017	0[a]	449	41	0	0	0[a]	0	92	123
2018	0[a]	313	8	0	0	0[a]	0	79	91
2019	0[a]	379	0	0	0	0[a]	0	18	48

[a] = Active management used to completely dissuade or limit the number of Caspian terns breeding at the site; -- = No data available

Table B.10. Estimated peak number of breeding pairs of double-crested cormorants at colony sites in the Columbia River estuary from 1997 to 2019.

	Columbia River Estuary - Double-crested Cormorant Breeding Pairs								
	Lower Estuary			Upper Estuary					
Year	Channel Markers[a]	East Sand Island	Desdemona Sands	Astoria-Megler Bridge	Rice Island	Miller Sands Spit	Channel Markers[b]	Lewis & Clark Bridge	Troutdale Towers
1997	--	5,023	130	0	1,141	0	64	--[f]	--
1998	--	6,285	120	0	795	0	70	--[f]	--
1999	--	6,561	--	0	0	0	--	--[f]	--
2000	--	7,162	61	0	0	0	150	--[f]	--
2001	--	8,120	0	0	150	12	75	--[f]	--
2002	--	10,230	0	0	53	0	129	--[f]	--
2003	--	10,646	0	0	211	0	183	--[f]	--
2004	--	12,480	0	6	0	0	194	--[f]	--
2005	--	12,287	0	14	0	0	208	--[f]	--
2006	--	13,738	0	7	35	41	162	--[f]	--
2007	--	13,771	0	11	0	90	160	--[f]	--
2008	--	10,950	0	20	0	129	216	--[f]	--
2009	--	12,087	0	24	0	0	235	--[f]	--
2010	--	13,596	0	63	0	0	254	--[f]	--
2011	--	13,045[c]	0	60	0	0	248	--[f]	--
2012	8	12,301[c]	0	139	0	0	245	--[f]	--
2013	7	14,916[c]	0	231	0	0	330	0	B
2014	5	13,626	0	333	0	0	318	11	80
2015	5	12,150[d]	0	425	0	0	319	122	B
2016	18	9,772[d]	0	549	0	0	300	102	B
2017	7	544[d]	0	834	0	0	204	147	B
2018	41	3,672[e]	0	1,737	0	0	293	201	--
2019	73	350[e]	0	3,542	0	0	219	139	--

[a] *= channel markers located between the mouth of the Columbia River (Rkm 0) and the Astoria-Megler Bridge (Rkm 22)*

[b] *= channel markers located from the Astoria-Megler Bridge (Rkm 22) to Fitzpatrick Island (Rkm 51)*

[c] *= pilot study to test efficacy of non-lethal techniques for deterring nesting on part of the colony area*

[d] *= Phase I of the management plan, which included culling of adults and oiling cormorant eggs in nests*

[e] *= Phase II of the management plan, which included non-lethal management to limit the area of cormorant nesting habitat*

[f] *= very few or no nests present*

Table B.11. Estimated peak number of breeding pairs for double-crested cormorants at colony sites in the Columbia Plateau region from 2003 to 2018.

	Columbia Plateau Region - Double-crested Cormorant Breeding Pairs									
	Lower Columbia River			Mid-Columbia River		Lower Snake River	Potholes Reservoir	Sprague Lake	Lenore Lake	Crab Creek
Year	Miller Rocks	Foundation Island	Crescent Island	Hanford Reach - Locke Island Complex	Okanogan River Delta	Lyons Ferry RR Bridge	Colony Complex	Harper Island	Shoal Island/ North Rock	Colony Complex
2003	--	200-300	0	--	--	--	--	--	0	--
2004	--	~300	0	--	20-30	--	300-500	--	0	--
2005	0	315	0	0	38	0	~800	0	0	--
2006	5	359	0	0	32	2	1160	0	0	--
2007	0	334	0	8	10	0	1015	0	0	
2008	0	360	0	0	33	0	~1000	38	0	--
2009	0	310	0	0	36	0	809	42	0	--
2010	0	308	5	0	26	0	~830	86	0	--
2011	2	318	15	0	32	0	~900	107	0	--
2012	0	390	0	0	40	0	~1000	146	0	--
2013	0	377	0	0	42	0	~800	174	0	--
2014	0	B	0	6	49	0	B	266	5	--
2015	0	B	0[a]	25	50	0	B	336	39	--
2016	0	B	0[a]	35	46	0	146	420	66	--
2017	0	B	0[a]	B	B	0	346	302	B	--
2018	0	B	0[a]	68	B	0	205	B	111	387

[a] = Active management used to completely dissuade or limit the number of Caspian terns breeding at the site; B = Breeding activity at the site but count not conducted; -- = Data not available

Table B.12. Estimated peak number of breeding pairs of Brandt's cormorants at colony sites in the Columbia River estuary from 1997 to 2018.

	Columbia River Estuary - Brandt's Cormorant Breeding Pairs		
	Upper Estuary	Lower Estuary	
Year	Astoria-Megler Bridge	East Sand Island – Pile Dike	East Sand Island
1997	0	B	0
1998	0	B	0
1999	0	B	0
2000	0	31	0
2001	0	30-50	0
2002	0	86	0
2003	0	97	0
2004	0	58	0
2005	0	62	0
2006	0	NA	44
2007	0	NA	288
2008	0	NA	508
2009	0	NA	684
2010	0	NA	985
2011	0	NA	1,491[a]
2012	0	NA	1,684[a]
2013	0	NA	1,523[a]
2014	0	NA	1,630
2015	0	NA	2,071[b]
2016	--	NA	1,515[b]
2017	--	NA	1,893[b]
2018	14	NA	2,120[b]

[a] = Active management used to limit the location of double-crested cormorants nesting at the site; [b] = Active management to limit the number double-crested cormorants nesting at the site; B = Breeding activity at the site but count not conducted; NA = Site not available as nesting habitat; -- = Data not available

Table B.13. Index counts of gulls (western/glaucous-winged gulls:ring-billed gulls) nesting at colony sites in the Columbia River estuary from 1997 to 2018. Counts are of the number of individual gulls present on the colony at the peak of the nesting period.

	Columbia River Estuary – Index Counts of Nesting Gulls			
	Upper Estuary			Lower Estuary
Year	Rice Island	Miller Sands Spit	Pillar Rock Sands	East Sand Island
1997	1,583 (1,583:0)	~1460 (1,268:< 200)	0	7,106 (7,106:0)
1998	1,576 (1,576:0)	~930 (742:< 200)	0	5,496 (5,496:0)
1999	--	--	--	--
2000	--	--	--	--
2001	--	--	--	--
2002	--	--	--	--
2003	--	--	--	--
2004	B (B:NB)	B (B:NB)	0	B (B:B)
2005	B (B:NB)	B (B:NB)	0	B (B:B)
2006	1,727 (1,727:0)	704 (704:0)	0	9,976 (8,587:1,389)
2007	B (B:NB)	B (B:NB)	0	B (B:B)
2008	B (B:NB)	B (B:NB)	0	B (B:B)
2009	2,049 (1,741:307)	160 (160:0)	0	8,409 (6,172:2,237)
2010	B (B:NB)	B (B:NB)	0	8,383 (6,966:1,417)
2011	B (B:NB)	B (B:NB)	0	8,720 (6,776:1,944)
2012	B (B:NB)	B (B:NB)	0	4,841 (3,369:1,472)
2013	B (B:NB)	0	0	7,256 (4,580:2,676)
2014	B (B:NB)	0	0	5,880 (3,595:2,285)
2015	B (B:NB)	0	0	6,047 (4,123:1,924)
2016	B (B:NB)	0	0	B (B:B)
2017	B (B:NB)	0	0	B (B:B)
2018	B (B:NB)	0	0	B (B:B)

B = Breeding activity at the site but count not conducted; NB = No breeding activity confirmed at the site and count not conducted; Total counts of individuals presented in the table as an index of breeding activity; -- = Data not available

Table B.14. Index counts of gulls (California gulls:ring-billed gulls) nesting at colony sites in the Columbia Plateau region from 1997 to 2018. Counts are of the number of individual gulls counted at the peak of the nesting period.

	Lower Columbia River						Mid-Columbia River		Potholes Reservoir	Banks Lake	Sprague Lake	Lenore Lake
Year	Little Memaloose Island	Three Mile Canyon Island	Miller Rocks	Blalock Islands Complex	Crescent Island	Badger Island	Island 20	Island 18	Goose Island	Twining/ Goose Islands	Harper Island	Shoal Island/ North Rock
1997	939 (939:0)	13,305 (*nd*)	3,783 (*nd*)	0	5,769 (*5,769:0*)	0	18,820 (*nd*)	14,495 (*nd*)	--	--	--	--
1998	357 (357:0)	11,102 (*nd*)	2,179 (*nd*)	0	4,597 (*4,597:0*)	0	22,348 (*nd*)	12,669 (*nd*)	--	--	--	--
1999	--	--	--	--	--	--	--	--	--	--	--	--
2000	--	--	--	--	--	--	--	--	--	--	--	--
2001	--	--	--	--	--	--	--	--	--	--	--	--
2002	--	--	--	--	--	--	--	--	--	--	--	--
2003	--	--	--	--	--	--	--	--	--	--	--	--
2004	0	B (*nd*)	B (*nd*)	0	B (*nd*)	0	B (*nd*)	B (*nd*)	B (*nd*)	--	--	--
2005	0	B (B:B)	B (B:B)	B (NB:B)	B (B:B)	0	B (B:NB)	B (NB:B)	B (B:B)	B (B:B)	B (B:B)	--
2006	0	B (B:B)	B (B:B)	B (NB:B)	B (B:B)	0	B (B:NB)	B (B:B)	B (B:B)	B (B:B)	B (B:B)	--
2007	0	B (B:B)	3,509 (*nd*)	B (NB:B)	5,601 (*nd*)	0	B (B:NB)	B (B:B)	B (B:B)	B (B:B)	B (B:B)	--
2008	0	B (*nd*)	4,443 (*nd*)	B (NB:B)	8,567 (*nd*)	0	20,999 (*nd*)	B (*nd*)	B (B:B)	B (B:B)	B (B:B)	--
2009	0	6,161 (5,868:293)	6,016 (5,272:744)	1,631 (0:1,631)	8,575 (8,575:0)	0	19,341 (9,946:9,395)	0	13,022 (2,481:10,541)	6,602 (3,737:2,865)	6,302 (1,798:4,504)	B (*nd*)
2010	0	B (*nd*)	5,533 (5,301:232)	B (*nd*)	B (*nd*)	0	B (*nd*)	0	B (*nd*)	B (*nd*)	B (*nd*)	0
2011	0	B (*nd*)	5,742 (*nd*)	B (*nd*)	8,018 (8,018:0)	0	B (*nd*)	0	11,392 (2,459:8,933)	B (*nd*)	B (*nd*)	--
2012	0	0	4,509 (*nd*)	8,989 (B:B)	7,187 (7,091:96)	0	B (*nd*)	0	12,005 (3,684:8,321)	B (*nd*)	B (*nd*)	--
2013	0	0	4,807 (4,757:50)	6,894 (4078:2816)	5,707 (5,555:152)	0	14,039 (8,977:5,062)	0	12,797 (3,007:9,790)	5,824 (*nd*)	5,625 (*nd*)	804 (*nd*)
2014	0	0	4,132 (*nd*)	6,020 (3,236:2,784)	6,404 (5,652:752)	0	14,475 (9,539:4,936)	0	14,334[a] (4,630:9,620)	B (*nd*)	B (*nd*)	B (*nd*)
2015	0	0	4,433 (4,368:65)	7,377 (4,756:2,621)	0[a]	3,740 (*nd*)	16,557 (12,009:4,548)	0	14,808[a] (5,180:9,628)	B (*nd*)	B (*nd*)	743 (0:743)
2016	0	0	3,734 (*nd*)	6,741 (B:B)	0[a]	4,126 (*nd*)	14,316 (*nd*)	0	13,273[a] (4,117:9,156)	B (*nd*)	B (*nd*)	B (*nd*)
2017	0	0	3,435 (*nd*)	4,163 (*nd*)	0[a]	4,505 (*nd*)	11,176 (*nd*)	0	11,225[a] (*nd*)	B (*nd*)	B (*nd*)	B (*nd*)
2018	0	0	4,284 (*nd*)	3,408 (*nd*)	0[a]	5,908 (*nd*)	13,069 (*nd*)	0	11,994[a] (*nd*)	B (*nd*)	B (*nd*)	B (*nd*)

[a] = Active management to completely dissuade or limit the number of Caspian terns breeding at the site; B = Breeding activity observed but count not conducted; NB = No breeding activity confirmed at the site and count not conducted; Total counts of individuals presented in the table as an index of breeding activity; -- = Data not available; nd = gull species not differentiated in counts

Table B.15. Estimated peak number of breeding pairs for American white pelicans at colony sites in the Columbia River estuary from 2010 to 2018.

	Columbia River Estuary - American White Pelican Breeding Pairs	
	Upper Estuary	
Year	Rice Island	Miller Sands Spit
2010	0	42
2011	0	194
2012	0	122
2013	0	104
2014	0	376
2015	0	B
2016	0	351
2017	102	76[a]
2018	398	246[a]

[a] = Colony failure occurred at Miller Sands Spit with subsequent re-nesting on Rice Island; B = Breeding activity at the site but count not conducted

Table B.16. Index counts of American white pelicans nesting at colony sites in the Columbia Plateau Region from 2004 to 2018. Counts are of the number of individual pelicans present on the colony at the peak of the nesting period.

	Columbia Plateau Region – Index Counts of Nesting American White Pelicans	
	Lower Columbia River	
Year	Badger Island	Crescent Island
2004	B	0
2005	1057	0
2006	1310	0
2007	913	0
2008	1349	0
2009	1754	0
2010	2048	50
2011	2228	0
2012	2075	0
2013	2077	0
2014	2447	0
2015	3267	0[a]
2016	3118	0[a]
2017	2513	0[a]
2018	3330	0[a]

[a] = Active management used to completely dissuade or limit the number of Caspian terns breeding at the site; B = Breeding activity at the site but count not conducted; Total counts of individuals presented in this table are an index of colony size

B.2 Predation Rates

Table B.17. Average annual recovery probabilities (95% credible intervals) of smolt PIT-tags on colonial waterbird breeding sites in the Columbia River basin during 1998-2006 (upper panel) and 2007-2018 (lower panel). Recovery probabilities were used to estimate predation rates. Recoveries are from Caspian terns (CATE), California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), Brandt's cormorants (BRCO), or American white pelicans (AWPE) nesting at colonies on Banks Lake islands (BLI), Potholes Reservoir islands (PTI), Lenore Lake islands (LLI), Sprague Lake island (Harper Island; SPL), Island 20 (I20), Hanford Reach island (HRI), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks (MRI), Rice Island (RCI), Miller Sands Spit (MSI), and East Sand Island (ESI). NA denotes years where an active colony existed, but smolt PIT tags were not recovered. NC denotes years when no colony was confirmed at that site in that year. Detection and deposition probabilities are those previously published by Evans et al. (2012, 2016a, 2019), Sebring et al. (2013), Hostetter et al. (2015), and Payton et al. (2019). Recovery probability is shown as the deposition probability multiplied by the annual weighted average of detection probability for CATE, DCCO, BRCO, and LAXX colonies. No estimate of deposition probability was available for AWPE, so estimates include detection probabilities only.

Colony	1998	1999	2000	2001	2002	2003	2004	2005	2006
RCI CATE	0.67 (0.52-0.71)[1]	0.67 (0.52-0.71)[1]	0.67 (0.52-0.71)[1]	0.67 (0.52-0.71)[1]	NC	NC	NC	NC	NC
RCI DCCO	0.48 (0.37-0.51)[1]	NA	NA	0.34 (0.3-0.39)[1]	NA	NC	NC	NC	0.34 (0.3-0.39)[1]
MSI DCCO	NC	NC	NC	NA	NC	NC	NC	NC	0.36 (0.32-0.4)
ESI CATE	NA	0.67 (0.52-0.71)[1]	0.69 (0.68-0.70)	0.67 (0.61-0.70)[1]	0.66 (0.60-0.70)[1]	0.62 (0.55-0.68)[1]	0.62 (0.55-0.68)	0.59 (0.58-0.61)	0.46 (0.44-0.48)
ESI DCCO	NA	0.35 (0.33-0.37)[1]	0.33 (0.31-0.35)[1]	0.35 (0.32-0.37)[1]	0.31 (0.29-0.34)[1]	0.33 (0.31-0.35)[1]	0.28 (0.26-0.3)[1]	0.33 (0.31-0.36)[1]	0.32 (0.29-0.35)[1]

Colony	2007	2008	2009	2010	2011	2012	2013	2014	2015	*2016*	*2017*	*2018*	*2019*
BLI CATE	0.53 (0.35-0.63)[1]	0.47 (0.37-0.56)	0.55 (0.49-0.6)	0.56 (0.48-0.62)	NA	NA	NA	0.55 (0.43-0.63)[1]	0.55 (0.48-0.6)	0.58 (0.53-0.63)	NC	NC	NC
PTI CATE	0.45 (0.38-0.51)	0.49 (0.46-0.52)	0.36 (0.33-0.4)	0.43 (0.41-0.45)	0.45 (0.42-0.47)	0.41 (0.38-0.44)	0.46 (0.43-0.49)	0.44 (0.38-0.53)[1]	NC	*0.53 (0.4-0.61)*	NC	NC	NC
PTI LAXX	NA	NA	NA	NA	NA	0.10 (0.8-0.13)	NA	NA	NA	NA	NA	NA	NA
PTI DCCO	NA	NA	NA	NA	NA	0.13 (0.11-0.15)	NA	NA	NA	NA	NA	NA	NA
LLI CATE	NC	NC	NC	NC	NC	NC	NC	NC	NC	*0.56 (0.42-0.65)*[1]	*0.55 (0.45-0.66)*[1]	*0.36 (0.22-0.51)*	*0.60 (0.55-0.65)*
SPL CATE	NC	NA	NA	NA	NA	0.44 (0.25-0.55)[1]	NA	NA	NA	*NA*	*NA*	*NA*	*NA*
HRI DCCO	NC	NC	NC	NC	NC	NC	NC	NA	NA	NA	NA	*0.84 (0.64-0.95)*[2]	*NA*
I20 LAXX	0.13 (0.08-0.15)[1]	NA	NA	NA	NA	NA	0.13 (0.12-0.14)	0.13 (0.11-0.15)[1]	0.13 (0.12-0.14)	*0.12 (0.11-0.14)*	*0.13 (0.12-0.14)*	*0.15 (0.14-0.15)*	*0.14 (0.13-0.14)*

Table B.17 (cont.)

FDI DCCO	0.34 (0.31-0.37)	0.37 (0.35-0.4)	0.36 (0.33-0.38)	0.33 (0.3-0.35)	0.27 (0.23-0.3)	0.20 (0.16-0.23)	NA	0.10 (0.06-0.14)	NA	*NA*	*NA*	*NA*	*NA*
BGI AWPE[2]	0.64 (0.58-0.71)	0.68 (0.62-0.74)	0.86 (0.8-0.91)	0.75 (0.69-0.82)	0.76 (0.6-0.92)[1]	0.71 (0.33-1)[1]	NA	0.75 (0.55-0.93)[1]	0.64 (0.26-0.97)[1]	*0.55* *(0.27-0.86)*[1]	NA	*0.58* *(0.49-0.68)*	*0.51* *(0.41-0.61)*
BGI LAXX	NC	NC	NC	NC	NC	NC	NC	NC	0.12 (0.10-0.15)[1]	*0.09* *(0.05-0.13)*[1]	*0.12* *(0.10-0.14)*	*0.06* *(0.04-0.07)*	*0.06* *(0.05-0.08)*
BGI CATE	NC	NC	NC	NC	NA	NA	NC	NC	NC	*NC*	*0.65* *(0.56-0.71)*[1]	*NC*	*NC*
CSI LAXX	0.13 (0.12-0.14)	0.13 (0.12-0.14)	0.12 (0.11-0.13	0.13 (0.12-0.14)	0.12 (0.1-0.13)	0.13 (0.12-0.14)	0.12 (0.11-0.13)	0.13 (0.12-0.14)	NC	NC	NC	NC	NC
CSI CATE	0.54 (0.53-0.56)	0.5 (0.48-0.52)	0.55 (0.53-0.57)	0.57 (0.54-0.59)	0.6 (0.56-0.63)	0.53 (0.49-0.57)	0.59 (0.55-0.62)	0.61 (0.58-0.64)	NC	NC	NC	NC	NC
CBI LAXX	NC	NC	NA	NA	NA	NA	0.12 (0.11-0.12)	0.14 (0.14-0.15)	0.14 (0.14-0.15)	*0.14* *(0.13-0.15)*	*0.14* *(0.14-0.15)*[1]	*0.14* *(0.14-0.15)*[1]	*0.15* *(0.14-0.15)*
CBI CATE	0.63 (0.59-0.66)	0.69 (0.66-0.71)	0.66 (0.62-0.69)	0.57 (0.39-0.66)[1]	0.57 (0.39-0.66)[1]	NA	0.57 (0.37-0.67)[1]	0.57 (0.37-0.67)[1]	0.56 (0.51-0.6)	*0.55* *(0.51-0.58)*	*0.4* *(0.36-0.44)*	*0.29* *(0.25-0.33)*	*0.25* *(0.21-0.29)*
MRI LAXX	0.13 (0.13-0.14)	0.13 (0.12-0.14)	0.13 (0.12-0.13)	0.13 (0.12-0.13)	0.13 (0.11-0.14)	0.13 (0.12-0.14)	0.13 (0.12-0.14)	0.13 (0.12-0.14)	0.14 (0.13-0.15)	0.13 (0.12-0.14)	0.12 (0.11-0.13)	0.13 (0.12-0.14)	0.13 (0.12-0.14)
RCI CATE	NC	NC	NC	NC	NA	NA	NA	NA	NA	NA	0.68 (0.61-0.7)	NA	NA
ESI CATE	0.64 (0.63-0.66)	0.66 (0.65-0.68)	0.65 (0.63-0.66)	0.6 (0.57-0.62)	0.56 (0.52-0.59)	0.55 (0.49-0.59)	0.42 (0.38-0.47)	0.47 (0.43-0.51)	0.6 (0.57-0.63)	0.56 (0.53-0.6)	0.51 (0.47-0.55)	0.55 (0.5-0.58)	NA
MSI DCCO	0.32 (0.26-0.37)	NA	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC	NC
ESI DCCO	0.29 (0.26-0.33)	0.35 (0.33-0.37)	0.34 (0.32-0.36)	0.39 (0.37-0.41)	0.37 (0.34-0.39)	0.37 (0.34-0.40)	0.35 (0.32-0.37)	0.39 (0.37-0.41)	0.41 (0.39-0.43)	0.31 (0.28-0.34)	0.36 (0.34-0.38)	0.47 (0.44-0.49)	NA
ESI BRCO	NA	NA	NA	NA	NA	0.37 (0.34-0.40)	0.35 (0.32-0.37)	0.39 (0.37-0.41)	0.41 (0.39-0.43)	NA	NA	NA	NA

[1] *Variation in detection probability partially inferred from other years (see Methods and Payton et al. 2019)*

[2] *Represents a minimum estimate* due to the lack of deposition probabilities (i.e. deposition probability was assumed to be 1.0)

Table B.18. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting at locations in the Columbia River estuary on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Bonneville Dam), Upper Columbia River (UCR; based on detection at Bonneville Dam), middle Columbia River (MCR; based on detection at Bonneville Dam), and Upper Willamette River (WR; based on detection at Sullivan Dam) during 1999-2018. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal 2%, 5%, 10%, 15%, and 20% are highlighted in yellow, light orange, dark orange, light red, and red, respectively.

	East Sand Island, Columbia River Estuary							
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
1999	0.4% (0.2-0.7)	0.5% (0.2-1.2)	0.2% (<0.1-0.8)	NA	NA	0.9% (0.3-2.6)	1.1% (0.8-2.2)	0.9% (0.6-1.7)
N	18,558	1,987	1,325			632	12,287	12,123
2000	4.6% (3.6-6.6)	3.3% (2.1-5.3)	2.2% (1.2-3.8)	NA	NA	NA	10.5% (8.4-15.0)	16.3% (12.8-22.9)
N	11,810	1,323	1,123				10,356	3,100
2001	14.0% (11.1-20.0)	6.4% (4.2-10.0)	13.2% (9.9-19.5)	NA	NA	15.0% (11.1-21.9)	33.9% (26.3-49.1)	NA
N	8,845	807	1,230			872	774	
2002	2.9% (2.3-4.1)	1.7% (1.2-2.6)	2.5% (1.9-3.5)	NA	NA	NA	21.9% (17.6-31.0)	14.2% (10.1-21.3)
N	30,617	4,899	20,493				7,331	561
2003	4.7% (3.7-6.9)	2.7% (2.0-4.0)	3.7% (2.9-5.3)	NA	NA	NA	26.0% (21.0-36.2)	19.0% (15.4-26.9)
N	28,150	6,234	30,723				8,553	27,918
2004	4.8% (3.6-7.0)	1.3% (0.6-2.6)	3.7% (2.9-5.4)	NA	NA	NA	25.8% (19.7-37.3)	14.1% (11.3-19.8)
N	4,816	929	9,533				803	6,040
2005	3.0% (2.2-4.4)	1.3% (0.6-2.6)	2.4% (1.6-3.8)	NA	NA	NA	28.3% (21.6-40.6)	15.1% (11.9-21.6)
N	5,935	1,121	2,518				753	5,610
2006	3.3% (2.4-5.0)	2.5% (1.7-3.9)	3.6% (1.8-6.6)	NA	NA	NA	27.5% (21.0-39.1)	23.4% (18.1-34.1)
N	5,570	4,057	731				1,100	2,064
2007	3.1% (2.5-4.4)	3.4% (2.3-5.3)	1.9% (1.2-3.2)	1.4% (0.8-2.5)	NA	18.7% (14.6-26.8)	22.6% (18.2-32.4)	15.7% (12.4-22.6)
N	23,830	2,005	2,268	1,505		2,234	6,391	3,042
2008	2.5% (1.9-3.6)	1.9% (1.5-2.7)	1.7% (1.0-2.9)	4.4% (3.2-6.7)	NA	13.5% (10.6-19.2)	14.2% (11.5-19.9)	16.7% (13.1-24.2)
N	11,425	24,136	1,662	2,509		2,291	19,572	2,513
2009	4.7% (3.7-6.9)	2.0% (1.5-2.9)	3.7% (2.5-5.6)	1.7% (1.2-2.7)	1.3% (0.7-2.2)	14.1% (11.1-20.0)	14.5% (11.9-20.1)	20.0% (15.6-29.3)
N	17,396	16,314	2,064	5,573	1,845	2,700	23,311	2,265
2010	3.4% (2.7-4.8)	0.7% (0.5-1.1)	2.9% (2.2-4.3)	1.8% (0.6-4.4)	1.6% (0.8-2.9)	11.9% (9.4-17.4)	14.3% (11.3-20.4)	13.7% (11.0-19.3)
N	*38,441*	*17,974*	*5,972*	*510*	*1,382*	*8,515*	*40,024*	*12,284*

Table B.18 (cont.)

2011	2.5% (1.8-3.6)	0.7% (0.5-1.1)	2.9% (1.4-5.3)	0.9% (0.3-2.0)	0.4% (0.1-1.3)	9.6% (6.6-14.7)	12.0% (9.4-17.3)	9.1% (6.9-13.4)
N	6,557	12,327	704	1,119	826	865	7,028	2,419
2012	2.2% (1.7-3.3)	0.7% (0.5-1.1)	1.2% (0.7-2.1)	0.7% (0.4-1.3)	2.1% (1.2-3.7)	9.4% (6.5-14.4)	10.2% (7.7-14.9)	7.5% (5.6-11.3)
N	17,929	10,742	3,227	3,731	1,457	1,084	4,768	3,357
2013	1.2% (0.8-1.8)	0.9% (0.5-1.6)	0.7% (0.3-1.4)	1.0% (0.5-1.8)	0.8% (0.3-2.0)	9.9% (7.0-15.3)	12.7% (9.6-18.5)	8.9% (6.6-13.4)
N	16,167	4,465	3,112	2,629	1,454	1,865	8,516	4,473
2014	1.1% (0.8-1.7)	1.0% (0.5-1.9)	1.4% (0.7-2.5)	1.2% (0.5-2.5)	1.6% (0.8-3.0)	9.5% (6.5-14.5)	8.6% (6.7-12.5)	11.4% (8.5-16.8)
N	14,828	2,800	2,297	1,587	1,739	1,119	8,812	3,841
2015	2.0% (1.5-2.9)	0.8% (0.4-1.5)	1.9% (1.3-2.9	0.4% (0.1-1.5)	1.6% (1.0-2.6)	7.8% (5.9-11.4)	10.2% (8.2-14.6)	10.5% (8.2-15.0)
N	20,245	2,629	5,943	768	3,311	3,927	16,451	6,004
2016	0.8% (0.6-1.2)	0.7% (0.3-1.3)	1.4% (0.9-2.1)	1.2% (0.4-3.2)	NA	8.8% (6.4-13.0)	6.1% (4.8-8.8)	7.5% (5.8-10.7)
N	21,874	2,887	5,939	604		2,086	14,473	8,123
2017	0.8% (0.5-1.2)	0.2% (0.1-0.5)	1.4% (0.9-2.3)	NA	NA	8.4% (5.6-13.1)	5.3% (3.9-7.7)	6.5% (4.7-9.6)
N	13,151	4,635	4,622			1,069	6,497	3,275
2018	1.4% (1.0-2.1)	1.3% (0.7-2.1)	1.4% (0.9-2.3)	NA	4.2% (2.9-6.4)	5.3% (3.8-8.0)	6.9% (5.3-10.2)	6.5% (4.8-9.7)
N	11,174	5,981	3,370		2,546	3,209	9,572	5,322

Rice Island, Columbia River Estuary

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
1998	2.5% (1.8-4.7)	2.4% (1.4-4.8)	3.3% (1.7-6.9)	NA	NA	NA	21.2% (16-42)	23.2% (15.8-42.2)
N	7,917	1,403	726				5,048	377
1999	3.8% (2.9-7.4)	2.8% (1.8-5.4)	2.1% (1.2-4.3)	NA	NA	15.6% (10.8-27.5)	17.1% (12.5-37.6)	13.3% (10.4-20.2)
N	18,558	1,987	1,325			632	12,287	12,123
2000	1.3% (1-2.6)	0.7% (0.3-1.7)	1.3% (0.6-2.8)	NA	NA	NA	3.1% (2.3-5.7)	2.9% (2-5.6)
N	11,810	1,323	1,123				10,356	3,100
2017	0.4% (0.2-0.6)	0.4% (0.2-0.7)	0.2% (0.1-0.5)	NA	NA	2.4% (1.3-4.1)	3.0% (2.3-4.4)	2.4% (1.7-3.7)
N	13,151	4,635	4,622			1,069	6,497	3,275

Table B.19. Estimated number of available PIT-tagged smolts ($\hat{N}$) and annual predation rates (95% credibility intervals) by Caspian terns nesting on East Sand Island in Columbia River estuary on ESA-listed salmonid populations originating from the Lower Columbia River (LCR) based on releases of tagged smolts both upstream and downstream of Bonneville Dam during 2007-2018. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal to 2%, 5%, 10%, 15%, and 20% are highlighted in yellow, light orange, dark orange, light red, and red, respectively.

	East Sand Island, Columbia River Estuary		
Year	LCR Chinook	LCR Coho	LCR Steelhead
2007	4.1% (3.2-5.6)	2.6% (1.4-4.3)	15.2% (11.4-20.7)
$\hat{N}$	13,229	943	7,260
2008	5.3% (4.1-6.9)	0.9% (0.6-1.5)	11.9% (8.7-16.5)
$\hat{N}$	16,850	3,130	8,375
2009	4.4% (3.3-6.1)	1.2% (0.7-1.8)	15.5% (11.3-22.0)
$\hat{N}$	24,022	3,099	5,589
2010	4.5% (3.5-6.2)	3.7% (2.6-5.1)	14.5% (11-20.1)
$\hat{N}$	33,461	6,059	6,049
2011	2.2% (1.6-3.0)	1.1% (0.6-1.6)	7.1% (5.2-10.1)
$\hat{N}$	22,374	5,694	7,825
2012	2.5% (1.7-3.4)	NA	10.4% (7.4-15.1)
$\hat{N}$	21,508		7,963
2013	2.4% (1.6-3.6)	NA	11.3% (7.7-15.9)
$\hat{N}$	14,046		9,984
2014	1.1% (0.7-1.6)	NA	10.5% (7.4-15.1)
$\hat{N}$	14,414		8,765
2015	0.6% (0.4-1.0)	NA	6.7% (4.9-9.1)
$\hat{N}$	12,633		14,335
2016	0.8% (0.5-1.2)	NA	8.4% (6.2-11.6)
$\hat{N}$	14,140		15,675
2017	1.0% (0.6-1.7)	2.5% (1.4-4.0)	5.8% (4.0-8.3)
$\hat{N}$	11,934	1,603	9,358
2018	1.8% (1.1-2.7)	6.7% (4.6-9.7)	8.7% (6.3-12.1)
$\hat{N}$	13,878	2,266	10,675

Table B.20 Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting at the Blalock Islands on ESA-listed salmonid populations originating from the Snake River (SR) and Upper Columbia River (UCR) based on detection at McNary Dam during 2007-2019. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than 2% and 5% are highlighted in yellow and light orange, respectively.

	Blalock Islands, John Day Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	<0.1%	0.1% (<0.1-0.2)	<0.1%	NA	0.9% (0.6-1.4)	1.0% (0.6-1.7)
N	74,905	7,374	6,764		7,683	3,111
2008	0.1% (0.1-0.2)	<0.1%	0.1% (<0.1-0.2)	NA	0.8% (0.6-1.2)	0.7% (0.4-1.2)
N	27,288	36,857	4,713		15,449	3,399
2009	0.3% (0.2-0.4)	<0.1%	0.2% (0.1-0.5)	<0.1%	0.6% (0.4-0.9)	0.5% (0.3-1.0)
N	60,155	43,461	3,982	2,088	29,877	3,663
2010	0.1% (<0.1-0.1)	<0.1%	0.1% (<0.1-0.1)	0.2% (<0.1-0.6)	0.9% (0.7-1.4)	0.9% (0.6-1.6)
N	52,129	29,587	10,456	1,327	17,806	4,161
2011	0.1% (<0.1-0.1)	0.1% (0.1-0.2)	<0.1%	0.3% (0.1-0.8)	0.1% (0.1-0.2)	0.1% (<0.1-0.3)
N	38,629	41,007	3,981	2,769	16,759	5,155
2013	<0.1%	0.1% (<0.1-0.1)	<0.1%	<0.1%	0.1% (<0.1-0.2)	0.2% (<0.1-0.5)
N	47,685	14,398	6,778	1,213	9,391	2,621
2014	0.1% (0.1-0.2)	0.3% (0.2-0.5)	0.2% (0.1-0.4)	0.4% (0.1-1.1)	0.4% (0.2-0.7)	0.6% (0.3-1.2)
N	41,109	10,293	4,611	1,922	10,389	2,686
2015	1.4% (1.1-2.2)	0.4% (0.4-0.8)	0.9% (0.5-1.5)	1.3% (0.7-2.5)	8.0% (6.0-11.4)	8.2% (5.9-12.4)
N	31,474	4,390	4,921	1,712	6,824	2,056
2016	0.3% (0.2-0.5)	0.6% (0.4-1.1)	0.2% (0.1-0.4)	2.3% (1.2-4.1)	3.9% (3.9-5.7)	3.1% (2.3-4.6)
N	47,573	6,726	11,320	1,095	14,332	7,414
2017	0.9% (0.6-1.3)	0.6% (0.4-1.1)	1.1% (0.7-1.8)	NA	3.4% (2.4-5.1)	4.2% (2.7-6.5)
N	17,215	9,230	6,517		5,795	2,536
2018	0.5% (0.3-0.9)	0.7% (0.4-1.4)	0.3% (0.1-0.8)	2.0% (0.4-6.1)	2.5% (1.4-4.5)	2.9% (1.5-5.2)
N	17,963	8,450	5,228	514	3,585	2,228
2019	0.4% (0.2-0.8)	1.3% (0.6-2.5)	0.9% (0.2-2.1)	1.4% (0.4-3.7)	3.0% (1.9-4.7)	5.9% (3.4-10.0)
N	11,225	3,395	2,838	1,167	5,878	1,671

Table B.21. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on Crescent Island and Badger Island in McNary Reservoir, Goose Island in Potholes Reservoir, an unnamed island in Potholes Reservoir, and an unnamed island in Lenore Lake on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Lower Monumental Dam) and Upper Columbia River (UCR; based on detection at Rock Island Dam) during 2007-2019. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal to 2%, 5%, 10%, 15%, and 20% are highlighted in yellow, light orange, dark orange, light red, and red, respectively.

Crescent Island, McNary Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	0.4% (0.3-0.6)	0.9% (0.4-1.7)	NA	NA	3.9% (3.1-5.6)	2.5% (1.7-3.8)
N	22,730	2,147		254	17,122	3,782
2008	0.9% (0.7-1.3)	1.6% (1.2-2.3)	NA	1.7% (0.6-3.7)	5.9% (4.7-8.5)	2.9% (2.1-4.3)
N	30,142	22,968		767	28,653	8,403
2009	1.5% (1.1-2.2)	1.1% (0.8-1.6)	0.2% (<0.1-1.2)	1.0% (0.5-1.7)	4.6% (3.7-6.6)	2.3% (1.7-3.5)
N	20,679	26,567	738	2,651	52,102	8,025
2010	0.6% (0.4-1.1)	1.3% (1.0-1.9)	0.9% (0.3-2.3)	1.5% (0.5-3.4)	5.5% (4.2-7.9)	1.8% (1.3-2.7)
N	5,790	28,067	929	566	7,913	8,382
2011	0.8% (0.6-1.2)	0.6% (0.5-0.9)	0.5% (0.1-1.2)	0.9% (0.7-1.4)	3.0% (2.3-4.3)	2.4% (1.8-3.6)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	0.7% (0.5-1.0)	0.6% (0.5-1.0)	0.2% (0.1-0.8)	2.4% (1.5-3.8)	3.1% (2.3-4.5)	1.2% (0.8-2.0)
N	41,258	24,772	1,812	2,884	25,841	6,845
2013	0.7% (0.5-1.1)	0.9% (0.6-1.5)	0.4% (<0.1-1.2)	1.2% (0.5-2.7)	3.5% (2.7-5.1)	2.9% (2.1-4.3)
N	14,859	4,773	992	848	9,696	6,019
2014	0.8% (0.6-1.1)	0.6% (0.4-1.0)	0.7% (0.2-2.1)	1.5% (0.8-2.8)	6.1% (4.8-8.9)	3.4% (2.5-4.8)
N	22,195	6,043	641	1,414	16,599	7,757

Badger Island, McNary Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2017	<0.1%	<0.1%	<0.1%	NA	0.4% (0.2-0.6)	0.5% (0.3-0.8)
N	27,977	9,769	2,681		24,247	7,644

Table B.21 (cont.)

Goose Island, Potholes Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	<0.1%	0.3% (<0.1-1.1)	NA	NA	0.1% (<0.1-0.2)	15.3% (9.8-27.7)
N	22,730	2,147			17,122	3,782
2008	<0.1%	<0.1%	NA	0.4% (<0.1-1.6)	<0.1%	11.1% (8.6-16.4)
N	30,142	22,968		767	28,653	8,403
2009	<0.1%	<0.1%	5.5% (2.7-10.7)	0.1% (<0.1-0.4)	0.1% (<0.1-0.1)	22.6% (17.2-33.7)
N	20,679	26,567	738	2,651	52,102	8,025
2010	<0.1%	<0.1%	2.0% (0.7-4.4)	0.3% (<0.1-1.9)	<0.1%	14.6% (11.0-21.8)
N	5,790	28,067	929	566	7,913	8,382
2011	<0.1%	<0.1%	0.6% (0.1-1.9)	<0.1%	<0.1%	12.9% (9.6-19.6)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	<0.1%	<0.1%	2.6% (1.2-5.4)	0.2% (<0.1-0.6)	0.2% (0.1-0.4)	18.4% (13.5-28.5)
N	41,258	24,772	1,812	2,884	25,841	6,845
2013	<0.1%	0.1% (<0.1-0.4)	2.5% (1.1-5.2)	0.1% (<0.1-1.1)	0.1% (0.1-0.4)	14.8% (11.4-21.6)
N	14,859	4,773	992	848	9,696	6,019
2014	<0.1%	0.1% (<0.1-0.8)	0.6% (0.1-2.2)	0.2% (<0.1-1.1)	<0.1%	2.9% (1.9-5.1)
N	22,195	6,043	641	1,414	16,599	7,757

Unnamed Island, Potholes Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2016	<0.1%	<0.1%	0.1% (<0.1-0.3)	<0.1%	<0.1%	4.1% (2.9-6.3)
N	38,633	5,461	1,956	522	20,729	7,003

Lenore Lake Island, Lenore Lake

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2015	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	4,471	1,393	766	1,262	2,400	7,222
2016	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	38,633	5,461	1,956	522	20,729	7,003
2017	<0.1%	<0.1%	0.3% (0.1-0.8)	NA	<0.1%	1.0% (0.6-2.0)

Table B.21 (cont.)

N	27,977	9,769	2,681		24,247	7,644
2018	<0.1%	<0.1%	0.1% (0.1-0.8)	<0.1%	<0.1%	0.8% (0.4-1.7)
N	19,986	8,753	2,090	1,443	19,632	7,511
2019	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)	<0.1%	1.0% (0.6-1.7)
N	18,757	7,501	1,885	1,675	28,813	4,401

Banks Lake Island, Banks Lake

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2008	<0.1%	<0.1%	NA	<0.1%	<0.1%	<0.1%
N	30,142	22,968		767	28,653	8,403
2009	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)
N	20,679	26,567	738	2,651	52,102	8,025
2010	<0.1%	<0.1%	<0.1%	0.2% (<0.1-1.2)	<0.1%	0.1% (<0.1-0.3)
N	5,790	28,067	929	566	7,913	8,382
2012	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)
N	41,258	24,772	1,812	2,884	25,841	6,845
2014	<0.1%	<0.1%	0.5% (<0.1-7.9)	0.1% (<0.1-0.6)	<0.1%	1.2% (0.3-6.4)
N	22,195	6,043	641	1,414	16,599	7,757
2015	<0.1%	<0.1%	0.2% (<0.1-0.9)	0.1% (0-0.5)	<0.1%	2.6% (1.8-3.9)
N	7,706	3,449	766	1,262	3,601	7,222
2016	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.2)
N	38,633	5,461	1,956	522	20,729	7,003

Table B.22. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on Harper Island in Sprague Lake on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Lower Granite Dam) and Upper Columbia River (UCR; based on detection at Rock Island Dam) in 2012. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates.

	Harper Island, Sprague Lake					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2012	<0.1%	<0.1%	<0.1%	<0.1%	0.2% (<0.1-1.3)	<0.1%
N	41,258	24,772	1,812	2,884	25,841	6,845

Table B.23. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting at locations in the Columbia River estuary on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Bonneville Dam), Upper Columbia River (UCR; based on detection at Bonneville Dam), middle Columbia River (MCR; based on detection at Bonneville Dam), and Upper Willamette River (UWR; based on detection at Sullivan Dam) during 1999-2018. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Predation rates greater than or equal to 2%, 5%, 10%, and 15% are highlighted in yellow, light orange, dark orange, and light red, respectively.

	East Sand Island, Columbia River Estuary							
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
1999	0.8% (0.5-1.3)	1.4% (0.6-3)	0.7% (0.1-1.9)	NA	NA	0.3% (<0.1-1.8)	2.4% (1.6-3.8)	1.9% (1.3-3.1)
N	18,558	1,987	1,325			632	12,287	12,123
2000	3.3% (2.3-5.3)	5.1% (2.9-9.3)	3.4% (1.6-6.8)	NA	NA	NA	10.6% (7.5-16.8)	6.0% (3.9-10.5)
N	11,810	1,323	1,123				10,356	3,100
2001	2.2% (1.4-3.5)	5.5% (2.9-10.4)	3.3% (1.7-6.3)	NA	NA	2.5% (1.0-5.7)	2.8% (1.1-6.1)	NA
N	8,845	807	1,230			872	774	
2002	1.8% (1.3-3.0)	1.4% (0.8-2.6)	2.2% (1.6-3.6)	NA	NA	NA	3.1% (2.0-5.1)	3.7% (1.4-8.6)
N	30,617	4,899	20,493				7,331	561
2003	1.7% (1.2-2.7)	1.1% (0.7-2.0)	1.4% (0.9-2.1)	NA	NA	NA	1.9% (1.2-3.0)	1.5% (1.0-2.4)
N	28,150	6,234	30,723				8,553	27,918
2004	5.1% (3.3-8.5)	1.9% (0.6-4.7)	4.7% (3.2-7.6)	NA	NA	NA	3.6% (1.4-8.0)	7.4% (5.1-11.8)
N	4,816	929	9,533				803	6,040
2005	4.8% (3.2-7.9)	3.6% (1.8-6.9)	4.5% (2.8-7.8)	NA	NA	NA	4.3% (2.0-8.6)	5.5% (3.7-8.8)
N	5,935	1,121	2,518				753	5,610
2006	5.2% (3.5-8.5)	2.7% (1.6-4.6)	4.7% (2.2-9.5)	NA	NA	NA	13.1% (8.2-22.7)	4.7% (2.8-8.2)
N	5,570	4,057	731				1,100	2,064
2007	1.7% (1.1-2.7)	1.6% (0.7-3.3)	2.7% (1.5-5.1)	1.0% (0.3-2.6)	NA	2.8% (1.5-5.2)	3.5% (2.3-5.8)	3.4% (2.1-6.1)
N	23,830	2,005	2,268	1,505		2,234	6,391	3,042
2008	3.5% (2.4-5.5)	2.6% (1.9-4.2)	3.6% (2.0-6.6)	3.3% (1.9-5.8)	NA	14.0% (9.5-23.2)	14.7% (10.6-23.2)	6.2% (4.0-10.4)
N	11,425	24,136	1,662	2,509		2,291	19,572	2,513
2009	6.8% (4.9-10.7)	4.5% (3.2-7.1)	2.7% (1.5-4.9)	1.4% (0.8-2.4)	5.7% (3.5-9.8)	14.9% (10.3-23.8)	16.6% (12.0-25.7)	7.2% (4.7-12.0)
N	17,396	16,314	2,064	5,573	1,845	2,700	23,311	2,265
2010	5.3% (3.9-8.4)	3.9% (2.7-6.1)	3.3% (2.3-5.4)	4.2% (1.6-9.2)	2.6% (1.3-4.9)	8.2% (5.8-13.1)	7.5% (5.5-12.0)	6.8% (4.9-10.6)
N	38,441	17,974	5,972	510	1,382	8,515	40,024	12,284
2011	4.3% (2.9-6.9)	1.9% (1.3-3.1)	5.6% (2.9-10.8)	0.4% (0.1-1.5)	4.8% (2.4-9.1)	7.8% (4.6-14.0)	5.3% (3.7-8.5)	11.4% (7.8-18.6)

N	6,557	12,327	704	1,119	826	865	7,028	2,419
2012	3.7% (2.6-6.0)	2.6% (1.8-4.2)	2.1% (1.2-3.7)	0.6% (0.3-1.3)	3.7% (2.0-6.9)	3.3% (1.7-6.4)	4.9% (3.2-8.1)	6.5% (4.3-10.8)
N	17,929	10,742	3,227	3,731	1,457	1,084	4,768	3,357
2013	3.6% (2.5-5.7)	2.2% (1.3-3.7)	3.0% (1.8-5.3)	1.0% (0.4-2.0)	3.3% (1.8-6.2)	2.1% (1.0-4.1)	2.5% (1.7-4.0)	3.4% (2.2-5.7)
N	16,167	4,465	3,112	2,629	1,454	1,865	8,516	4,473
2014	8.5% (6.1-13.2)	2.4% (1.5-4.2)	6.1% (3.9-10.1)	1.8% (0.9-3.6)	4.5% (2.7-7.7)	6.4% (3.7-10.7)	7.8% (5.6-12.0)	10.4% (7.3-16.3)
N	14,828	2,800	2,297	1,587	1,739	1,119	8,812	3,841
2015	14.5% (10.5-22.4)	8.7% (6.0-14.0)	8.3% (5.9-12.9)	2.4% (0.9-5.2)	2.4% (1.5-4.1)	12.4% (8.8-19.2)	12.8% (9.3-19.6)	10.5% (7.6-16.2)
N	20,245	2,629	5,943	768	3,311	3,927	16,451	6,004
2016[a]	5.1% (3.7-8.1)	2.1% (1.1-3.9)	3.5% (2.3-5.7)	0.4% (0-2.1)	NA	2.7% (1.4-4.9)	6.8% (4.8-10.7)	5.1% (3.6-8.2)
N	21,874	2,887	5,939	604		2,086	14,473	8,123
2017[a]	0.7% (0.4-1.1)	0.1% (0-0.2)	0.4% (0.1-0.8)	NA	NA	0.7% (0.1-2.1)	0.4% (0.2-0.8)	1.4% (0.8-2.7)
N	13,151	4,635	4,622			1,069	6,497	3,275
2018[a]	0.5% (0.3-0.8)	0.9% (0.5-1.6)	0.6% (0.3-1.2)	NA	0.9% (0.5-1.9)	0.4% (0.1-1.0)	0.5% (0.3-0.9)	0.7% (0.4-1.4)
N	11,174	5,981	3,370		2,546	3,209	9,572	5,322

Rice Island, Columbia River Estuary

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
1998	1.6% (1.0-3.1)	1.7% (0.8-3.9)	2.4% (1.0-5.8)	NA	NA	NA	5.3% (3.5-10.8)	NA
N	7,917	1,403	726				5,048	
2001	0.4% (0.2-0.8)	1.1% (0.2-3.3)	0.7% (0.1-2.0)	NA	NA	0.2% (<0.1-1.5)	1.0% (0.1-3.0)	NA
N	8,845	807	1,230			872	774	
2006	0.1% (<0.1-0.4)	<0.1%	0.3% (<0.1-2)	NA	NA	NA	0.2% (<0.1-1.5)	0.2% (<0.1-0.8)
N	5,570	4,057	731				1,100	2,064

Miller Sands Spit, Columbia River Estuary

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2006	0.1% (<0.1-0.4)	<0.1%	0.3% (<0.1-1.9)	1.2% (<0.1-8.5)	NA	1.1% (<0.1-6.9)	0.2% (<0.1-1.4)	0.2% (<0.1-0.9)
N	5,570	4,057	731	461		201	1,100	2,064
2007	0.3% (0.1-0.6)	0.3% (<0.1-1.2)	0.1% (<0.1-0.6)	0.3% (<0.1-2.1)	NA	0.3% (<0.1-1.2)	0.9% (0.4-1.8)	0.6% (0.2-1.5)
N	23,830	2,005	2,268	1,505		2,234	6,391	3,042

[a] *Minimum estimates of predation rate due to multiple colony dispersal events (2016 and 2017) or the late arrival of double-crested cormorants on East Sand Island (2018)during the peak smolt out-migration period*

Table B.24. Estimated number of available PIT-tagged smolts ($\widehat{N}$) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting on East Sand Island in the Columbia River estuary on ESA-listed salmonid populations originating from the Lower Columbia River (LCR) based releases of tagged smolts both upstream and downstream of Bonneville Dam during 2007-2018. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal to 2%, 5%, 10%, 15%, and 20% are highlighted in yellow, light orange, dark orange, light red, and red, respectively.

	East Sand Island, Columbia River Estuary		
Year	LCR Chinook	LCR Coho	LCR Steelhead
2007	22.1% (15.5-33.5)	8.1% (4.2-14.1)	2.6% (1.5-4.0)
$\widehat{N}$	13,229	943	7,260
2008	51.0% (38.3-59.0)	17.8% (11.6-23.8)	9.0% (5.9-13.4)
$\widehat{N}$	16,850	3,130	8,375
2009	25.0% (18.2-34.3)	12.5% (8.0-18.6)	6.3% (3.9-9.8)
$\widehat{N}$	24,022	3,099	5,589
2010	31.7% (22.3-44.2)	18.1% (13.0-26.1)	4.5% (2.8-6.7)
$\widehat{N}$	33,030	6,060	6,050
2011	18.5% (15.3-22.4)	17.8% (12.7-26.6)	5.7% (3.8-8.5)
$\widehat{N}$	22,374	5,694	7,825
2012	19.8% (13.2-28.6)	NA	4.0% (2.6-6.4)
$\widehat{N}$	21,508		7,963
2013	26.8% (18.7-39.3)	NA	1.7% (1.1-2.7)
$\widehat{N}$	14,046		9,984
2014	22.0% (16.5-30.7)	NA	7.7% (4.9-11.5)
$\widehat{N}$	14,414		8,765
2015	18.9% (12.6-29.6)	NA	11.2% (7.5-16.6)
$\widehat{N}$	12,633		14,335
2016	5.7% (3.9-8.4)	NA	3.1% (2.1-4.8)
$\widehat{N}$	14,140		15,675
2017	1.0% (0.5-1.7)	0.2% (<0.1-0.7)	0.6% (0.3-1.0)
$\widehat{N}$	11,934	1,603	9,358
2018	7.3% (4.8-11.6)	0.3% (<0.1-0.8)	0.6% (0.3-1.0)
$\widehat{N}$	13,878	2,266	10,675

[a] Minimum estimates of predation rate due to multiple colony dispersal events (2016 and 2017) or the late arrival of double-crested cormorants on East Sand Island (2018) during the peak smolt out-migration period

Table B.25. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting at Foundation Island in McNary Reservoir, an unnamed island in Hanford Reach of the middle Columbia River, an unnamed island in Lenore Lake, and in northern Potholes Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Lower Monumental Dam) and Upper Columbia River (UCR; based on detection at Rock Island Dam) during 2007-2019. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal to 2% are highlighted in yellow.

Foundation Island, McNary Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	1.5% (1.0-2.4)	1.9% (1.0-3.6)	NA	NA	4.7% (3.4-7.5)	< 0.1%
N	22,730	2,147			17,122	3,782
2008	2.0% (1.4-3.1)	0.9% (0.6-1.4)	NA	2.4% (1.0-5.3)	4.7% (3.4-7.3)	0.3% (0.1-0.6)
N	30,142	22,968		767	28,653	8,403
2009	1.8% (1.3-2.9)	1.1% (0.7-1.7)	0.2% (<0.1-1.4)	4.3% (2.7-7.1)	3.6% (2.6-5.6)	0.2% (0.1-0.5)
N	20,679	26,567	738	2,651	52,102	8,025
2010	2.4% (1.5-3.9)	1.1% (0.7-1.7)	0.2% (<0.1-1.2)	3.8% (1.5-8.3)	3.6% (2.5-5.9)	0.2% (0.1-0.4)
N	5,790	28,067	929	566	7,913	8,382
2011	1.4% (1.0-2.4)	1.2% (0.8-1.9)	0.5% (0.1-1.8)	1.6% (1.0-2.8)	4.3% (2.9-7.0)	0.3% (0.1-0.6)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	0.9% (0.6-1.5)	0.6% (0.3-1.1)	0.5% (0.1-1.9)	4.1% (2.3-7.6)	2.4% (1.5-3.9)	0.5% (0.2-1.1)
N	41,258	24,772	1,812	2,884	25,841	6,845
2014	1.1% (0.5-2.1)	0.6% (0.2-1.9)	1.0% (<0.1-6.2)	2.8% (0.7-8.0)	1.8% (0.9-3.4)	0.2% (<0.1-0.8)
N	22,195	6,043	641	1,414	16,599	7,757

Hanford Reach island, Middle Columbia River

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2018	<0.1%	<0.1%	0.2% (0.1-0.7)	<0.1%	<0.1%	0.2% (0.1-0.4)
N	19,986	8,753	2,090	1,443	19,632	7,511

Lenore Lake island, Lenore Lake

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2017	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	27,977	9,769	2,681	304	24,247	7,644

Table B.25 (cont.)

2019	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	18,757	7,501	1,885	1,675	28,813	4,401

Northern Potholes Reservoir, Potholes Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2012	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.3% (<0.1-0.8)
N	41,258	24,772	1,812	2,884	25,841	6,845

Table B.26. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by California and ring-billed gulls nesting Miller Rocks in The Dalles Reservoir and the central Blalock Islands in John Day Reservoir on ESA-listed salmonid populations originating from the Snake River (SR) and Upper Columbia River (UCR) based on detection at McNary Dam during 2007-2019. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal to 2%, 5%, and 10% are highlighted in yellow, light orange, and dark orange, respectively.

Miller Rocks, The Dalles Reservoir						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	1.2% (0.8-1.7)	3.3% (2.1-5.4)	2.2% (1.3-3.8)	NA	9.9% (6.9-14.6)	8.7% (5.5-13.9)
N	74,905	7,374	6,764		7,683	3,111
2008	2.3% (1.5-3.4)	2.7% (1.9-4.0)	1.6% (0.8-3.2)	NA	9.2% (6.6-13.4)	7.2% (4.5-11.6)
N	27,288	36,857	4,713		15,449	3,399
2009	2.2% (1.5-3.2)	3.7% (2.6-5.4)	2.5% (1.2-4.6)	9.1% (5.3-15.1)	9.8% (7.1-14.3)	7.4% (4.6-11.8)
N	60,155	43,461	3,982	2,088	29,877	3,663
2010	1.8% (1.2-2.7)	1.0% (0.6-1.5)	1.7% (1.0-2.9)	4.1% (1.6-8.8)	9.1% (6.5-13.2)	7.0% (4.4-11.2)
N	52,129	29,587	10,456	1,327	17,806	4,161
2011	0.8% (0.5-1.3)	0.6% (0.4-1.0)	1.0% (0.3-2.3)	2.0% (0.8-4.2)	5.0% (3.5-7.6)	3.5% (2.0-5.9)
N	38,629	41,007	3,981	2,769	16,759	5,155
2012	0.6% (0.4-1.0)	0.7% (0.4-1.3)	1.5% (0.8-2.9)	6.3% (3.6-10.9)	4.6% (3.0-7.2)	7.2% (4.4-11.6)
N	40,168	25,017	6,800	2,492	8,840	3,804
2013	1.1% (0.7-1.7)	2.4% (1.6-3.8)	1.9% (1.0-3.4)	6.4% (3.0-12.5)	6.4% (4.4-9.7)	11.7% (7.4-18.4)
N	47,685	14,398	6,778	1,213	9,391	2,621
2014	1.0% (0.7-1.6)	1.8% (1.0-3.0)	1.3% (0.6-2.7)	4.4% (2.1-8.3)	5.3% (3.6-8.0)	6.1% (3.5-10.4)
N	41,109	10,293	4,611	1,922	10,389	2,686
2015	1.7% (1.1-2.6)	2.6% (1.4-4.6)	3.5% (2.1-6.0)	7.4% (4.1-13.1)	9.7% (6.6-14.6)	13.2% (8.3-21.1)
N	31,474	4,390	4,921	1,712	6,824	2,056
2016	1.2% (0.8-1.9)	1.0% (0.4-2.1)	2.5% (1.6-4.0)	6.4% (2.9-12.8)	6.7% (4.6-9.9)	10.1% (7.0-15.2)
N	47,573	6,726	11,320	1,095	14,332	7,414
2017	0.6% (0.3-1.1)	0.8% (0.4-1.7)	2.2% (1.2-3.9)	NA	7.0% (4.3-11.0)	6.9% (3.8-12.0)
N	17,215	9,230	6,517		5,795	2,536
2018	0.4% (0.2-0.9)	2.1% (1.2-3.5)	1.2% (0.5-2.4)	7.0% (2.3-16.4)	3.3% (1.8-6.0)	8.3% (4.8-13.9)
N	17,963	8,450	5,228	514	3,585	2,228

Table B.26 (cont.)

2019	0.9% (0.5-1.7)	3.2% (1.7-5.8)	2.4% (1.1-4.9)	5.9% (2.6-11.8)	5.1% (3.2-8.1)	6.5% (3.4-11.8)
N	11,225	3,395	2,838	1,167	5,878	1,671
			Central Blalock Islands, John Day Reservoir			
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2013	0.1% (<0.1-0.2)	0.3% (0.1-0.8)	0.4% (0.1-1.2)	1.9% (0.4-5.6)	1.2% (0.6-2.3)	1.2% (0.3-3.1)
N	47,685	14,398	6,778	1,213	9,391	2,621
2014	0.2% (0.1-0.4)	0.4% (0.1-0.9)	0.4% (0.1-1.1)	1.0% (0.2-2.7)	1.5% (0.9-2.5)	2.5% (1.2-4.9)
N	41,109	10,293	4,611	1,922	10,389	2,686
2015	0.2% (0.1-0.4)	0.7% (0.2-1.7)	0.6% (0.2-1.5)	1.4% (0.4-3.7)	2.6% (1.6-4.2)	6.8% (3.9-11.4)
N	31,474	4,390	4,921	1,712	6,824	2,056
2016	0.1% (0.1-0.3)	0.5% (0.2-1.2)	0.2% (0-0.5)	3.7% (1.4-8.2)	3.5% (2.4-5.2)	6.3% (4.4-9.2)
N	47,573	6,726	11,320	1,095	14,332	7,414
2017	0.2% (0.1-0.5)	0.2% (<0.1-0.5)	0.2% (0-0.7)	NA	2.5% (1.5-4.2)	5.2% (3-8.8)
N	17,215	9,230	6,517		5,795	2,536
2018	0.2% (0.1-0.4)	0.3% (0.1-0.8)	0.6% (0.2-1.5)	2.2% (0.3-8)	3.3% (1.8-5.8)	3.1% (1.5-6)
N	17,963	8,450	5,228	514	3,585	2,228
2019	02% (0.1-0.5)	0.3% (0.1-1.1)	0.3% (0.1-1.2)	1.4% (0.3-4.2)	1.6% (0.8-2.8)	1.5% (0.4-3.8)
N	11,225	3,395	2,838	1,167	5,878	1,671

Table B.27. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by California and ring-billed gulls nesting on Crescent Island and Badger islands in McNary Reservoir, Island 20 in the middle Columbia River, and Goose Island in Potholes Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Lower Monumental Dam) and Upper Columbia River (UCR; based on detection at Rock Island Dam) during 2007-2018. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. Predation rates greater than or equal to 2%, 5%, and 10% are highlighted in yellow, light orange, and dark orange, respectively.

Crescent Island, McNary Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	0.6% (0.3-1.1)	0.6% (0.1-2.3)	NA	NA	4.1% (2.7-6.5)	5.9% (3.5-10.1)
N	22,730	2,147			17,122	3,782
2008	0.9% (0.5-1.4)	0.6% (0.3-1.1)	NA	1.7% (0.2-6.1)	4.0% (2.8-5.9)	3.0% (1.8-4.8)
N	30,142	22,968		767	28,653	8,403
2009	0.9% (0.5-1.6)	0.7% (0.4-1.1)	0.8% (<0.1-4.7)	2.6% (1.1-5.4)	4.7% (3.4-6.8)	7.5% (5.0-11.4)
N	20,679	26,567	738	2,651	52,102	8,025
2010	1.4% (0.7-2.8)	0.3% (0.2-0.6)	0.5% (<0.1-3.3)	0.9% (<0.1-5.4)	5.1% (3.3-7.9)	7.8% (5.3-11.7)
N	5,790	28,067	929	566	7,913	8,382
2011	1.1% (0.7-1.7)	0.7% (0.4-1.2)	0.4% (<0.1-2.4)	1.6% (0.9-2.8)	3.4% (2.4-5.1)	3.6% (2.2-5.9)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	1.1% (0.7-1.7)	0.6% (0.3-1.2)	1.3% (0.3-4.0)	1.3% (0.4-3.2)	5.2% (3.5-8.0)	4.7% (2.9-7.8)
N	41,258	24,772	1,812	2,884	25,841	6,845
2013	1.0% (0.5-1.8)	0.8% (0.2-2.0)	0.6% (<0.1-3.6)	2.8% (0.6-8.3)	5.8% (3.8-8.9)	6.1% (3.8-9.7)
N	14,859	4,773	992	848	9,696	6,019
2014	1.1% (0.6-1.8)	0.3% (<0.1-0.9)	2.1% (<0.3-7.5)	3.1% (1.1-7.0)	5.5% (3.8-8.2)	6.8% (4.6-10.4)
N	22,195	6,043	641	1,414	16,599	7,757

Badger Island, McNary Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2015	0.1% (<0.1-0.5)	0.1% (0-0.9)	0.5% (<0.1-3.3)	1.1% (0.2-4.8)	2.9% (1.3-6.4)	5.2% (3.2-9.2)
N	7,706	3,449	766	1,262	3,601	7,222
2016	0.2% (0.1-0.4)	<0.1%	0.9% (0.1-3.3)	1.2% (0.1-7.7)	1.1% (0.6-1.9)	4.3% (2-13.8)
N	38,633	5,461	1,956	522	20,729	7,003

Table B.27 (cont.)

2017	0.2% (<0.1-0.4)	0.4% (0.1-1.0)	0.6% (0.1-2.1)	NA	1.0% (0.6-1.8)	1.3% (0.6-2.6)
N	27,977	9,769	2,681		24,247	7,644
2018	1.0% (0.5-1.8)	1.0% (0.4-2.1)	1.1% (0.2-4.1)	4.0% (1.1-9.5)	4.3% (3-6.7)	4.8% (2.8-8.0)
N	19,986	8,753	2,090	1,443	19,632	7,511
2019	1.2% (0.6-2.1)	1.9% (0.8-8.4)	3.6% (1.1-8.8)	3.1% (0.9-8.3)	5.6% (3.9-8.4)	10.9% (6.7-17.7)
N	18,757	7,501	1,885	1,675	28,813	4,401

Island 20, Middle Columbia River

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2013	0.3% (0.1-0.7)	0.1% (<0.1-0.7)	0.5% (<0.1-3.2)	0.6% (<0.1-3.6)	0.7% (0.3-1.4)	1.4% (0.6-2.7)
N	14,859	4,773	992	848	9,696	6,023
2014	0.2% (0.1-0.5)	0.2% (<0.1-0.9)	0.8% (<0.1-5.1)	0.4% (<0.1-2.3)	0.6% (0.3-1.1)	1.6% (0.8-3.1)
N	22,195	6,043	641	1,414	16,599	7,757
2015	0.3% (0.1-0.8)	0.1% (0-0.8)	0.6% (<0.1-3.7)	NA	2.4% (1.2-4.5)	7.9% (5.3-12.0)
N	7,706	3,449	766	1,262	3,601	7,222
2016	0.2% (0.1-0.4)	<0.1%	0.2% (<0.1-3.8)	0.9% (<0.1-5.8)	1.2% (0.7-2.0)	5.7% (3.7-8.9)
N	38,633	5,461	1,956	522	20,792	7,003
2017	0.2% (<0.1-0.4)	0.2% (<0.1-0.6)	0.1% (<0.1-1.0)	NA	1.7% (1.1-2.6)	3.0% (1.8-4.9)
N	27,977	9,769	2,681		24,247	7,644
2018	0.1% (<0.1-0.3)	0.3% (0.1-0.8)	0.2% (<0.1-1.3)	0.8% (0.1-2.8)	1.3% (0.8-2.1)	1.1% (0.5-2.0)
N	19,986	8,793	2,090	1,443	19,632	7,511
2019	0.2% (0.1-0.4)	0.2% (0.1-0.7)	0.2% (0.1-1.5)	0.2% (<0.1-1.5)	2.8% (1.9-4.2)	7.4% (4.8-11.7)
N	18,757	7,501	1,885	1,675	28,813	4,401

Goose Island, Potholes Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2012	<0.1%	<0.1%	1.3% (0.3-4.0)	<0.1%	0.1% (<0.1-0.3)	2.8% (1.1-5.6)
N	41,258	24,772	1,812	2,884	25,841	6,845

Table B.28. Number of available PIT-tagged smolts (N) and annual predation rates (95% Credibility Intervals) by Brandt's cormorants nesting on East Sand Island in the Columbia River estuary on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Bonneville Dam), Upper Columbia River (UCR; based on detection at Bonneville Dam), middle Columbia River (MCR; based on detection at Bonneville Dam), and Upper Willamette River (UWR; based on detection at Sullivan Dam) during 2012-2015. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates.

	East Sand Island, Columbia River Estuary							
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2012	<0.1%	0.1% (<0.1-0.1)	<0.1%	0.2% (<0.1-0.4)	<0.1%	0.2% (<0.1-0.6)	<0.1%	0.1% (<0.1-0.3)
N	17,929	10,742	3,227	3,731	1,457	1,084	4,768	3,357
2013	0.2% (0.1-0.3)	0.1% (<0.1-0.2)	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.2)	0.1% (<0.1-0.3)	0.1% (<0.1-0.5)
N	16,167	4,465	3,112	2,629	1,454	1,865	8,516	4,473
2014	0.3% (0.2-0.5)	0.2% (0.1-0.7)	0.4% (0.1-1.2)	0.1% (<0.1-0.6)	0.6% (0.2-0.5)	0.4% (<0.1-1.5)	0.2% (0.1-0.5)	0.4% (0.1-0.9)
N	14,828	2,800	2,297	1,587	1,739	1,119	8,812	3,841
2015	0.4% (0.2-0.7)	0.4% (0.1-1.0)	0.4% (0.2-0.9)	0.2% (<0.1-1.4)	0.3% (0.1-0.7)	0.5% (0.2-1.1)	0.4% (0.2-0.7)	0.6% (0.3-1.2)
N	20,245	2,629	5,943	768	3,311	3,927	16,451	6,004

Table B.29. *Number of available PIT-tagged smolts (N) and annual minimum predation rates (95% credibility intervals) by American white pelicans nesting on Badger Island in McNary Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Lower Monumental Dam) and Upper Columbia River (UCR; based on detection at Rock Island Dam) during 2007-2019. NA indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates of predation rates. PIT tag deposition probabilities for white pelicans were lacking, so estimates represent minimum estimates of predation rates (see Methods).*

	Badger Island, McNary Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	<0.1%	<0.1%	NA	NA	0.3% (0.2-0.4)	0.1% (<0.1-0.3)
N	22,730	2,147			17,122	3,782
2008	<0.1%	<0.1%	NA	0.1% (<0.1-0.7)	0.2% (0.1-0.3)	0.1% (<0.1-0.2)
N	30,142	22,968		767	28,653	8,403
2009	0.2% (0.1-0.2)	0.1% (<0.1-0.1)	0.1% (<0.1-0.6)	<0.1%	0.3% (0.3-0.4)	0.3% (0.2-0.4)
N	20,753	27,198	738	2,651	52,220	8,025
2010	0.1% (<0.1-0.2)	0.1% (<0.1-0.1)	0.1% (<0.1-0.5)	0.1% (<0.1-0.8)	0.5% (0.3-0.7)	0.1% (<0.1-0.2)
N	5,790	28,067	929	566	7,913	8,382
2011	0.3% (0.2-0.5)	0.1% (0.1-0.2)	0.1% (<0.1-0.3)	0.4% (0.3-0.6)	0.4% (0.3-0.7)	0.1% (<0.1-0.2)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	0.1% (<0.1-0.3)	0.1% (<0.1-0.3)	0.2% (<0.1-1.2)	<0.1%	0.2% (0.1-1.3)	0.1% (<0.1-0.7)
N	41,258	24,772	1,812	2,884	25,841	6,845
2014	<0.1%	<0.1%	0.1% (<0.1-0.9)	0.1% (<0.1-0.4)	0.1% (0.1-0.3)	<0.1%
N	22,195	6,043	641	1,414	16,599	7,757
2015	0.1% (<0.1-0.2)	<0.1%	0.1% (<0.1-0.7)	NA	0.2% (0.1-0.6)	0.1% (<0.1-0.2)
N	4,471	1,393	766		2,400	7,222
2016	<0.1%	<0.1%	<0.1%	0.2% (<0.1-1.0)	<0.1%	<0.1%
N	38,633	5,461	1,956	522	20,729	7,003
2018	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	19,986	8,753	2,090	1,443	19,632	7,511
2019	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.4)	0.1% (<0.1-0.2)	<0.1%
N	18,757	7,501	1,885	1,675	28,813	4,401

Table B.30. Annual predation rates (95% credible interval) of ESA-listed PIT-tagged salmonids collected at Lower Granite, Little Goose, and Lower Monumental dams on the Snake River (SR) and released from barges downstream of Bonneville Dam by double-crested cormorants and Caspian terns nesting on East Sand Island in the Columbia River estuary during 2006-2018. NA denote insufficient sample sizes (< 500 PIT-tagged fish) for generating reliable estimates of predation rates. Predation rates greater than or equal to 2%, 5%, 10%, 15%, and 20% are highlighted in yellow, light orange, dark orange, light red, and red, respectively.

Year	**East Sand Island Caspian terns**				**East Sand Island Double-crested cormorants** [a]			
	SR Sp/Su Chinook	SR Fall Chinook	SR Sockeye	SR Steelhead	SR Sp/Su Chinook	SR Fall Chinook	SR Sockeye	SR Steelhead
2006	4.0% (3.2-5.6)	1.8% (1.4-2.6)	NA	22.7% (18.2-31.1)	4.9% (3.5-7.7)	1.7% (1.2-2.6)	NA	8.1% (5.9-12.8)
N	*78,532*	*48,661*		*70,988*	*78,532*	*48,661*		*70,988*
2007	2.3% (1.8-3.4)	3.0% (1.6-5.5)	NA	16.7% (13.4-24.5)	2.1% (1.4-3.3)	0.9% (0.1-3.4)	NA	3.9% (2.7-6.1)
N	*32,184*	*607*		*45,276*	*32,184*	*607*		*45,276*
2008	4.2% (3.4-5.9)	1.6% (1.2-2.2)	NA	18.7% (15.2-26.1)	3.9% (2.8-6.1)	5.3% (3.9-8.2)	NA	6.0% (4.3-9.1)
N	*95,267*	*48,039*		*65,097*	*95,267*	*48,039*		*65,097*
2009	4.3% (3.5-6.3)	1.8% (1.4-2.6)	1.1% (0.8-1.6)	16.1% (13.1-23.1)	6.8% (4.9-10.3)	5.8% (4.2-8.9)	8.9% (6.4-13.8)	10.7% (7.8-16.8)
N	*51,805*	*34,407*	*10,167*	*22,627*	*51,805*	*34,407*	*10,167*	*22,627*
2010	3.6% (2.9-5.1)	0.9% (0.7-1.3)	NA	14.9% (12.0-21.2)	4.7% (3.4-7.2)	5.3% (3.8-8.1)	NA	9.4% (6.8-14.3)
N	*40,996*	*46,843*		*32,904*	*40,996*	*46,843*		*32,904*
2011	1.9% (1.5-2.7)	0.5% (0.4-0.8)	0.4% (0.2-0.7)	9.2% (7.3-13.0)	3.6% (2.6-5.6)	4.0% (2.9-6.2)	8.6% (6.2-13.5)	6.5% (4.8-10.1)
N	*64,858*	*53,093*	*7,038*	*26,862*	*64,858*	*53,093*	*7,038*	*26,862*
2012	2.4% (1.8-3.4)	1.0% (0.8-1.5)	1.0% (0.7-1.5)	8.2% (6.5-12.0)	2.7% (1.9-4.2)	6.6% (4.8-10.3)	6.2% (4.4-9.7)	4.4% (3.1-6.9)
N	*38,963*	*41,537*	*14,013*	*30,542*	*38,963*	*41,537*	*14,013*	*30,542*
2013	1.1% (0.8-1.6)	1.3% (0.6-2.5)	0.5% (0.3-0.9)	8.9% (6.8-13.3)	4.0% (2.9-6.3)	9.7% (6.6-15.5)	1.3% (0.8-2.1)	4.4% (3.2-6.8)
N	*49,592*	*2,106*	*9,280*	*32,490*	*49,592*	*2,106*	*9,280*	*32,490*
2014	1.1% (0.8-1.6)	0.9% (0.4-2.0)	0.8% (0.4-1.3)	9.5% (7.4-13.4)	8.4% (6.2-13.2)	4.4% (2.6-7.6)	7.6% (5.4-12.0)	8.5% (6.2-13.1)
N	*66,759*	*1,539*	*5,839*	*33,327*	*66,759*	*1,539*	*5,839*	*33,327*
2015	1.3% (1.0-2.0)	2.1% (1.6-3.1)	2.4% (1.7-3.6)	8.9% (7.0-12.8)	16.1% (11.7-24.8)	5.3% (3.8-8.3)	7.8% (5.4-12.3)	9.3% (6.7-14.5)
N	*20,575*	*8,347*	*4,357*	*10,461*	*20,575*	*8,347*	*4,357*	*10,461*
2016	0.8% (0.6-1.1)	1.1% (0.8-1.6)	5.9% (4.2-8.7)	11.3% (8.9-16.2)	6.3% (4.5-9.7)	0.1% (0-0.2)	0.2% (0-0.7)	2.9% (2.0-4.5)
N	43,068	10,948	2,829	13,608	43,068	10,948	2,829	13,608
2017	0.8% (0.6-1.3)	0.3% (0.2-0.5)	2.3% (1.3-4.0)	6.4% (5.0-9.2)	0.4% (0.3-0.7)	0.2% (0.1-0.3)	1.4% (0.6-3.0)	0.9% (0.6-1.4)
N	32,395	13,205	1,589	28,964	32,395	13,205	1,589	28,964
2018	1.1% (0.9-1.7)	0.4% (0.3-0.7)	1.5% (1.1-2.4)	6.1% (4.8-8.8)	0.3% (0.2-0.4)	1.1% (1.3-3.0)	1.1% (0.7-1.8)	0.5% (0.3-0.8)
N	66,723	17,402	10,087	44,241	66,723	17,402	10,087	44,241

[a] Minimum estimates of predation rate due to multiple colony dispersal events (2016 and 2017) or the late arrival of double-crested cormorants on East Sand Island (2018) during the peak smolt out-migration period

APPENDIX C: East Sand Island Caspian Tern Predictive Predation Rate Analysis

C.1. Background

Passive integrated transponder (PIT) tags implanted in juvenile salmonids (*Oncorhynchus spp.)* and subsequently recovered on bird colonies have been used to estimate predation rates by Caspian terns (*Hydroprogne caspia*) breeding on East Sand Island in the Columbia River estuary for over 20 years (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2019). Although these studies provide valuable information on predation impacts and the efficacy of management actions aimed at reducing predation by terns, recoveries of PIT tags on East Sand Island are unlikely to continue in perpetuity due to the cost of collecting and analyzing PIT tag data. In lieu of PIT tag recovery, and in order to continue to monitor predation impacts by Caspian terns on East Sand Island into the future, it may be possible to predict predation rates based on estimates of the size (number of breeding adults) of the East Sand Island tern colony. For instance, previous studies have shown a statistically significant, linear relationship between annual measures of colony size and annual measures of predation rates on PIT-tagged steelhead (*O. mykiss*), with reductions in predation rates generally commensurate with reductions in colony size (Evans et al. 2016; see also *Chapter 1*). Previous research, however, also indicates that colony size is just one of several factors that explain variation in predation rates on juvenile salmonids by Caspian terns breeding on East Sand Island. For example, previous research has linked variation in predation rates on juvenile salmonids with colony productivity (number of young raised), the abundance or density of juvenile salmonids, Columbia River discharge and spill at hydroelectric dams, and large-scale climate indices like the North Pacific Gyre Oscillation and upwelling (Hostetter et al. 2012, Lyons et al. 2014, Evans et al. 2016; see also *Chapter 9* and *Chapter 10*). Results from these studies indicate a dynamic and complex set of environmental and biological factors influence smolt susceptibility to predation by Caspian terns in the estuary.

As part of this study, we developed and tested the accuracy and precision of various predictive predation rate models in order to present a "best" model that could potentially be used in lieu of PIT tag recovery efforts on East Sand Island in the future. These models entailed environmental and biological conditions (covariates) previously linked with variation in predation rates on juvenile salmonids by Caspian terns nesting on East Sand Island and focused on predation impacts to Snake River and Upper Columbia River steelhead; the salmonid species and populations most susceptible to Caspian tern predation and the species that was the primary impetus for the *Caspian Tern Management Plan in the Columbia River Estuary* (USFWS 2005). Covariates were selected *a priori* based on previous research and are readily available to managers and researchers via online datasets and repositories.

C.2. Methods

C.2.1 Environmental & Biological Covariates

We used data on East Sand Island Caspian tern colony size (2000–2018), steelhead abundance, river conditions (discharge and spill), and large-scale climate indices to determine which suite of factors best explained variation in PIT tag derived Snake River and Upper Columbia River steelhead predation rates. Caspian tern colony size was assessed as peak annual colony size (number of breeding pairs) and as the average weekly colony size (number of adults counted on-colony). Colony size data were obtained from Bird Research Northwest (http://www.birdresearchnw.org). Peak colony size is estimated via high resolution aerial photography and weekly colony size via an observation blind located adjacent to the colony (Adkins et al. 2014). The explanatory power of weekly measures of colony size must be weighed against the increased effort and cost needed to conduct more intensive monitoring during the breeding season to collect these data.

The steelhead smolt passage index was measured at Bonneville Dam and was used as a representation of steelhead abundance in the Columbia River estuary. Data were obtained from the Fish Passage Center (http://www.fpc.org). Percent spill at Bonneville Dam was also used to represent potential changes in smolt abundance and travel times in the estuary and data were obtained from the Columbia Basin Research website maintained by University of Washington (http://www.cbr.washington.edu/dart). River discharge, another potential factor regulating fish abundance and travel times in the estuary, was measured at Port Westward, near Quincy, OR, the closest publicly available gauge to East Sand Island (station previously named Beaver Army Terminal [BT], river km [Rkm] 87; obtained from the U.S. Geological Survey [https://waterdata.usgs.gov/nwis]).

Large-scale climate indices, which are believed to regulate the availability of alterative prey in the estuary, included the Multivariate El Niño/Southern Oscillation (ENSO) Index Version 1 (MEI.v1; https://www.esrl.noaa.gov/psd/enso/mei.old/table.html) and Version 2 (MEI.v2; https://www.esrl.noaa.gov/psd/enso/mei/), the Pacific Decadal Oscillation Index (PDO; http://research.jisao.washington.edu/pdo/PDO.latest), and the North Pacific Gyre Oscillation Index (NPGO; http://o3d.org/npgo/data/NPGO.txt). Three upwelling indices were also considered, the Bakun upwelling index (http://www.cbr.washington.edu/dart) and two newer indices, the Coastal Upwelling Transport Index (CUTI) and the Biologically Effective Upwelling Transport Index (BEUTI). Both newer upwelling indices are available online at http://mjacox.com/upwelling-indices/.

For environmental factors with multiple quantifying indices (e.g., MEI and upwelling), only one of the metrics was considered for any given model. Daily values of each covariate were summarized by weekly averages or totals, if considered in a model with weekly tern colony counts or repeated for the appropriate weeks if data were only available at larger time series (e.g., bi-weekly). Annual values were restricted to the steelhead smolt passage period at Bonneville Dam, roughly early-April through late-June, and considered only in models that included peak colony counts.

Finally, it should be noted that in addition to the covariates tested herein, several other variables have been linked to variation in East Sand Island Caspian predation rates but were not included in the study because they would require additional data collection and analysis, making their inclusion cost prohibitive given the goals of the study (i.e. to develop a model in lieu of field studies). For instance, measures of colony productivity (number of young raised), diet composition data on the availability of alternative prey (e.g., Pacific herring [*Clupea pallasii*] and Northern anchovy {*Engraulis mordax*}), data on fish condition, and other factors have all been linked to variation in predation rates (Hostetter et al. 2012, Lyons et al. 2014, Evans et al. 2016, Payton et al. 2016), but collection of these data would require field studies each year at cost similar to or greater than that of PIT tag recovery.

C.2.2 Statistical Models

To determine to what degree annual covariates explained variation in predation rates and could thus be used to predict predation rates, we let n_y represent the number of PIT-tagged steelhead detected at Bonneville Dam (BON) in year y, we let z_y represent the number of those fish subsequently consumed by Caspian terns on East Sand Island (ESI), and we let r_y represent the number of tags recovered on the ESI tern colony following the breeding season. The statistical model employed can be expressed as a state-space expansion of a logistic regression model, where $\mathrm{X_y}$ represents the values of covariates associated with year y and β_{annual} represents the effects of the covariates under consideration on predation rates. That is, we assume:

$$z_y \sim binomial\left[n_y, logit^{-1}\left(\mathrm{X_y}\beta_{annual}\right)\right] \text{ and}$$

$$r_y \sim binomial(z_y, \bar{\psi}_y * \phi_y);$$

where ϕ_y represents the probability of a tern depositing a consumed tag on the ESI colony (modelled using an informative prior; Hostetter et al. 2015) and $\bar{\psi}_y$ represents probability of recovering a deposited tag from ESI averaged across the year.

To expand this model to incorporate weekly covariates, we let $V_{y,w}$ represents the covariate values associated with week w of year y and introduce β_{weekly} to represent the effects of these covariate on predation rates.

$$z_{y,w} \sim binomial\left[n_{y,w}, logit^{-1}\left(\mathrm{X_y}\beta_{annual} + \mathrm{V_{y,w}}\beta_{weekly}\right)\right] \text{ and}$$

$$r_{y,w} \sim binomial(z_{y,w},\ \phi_y * \psi_{y,w});$$

where $\psi_{y,w}$ represents the probability that a consumed and deposited tag released from BON in week w of year y is subsequently recovered following the breeding. $\psi_{y,w}$ is informed by test tags intentionally sown on the colony throughout each year (see also Evans et al. 2012). That is, letting $m_{y,i}$ represent the number of test tags intentionally sown in week i of year j and $t_{y,i}$ represent the subset of those tags subsequently recovered, we assume

$$t_{y,i} \sim \text{binomial}(m_{y,i}, \psi_{y,i}) \text{ and}$$

$$\psi_{y,i} = logit^{-1}\left[a_y + b_y * (i - 22)\right].$$

Assuming the middle week of each year is approximately 22, we centered the estimation of the a and b parameters about that week. Covariate values were all normalized in an attempt to stabilize the geometry of posterior distribution and standardize the prior distributions employed for the covariate parameter vectors. As such, we assigned $\text{normal}(0,1)$ for all βs. The priors associated with the rest of the parameters are equivalent to those used in all avian predation rate models (see also Payton et al. 2019).

All models were implemented using the software STAN accessed through R version 3.1.2 (R Development Core Team 2014) using the rstan package (version 2.8.0; Stan Development Team 2015). To simulate random draws from the joint posterior distribution we ran four Hamiltonian Monte Carlo (HMC) Markov Chain processes. Each chain contained 4,000 adaptation iterations, followed by 4,000 posterior iterations. Posterior iterations were then thinned by a factor of 4. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013) and all accepted chains reported zero divergent transitions. Reported estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% CRI).

C.2.3 Model Evaluation

A comparison of predictive errors among competing models was performed using a Pareto-Smoothed-Importance-Sampling Leave-One-Out (PSIS-LOO) cross-validation technique (Vehtari et. al. 2017). Briefly, LOO partitions the data into subsets, and iteratively trains the model with all the data save for one subset and then evaluates the fit of the model to the left-out, or validation portion, of the data. For this dataset, the data was partitioned by years. PSIS allows for this process to be approximated with a less computationally involved algorithm. Model 'fit' was quantified by the expected log pointwise predictive density (ELPD); roughly equivalent to the log of the mean likelihood of recovering the observed number of PIT tags annually.

To assist in assessing the overwhelming number of models possible given the number of combinations of covariates, we prioritized assessing annual explanatory variables (covariates) and assumed the necessary inclusion of annual colony size. The distributions for each pseudo-residual from each year's annual predation rate were then plotted vs the explanatory values to check for non-linearity. All possible combinations of annual explanatory variables were assessed with the models ranked by fitness (based on ELPD). Only models in which all covariate parameters were significantly different from zero were considered in the second stage of the analysis. The second stage involved investigating the inclusion of the additional explanatory variables measured at finer temporal scales (i.e. monthly or weekly). All possible combinations were again evaluated with models ranked by fitness.

Predicted posterior distributions and the resulting predicted predation rates for each year were then constructed according to each of the best fitting models (informed by all years save the year in question; not approximated using PSIS). We then graphically compared predicted predation rates with previously calculated empirical predation rate estimates to evaluate any potential loss in precision, increase in bias, and other relative comparisons of model fit or performance.

C.3 Results

Sample sizes of PIT-tagged Snake River (SR) steelhead averaged 17,419 smolts annually, ranging from 1,747 to 60,823 tagged smolts. Over the 19-year time series, empirical predation rates or probabilities (proportion of available fish consumed) by Caspian terns nesting at East Sand Island averaged 0.171 (95% CRI = 0.157–0.186), ranging from 0.056 (0.038–0.075) to 0.353 (0.257–0.472). Sample sizes for Upper Columbia River (UCR) steelhead averaged 5,371 tagged smolts annually, ranging from just 73 to 27,905 tagged smolts. Empirical predation rates on UCR steelhead by East Sand Island terns averaged 0.147 (0.134–0.159), ranging from 0.068 (0.046–0.094) to 0.35 (0.204–0.519).

Predictive predation rate modelling results, those that explored the incorporation of covariate metrics, are provided below separately for SR and UCR steelhead, respectively:

C.3.1 Snake River Steelhead Trout

Other than measures of annual colony size, variation in SR steelhead predation rates were not found to have a statistically significant relationship with any other annual covariate metric. Explanatory variables measured at finer temporal scales, however, were found to significantly improve the model's explanation of weekly variation in SR steelhead predation rates. Aggregated by year, these variables were additionally effective in explaining some of the residual variation in annual predation rates left unexplained by measures of annual colony size alone. Several combinations of these covariates were found to approximately estimate annual predation rates, with little statistical difference in their explanatory power measured among them. The most common covariates in the 20 best fitting models were steelhead passage index and weekly colony counts (both included in 19 models) and MEI (included in 17 models). *Table*

C.1 depicts the best four SR steelhead models, with model effects provided in *C.6 Supplements* section, *Table C.S1*.

Table C1. Best four models for predicting annual predation rates in Snake River steelhead, with the associated difference in expected log predictive distribution from the best model, along with an estimate of the standard error of this difference.

	Model	ΔELPD	SE(Δ)
1	Colony Count + STHD Index + Discharge (@BT) + MEI + NPGO + Upwelling	0	0
2	Colony Count * STHD Index + Discharge (@BT) + MEI + Upwelling	-0.30	0.45
3	Colony Count + STHD Index + Spill (@BON) + MEI + NPGO + Upwelling	-0.73	0.94
4	Colony Count * STHD Index + Spill (@BON) + MEI + PDO + NPGO + Upwelling	-0.88	0.73
0	(annual colony size only)	-14.38	6.44

We estimate that annual colony size alone was able to explain 52.0% (25.1–72.0) of the variation in annual SR steelhead predation rates. On average, the best annual model overestimated predation nominally by 0.01 (-0.18–0.06) or by a factor of 1.22 (0.5–1.75). The predictive model performed best in 2014, where we estimate it only overestimated the empirical rate by <0.01 (-0.09–0.08), equal to underestimating the empirical rate by a factor of 1.02 (0.41–1.84). At its worst, in 2001, the predicted predation rate was nominally 0.18 (-0.03–0.38) less than the empirical rate, equal to underestimating the empirical rate by a factor of 0.48 (0.08–0.97). The model's predicted predation rate was nominally 0.06 (-0.01–0.15) greater than the empirical rate in 2016, equal to overestimating the empirical rate by a factor of 1.63 (0.81–2.91). Differences in predictive annual models and PIT tag informed (i.e. empirical) predation rates in SR steelhead are provided in *Figure C.1*.

After introducing weekly covariates, the accuracy of the predictive estimates still varied widely from year to year without a consistent over- or under-bias apparent. On average, the best weekly-based model overestimated predation nominally by 0.01 (-0.12–0.08) or by a factor of 1.17 (0.41–2.13). The predictive model performed best in 2002, where we estimate it only underestimated the empirical rate by <0.01 (-0.09–0.09), equal to underestimating the empirical rate by a factor of 1.00 (0.45–1.70). At its worst, the model predicted predation rate was nominally 0.12 (0.03–0.21) less than the empirical rate in 2003, equal to underestimating the empirical rate by a factor of 0.39 (0.2–0.74). The model's predicted predation rate was nominally 0.08 (-0.01–0.18) greater than the empirical rate 2000, equal to overestimating the empirical rate by a factor of 1.63 (0.88–2.44). Differences in predictive weekly models and PIT tag informed predation rates in SR steelhead are provided in *Figure C.1*.

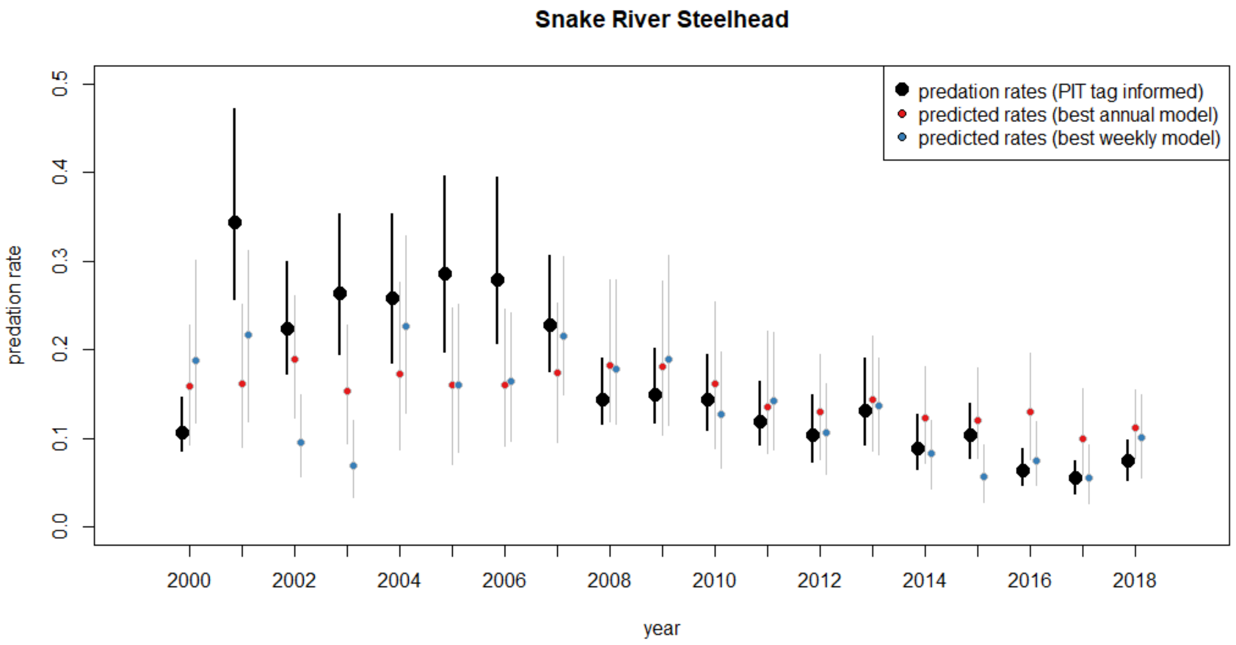

Figure C.1. Predation rates on Snake River steelhead as estimated from recovered PIT tags and predicted predation rates based on the best fitting model employing annually covariates and the best fitting model employing both annually and weekly covariates. Dots represent point estimates with the associated vertical lines representing 95% credible intervals. The components of the models are enumerated in Table C.1 above.

C.3.2 Upper Columbia River Steelhead Trout

Analogous to results in SR steelhead, other than annual colony size, variation in annual predation rates on UCR steelhead were not found to have a statistically significant relationship with any of the other annual covariate metrics evaluated. Several explanatory variables measured at temporally finer scales, however, were found to significantly improve the model's explanation of variation in predation rates. Aggregated over each year, these variables were additionally effective in explaining some of the residual variation in annual predation rates left unexplained by measures of annual colony size alone. Several combinations of these covariates were found to approximately estimate annual predation rates, with little statistical difference in their explanatory power measured among them. The most common covariates in the 20 best fitting models were discharge (as measured at Port Westward, near Quincy, Oregon; included in 17 models), MEI (included in 15 models), and spill at BON (included in 12 models). *Table C.2* depicts the best four UCR steelhead models, with model effects provided in *C.6 Supplements, Table C.S2*.

Table C.2. Table describing the best four models for predicting annual Upper Columbia River steelhead predation rates, with the associated difference in expected log predictive distribution from the best model along with an estimate of the standard error of this difference.

	Model	ΔELPD	SE(Δ)
1	Colony Count * STHD Index + Discharge (@BT) + MEIv2 + PDO + NPGO + BEUTI	0	0
2	STHD Index + Discharge (@BT) + MEI + PDO + Upwelling	-0.44	1.13
3	STHD Index + Discharge (@BT) + Spill (@BON) + MEI + PDO + NPGO + Upwelling	-0.52	1.04
4	STHD Index + Discharge (@BT) + Spill (@BON) + MEI	-0.56	1.1
0	(annual colony size only)	-12.52	1.11

We estimated that measures of annual colony size alone were able to explain 70.6% (38.4–94.4) of the variation in annual predation rates in UCR steelhead. On average, the best annual model overestimated predation nominally by 0.01 (-0.19–0.06) or by a factor of 1.23 (0.49–1.82). This predictive model performed best in 2014, where it underestimated the empirical rate by <0.01 (-0.07–0.09), equal to underestimating the empirical rate by a factor of 0.98 (0.41–1.71). At its worst, in 2001, the predicted predation rate was nominally 0.19 (0–0.37) less than the empirical rate, equal to underestimating the empirical rate by a factor of 0.46 (0.14–0.93). The model's predicted predation rate was nominally 0.06 (-0.03–0.16) greater than the empirical rate 2013, equal to overestimating the empirical rate by a factor of 1.65 (0.57–2.84). Differences in predictive annual models and PIT tag informed predation rates in UCR steelhead are provided in *Figure C.2* above.

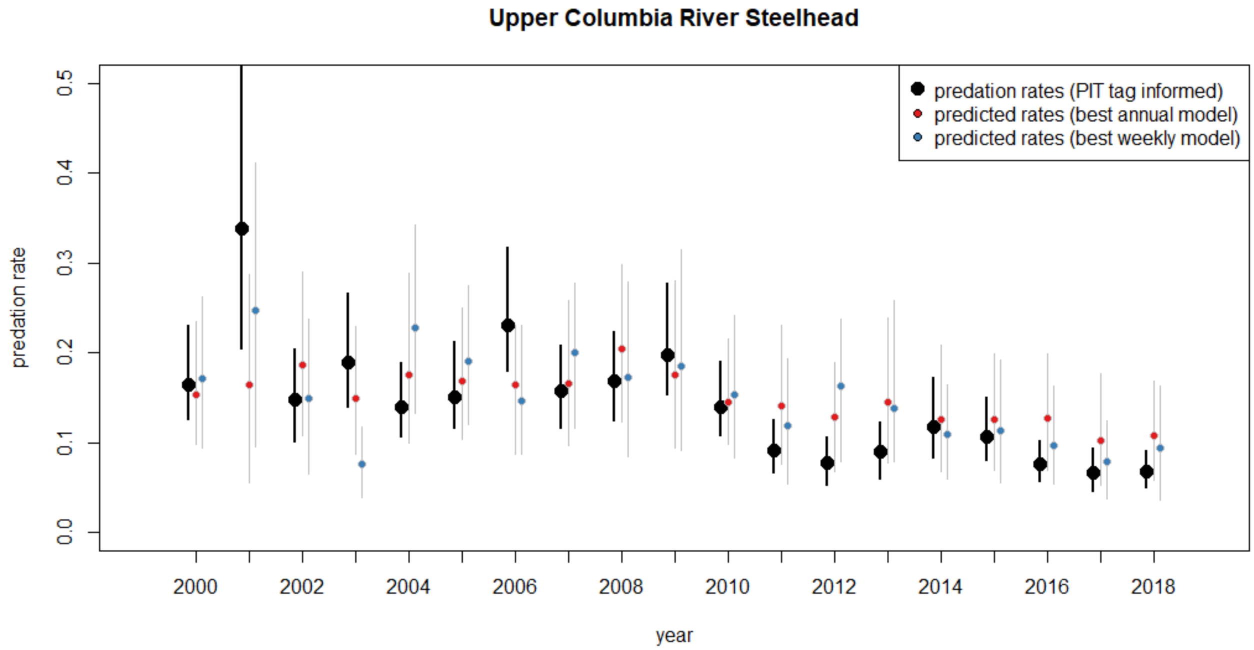

Figure C.2. Predation rates on Upper Columbia River steelhead as estimated from recovered PIT tags and predicted predation rates based on the best fitting model employing annually covariates and the best fitting model employing both annually and weekly covariates. Dots represent point estimates with the associated vertical lines representing 95% credible intervals. The components of the models are enumerated in Table C.2 above.

After introducing weekly covariates, the accuracy of the predictive estimates still varied widely from year to year without a consistent over- or under-bias apparent. On average, the best weekly-based model overestimated predation nominally by 0.01 (-0.16–0.10) or by a factor of 1.27 (0.57–2.19). This predictive model performed best in 2002, where it only underestimated the empirical rate by -0.01 (-0.11–0.09), equal to underestimating the empirical rate by a factor of 0.97 (0.59–1.45). At its worst, in 2003, predicted predation rate was nominally 0.15 (0–0.31) less than the empirical rate, equal to underestimating the empirical rate by a factor of 0.56 (0.29–0.94]). The weekly model's predicted predation rate was nominally 0.09 (0–0.22) greater than the empirical rate in 2012, equal to overestimating the empirical rate by a factor of 1.60 (0.97–2.64). Differences in predictive weekly models and PIT tag informed predation rates in UCR steelhead are provided in *Figure C.2* above.

Programmatic code and annotated instructions to produce predicted predation rate estimates from the top SR and UCR annual and weekly models were provided to the U.S. Army Corps. Average annual predation rate estimates based on measures of peak colony size alone can simply be derived by multiplying measures of colony size by the per capita (per breeding pair) predation rate, which were 0.000021 (0.000020–0.000023) and 0.000018 (0.000016–0.000020) for SR and UCR steelhead, respectively.

C.4 Conclusions

Results of this study indicate that models that incorporate estimates of the size of the East Sand Island Caspian tern colony could be used to predict predation rates on SR and UCR steelhead. The inclusion of additional explanatory variables or covariates, particularly steelhead abundance, river discharge, and ocean indices, were able to improve the predictability of these models by explaining additional variation in predation rates. Furthermore, models that incorporated intra-annual (weekly) covariate values often explained more variation in predation rates than models that relied solely on annual values. An evaluation of model performance, however, indicated there were often statistically significant differences in predicted predation rates (covariate informed estimates) compared with empirically derived estimates of predation (i.e. PIT tag informed estimates). In some cases, predicted predation rates underestimated or overestimated actual predation rates by factor of 0.39 or 1.63, respectively, depending on the steelhead population and year. Predicted estimates were also less precise (i.e. wider credible intervals) compared with PIT tag informed estimates of predation. For instance, even when predation rates were relatively low, like those on UCR steelhead in 2018, 95% CRI associated with predicted predation rates ranged widely from 0.04 (lower bound) to 0.17 (upper bound) but were just 0.06 (lower bound) to 0.09 (upper bound) in PIT tag informed estimates of predation.

Results of this and other studies indicate that colony size, although an important factor, is not the only factor associated with variation in steelhead predation rates by Caspian terns on East

Sand Island (Lyons et al. 2014, Evans et al. 2016; see also *Chapter 10*). By itself, an annual measure of colony size for East Sand Island terns explained 52.0% and 70.6% of observed variation in annual SR and UCR steelhead predation rates, respectively. Evans et al. (2016) concluded that a dynamic and complex suite of biotic and abiotic factors were associated with juvenile salmonid susceptibility to Caspian tern predation in the Columbia river estuary. In the present study, several similar environmental factors were found to be important in predicting Caspian tern predation rates on both Snake River and Upper Columbia River steelhead. The most common shared variables in top models were steelhead index, river discharge, upwelling and MEI. Previous studies indicate there is a strong relationship between the abundance of juvenile salmonids and Caspian tern predation rates, with predation rates decreasing as the number of available smolts increases (Hostetter et al. 2012, Evans et al. 2016; see also *Chapter 9*). Previous studies also indicate that river flows are related to smolt travel times (Zabel 2002, Harnish et al. 2012) and consequently the amount of time fish are susceptible predation by Caspian terns (Payton et al. 2016). Coastal upwelling presumably influences the abundance of marine forage fishes, an alternative food source for Caspian terns on East Sand Island (Lyons 2010, Lyons et al. 2014). Fluctuations in large scale climate indices, like the MEI, may also create conditions that influence the abundance and availability of marine forage fish in the estuary (Peterson and Burke 2013). Irrespective of the influence of these covariates, however, differences in expected log predictive densities among the top fitting models were all relatively equivalent statistically. This suggest there was no single preferred or 'best' model and/or that the environmental variables included in the study were rough approximations or indices of the underlying mechanisms that regulate steelhead susceptibility to Caspian tern predation in the Columbia River estuary.

Although predicted predation rates were often inaccurate and imprecise relative to PIT tag informed estimates, predicted estimates generally followed inter-annual trends in predation by Caspian terns on East Sand Island. As such, the predicted predation rate model developed herein could serve as an alternative, albeit gross, method of estimating predation rates in lieu of PIT tag recovery on East Sand Island. There are several assumptions and caveats, however, that should be carefully considered before using predictive models to estimate predation rates. First, future covariate values should remain within the range of observed values used in the present study; extrapolating predictions to conditions outside of these limits could bias predicted predation rates to an unknown degree. Secondly, the accuracy of colony counts depends on the frequency of colony monitoring efforts on East Sand Island, hence future colony monitoring efforts should follow previously established methods. Thirdly, there are several other factors that have been linked to variation in predation rates, including smolt travel times, fish condition, river turbidity and salinity, and colony productivity. These factors were not considered herein due the impracticality of collecting these data each year. Substantial changes in these other factors in the future, could influence the accuracy of predicted predation rates to an unknown degree. Finally, predicted models apply only to SR and UCR steelhead and to Caspian terns breeding on East Sand Island; use of these models in other salmonid species and populations and at other colonies (e.g., East Sand Island double-crested cormorants) would likely result in substantially biased estimates of predation.

In summary, the value of using predictive models to estimate East Sand Island Caspian tern predation rates on steelhead in lieu of PIT tag derived estimates will depend on the degree to which accurate and precise estimates of predation are needed or required by resource managers and researchers. Predicted estimates based on colony size and other covariates were significantly different than PIT tag informed estimates in many, but not all years, and predicted estimates were less precise. None-the-less, the use of these predicted models could provide an alternative method to estimate predation rates if PIT tag recovery efforts are not possible or are cost-prohibited on an annual basis.

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C.6 Supplements

Table C.S1. Effects table for Snake River steelhead model 1, model 2, and model 0. Values for each explanatory variable were normalized (i.e. [value-mean]/se) for easier fit and to assist in comparisons of relative importance.

Model	Effect	Estimate	95% CRI	p
1	(intercept)	-1.61	(-1.72, -1.46)	<0.001
	Annual Colony Size	0.28	(0.15, 0.34)	<0.001
	Weekly Colony Size	0.15	(0.02, 0.32)	0.011
	Weekly STHD Index	-0.15	(-0.22, -0.07)	<0.001
	Weekly Discharge (@ BT)	-0.10	(-0.20, 0.01)	0.022
	Weekly MEI	-0.19	(-0.33, -0.06)	0.002
	Weekly NPGO	-0.01	(-0.15, 0.14)	0.217
	Weekly Upwelling	0.10	(-0.01, 0.19)	0.014
2				
	(intercept)	-1.61	(-1.74, -1.47)	<0.001
	Annual Colony Size	0.27	(0.17, 0.34)	<0.001
	Weekly Colony Size	0.15	(0.01, 0.31)	0.009
	Weekly STHD Index	-0.16	(-0.24, -0.06)	<0.001
	Colony Size*STHD Index	-0.01	(-0.10, 0.08)	0.188
	Weekly Discharge (@ BT)	-0.11	(-0.22, 0.01)	0.015
	Weekly MEI	-0.19	(-0.33, -0.07)	0.001
	Weekly Upwelling	0.08	(-0.01, 0.17)	0.016
0	(intercept)	-1.59	(-1.77, -1.48)	<0.001
	Annual Colony Size	0.22	(0.17, 0.33)	<0.001

Table C.S2. Effects table for Upper Columbia River steelhead model 1, model 2, and model 0. Values for each explanatory variable were normalized for easier fit and to assist in comparisons of relative importance.

Model	Effect	Estimate	95% CRI	p
1	(intercept)	-1.69	(-1.86, -1.52)	<0.001
	Annual Colony Size	0.20	(0.04, 0.32)	0.005
	Weekly Colony Size	0.08	(-0.11, 0.26)	0.108
	Weekly STHD Index	-0.29	(-0.38, -0.18)	<0.001
	Colony Size*STHD Index	0.09	(-0.02, 0.16)	0.015
	Weekly Discharge (@ BT)	-0.20	(-0.34, -0.07)	0.001
	Weekly MEI v2	0.13	(0, 0.31)	0.013
	Weekly PDO	-0.11	(-0.28, 0.05)	0.049
	Weekly NPGO	0.04	(-0.13, 0.23)	0.148
	Weekly BEUTI	-0.10	(-0.21, 0.02)	0.020
2				
	(intercept)	-1.64	(-1.77, -1.48)	<0.001
	Annual Colony Size	0.24	(0.13, 0.33)	<0.001
	Weekly STHD Index	-0.22	(-0.30, -0.13)	<0.001
	Weekly Discharge (@ BT)	-0.20	(-0.32, -0.05)	0.002
	Weekly MEI	-0.17	(-0.34, -0.02)	0.006
	Weekly PDO	0.02	(-0.14, 0.18)	0.222
	Weekly Upwelling	0.12	(0.01, 0.23)	0.009
0	(intercept)	-1.64	(-1.77, -1.48)	<0.001
	Annual Colony Size	0.25	(0.18, 0.3)	<0.001

APPENDIX D: Annotated Bibliography

An annotated bibliography of key references cited in the Avian Predation Synthesis Report have been compiled in a free open-source reference manager program call Zotero (https://www.zotero.org/). This interactive program can be accessed online or downloaded and synced with the "Avian Predation" library that has being created. The Avian Predation library contains literature pertinent to avian predation concerns in the Columbia River basin and additional references from the Avian Predation Synthesis Report.

The URL to access the Avian Predation library is: https://www.zotero.org/groups/2464268/avian_predation/library. This library is publicly viewable and searchable, but one must create a Zotero account and join the Avian Predation group to have access to the stored pdfs. Each entry includes the standard citation information, including the abstract or executive summary. Each entry also includes a link to a PDF of the publication, if publicly available online. If the pdf is not publicly available, the publisher's URL of where the publication can be accessed online is provided. To search the library, use the drop-down menu of the search bar to select "Title, Creator, Year + Full-Text Content", and then enter the key words of interest into the search bar. The entire library can be exported as a bibliography and imported into Microsoft Word. Individual citations may also be exported as needed. A pdf (printed) version of the annotated bibliography will also be provided to the U.S. Army Corps for archival purposes as part of the Final Synthesis Report.

If you should have questions, or need further assistance with the annotated bibliography, please contact Ken Collis at ken@realtimeresearch.com