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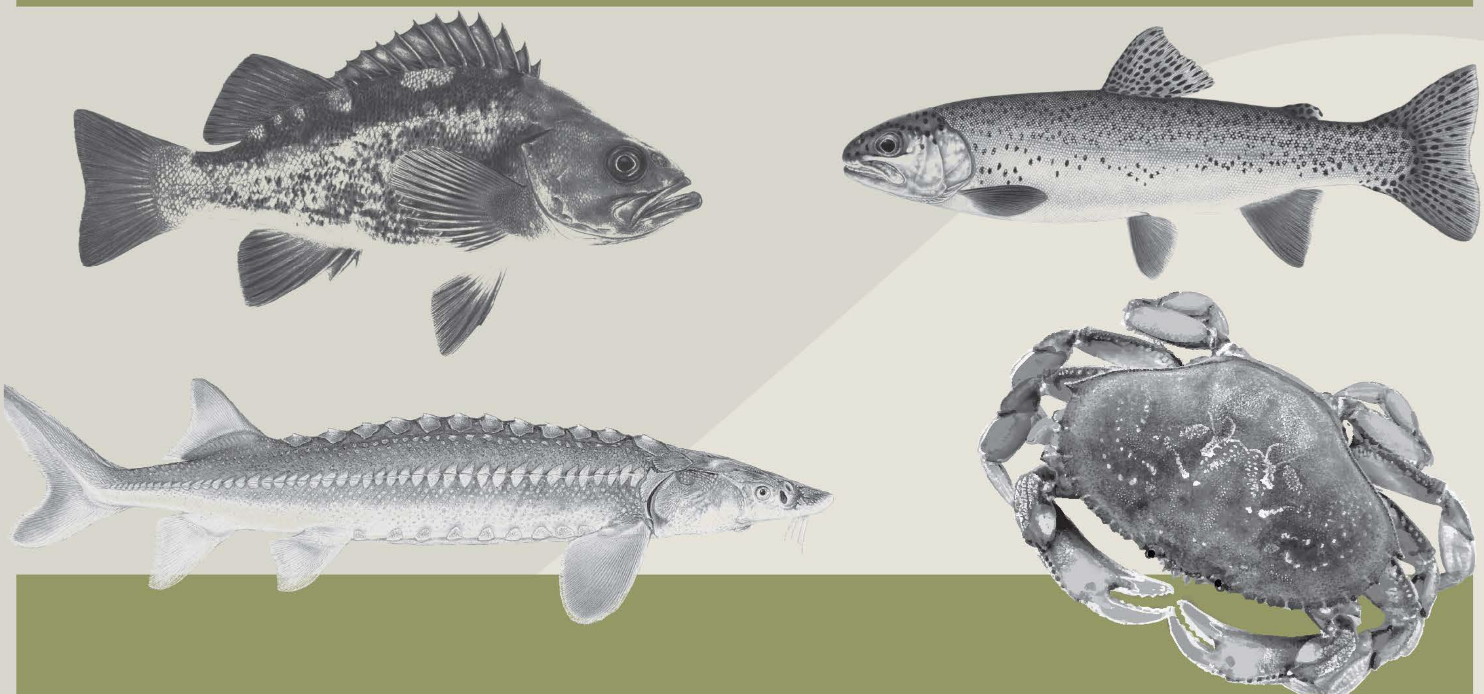

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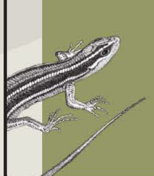

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A Status Assessment of the Double-crested Cormorant (*Nannopterum auritum*) in the Columbia River Estuary and Implications for Predation on Outmigrating Juvenile Salmonids

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Cover: Adult double-crested cormorant photographed on the Astoria-Megler Bridge colony in June, 2019. This individual was banded as a chick on the East Sand Island colony. Photograph taken by Tim Lawes, Oregon State University.

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Executive Summary

The double-crested cormorant (*Nannopterum auritum*) is a locally common, piscivorous (fish-eating) bird native to the Pacific Northwest whose abundance in the Columbia River estuary has grown substantially since the early 1980s. This increase has been part of a broader trend across North America following the species' protection under the Migratory Bird Treaty Act and the effective ban on the use of DDT in the United States, both occurring in 1972. Although not an original cause of declines in salmon and steelhead (genus *Oncorhynchus*; collectively, salmonids) in the Columbia River basin, predation of juvenile fish by double-crested cormorants is now one of many factors that potentially impedes recovery of basin salmonids listed under the federal Endangered Species Act (ESA). To reduce predation on ESA-listed juvenile salmonids, the U.S. Army Corps of Engineers (Corps) implemented a management plan during 2015–2020 to reduce double-crested cormorant abundance on East Sand Island, a human-modified island near the mouth of the Columbia River estuary that supported about 97% of all nesting pairs within the estuary during 2004–2014. Concurrent with implementation of the management plan on East Sand Island (ESI management plan), a major shift occurred in the distribution of double-crested cormorants within the estuary. This shift likely diminished the plan's intended survival benefit for juvenile salmonids, and may even have increased double-crested cormorant predation rates on juvenile salmonids relative to the period prior to management.

Status of the Double-crested Cormorant in the Columbia River Estuary

The abundance and distribution of double-crested cormorants breeding within the Columbia River estuary have varied substantially in recent decades. Estuary-wide abundance grew from 131 breeding pairs when the estuary was first surveyed in 1979–1980 to an average 13,337 breeding pairs during 2004–2014, the period of peak double-crested cormorant abundance in the estuary. However, during implementation of the ESI management plan during 2015–2020 (management period), the number of double-crested cormorants breeding on East Sand Island declined substantially, from an average 12,982 breeding pairs during 2004–2014 to an average 2,116 pairs during 2018–2020. No sustained breeding activity was observed on East Sand Island in 2020. Concurrent with management on East Sand Island, the colony located on the Astoria-Megler Bridge, located 12 km upstream of East Sand Island, grew from 333 breeding pairs in 2014 to 5,081 pairs in 2020. The aggregate total at other estuary colony sites grew from 414 pairs to 843 pairs during this same period. Overall, the estimated abundance of double-crested cormorants across the Columbia River estuary in 2020 was 5,924 breeding pairs, reflecting a decline of about 56% compared to the 2004–2014 peak abundance period. Along with this decline, the breeding distribution of double-crested cormorants shifted from the marine zone in the lower estuary, where East Sand Island is located, to colony sites farther upriver, where salmonids constitute a much larger proportion of the double-crested cormorant diet. Only about 3% of estuary-wide breeding abundance occurred upriver of the marine zone during the 2004–2014 peak abundance period, compared to over 99% in 2020. The available evidence strongly suggests the recent redistribution of double-crested cormorants within the estuary was primarily a result of recruitment of individuals displaced from the East Sand Island colony during implementation of the ESI management plan. Further, the weight of evidence suggests implementation of the ESI management plan was a pre-eminent causal factor in the decline and collapse of the East Sand Island colony, acting directly by reducing double-crested cormorant fidelity to East Sand Island, and indirectly by decreasing the resilience of double-crested cormorants to colony disturbances by bald eagles and other potential stressors, and by failing to include a clear plan for adaptive management to deter emigration from East Sand Island to the nearby Astoria-Megler Bridge colony.

The future status of double-crested cormorants in the Columbia River estuary will likely be tied closely to availability of human-built or human-altered breeding habitat (e.g. modified or constructed islands, bridges, and navigation markers), where all known estuary colonies have been located. The Astoria-Megler Bridge colony is currently unmanaged and is capable of supporting thousands of breeding pairs. In contrast, the other two colonies with comparably large amounts of breeding habitat, Rice Island and East Sand Island, are anticipated to be managed by the Corps under Biological Opinions written by the National Marine Fisheries Service to address channel maintenance and hydrosystem operation. Under current Biological Opinions, no breeding will be allowed on Rice Island, and no more than 5,939 breeding pairs will be allowed to nest on East Sand Island. Most of the remaining historical colony sites in the estuary are currently unmanaged, but these sites either seem to offer limited nesting habitat or would likely be constrained in size by available food, which is more abundant near the mouth of the estuary than farther upriver.

Implications for Salmonid Recovery

The recent redistribution of double-crested cormorants within the Columbia River estuary has complicated efforts by managers to assess whether the ESI management plan has improved survival of juvenile salmonids. In most cases, predation impacts are currently estimated by the relocation of passive integrated transponder (PIT) tags from previously marked fish that are deposited by double-crested cormorants on their colonies. Within the estuary, most PIT tag relocation effort has been restricted to the East Sand Island colony, where the vast majority of double-crested cormorants has historically nested. However, because most double-crested cormorants breeding in the Columbia River estuary now occur upriver of East Sand Island at other colonies (non-ESI colonies), where PIT tags are not consistently recovered, it is necessary to estimate predation impacts using alternative methods.

I estimated the potential estuary-wide impact of double-crested cormorant predation on juvenile steelhead (*Oncorhynchus mykiss*), a salmonid species vulnerable to double-crested cormorant predation in the Columbia River estuary. I created a simple model using available data that quantified susceptibility of juvenile steelhead to predation by double-crested cormorants nesting within three different estuary salinity zones. Based on previous work, I assumed double-crested cormorants breeding in the freshwater and mixing zones of the estuary consumed 8.6 and 4.3 times more steelhead per capita, respectively, than individuals breeding in the marine zone, where East Sand Island is located. Results indicated that individuals breeding within the freshwater and mixing zones accounted for an average 17% of annual estuary-wide predation by double-crested cormorants (estuary-wide predation) during the 2004–2014 peak abundance period; however, they accounted for an estimated average 73% of estuary-wide predation during the 2015–2020 management period, and >99% of estuary-wide predation in 2020. Associated with this spatial shift in predation, the magnitude of estuary-wide double-crested cormorant predation increased substantially. Estuary-wide predation in 2020 was equivalent to 28,843 pairs on East Sand Island, about 182% of predation during the 2004–2014 peak abundance period.

Conclusions

As a result of the recent shift in double-crested cormorant distribution to colony sites upriver of East Sand Island, estuary-wide predation of ESA-listed salmonids may be equivalent or higher compared to the period prior to implementation of the ESI management plan. Although empirical data are limited, it is likely most double-crested cormorant predation on juvenile salmonids within the Columbia River estuary is now associated with the Astoria-Megler Bridge colony. Reducing or eliminating use of this colony would therefore reduce estuary-wide double-crested cormorant predation considerably, as long as such management does not cause redistribution of double-crested cormorants to new estuary colony sites. However, potential future management of double-crested cormorants in the Columbia River estuary could affect the status of the species' population in the western conterminous United States and southern Canada (western population), which has already declined by perhaps 38% since management began on East Sand Island. Conservation planning for the western population of double-crested cormorants would benefit from an updated regional status assessment, an action previously recommended by researchers to occur following management on East Sand Island.

Section 1: Introduction

Predation of juvenile fish by colonial waterbirds is one of many factors that potentially impedes recovery of Columbia River salmon and steelhead (genus *Oncorhynchus*; collectively, salmonids) listed under the federal Endangered Species Act (ESA). The double-crested cormorant (*Nannopterum auritum*) is a species whose abundance in the Columbia River estuary has increased considerably since the late 1970s, and its predation of ESA-listed salmonids in the estuary is a concern for fisheries managers. In recent years double-crested cormorants nesting in the Columbia River estuary have been estimated to annually consume up to 17% of available juvenile salmonids from individual ESA-listed runs originating upstream of Bonneville Dam (Evans et al. 2019), the lowermost dam in the Columbia River basin, and up to 51% of individual ESA-listed runs associated with the lower Columbia River (Roby et al. 2021), which includes the reach of river near Bonneville Dam to the mouth of the river. Although predation by double-crested cormorants and other avian predators was not an original cause of salmonid declines in the Columbia River basin (NRC 1996), current levels of avian predation may reduce the likelihood of timely recovery for depressed salmonid runs, toward which billions of dollars have been spent in recent decades (NWPCC 2019).

To address potential impacts of double-crested cormorants to ESA-listed salmonids, the National Marine Fisheries Service (NMFS) called for limiting double-crested cormorant predation under several Biological Opinions associated with operation of the Federal Columbia River Power System (FCRPS; NMFS 2008, 2010, 2014), which comprises a series of dams located throughout the Columbia River basin. The 2014 Supplemental Biological Opinion (NMFS 2014) stated in Reasonable and Prudent Alternative (RPA) 46:

> *The FCRPS Action Agencies will develop a cormorant management plan (including necessary monitoring and research) and implement warranted actions to reduce cormorant predation in the estuary to Base Period levels (no more than 5,380 to 5,939 nesting pairs on East Sand Island).*

To fulfill the requirements of RPA 46, the U.S. Army Corps of Engineers (Corps) developed a plan to manage the double-crested cormorant breeding colony on East Sand Island, a Corps-administered site that until recently supported most breeding double-crested cormorants in the Columbia River estuary. This plan, entitled *Double-crested Cormorant Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (ESI management plan; USACE 2015), specified a reduction in the number of double-crested cormorants nesting on East Sand Island from a pre-management baseline of 12,917 breeding pairs to 5,380–5,939 breeding pairs. Although the most recent Biological Opinion does not include a provision that enjoins the FCRPS agencies to meet the requirements of RPA 46, the FCRPS agencies proposed to continue actions described in the ESI management plan (NMFS 2020).

Following implementation of the ESI management plan beginning in 2015, the abundance of double-crested cormorants dramatically declined at the East Sand Island colony, but increased concurrently at the nearby Astoria-Megler Bridge colony, located 12 km upriver of East Sand Island, as well as other estuary colonies. This shift in double-crested cormorant distribution has likely offset the ESI management plan's anticipated benefit to ESA-listed salmonids. However, a comprehensive characterization of this distribution shift is currently lacking. Similarly, potential changes in double-crested cormorant predation impacts related to this distribution shift are unclear.

The intent of this report is to clarify the status of the double-crested cormorant within the Columbia River estuary and the associated impacts to salmonids in the basin. It provides a brief background of factors related to double-crested cormorant colony-site selection and diet; summarizes historical and recent changes in distribution and abundance across the Columbia River estuary; investigates the potential causes of recent changes; assesses current and future avian impacts to survival of juvenile salmonids; and concludes with considerations for future management.

Section 2: Background

The double-crested cormorant is a large (1.2–2.5 kg), piscivorous bird that typically breeds colonially. The species is native to much of North America, including the region west of the Continental Divide in the conterminous United States and southern Canada (Wires and Cuthbert 2006); individuals breeding within this region constitute the western population (Adkins et al. 2014). Breeding double-crested cormorants have high energy demands and thus require access to reliable, abundant sources of food. An adult double-crested cormorant consumes between about 350 g to 800 g of food per day during the breeding season (Lyons 2010, Göktepe et al. 2012), in addition to food provided to growing chicks. Consequently, large colonies can consume several thousand tons of fish annually, and thus are often associated with marine or estuary habitats because of the high abundance of forage fish these habitats support. Double-crested cormorants are considered generalist foragers, consuming prey species roughly in proportion to their abundance rather than preferring specific prey types (Dorr et al. 2021). Consistent with this, recent work indicates the diet of double-crested cormorants in regional estuaries tends to be composed largely of marine or estuarine fish species (Lawes et al. 2021, Oregon Department of Fish and Wildlife [ODFW] unpubl. data), which tend to predominate in these systems. However, freshwater-derived fish such as outmigrating salmonids can be consumed in large numbers in some situations and appear particularly vulnerable when they are highly accessible relative to other prey types (Lyons et al. 2014a, Weitkamp et al. 2016).

The Columbia River estuary, considered here as the tidally influenced reach from the river mouth to Bonneville Dam (river km [RKM] 234; Simenstad et al. 2011), supports a high abundance of fish prey (Bottom and Jones 1990), which is a major factor in its attractiveness to double-crested cormorants (Peck-Richardson et al. 2018). The composition and abundance of prey across the estuary varies along a salinity gradient (Bottom and Jones 1990), with average salinity highest at the mouth of the estuary and decreasing progressively in direct relation to distance from the river mouth until about RKM 55, where the influence of ocean-derived salinity ends (Simenstad et al. 1990). While this gradient is continuous and varies with respect to season, rainfall, tides, and other factors, the estuary can be thought of as having three salinity zones for purposes of comparison: marine, mixing, and freshwater zones (Figures 1a, 1b; Bottom and Jones 1990, Simenstad et al. 1990, Anderson et al. 2004). In general, density and overall abundance of potential prey fishes is considerably higher within the marine and mixing zones compared with the freshwater zone (Bottom and Jones 1990). This pattern of prey abundance is consistent with an observed relationship between abundance of nesting double-crested cormorants and the salinity zone in which their colonies are located. The maximum observed sizes for double-crested cormorant colonies within the marine, mixing, and freshwater zones are 14,916 breeding pairs (East Sand Island in 2013), 5,081 breeding pairs (Astoria-Megler Bridge in 2020), and 1,444 breeding pairs (Rice Island in 1992), respectively. Importantly, the pattern of high prey abundance within the marine and mixing zones appears far less pronounced during April–June compared with July–September (Bottom and Jones 1990), reflecting the overall reduced abundance of marine-oriented species during seasonally high river flows during spring (Weitkamp et al. 2012). During April–June, fish density in the lower freshwater zone may be nearly equivalent to that of the marine zone, and about 50% of that in the mixing zone (Bottom and Jones 1990). This suggests avian predators may be particularly attracted to the upper to middle portion of the estuary during spring, a conclusion consistent with available research findings (Lyons et al. 2007).

Like colonial birds in general, double-crested cormorants are constrained by the need to provision their young with food captured some distance from their nesting area. Thus, they are motivated to minimize energy and time expenditure by foraging close to their colony sites, all other things being equal (Schoener 1971). Further, as a presumed consequence of energetically costly flight for birds with high wing-loading (Pennycuick 1975), such as cormorant species, the observed average foraging range for double-crested cormorants is relatively low compared to other piscivorous waterbirds. The average commuting distance for double-crested cormorants breeding on East Sand Island has been estimated to be about 9 and 16 km for females and males, respectively (Anderson et al. 2004). It is not surprising, therefore, that the double-crested cormorant diet at colonies along the Columbia River estuary's salinity gradient appears to align with the spatial distribution of fish species along this gradient, a pattern that is shared by other piscivorous birds that breed in the estuary. Avian predators associated with colonies located in the estuary's freshwater zone consume far more juvenile salmonids as a proportion of their diet compared with those on East Sand Island, located in the marine zone (Collis et al. 2002, Roby et al. 2002, Cramer et al. 2021). An explanation for this discrepancy is the differential abundance of alternative food sources across the estuary (Collis et al. 2002, Roby et al. 2002). Within the freshwater zone fewer food sources besides juvenile salmonids are available compared with the marine zone, where forage fishes such as northern anchovy (*Engraulis mordax*) and clupeid species are often highly abundant, and therefore tend to dominate the forage base available for birds.

The double-crested cormorant is generally constrained to nest in habitats inaccessible to mammalian predators. Thus, double-crested cormorants within the western population have historically nested in colonies located on islands, cliffs, and tall trees. However, human-built structures such as bridges, navigation markers, and power transmission towers are increasingly being used by double-crested cormorants across the western population (Adkins and Roby 2010, ODFW unpubl. data, U.S. Fish and Willdife Service [USFWS] unpubl. data), including within the Columbia River basin (Naughton et al. 2007, Roby et al. 2021). Within the Columbia River estuary, all 11 known historical double-crested cormorant colony sites are human-built or human-modified (Tables 1, 2). The exclusive use of human-built or human-modified structures for nesting in the estuary suggests natural habitats capable of supporting large colonies may be limited or absent. The Corps administers and currently manages avian predators at three known historical colony sites within the Columbia River estuary: East Sand Island (RKM 8), Rice Island (RKM 34), and Miller Sands Spit (RKM 38), all islands that have been either created or substantially modified for operation and maintenance of the Columbia River navigation channel. The remaining eight historical colonies or colony complexes are currently unmanaged and are administered by a variety of entities, including the U.S. Coast Guard (navigation markers), states of Oregon and Washington (bridges), and Bonneville Power Administration/regional power utilities (power transmission towers).

To date, active management of double-crested cormorant abundance within the estuary has essentially been limited to the East Sand Island colony. The Corps has committed to limit the size of this colony to 5,380–5,939 breeding pairs, compared to an average 12,917 pairs during 2004–2013, the pre-management baseline period identified in USACE (2015). The reduction in colony size was planned to occur primarily by means of lethal take of adult individuals and eggs over the course of four consecutive years (phase 1) followed by restricting available nesting habitat by modifying a portion of the island to allow for tidal flooding (phase 2). A full account of planned and implemented management under the ESI management plan is presented in Appendix A. The two other Corps-managed colony sites, Rice Island and Miller Sands Spit, are both currently used as dredge material placement sites, and the Corps dissuades avian predators from nesting on these islands as part of a Biological Opinion associated with maintenance of the river channel (NMFS 2012). However, double-crested cormorants have not attempted to nest on either of these islands since 2008.

Section 3: Status of the Double-crested Cormorant in the Columbia River Estuary

3.1. Introduction

The most recent regional double-crested cormorant status assessment documented the growth of the species' western population since the mid-1990s (Adkins and Roby 2010). This growth reflected a broader trend of population recovery as abundance grew across the United States in response to federal protection under the Migratory Bird Treaty Act and the banning of DDT (both in 1972; Wires and Cuthbert 2006). However, in recent years major changes have occurred to double-crested cormorant abundance and distribution within the Columbia River estuary, which supported roughly 40% of the western population during 2008 and 2009, when a regional census was last conducted (Adkins and Roby 2010, Adkins et al. 2014). Understanding these changes is important for estimating potential changes in double-crested cormorant predation impacts on juvenile salmonids within the Columbia River basin, and for management of the western population. The purpose of this section is to synthesize available information regarding double-crested cormorant abundance at estuary colony sites; to evaluate potential causes for recent changes, with an emphasis on the effects of implementation of the ESI management plan; and to identify potential future scenarios for double-crested cormorant status in the Columbia River estuary.

3.2. Methods

I compiled all available survey data for double-crested cormorant colonies located along the Columbia River from the river mouth (RKM 0) to Bonneville Dam (RKM 234), the entire extent of the Columbia River estuary (Simenstad et al. 2011). I conducted a literature search and identified and compiled all available annual survey data from published sources. Additional unpublished data were provided by regional biologists and managers. I included all survey data that reflected complete counts of breeding double-crested cormorants at or near the presumed annual peak of colony attendance. In cases where the same colony was surveyed several times in a season, peak abundance of active nests was used as a measure of annual abundance (Pacific Flyway Council 2013). Additional details on methods are provided in Appendix B.

3.3. Results and Discussion

3.3.1. Abundance and Distribution: Historical–2020

Double-crested cormorants appear to have occurred historically in the Columbia River estuary, but their past abundance and breeding status in the estuary is unclear (Wires and Cuthbert 2006). No quantitative information on breeding double-crested cormorant abundance in the estuary was found for years prior to the first breeding surveys in 1979–1980.

Survey data for double-crested cormorant colonies within the Columbia River estuary were found for the period 1979–2020 and are compiled in Appendix B. During the first survey effort in 1979–1980, 131 nesting pairs were detected at a colony site in Trestle Bay in the lower estuary, and at least 10 nesting pairs were present at unspecified navigation markers west of Miller Sands (CREST 1984). Double-crested cormorant abundance progressively grew in the estuary following the first survey years, with at least part of this growth likely a result of immigration from colony sites outside the estuary (Carter et al. 1995, Lawes et al. 2021). In 1988, colonies were first noted on East Sand Island and Rice Island, although their sizes for that year are unknown (Carter et al. 1995); both these colonies grew rapidly and together supported most nesting double-crested cormorants in the estuary during the 1990s (Figure 2). By 1999, the East Sand Island colony had grown to 6,561 pairs, while Rice Island, which had supported up to a maximum of 1,444 pairs during the 1990s, had been abandoned, likely because of human activity associated with the managed relocation of Caspian terns (*Hydroprogne caspia*) from Rice Island to East Sand Island (Lawes et al. 2021). Throughout 1991–1998, substantial double-crested cormorant breeding effort occurred across both the marine and freshwater zones of the estuary, although by 1999 the vast majority occurred in the marine zone following decline of the Rice Island colony (Figure 3).

Double-crested cormorant abundance continued to grow on East Sand Island during the early 2000s, reaching a long-term peak beginning in roughly 2004, and extending until 2014, the year prior to implementation of the ESI management plan (peak abundance period; Figure 4). A slightly shorter period, 2004–2013, was used by federal managers as a pre-management abundance baseline associated with the ESI management plan (USACE 2015). East Sand Island supported 39% of all breeding pairs in the western population at the time of the last status assessment (Adkins and Roby 2010) and was also the largest known breeding colony for the species anywhere in its North American range during at least a portion of the pre-management period (Adkins et al. 2014). The Rice Island colony was only intermittently active after the year 2000, supporting up to 211 breeding pairs during four years of nesting activity. The colony on the Astoria-Megler Bridge was first noted in 2004, when it supported 6 breeding pairs; it subsequently grew to 333 breeding pairs by 2014 (Figure 5). Aggregate annual abundance at other estuary colony sites during the 2004–2014 peak abundance period averaged 270 pairs, although nesting abundance increased during 2010–2014, mainly because of new colony formation on power transmission towers near Troutdale, Oregon, and the Lewis and Clark Bridge near Longview, Washington (Figure 6). By 2014, estuary colony sites besides East Sand Island composed an aggregate total of 747 breeding pairs. Overall, average estuary-wide double-crested cormorant abundance during the 2004–2014 peak abundance period was 13,337 breeding pairs, and 97% of breeding effort was focused in the marine zone of the estuary, nearly all associated with the large colony on East Sand Island.

The overall abundance of breeding double-crested cormorants in the Columbia River estuary declined substantially during 2015–2020, concurrent with implementation of the ESI management plan (management period). At the same time, the species' distribution shifted from colonies in the marine zone to those in the mixing and freshwater zones of the estuary (Figure 3). The number of breeding pairs on East Sand Island declined from an average of 12,982 pairs during the 2004–2014 peak abundance period to an average 7,489 pairs during 2015–2017 (phase I of the ESI management plan), to only 2,116 pairs during 2018–2020 (phase II of the ESI management plan; Table 3). No sustained double-crested cormorant breeding effort was observed on East Sand Island in 2020 (USACE unpubl. data), apparently the first year since prior to 1988 when meaningful nesting activity did not occur (Carter et al. 1995). In contrast to East Sand Island, the Astoria-Megler Bridge colony grew rapidly during the 2015–2020 management period, from 333 nesting pairs in 2014 to 5,081 pairs in 2020 (Figure 5). The aggregate total for other colony sites in the estuary also increased somewhat (Figure 6), from 414 breeding pairs in 2014 to 843 pairs in 2020. Abundance in the marine zone declined from an average 12,983 pairs during the 2004–2014 peak abundance period to 81 pairs in 2020. Abundance within the freshwater and mixing zones increased from an average 354 pairs during the 2004–2014 peak abundance period to 5,843 pairs in 2020. Overall, estuary-wide abundance of breeding double-crested cormorants declined from an average of 13,337 breeding pairs during the 2004–2014 peak abundance period to 5,924 pairs in 2020 (Figure 2), representing a decline of 56%. About 3% of estuary-wide breeding abundance occurred in the mixing and freshwater zones during the 2004–2014 peak abundance period, compared to over 99% in 2020.

3.3.2. Causes for Recent Changes in Distribution

3.3.2.1. Dispersal of Breeding-age Individuals from East Sand Island

Evidence suggests dispersal of breeding-age individuals from the East Sand Island colony and subsequent recruitment to other estuary colonies (non-ESI colonies) accounted for most of the recent change in distribution of double-crested cormorants in the Columbia River estuary. Such dispersal was anticipated by researchers as a response to management (Courtot et al. 2012, BRNW 2013, Roby et al. 2014, USACE 2015, Peck-Richardson 2017), but an adaptive response to address this dispersal did not occur. Several lines of supporting evidence are detailed in this section: 1) observed and expected dispersal from East Sand Island and associated loss of fidelity to this colony; 2) known connectivity between the East Sand Island and the Astoria-Megler Bridge colonies; 3) implausibility that rapid growth at non-ESI colonies could have occurred absent dispersal from East Sand Island; and 4) observation of marked birds at non-ESI colony sites following dispersal from East Sand Island.

First, the growth of non-ESI colonies is consistent with the timing and scale of double-crested cormorant dispersal from the East Sand Island colony and the abrupt decline in abundance there. The first observed dispersal events began prior to implementation of the ESI management plan, when researchers deliberately restricted large portions of available breeding habitat on East Sand Island to determine the feasibility of colony relocation. During this research in 2011–2013, double-crested cormorants that were radio- and satellite-tagged on East Sand Island dispersed widely across the Columbia River estuary and Pacific Northwest region (Roby et al. 2012, 2013, 2014, Peck-Richardson 2017). Within the estuary, marked birds were frequently detected at the growing Astoria-Megler Bridge colony (USACE 2015, Peck-Richardson 2017), as well as new and incipient colony sites at power transmission towers near Troutdale (Troutdale Towers) and at the Longview Bridge in Washington (Peck-Richardson 2017). In addition to detections of marked birds, thousands of unmarked double-crested cormorants were observed using the Astoria-Megler Bridge as a roost site at various times during 2012 and 2013 (Roby et al. 2013, 2014). However, most individuals returned to East Sand Island within several weeks following experimental colony disturbances (Roby et al. 2013, 2014).

In contrast to limited dispersal associated with research, much larger-scale dispersal was observed or inferred during implementation of the ESI management plan. During 2016, an unprecedented complete colony abandonment occurred on East Sand Island in mid-May and extended until late June; ca. 9,000 breeding pairs, most apparently attending active nests, left the colony (Anchor QEA 2017; USACE unpubl. data). Subsequently, observers noted up to 11,000 individuals roosting on the Astoria-Megler Bridge (Anchor QEA 2017), presumably having dispersed there from East Sand Island. During 2017, dispersal events were observed on East Sand Island during mid-May (ca. 3,400 pairs) and early June (ca. 3,000 pairs), and the colony supported a peak abundance of only 544 nests (Turecek et al. 2018). However, up to 16,000 non-breeding double-crested cormorants were observed in the estuary during aerial surveys in 2017, with up to 10,000 individuals observed roosting on the Astoria-Megler Bridge during some periods (USACE unpubl. data). Most of these birds were presumably breeding-age individuals associated with the East Sand Island colony. In 2018, managers restricted the area available for double-crested cormorant nesting on East Sand Island, and it appears thousands of individuals were likely forced to disperse to other colonies beginning in that year (Appendix C). In 2018, as in the previous two years, thousands of apparently non-breeding double-crested cormorants presumably associated with the East Sand Island colony were observed roosting on the Astoria-Megler Bridge (ODFW unpubl. data). In 2019, only up to 1,600 breeding pairs attempted to nest on East Sand Island, although up to 5,000–6,000 individuals were observed roosting on the colony site and adjacent beaches during June and July (USACE unpubl. data). Limited survey data in 2019 indicated frequent dispersal events throughout the breeding season and a peak of only 350 active nests (USACE unpubl. data). As during 2016–2018, thousands of apparently non-breeding double-crested cormorants were observed roosting on the Astoria-Megler Bridge in 2019, presumably associated with the East Sand Island colony (ODFW unpubl. data). During 2020, no sustained breeding activity was observed on East Sand Island, although hundreds to about 1,000 individuals were observed on or near the island at various times (USACE unpubl. data). In contrast to 2016–2019, relatively few apparently non-breeding double-crested cormorants were observed on the Astoria-Megler Bridge during 2020 (ODFW unpubl. data), suggesting individuals previously displaced from East Sand Island were either nesting at the Astoria-Megler Bridge colony, nesting at other colony sites, or were no longer part of the regional breeding population (e.g. individuals that did not nest or had died).

Additional evidence for the connection between growth of non-ESI colonies and dispersal from East Sand Island consists of previous research findings. Double-crested cormorant fidelity to the estuary was documented by recovery of banded individuals (Clark et al. 2006) and individuals marked with radio- and satellite tags (Roby et al. 2013, 2014, USACE 2015, Peck-Richardson 2017). In particular, strong connectivity between the East Sand Island and Astoria-Megler Bridge colonies was documented (USACE 2015, Peck-Richardson 2017). Moreover, additional research on dispersal patterns for double-crested cormorants (Duerr et al. 2007, Strickland et al. 2011, Courtot et al. 2012) and other colonial waterbirds (Aebischer 1995, Serrano and Tella 2003, Henaux et al. 2007) indicated double-crested cormorants dispersing from East Sand Island would have tended to occupy nearby colony sites rather than prospect for sites farther away. Overall, evidence from a variety of sources indicated that colony sites closest to East Sand Island, and especially the Astoria-Megler Bridge, would have been likely to grow in the event of double-crested cormorant dispersal from East Sand Island. Consequently, growth of the Astoria-Megler Bridge colony was specifically identified in the ESI management plan as a likely outcome of dispersal from East Sand Island (USACE 2015).

Another line of evidence for the connection between growth of non-ESI colonies and dispersal from East Sand Island relates to the extremely rapid growth of non-managed colony sites. Growth rates for non-managed colony sites in aggregate ($\lambda = 1.33$) and for the Astoria-Megler Bridge in particular ($\lambda = 1.56$) far exceed credible rates of growth for colonies experiencing purely intrinsic growth (i.e. growth not supplemented by immigration) for colonies in the western population (λ = 1.16; Appendix D). Such high growth rates are impossible to explain unless immigration has been a dominant influence. Further, the large size of the East Sand Island colony prior to management, and thus the large number of potential immigrating individuals associated with it, is consistent with the extreme rate of growth at non-managed colony sites. The colony on East Sand Island was by far the largest nearby source of potential recruits to non-ESI colonies, supporting 76% of the coastal double-crested cormorant population in Washington, British Columbia, and Oregon during the most recent population census (Adkins and Roby 2010).

Finally, field observations of marked individuals indicate double-crested cormorants previously associated with East Sand Island immigrated to other colony sites in the estuary. During management feasibility studies on East Sand Island, five marked individuals appeared to relocate to other colonies following their capture and tagging on East Sand Island. In 2012, two radio-tagged individuals relocated to the Astoria-Megler Bridge where they presumably re-nested (Roby et al. 2013). In 2013, three satellite-tagged double-crested cormorants relocated to new estuary colonies, where they presumably re-nested, including two at the Astoria-Megler Bridge colony and one at the Troutdale Towers colony (Peck-Richardson, 2017). The dispersal of these marked individuals occurred despite no major restriction to available breeding habitat on East Sand Island during these studies. Further, during implementation of the ESI management plan, double-crested cormorants previously banded on East Sand Island during 2012 and 2013 were observed using the Astoria-Megler Bridge, and multiple banded individuals were observed breeding there during 2017 and 2018 (Turecek et al. 2019). Additionally, four banded double-crested cormorants were observed opportunistically on the Astoria-Megler Bridge in 2019 (ODFW unpubl. data), although it is unclear whether these latter birds were nesting. While the number of bands sighted on the bridge is small, based on the time lag since individuals were first marked, even a small number of detections probably represents a significant fraction of banded birds still alive.

3.3.2.2. Dispersal of Surplus Recruits and Intrinsic Growth

Productivity associated with the East Sand Island colony, a measure of the number of chicks fledged per nest, may have been a contributing factor to recent growth of non-ESI colonies, but likely played only a minor role. During 2004–2013, the pre-management baseline period (USACE 2015), the East Sand Island colony produced an estimated minimum 1,150 breeding-age individuals (575 breeding pairs) annually above its population replacement level (Appendix E), suggesting a ready source of recruits available to immigrate to nearby colonies. Since double-crested cormorant abundance on East Sand Island was generally stable during this period, we can infer that any “surplus” individuals (*sensu* Pulliam 1988) attempted to breed elsewhere, with some probably dispersing to nearby colony sites in the estuary. However, dispersal of recruits derived from East Sand Island cannot explain the rapid recent growth of non-ESI colonies because 1) such growth would be expected to occur gradually throughout the pre-management period, rather than abruptly during the management period, and 2) observed growth of non-ESI colonies during 2018–2020 far exceeded the credible productivity of East Sand Island. Productivity on East Sand Island was apparently low throughout 2015–2020 (Appendix A), indicating few recruits would have been available to disperse in subsequent years.

Growth of non-ESI colonies could also have resulted from intrinsic productivity (i.e. subadults recruiting to their natal colonies). However, assuming demographic rates used for population modelling in the ESI management plan, intrinsic growth for the Astoria-Megler Bridge colony could only explain about 12% of its total growth during 2010–2019, and 18% of growth of all non-managed colonies during the same period (Appendix D). Thus, intrinsic growth could represent only a small proportion of the observed growth of non-ESI colonies in the Columbia River estuary.

Notably, despite the apparent limited importance of productivity to the observed rapid growth of non-ESI colonies during the management period, it is nevertheless likely double-crested cormorant abundance at these colonies would have grown over the long term regardless of the outcome of the ESI management plan. This is because the ESI management plan only sought to limit breeding habitat at one of many potential estuary colony sites, and therefore productivity-related growth at non-ESI colonies was unconstrained. Thus, gradual growth of non-ESI colonies would have been expected regardless of whether dispersal from the East Sand Island colony occurred during the management period. This fact is especially relevant for the Astoria-Megler Bridge, which was specifically identified as a colony site with substantial unused breeding habitat, and therefore had considerable potential for growth (USACE 2015).

3.3.2.3. Immigration from outside the Columbia River estuary

It appears unlikely immigration from colonies outside of the Columbia River estuary significantly contributed to growth of non-managed colony sites within the estuary. Most colonies displaying the strongest connectivity with East Sand Island are located in coastal areas of Washington and British Columbia (Courtot et al. 2012), suggesting this region would be the most likely source of immigrating individuals. However, during the last region-wide census, colonies in coastal British Columbia and Washington only supported a combined total of about 1,200 breeding pairs (Adkins et al. 2014), and recent surveys do not seem to indicate major declines in these areas potentially associated with large-scale dispersal (USFWS unpubl. data). Likewise, in coastal Oregon, where the breeding population south of the Columbia River estuary has averaged about 1,900 pairs during 2009–2019 (Adkins et al. 2014; ODFW unpubl. data), no declines have been observed on a scale that would indicate immigration to the Columbia River estuary (ODFW unpubl. data). Finally, it is unlikely large numbers of double-crested cormorants immigrated to the Columbia River estuary from outside the Pacific Northwest, both because of the apparent low connectivity of these areas with the East Sand Island colony (Courtot et al. 2012) and because of their distance from the Columbia River estuary.

3.3.3. Causes of Dispersal from East Sand Island

3.3.3.1. Pre-management Period

Observed dispersal from the East Sand Island colony during 2011–2013 was associated with colony disturbances by researchers and bald eagles and was likely facilitated by increased social attraction associated with growing non-ESI colonies. Research on East Sand Island during 2008–2013 involved experimentally disturbing limited portions of the colony area to evaluate the response of double-crested cormorants (BRNW 2013, Roby et al. 2014). Concurrent with this research, the number of bald eagles near the colony increased by ca. 20% per year during 2008–2012, and the number of colony disturbances caused by eagles was especially high in 2011 and 2012 (Lawes et al. 2021). Colony disturbances by researchers and eagles were associated with dispersal and irregular colony attendance by radio-tagged double-crested cormorants in 2011 (Roby et al. 2012) and were associated with dispersal of radio- and satellite-tagged individuals and simultaneous occurrence of thousands of roosting double-crested cormorants at the Astoria-Megler Bridge in 2012 and 2013 (Roby et al. 2013, 2014). Based on research for other colonial birds, this dispersal would have tended to increase the likelihood of prospecting for new colony sites (Carney and Sydeman 1999, Henaux et al. 2007, Fernández-Chacón et al. 2013). The large number of double-crested cormorants observed roosting on the Astoria-Megler Bridge during experimental dissuasion activities during 2012 and 2013 seems to have reflected such prospecting behavior, which is supported by the rapid growth of this colony during subsequent years.

In addition to disturbances that may have deterred some double-crested cormorants from nesting on East Sand Island, the initial growth of non-ESI colony sites during the pre-management period likely created a situation that was favorable for subsequent growth. Like other colonial waterbirds, double-crested cormorants are gregarious, and they can thus be attracted to new habitats merely by the presence of other individuals. Such “social attraction” appears to play an important role in site selection for colonial birds (Schjørring et al. 1999, Doligez et al. 2002, Henaux et al. 2007), including double-crested cormorants (Suzuki et al. 2015). The presence of growing colonies near East Sand Island that were apparently free of disturbance was likely a factor in their increasing use by double-crested cormorants throughout the pre-management period, especially during periods of colony disturbances on East Sand Island.

3.3.3.2. Management period

Dispersal from the East Sand Island colony during the management period coincided with the presence of numerous stressors. These included lethal take of 5,576 adult double-crested cormorants and destruction of 6,181 nests associated with the East Sand Island colony during 2015–2017 (Lawes et al. 2021, Appendix A), substantial modification of the size and shape of East Sand Island to reduce available nesting habitat (Lawes et al. 2021, Appendix A), dissuasion of individuals attempting to nest in undesired areas on East Sand Island (Lawes et al. 2021, Appendix A), likely unintended colony disturbance by managers on East Sand Island (Lawes et al. 2021), colony disturbances by bald eagles on East Sand Island (Turecek et al. 2018, 2019, Lawes et al. 2021, USACE unpubl. data, Appendix A), and potentially poor foraging conditions within the estuary during at least one year (Turecek et al. 2018). Overall, it appears likely the cumulative effect of some or all of these stressors led to the decline of the East Sand Island colony and subsequent dispersal of double-crested cormorants to other colony sites in the estuary (Lawes et al. 2021).

The first two years of management plan implementation (2015–2016; Appendix A) resulted in high mortality for adults (culling) and depressed nest productivity (nest destruction, colony abandonment), which together constituted cues likely perceived by double-crested cormorants as a decline in the quality of East Sand Island as a colony site, as suggested by previous work with double-crested cormorants and other colonial waterbirds (Schjørring et al. 1999, Boulinier et al. 2008, Strickland et al. 2011). Subsequently, during 2017–2020, the rate of bald eagle disturbance was apparently high on the reduced-size colony (Turecek et al. 2018, 2019, USACE unpubl. data), and associated breeding productivity was low (Appendix A, USACE unpubl. data). Concurrently, and in contrast to the East Sand Island colony, the nearby Astoria-Megler Bridge colony experienced no major observed disturbances during the management period (ODFW unpubl. data), and available data indicate productivity was high, apparently as high or higher than at the East Sand Island colony prior to management (Turecek et al. 2019; ODFW unpubl. data; Appendix A). Consequently, the optimal colony site in the Columbia River estuary from a standpoint of reproductive fitness (Stearns et al. 1989) apparently shifted during implementation of the ESI management plan from East Sand Island to the nearby Astoria-Megler Bridge. This shift was likely a major contributing factor to the rapid decline of the East Sand Island colony and was likely self-reinforcing. The presence of a thriving colony on the Astoria-Megler Bridge would have enticed stressed individuals from the East Sand Island colony, which in turn would have caused the Astoria-Megler Bridge colony to grow larger, which in turn would have further increased the attractiveness of this colony through the mechanism of social attraction (Suzuki et al. 2015). The erratic and seasonally delayed breeding effort on East Sand Island during the management period (Lawes et al. 2021), which coincided with individuals apparently prospecting for nesting sites on the Astoria-Megler Bridge (ODFW unpubl. data), lends additional support to a shift in the optimal colony site within the Columbia River estuary as perceived by double-crested cormorants.

One contributing factor to dispersal merits special attention because of its importance to colony failures on East Sand Island during 2017–2020: colony disturbances caused by bald eagles. Although it is possible bald eagle disturbances had effects on the colony independent of management, it is more likely that management decreased the colony's resilience to bald eagle attacks, a scenario unanticipated in the management plan, although identified by researchers in other sources (Adkins et al. 2014, Peck-Richardson 2017). One advantage of colonial breeding is thought to be a buffering effect associated with: 1) predator-swamping, whereby the sheer number of nests present at a colony is sufficient to reduce the probability of any given individual or nest being preyed upon (Ims 1990), and 2) the "selfish herd" effect (Hamilton 1971), where configuration of nests would advantage individuals nesting near a maximum number of neighbors and away from colony edges. Presumably, below some threshold level of nest abundance this buffering effect does not sufficiently mitigate predation risk and, consequently, colony abandonment becomes more likely. Further, disturbance and associated stress caused by bald eagles would have been expected to be additive or synergistic to the considerable stress associated with management (Lawes et al. 2021). The fact that bald eagles contributed to serious breeding failures on East Sand Island only after considerable culling and egg-oiling occurred suggests management was an important underlying factor associated with bald eagle disturbances. In addition, presumed poor forage availability in 2017 (Turecek et al. 2018) may have interacted with bald eagle attacks to further reduce double-crested cormorant breeding effort on East Sand Island; this is because parents may have been unlikely to trade off the risk of bald eagle predation associated with nesting for the probable low reproductive payoff associated with poor foraging conditions (Drent and Daan 1980).

An additional management-related cause of dispersal was the apparent displacement of breeding-age double-crested cormorants when the colony area on East Sand Island was restricted during the beginning of phase 2 of management in 2018. Thousands of individuals were apparently precluded from nesting on East Sand Island in 2018 and were thus forced to breed elsewhere (Appendix C). This scenario is consistent with the rapid increase in size of the Astoria-Megler Bridge colony during phase 2 of the ESI management plan, when the colony grew from 834 breeding pairs in 2017 to 5,081 breeding pairs by 2020.

In summary, it is likely the cumulative influence of a variety of stressors during the management period ultimately caused the decline and collapse of the East Sand Island colony and associated dispersal of double-crested cormorants to new colony sites. However, available information suggests implementation of the ESI management plan was a pre-eminent causal factor, acting directly by reducing double-crested cormorant fidelity to East Sand Island and reducing the amount of nesting habitat available there, and indirectly by decreasing the resilience of double-crested cormorants to disturbances by bald eagles and other potential stressors, and by failing to include a clear plan for adaptive management to deter emigration from East Sand Island to the nearby Astoria-Megler Bridge colony, which was specifically identified in the management plan as a likely destination for individuals seeking new habitat (USACE 2015).

3.3.4. Potential Future Status

The future status of double-crested cormorants in the Columbia River estuary seems likely to be tied closely to availability of human-built or human-altered breeding habitat, where all known colonies in the estuary were located during 1979–2020. The majority of estuary habitat historically used by double-crested cormorants is located at three human-built or human-altered colony sites: East Sand Island and Rice Island, both managed by the Corps, and one unmanaged site, the Astoria-Megler Bridge, which is administered by the Oregon Department of Transportation. Based on recent observed colony declines and existing management (USACE 2015), it seems likely that managers will be able to maintain double-crested cormorant abundance on East Sand Island to fewer than 5,939 breeding pairs, as called for in the ESI management plan. For Rice Island, double-crested cormorants seem unlikely to re-establish a colony if federal managers continue ongoing management associated with the Corps' channel maintenance operations (NMFS 2012). However, the future size of the Astoria-Megler Bridge colony is less certain because it is currently unmanaged by any entity. Although the ESI management plan described potential actions to address colony growth on the Astoria-Megler Bridge (Table 4), they have not been implemented, and the size of this colony has therefore remained unrestricted.

During the management period, colonies besides the three noted above supported an aggregate annual maximum of 843 breeding pairs (in 2020), about three times higher than the average during the 2004–2014 peak abundance period. However, continued growth of these colonies is uncertain because of potential habitat and food constraints in the estuary freshwater zone, where most of them are located. Besides historical colony sites, currently unused potential breeding sites are scattered across the estuary. Examples of these habitats include large trees associated with Cape Disappointment State Park (WA), Lewis and Clark National Historical Park (OR), Fort Stevens State Park (OR), and at various islands throughout the estuary, especially those offering nesting trees and absence of mammalian predators; various bridges; and tens of navigation markers located throughout the estuary. Those colony sites located within average foraging range (roughly 30 km) of the marine and mixing zones of the estuary would have the highest likelihood of reaching a size greater than several hundred breeding pairs, owing to abundant marine-derived fish available near these locations. To date, no estuary colony more than 30 km from marine or mixing zone habitats has exceeded 400 breeding pairs (Appendix B).

Several double-crested cormorant abundance scenarios are possible for the Columbia River estuary based on presumed habitat capacity at identified colony sites. Abundance under these scenarios can be compared with the projected estuary-wide abundance of 5,380-5,939 pairs under RPA 46.

> *Scenario 1*: Under the presumed peak managed colony size for East Sand Island (5,939 pairs, USACE 2015), estimated habitat capacity for the Astoria-Megler Bridge (10,950 pairs, ODFW unpubl. data), and most recent abundance for other non-ESI colony sites (843 pairs), the Columbia River estuary could support at least 17,732 breeding pairs in the future. This abundance is roughly 133% of abundance during the 2004–2014 peak abundance period. Approximately 66% of the estuary population would be located in the mixing or freshwater zones of the estuary, in contrast to an average of only 3% during the 2004–2014 peak abundance period.
>
> *Scenario 2*: Recent partial (2016, 2018) and near-complete or complete (2017, 2019) breeding failures on East Sand Island, and the absence of a sustained breeding effort there in 2020, suggest this site may no longer be able to reliably support a major breeding colony. Future abundance may therefore be dominated by the Astoria-Megler Bridge colony and other colonies; the estimated habitat capacity of these colony sites in aggregate is 11,793 breeding pairs, about 88% of estuary-wide abundance during the 2004–2014 peak abundance period. Under this scenario, more than 99% of double-crested cormorant abundance in the estuary could occur in the freshwater and mixing zones.
>
> *Scenario 3*: Ecological or anthropogenic constraints on abundance at East Sand Island, the Astoria-Megler Bridge, and other colony sites result in an estuary-wide double-crested cormorant population lower than possible strictly based on amount of breeding habitat present, but possibly higher than the 5,380–5,939 breeding pairs specified for East Sand Island under the ESI management plan.

Regardless of which scenario is realized, it seems likely the status of double-crested cormorants in the Columbia River estuary during the immediate future will be substantially different than anticipated under RPA 46 and the ESI management plan. In any case, the abundance scenarios above, when integrated with estimated predation rates across different salinity zones in the estuary, can be used to estimate potential future predation impacts, a topic investigated further in Section 4.

There are several uncertainties related to constraints on future double-crested cormorant abundance in the Columbia River estuary. First, the willingness of double-crested cormorants to colonize currently unused sites within the estuary is unclear. Previous research failed to show a strong connection between the East Sand Island colony and other colony sites within the estuary, except for the Astoria-Megler Bridge (Peck-Richardson 2017). Besides the Astoria-Megler Bridge, double-crested cormorants appeared most strongly connected with colony sites outside of the Columbia River basin, suggesting that individuals displaced as a result of habitat saturation or management at existing sites may leave the estuary entirely. Second, “top-down” factors such as colony disturbance or threat of predation could preclude double-crested cormorant use of currently unused colony sites. Double-crested cormorants are sensitive to colony disturbance and will abandon colonies they perceive unsafe (Ellison and Cleary 1978, Strickland et al. 2011). Double-crested cormorants nesting in trees or on open ground along the Oregon Coast have appeared particularly sensitive to bald eagle attacks (ODFW unpubl. data). There is little reason to suspect this sensitivity would not also apply to the marine and mixing zones of the Columbia River estuary, where it seems these habitat types compose a significant portion of heretofore-unused habitat. Further, human disturbance of colonies of double-crested cormorants (Lawes et al. 2021) and American white pelicans (*Pelecanus erythrorhynchus;* Lawes and Roby 2018) has been noted in the estuary in the past and may be a limiting factor for some estuary sites in the future, as it apparently has for other colonies within the western population (Adkins et al. 2014). Finally, availability of forage within the estuary varies annually and across longer timescales (Weitkamp et al. 2012, 2016). Future constraints on available food therefore may vary compared with current levels, thereby driving double-crested cormorant abundance through so-called “bottom-up” effects (Collar et al. 2017).

Overall, it seems likely double-crested cormorants will continue to be attracted to the Columbia River estuary because of its highly abundant forage resources. However, constraints related to nesting habitat and colony disturbances will likely determine the abundance of breeding double-crested cormorants within the estuary. Because of the importance of marine-derived forage fishes to the double-crested cormorant diet, abundance at an apparently limited number of potential colony sites within foraging range of the estuary’s marine and mixing zones, where such forage is abundant, will probably drive future abundance across the entire Columbia River estuary.

Section 4: Implications for Outmigrating Juvenile Salmonids

4.1. Introduction

The recent redistribution of double-crested cormorants within the Columbia River estuary presents a challenge to managers interested in how double-crested cormorant predation has changed following implementation of the ESI management plan. During the 2004–2014 peak abundance period, 97% of breeding double-crested cormorants within the estuary nested on East Sand Island, therefore monitoring this colony site was sufficient to estimate nearly all predation impacts across the estuary. However, in recent years nearly all nesting has occurred at colonies where predation impacts are not monitored, most located upriver from East Sand Island within the estuary freshwater and mixing zones (Section 2). In light of the mismatch between the limited scope of predation monitoring and current double-crested cormorant distribution, the goal of this section is to summarize available information on predation impacts for non-ESI colony sites and model current potential estuary-wide impacts given available data.

4.2. Available Data

The impact of double-crested cormorant predation on survival of juvenile salmonids is well understood for the East Sand Island colony, but data is limited for non-ESI colonies. For East Sand Island, annual predation estimates for various ESA-listed runs have been calculated since 1999 (Roby et al. 2021). These predation estimates are derived from salmonids tagged with Passive Integrated Transponder (PIT) tags that are subsequently consumed by double-crested cormorants and excreted on the colony, and later recovered by researchers. Information on predation impacts for non-ESI colonies, however, is limited because attempts to recover or analyze PIT tags have occurred only intermittently at a few colony sites.

Despite limited study, available data have consistently shown higher per capita predation rates on ESA-listed salmonids for double-crested cormorants nesting at colonies within the freshwater zone of the Columbia River estuary compared with the marine zone. PIT tags recovered during previous years at two freshwater zone colonies, Rice Island (2001, 2006) and Miller Sands Spit (2006, 2007), were recently analyzed and compared to predation rates on East Sand Island during the same year. Average per capita predation rates for Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus kisutch*), and steelhead (*Oncorhynchus mykiss*) were 5.1, 7.7 and 7.3 times higher on Rice Island compared with East Sand Island, respectively (Cramer et al. 2021). Average per capita predation rates for Chinook salmon, sockeye salmon (*Oncorhynchus nerka*), coho salmon, and steelhead were 6.3, 17.9, 7.7, and 9.9 times higher on Miller Sands Spit compared with East Sand Island, respectively (Cramer et al. 2021). Such a major difference in predation rates is remarkable because Rice Island and Miller Sands Spit are located only 26 and 30 river kilometers upriver of East Sand Island, respectively. In addition to the study by Cramer et al. (2021), thousands of PIT tags were recovered annually from the Rice Island double-crested cormorant colony in 1996–1998 (Collis et al. 2001), although no PIT-tag data for the East Sand Island colony are available for comparison for those years. In addition to study of colonies in the freshwater zone, a recent study suggests high predation rates on PIT tagged salmonids by double-crested cormorants on the Astoria-Megler Bridge, within the mixing zone, although this study was not able to compare predation rates with the East Sand Island colony during the same outmigration year (Evans et al. 2022).

Studies of the diets of double-crested cormorants and other colonial piscivorous birds suggest the same pattern of higher predation on juvenile salmonids in the freshwater zone compared with the marine zone. Collis et al. (2002) found the diet of double-crested cormorants nesting at Rice Island and nearby channel markers consisted of 46% salmonids in 1997 and 1998, in contrast to only 16% salmonids for double-crested cormorants nesting on East Sand Island during the same period. Roby et al. (2002) found the diet of Caspian terns breeding on Rice Island during 1999 and 2000 consisted of 77% and 90% juvenile salmonids, respectively, while the diet of terns breeding on East Sand Island during 1999, 2000, and 2001 consisted of only 46%, 47%, and 33%, respectively. Similarly, Collis et al. (2002) found that glaucous-winged x western gull hybrids (*Larus glaucescens* x *occidentalis*) nesting on Rice Island and Miller Sands Spit consumed 52% salmonids in 1997–1998, in contrast to only 16% salmonids for double-crested cormorants nesting on East Sand Island during the same period.

4.3. Methods

I estimated the potential estuary-wide impact of double-crested cormorant predation on juvenile steelhead, one of the salmonid species most vulnerable to avian predation in the Columbia River estuary (Roby et al. 2021). I derived salinity zone-specific relative predation susceptibility (RPS) from the PIT tag-based analysis of Cramer et al. (2021), who calculated RPS by dividing annual per capita PIT tag consumption at a given colony by the per capita PIT tag consumption at East Sand Island during the same year. For the freshwater zone, I took the average of annual point estimates for colony-specific RPS for both colonies in the freshwater zone for which data was available, Rice Island and Miller Sands Spit; the resulting RPS was 8.6. I assumed RPS for mixing zone colonies would be 4.3, a value intermediate between the freshwater and marine zones. I assumed RPS across the marine zone was 1.0, the same as for East Sand Island. I multiplied annual zone-specific double-crested cormorant abundance (Section 3) by zone-specific relative predation susceptibility and summed these values to determine annual estuary-wide predation impacts. The resulting estimates are represented in terms of the number of breeding pairs on East Sand Island that would cause an equivalent level of predation (ESI predation equivalents) compared with aggregate predation across estuary colonies.

Since no PIT tag or diet data were available for colonies within the mixing zone, I conducted an additional analysis to determine whether the assumed RPS value of 4.3 for this zone was a credible estimate. I used data from a two-year telemetry study conducted on breeding double-crested cormorants captured on East Sand Island (Anderson et al. 2004) to determine the degree to which energetic constraints related to commuting to the freshwater zone may be lessened for double-crested cormorants breeding at the Astoria-Megler Bridge (mixing zone) vs East Sand Island (marine zone). I obtained telemetry data that reflected the distance double-crested cormorants commuted from East Sand Island for both sexes, and then modeled a continuous distribution for commuting distance that assumed a 1:1 sex ratio, using a generalized additive model using the mcgv package in the computer software R. I assumed the resulting distribution reflected energetic constraints imposed by flight costs related to morphology of cormorant species (Elliot et al. 2013); these flight costs would be associated with commuting to and from colony sites in the Columbia River estuary. I then applied this modelled distribution to the Astoria-Megler Bridge colony to determine double-crested cormorant accessibility of freshwater zone habitats relative to individuals breeding on East Sand Island. I used distance measured in a straight line between points to reflect the most energetically efficient commuting route.

4.4. Results

During the 2004–2014 peak abundance period, double-crested cormorants breeding within the freshwater and mixing zones accounted for an average of 17% of annual estuary-wide predation by double-crested cormorants (estuary-wide predation); however, they accounted for an estimated average of 73% of estuary-wide predation during the 2015–2020 management period, and >99% of estuary-wide predation in 2020. Associated with this spatial shift, estuary-wide impacts increased relative to the pre-management period. In 2020, estimated predation within the marine, mixing, and freshwater zones reflected 81, 21,848, and 6,553 ESI predation equivalents, respectively. Estimated estuary-wide predation in 2020 was equivalent to 28,483 pairs on East Sand Island, about 182% of predation during the 2004–2014 peak abundance period, which averaged 15,670 ESI predation equivalents (Figure 7).

Based on the modelled distribution of commuting distance, at least 43% of foraging trips for double-crested cormorants nesting on the Astoria-Megler Bridge would meet or exceed the distance to the former colony site on Rice Island (12 km; Figure 8). In contrast, only 18% of foraging trips for individuals nesting on East Sand Island would meet or exceed the distance from East Sand Island to Rice Island (21 km). The difference between colonies was greater for the network of pile dikes upstream of Rice Island, an apparently preferred foraging location (Collis et al. 2002, Lyons et al. 2007). According to the model, 35% of foraging trips from the Astoria-Megler Bridge colony would meet or exceed the distance to these pile dikes (15 km), compared with only 8% of flights from East Sand Island (24 km). Overall, the modelled distribution predicts that accessibility of two areas within the freshwater zone are 2.4 and 4.4 times higher for individuals nesting on the Astoria-Megler Bridge compared with East Sand Island. I concluded that the RPS value of 4.3 for the mixing zone was a credible estimate.

4.5 Discussion

The results of this analysis reveal a major shift in double-crested cormorant predation from the marine zone to the freshwater and mixing zones in recent years and a concomitant increase in predation impacts in 2020 relative to the peak abundance period (Figure 7). These changes largely reflect the redistribution of double-crested cormorants during the 2015–2020 management period from the East Sand Island colony to colonies primarily located upriver of East Sand Island (Section 3). Further, these results suggest future predation within the Columbia River estuary could exceed that modeled for 2020, which experienced higher levels of predation (in terms of ESI predation equivalents) than any previous year. At its estimated carrying capacity of 10,950 breeding pairs (ODFW unpubl. data) and assuming an RPS value of 4.3, the Astoria-Megler Bridge colony alone could have the equivalent impact of 47,085 breeding pairs on East Sand Island. The predation impact for other estuary colonies may be important as well. In 2020, colonies besides the Astoria-Megler Bridge and East Sand Island reflected an estimated 6,634 ESI predation equivalents, a higher level of predation than anticipated by the ESI management plan for the entire estuary (5,380–5,939 breeding pairs; USACE 2015). Altogether, future predation impacts associated with the Astoria-Megler Bridge and other non-ESI colonies could represent an ESI predation equivalent of 53,719 breeding pairs, 3.4 times higher than predation during the 2004–2014 peak abundance period and 9.0–10.0 times higher than the predation target for the ESI management plan (5,380–5,939 ESI predation equivalents).

My analysis for this section relied on several assumptions: 1) RPS values for sampled colonies were representative of non-sampled colonies, 2) RPS values for estuary colonies were the same within each salinity zone, 3) RPS values for salinity zones were related linearly and inversely to salinity across the three zones, and 4) per capita predation rates were independent of double-crested cormorant abundance. I acknowledge that empirical verification for these assumptions is lacking. Further, the work of Cramer et al. (2021) was only based on three years of data and involved only two sampled colonies in the freshwater zone and one in the marine zone.

Additionally, sampled colonies in the freshwater zone were composed of 150 breeding pairs or fewer. Extrapolating impacts across the estuary from such a limited sample is necessarily tenuous and imprecise. The results of this analysis therefore represent potential estuary-wide impacts based on limited available data, and do not necessarily reflect actual impacts. Nevertheless, despite its limitations, this analysis represents the first attempt to quantify estuary-wide predation impacts associated with recent changes to the status of double-crested cormorants in the Columbia River estuary, and clearly illustrates the potential for major conservation implications for ESA-listed salmonids.

The results of my analysis of energetic constraints related to commuting are consistent with an RPS estimate of 4.3 for the mixing zone, the zone that accounted for the vast majority of double-crested cormorant abundance in 2020. This analysis involved several assumptions that are supported by previous work. First, I assumed double-crested cormorants would be motivated to use upper estuary habitats for foraging during April–mid-June, the period when most outmigrating salmonids pass through the estuary. This assumption is supported by previous research that shows: 1) high fish density in the freshwater and mixing zones relative to the marine zone during spring (Bottom and Jones 1990), and 2) substantial double-crested cormorant use of the freshwater zone (Anderson et al. 2004, Peck-Richardson et al. 2018), especially during spring (Lyons et al. 2007). Second, I assumed individuals foraging in the freshwater and mixing zones would be more likely to consume salmonids compared to individuals foraging in the marine zone. This is supported by the high relative abundance of salmonids relative to other fish in the freshwater zone (Bottom and Jones 1990) and the high percentage of salmonids in the diet of avian predators nesting in the freshwater zone (Collis et al. 2002). Third, I assumed energetic demands related to commuting from the colony site constrain use of the freshwater zone by double-crested cormorants in direct proportion to the distance their colonies are located downstream of this zone. This assumption is supported by the high energetic cost of flight related to the morphology of cormorant species (Elliott et al. 2013) and fundamental ecological theory related to the interplay between animal energy budgets and feeding strategies (Schoener 1971).

While the analysis in this section provides information on potential predation impacts of double-crested cormorants, additional work is necessary to verify these impacts. PIT tag-based predation estimates for the Astoria-Megler Bridge colony in particular would improve our understanding of estuary-wide predation impacts given the large size of this colony and its current growth trend. However, many nests on the bridge are situated in areas where tag recovery would be difficult or impossible. Thus, an approach involving sampling accessible areas of the colony may be required to obtain PIT-tag based estimates. In 2021, a first attempt at such a study was conducted, and results suggested predation rates on salmonids similar to for the East Sand Island colony prior to management (Evans et al. 2022). However, for many other estuary colonies PIT tag recovery may be difficult because of poor colony accessibility to workers (e.g. navigation markers) and potential low tag recovery rates for colonies located on structures over water. Further, the large number of colonies or sub-colonies (19 discrete colony or sub-colony sites active in 2020) would require a substantially expanded monitoring program to verify their impacts.

Although annual PIT tag monitoring at each estuary colony would be an ideal way to measure annual predation impacts, the alternative method of using ESI predation equivalents is advantageous for several reasons. First, it allows for estimation of predation impacts without annual PIT tag sampling of each of many active colony sites across the estuary. Second, the estuary-wide predation goal of RPA 46 is expressed in terms of the abundance of breeding pairs on East Sand Island. Thus, predation measured in terms of ESI predation equivalents can be directly compared with predation targets outlined in RPA 46 (predation equivalent to 5,380–5,939 breeding pairs on East Sand Island). Finally, predation rates vary widely across years, at least for the East Sand Island colony; therefore, a long-term monitoring plan would be needed to obtain a reliable estimate of average predation rates. In contrast, measuring predation in terms of ESI predation equivalents would be robust to annual variation in predation rates. However, there are two important caveats to this method: 1) sufficient breeding effort must exist on East Sand Island to allow for comparison with other colonies within a given year and 2) the timing of breeding needs to be similar among colonies to allow for valid comparisons, since predation rates tend to vary across weeks (Hostetter et al. 2021).

Overall, the results of this study suggest that recent management of the East Sand Island colony failed to meet its ultimate objective: to reduce double-crested cormorant predation across the Columbia River estuary to a level equivalent to 5,380–5,939 breeding pairs on East Sand Island (USACE 2015). Instead, double-crested cormorant predation of ESA-listed salmonids may be substantially higher than prior to management plan implementation and could continue to increase if double-crested cormorants occupy additional habitat on the Astoria-Megler Bridge and other non-ESI colony sites. Overall, it appears additional work is necessary to meet the fish survival objectives reflected in RPA 46 and the ESI management plan.

Section 5: Conclusions

Major changes in double-crested cormorant abundance and distribution have occurred within the Columbia River estuary since the early 2010s, and especially since implementation of the ESI management plan beginning in 2015. A wide body of evidence suggests these changes have resulted in large part from a variety of stressors associated with implementation of the ESI management plan. In response, double-crested cormorant abundance has increased dramatically at the Astoria-Megler Bridge and to a lesser degree at other estuary colony sites. Although the overall number of double-crested cormorant pairs nesting within the Columbia River estuary has declined about 56% since management began in 2015, it is unlikely predation impacts on juvenile salmonids have similarly declined. Rather, double-crested cormorant predation on juvenile steelhead, and presumably other salmonid species, may currently exceed pre-management levels, with even higher predation possible if double-crested cormorants completely occupy available habitat on the Astoria-Megler Bridge colony. The available data strongly suggest that additional work is needed to meet the objective of improved fish survival reflected in RPA 46 and the ESI management plan.

The failure of managers to limit double-crested cormorant dispersal from East Sand Island appears a primary enabling factor in their redistribution within the Columbia River estuary. This failure reflects a mismatch between the objective of improving estuary-wide survival of juvenile salmonids and a management response that has been limited to only one of many potential estuary colony sites. Similarly, monitoring of double-crested cormorant predation rates within the estuary has focused on East Sand Island and has not expanded sufficiently in response to increasing abundance at non-ESI colonies; however, the recent effort to develop a monitoring procedure for the Astoria-Megler Bridge colony (Evans et al. 2022) is an important first step toward correcting this monitoring gap. Nevertheless, predation impacts associated with non-ESI colonies are still incompletely understood. A credible long-term strategy to reduce double-crested cormorant predation impacts must be based on clear objectives regarding acceptable levels of predation on an estuary-wide scale and must include provisions that allow for adaptive monitoring and management of numerous potential colony sites. Because colony sites within the estuary are administered by a variety of federal and state entities, interagency coordination will be essential if managers wish to address double-crested cormorant predation issues within the Columbia River estuary in the future.

Currently, the majority of double-crested cormorant predation within the Columbia River estuary is likely associated with the Astoria-Megler Bridge colony; eliminating use of this colony would therefore reduce estuary-wide double-crested cormorant predation considerably.

However, given the estuary's attractiveness to double-crested cormorants in recent decades, management at the bridge would likely cause dispersal of individuals to new estuary colony sites, which would need to be monitored and managed as needed. Within the freshwater zone, inconsistent access to highly abundant fish at most constituent colony sites (Section 2) suggest a further increase in abundance may be unlikely in this zone. An exception could be the zone's lower margin, which lies within foraging range of abundant food resources downriver. Potential colony sites here include several islands, channel markers, and trees on and near Tongue Point and other areas. However, the Corps is anticipated to continue dissuading double-crested cormorant nesting at several islands in the freshwater zone under an existing management plan for channel maintenance (NMFS 2012), including Rice Island, the site of by far the largest historical colony in the freshwater zone. Within the estuary mixing zone, available food is manifestly able to support thousands of double-crested cormorants (e.g. Astoria-Megler Bridge colony), but colony sites in this zone appear limited. Therefore, substantial dispersal to additional mixing zone colonies seems unlikely. In contrast, double-crested cormorants appear highly likely to disperse to marine zone colonies if the Astoria-Megler Bridge colony is managed. This dispersal would be driven by the presence of abundant food and nesting habitat within the marine zone. Double-crested cormorants may be especially likely to disperse to East Sand Island, which appears to represent the majority of marine zone nesting habitat and is at least theoretically capable of supporting at least 5,380–5,939 breeding pairs following the recent modification of the island by managers (USACE 2015). However, recent breeding failures at the East Sand Island colony associated with disturbances by bald eagles suggest this colony may not be able to persist at a reduced size without management that would reduce the frequency and intensity of such disturbances. Nevertheless, recolonization of East Sand Island would be ideal from a management point-of-view because individuals nesting there would tend to have much lower per capita impacts on juvenile salmonids compared with colonies located within the freshwater and mixing zones, and methods for monitoring predation rates at this colony have already been established. Double-crested cormorant abundance could also increase at other historical colony sites within the marine zone, as well as new sites that potentially include scattered navigation markers, as well as trees associated with Cape Disappointment and Fort Columbia state parks in Washington, and trees associated with Fort Stevens State Park in Oregon.

The Columbia River estuary's attractiveness to double-crested cormorants and other piscivorous birds is largely a function of its abundant food resources. A substantial proportion of available food is composed of juvenile salmonids (Bottom and Jones 1990, Weitkamp et al. 2012), most of which are hatchery-reared (ISAB 2011). As a consequence of hatchery inputs, the overall number and biomass of juvenile salmonids within the Columbia River estuary may be several times higher today than historically (ISAB 2011), which is remarkable given how much lower returning adult salmonid abundance is relative to historical levels. The high abundance of hatchery fish may represent a food subsidy to double-crested cormorants and other avian predators, sustaining them during periods when abundance of schooling marine forage fish is low (Phillips et al. 2017). This may be especially true when other food sources may be limited within the estuary, such as during spring (Bottom and Jones 1990), or during years marked by high river flows (Weitkamp et al. 2012, Lyons et al. 2014a). Consequently, an evaluation of hatchery practices may be warranted to determine the feasibility of reducing salmonid vulnerability to avian predation, which in turn may reduce the fidelity of avian predators to the estuary, especially if available food in spring is a potential limiting factor. Of particular concern may be hatchery releases of coho and Chinook salmon that occur within the Columbia River estuary. These releases appear particularly susceptible to double-crested cormorant predation based on observations at fish release sites (ODFW unpubl. data) and predation rate estimates for PIT-tagged runs (Roby et al. 2021, ODFW unpubl. data).

Although our scientific understanding of avian predation in the Columbia River basin is rapidly advancing, the effect of avian predation on salmonid runs is still unclear, largely because of the inherent challenge in demonstrating clear cause-and-effect relationships in complex, multispecies food webs (Sih et al. 1998, Yodzis 2001). Nevertheless, managers must make decisions in the face of scientific uncertainty, relying on the weight of accumulated evidence and using the precautionary principle as needed to protect sensitive fish populations. The weight of accumulated evidence to date indicates some avian predators, including double-crested cormorants, may exert at least some effect on life-cycle scale survival for some Columbia River basin salmonid runs (Lyons et al. 2014b, Evans et al. 2016, Evans et al. 2019, NMFS 2020, Payton et al. 2020, ISAB 2021, Payton et al. 2021; but see Haeseker et al. 2020 and DeHart 2021). However, despite the high abundance of double-crested cormorants within the estuary in recent years, the species represents only a fraction of the total number of smolt predators in the Columbia River estuary and plume. Within the estuary, Caspian terns are abundant and consume substantial numbers of juvenile salmonids, even after extensive management of their primary colony site on East Sand Island (Roby et al. 2021). Within the plume, the number of common murres (*Uria aalge*) and sooty shearwaters (*Ardenna grisea*) is more than an order of magnitude greater than the combined number of double-crested cormorants and Caspian terns in the estuary (Phillips et al. 2017). The abundance of non-avian predators in the estuary and plume is also high, with large numbers of Pacific harbor seals (*Phoca vitulina*; Jeffries et al. 2015) and various piscivorous fishes (Emmett and Krutzikowsky 2008) present during the spring smolt outmigration. Based on the diversity and abundance of these other predator species, predation by double-crested cormorants and Caspian terns—the only two predators whose impacts on juvenile salmonids are annually quantified in the estuary—may constitute a minority of aggregate predation across the estuary and plume. It follows that potential benefits to salmonids from double-crested cormorant management may be modest and therefore difficult to empirically verify. Moreover, the complexity of the predator community in the estuary and plume suggests management of double-crested cormorants could result in a compensatory increase in predation by other species, a phenomenon observed among some predator manipulation studies involving complex predator guilds (Errington 1946, Errington 1967, Ellis-Felege et al. 2012). Nevertheless, even a small or uncertain survival gain for salmonids is potentially important as part of a broad-based recovery strategy for salmonid runs in the Columbia River basin.

Finally, potential future management of double-crested cormorants in the Columbia River estuary may affect the species' regional population, which has already declined by perhaps 38% since management began on East Sand Island (USFWS 2020). In 2014, the year prior to implementation of the ESI management plan, the point estimate for the western population was 36,719 breeding pairs (95% CI = 33,562–39,875), while in 2019 it was 22,889 breeding pairs, although the precision associated with this estimate is low (95% CI = 15,925–29,855; USFWS 2020). The ESI management plan implicitly assumed that a large proportion of the regional double-crested cormorant population would be supported by the East Sand Island colony following plan implementation (USACE 2015). However, the recent collapse of this colony and the apparent failure by double-crested cormorants to reestablish a productive breeding colony there since at least 2018 (Turecek et al. 2019, USACE unpubl. data) suggest it may no longer represent a viable colony site. Thus, potential management of the Astoria-Megler Bridge colony and other non-ESI colony sites could eventually result in double-crested cormorants dispersing out of the estuary entirely instead of successfully recolonizing East Sand Island. It is unclear, however, whether successful dispersal to colonies outside of the Columbia River estuary would be likely, given the apparently tenuous status of colonies in many areas within the region (Adkins et al. 2014). Thus, future double-crested cormorant abundance within the region could decline to a level lower than originally predicted by the ESI management plan. Further, the poor precision of recent population estimates for the western population (USFWS 2020) and potential issues associated with recent monitoring of the western population (ODFW letter to USFWS, July 20, 2020) seem likely to complicate regional management decisions. An updated regional status assessment would benefit conservation planning for this species by clarifying regional population dynamics associated with the recent rapid population decline in the Columbia River estuary. A regional status assessment would ideally include a breeding census of the western population, an action previously recommended by researchers to occur following major changes to the status of the East Sand Island colony (Adkins et al. 2014).

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[1] As written in original source. The correct spelling is J. MacDonald.

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Figures and Tables

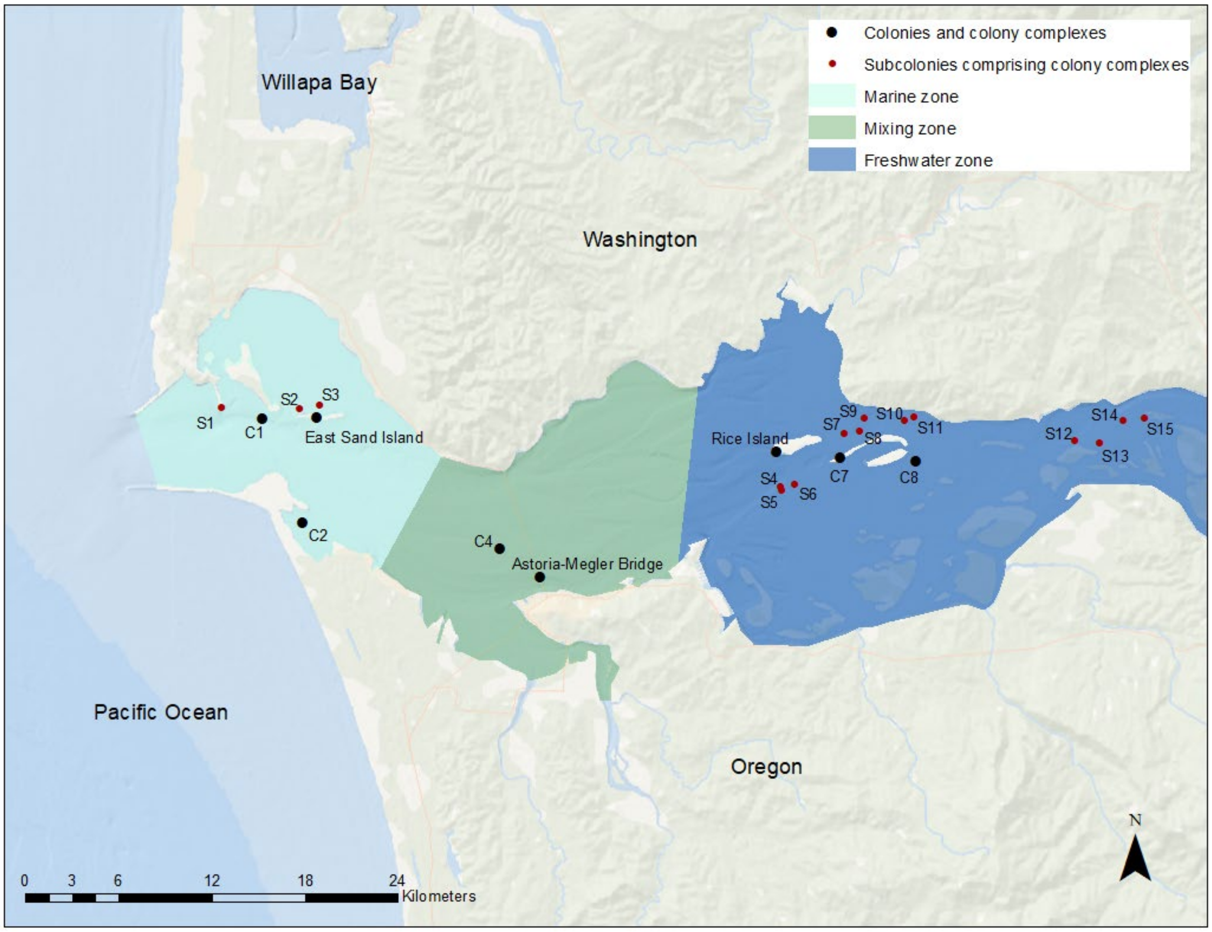

Figure 1a. Locations of double-crested cormorant colonies and colony complexes along the lower 55 km of the Columbia River estuary relative to salinity zones based on Simenstad et al. (1990) as modified by Anderson et al. (2004). Colony and sub-colony labels refer to colony names or ID codes in Tables 1 and 2.

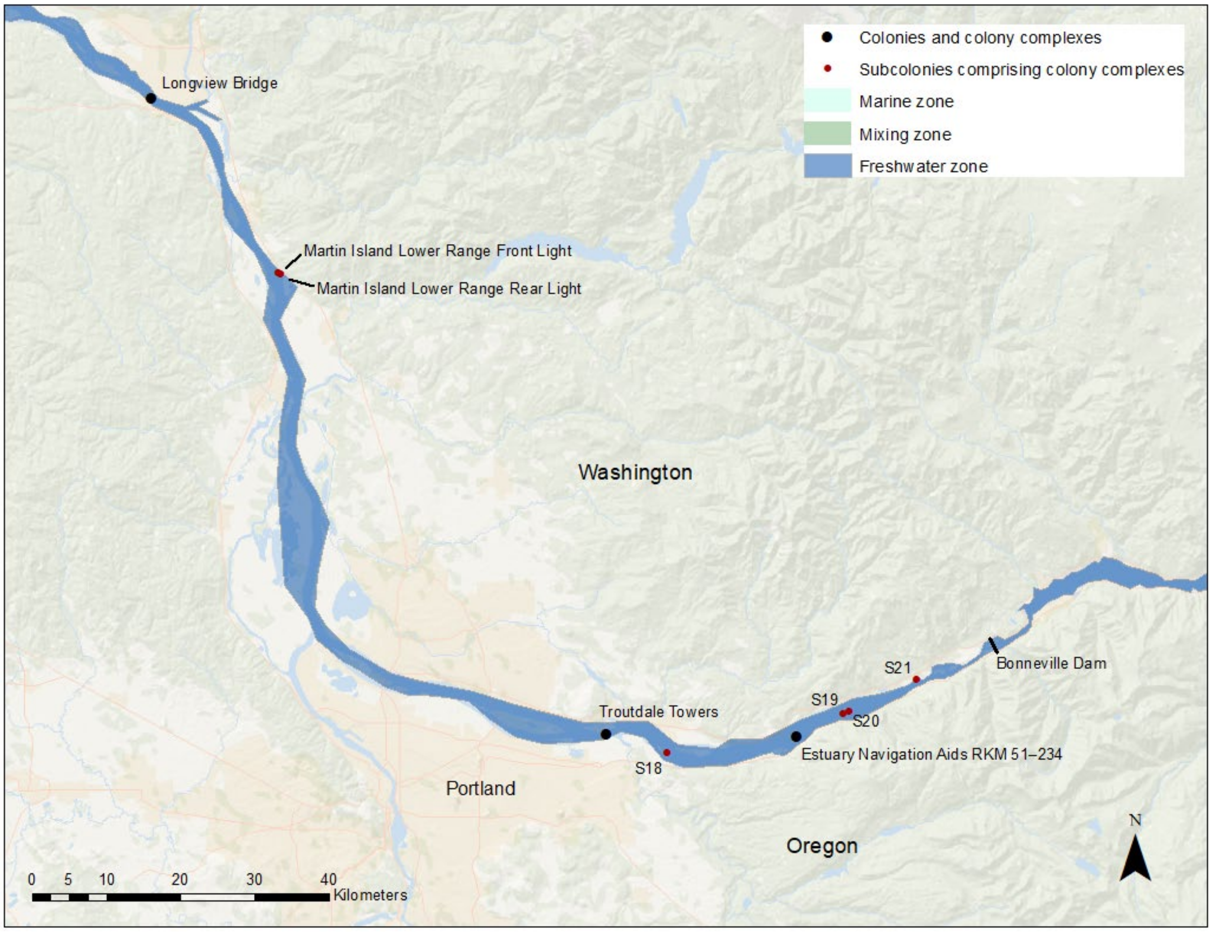

Figure 1b. Locations of double-crested cormorant colonies and colony complexes in the Columbia River estuary from river km 55 to Bonneville Dam (river km 234) relative to salinity zones based on Simenstad et al. (1990) as modified by Anderson et al. (2004). Colony and sub-colony labels refer to colony names or ID codes in Tables 1 and 2.

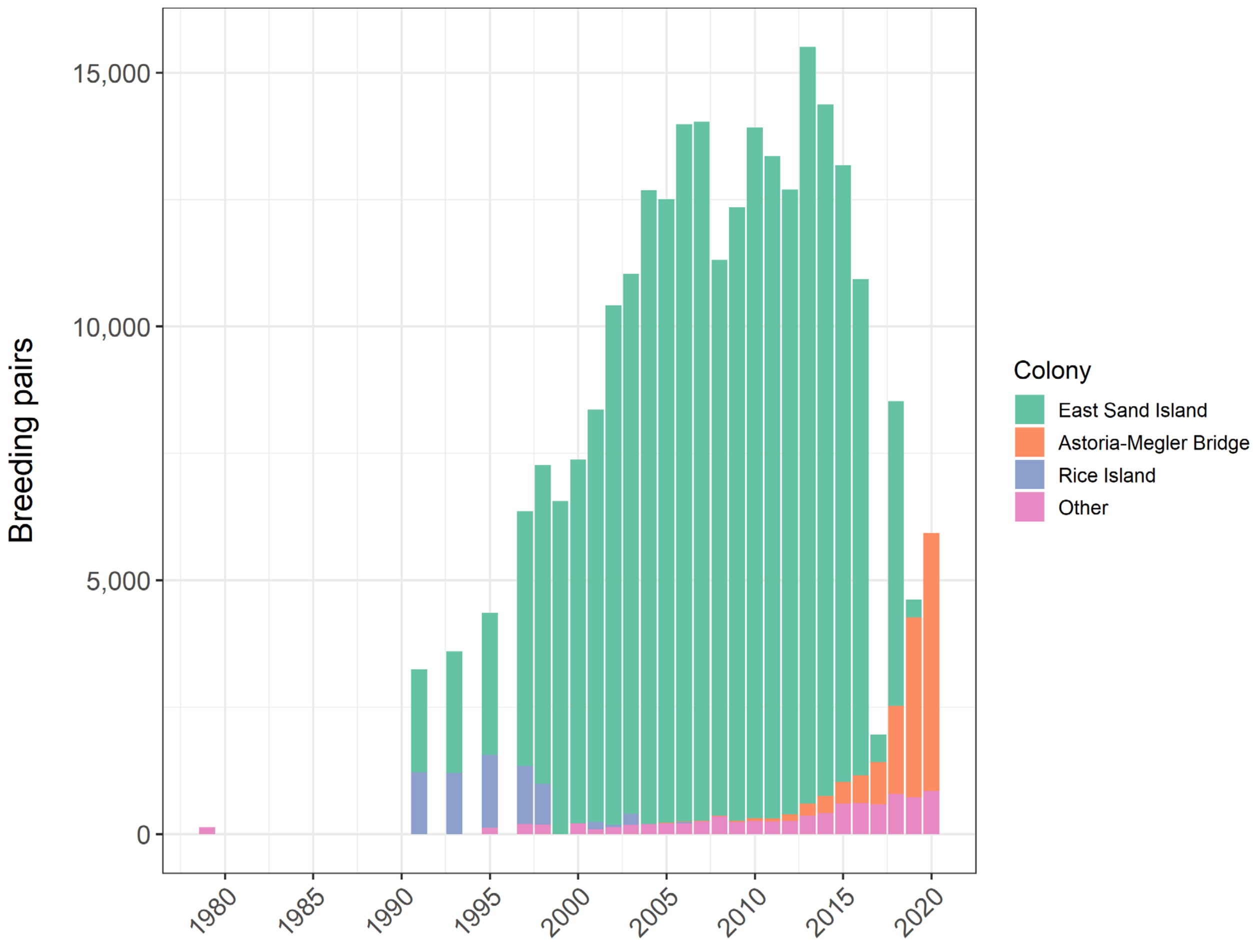

Figure 2. Number of double-crested cormorant breeding pairs nesting among all colony sites within the Columbia River estuary, 1979–2020. Graph only includes years when survey effort was presumed to reflect estuary-wide double-crested cormorant abundance. See Appendix B for data.

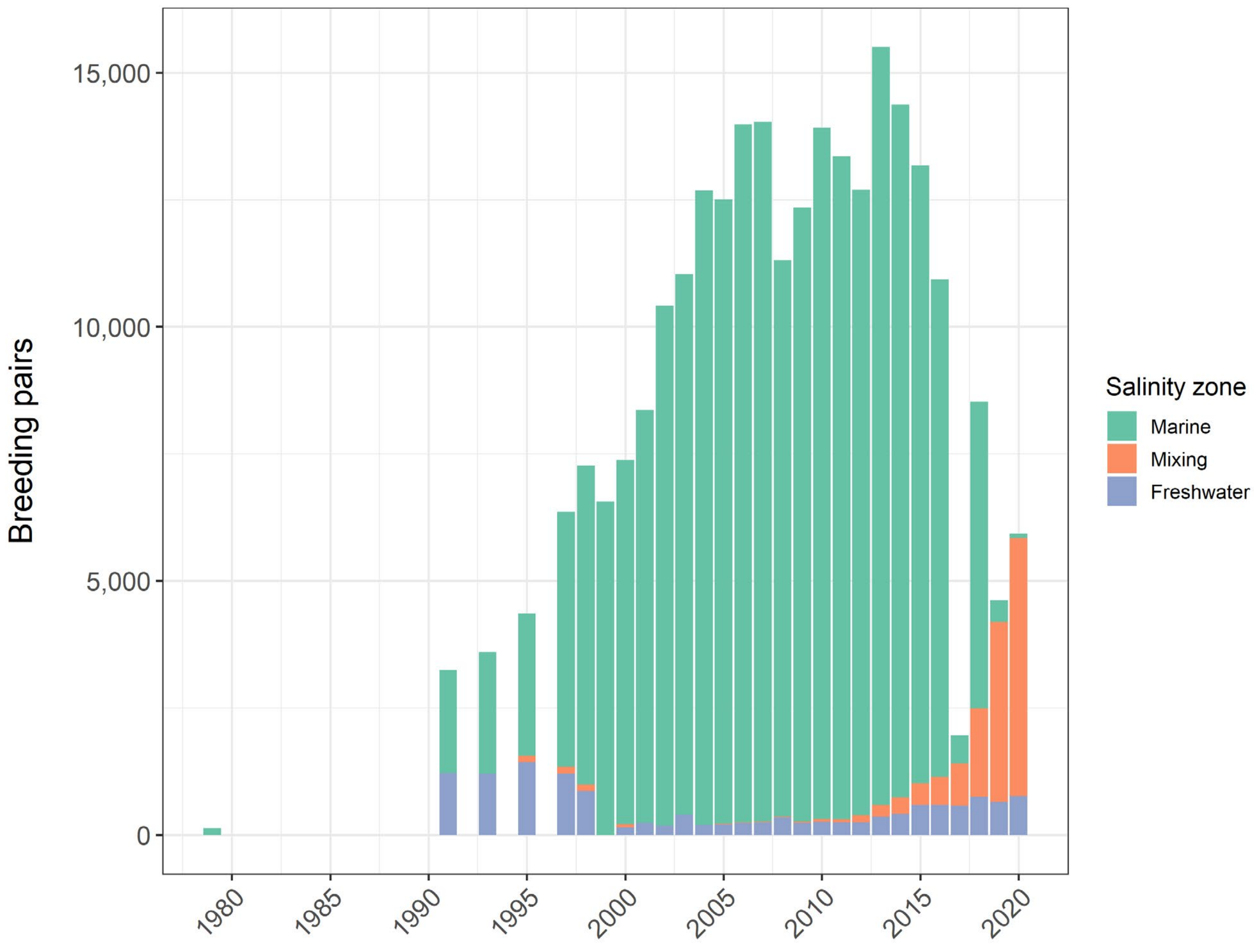

Figure 3. Number of double-crested cormorant breeding pairs nesting within three salinity zones of the Columbia River estuary, 1979–2020. Graph only includes years when survey effort was presumed to reflect estuary-wide double-crested cormorant abundance. See Appendix B for data.

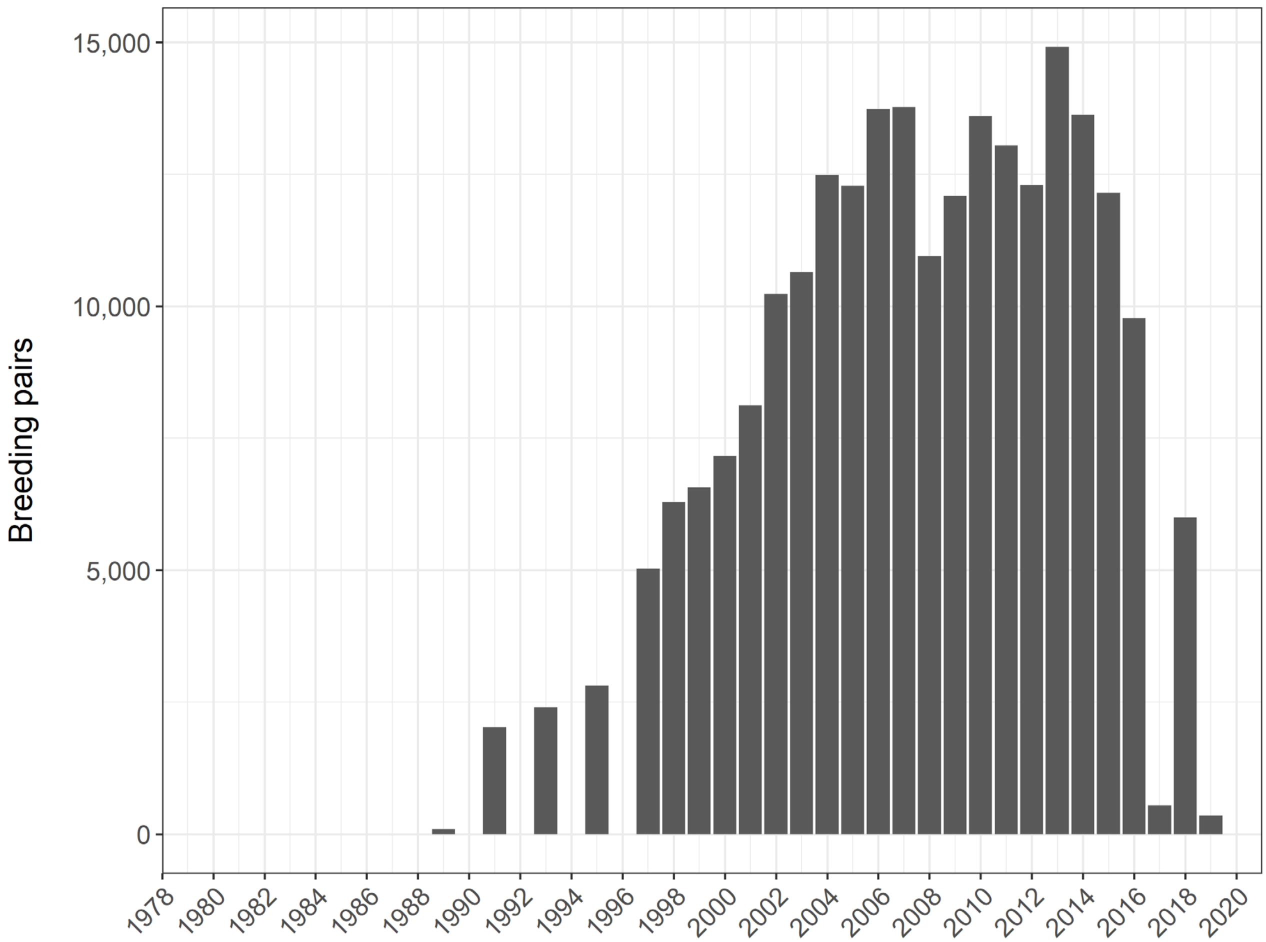

Figure 4. Number of double-crested cormorant breeding pairs nesting at East Sand Island in the Columbia River estuary during 1979–2020. See Appendix B for data.

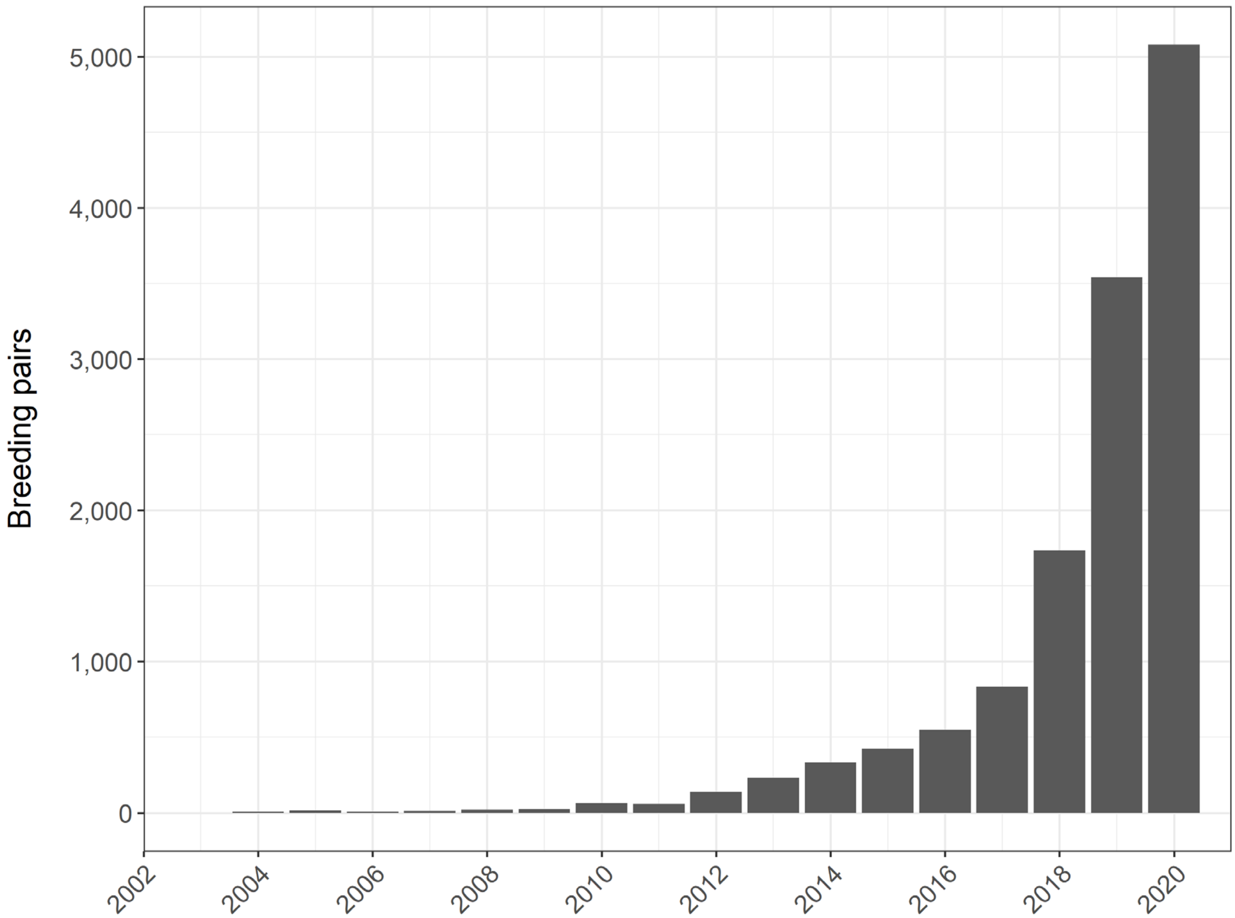

Figure 5. Number of double-crested cormorant breeding pairs nesting at the Astoria-Megler Bridge colony in the Columbia River estuary during 1979–2020. See Appendix B for data.

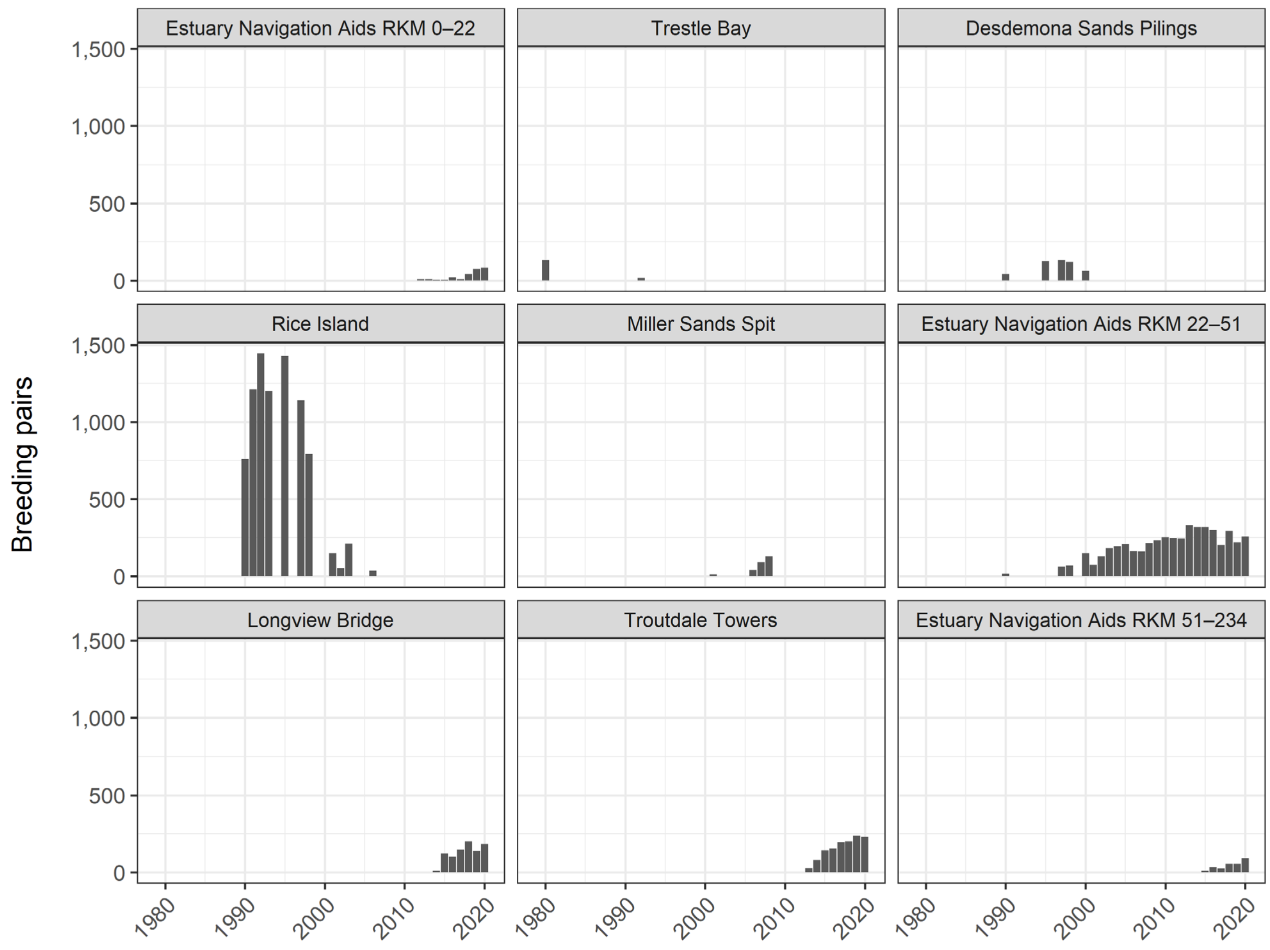

Figure 6. Number of double-crested cormorant breeding pairs nesting in the Columbia River estuary during 1979–2020 at known colony sites, excluding East Sand Island and the Astoria-Megler Bridge. See Appendix B for data.

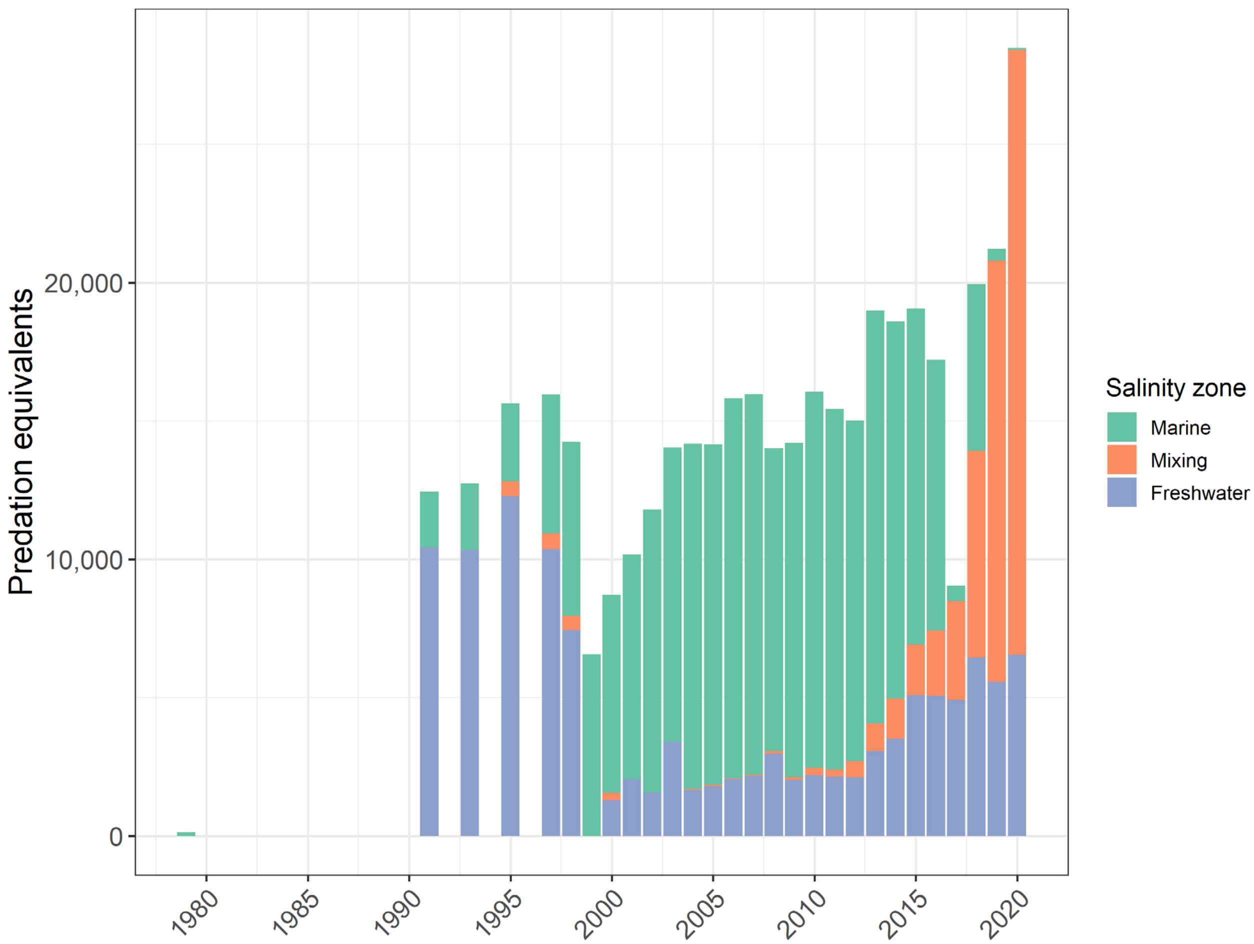

Figure 7. Estimated predation impact on juvenile steelhead for double-crested cormorants breeding in three salinity zones within the Columbia River estuary. Predation expressed as the number of breeding pairs on East Sand Island that would cause equivalent predation impacts (predation equivalents). Graph only includes years when survey effort was presumed to reflect estuary-wide double-crested cormorant abundance.

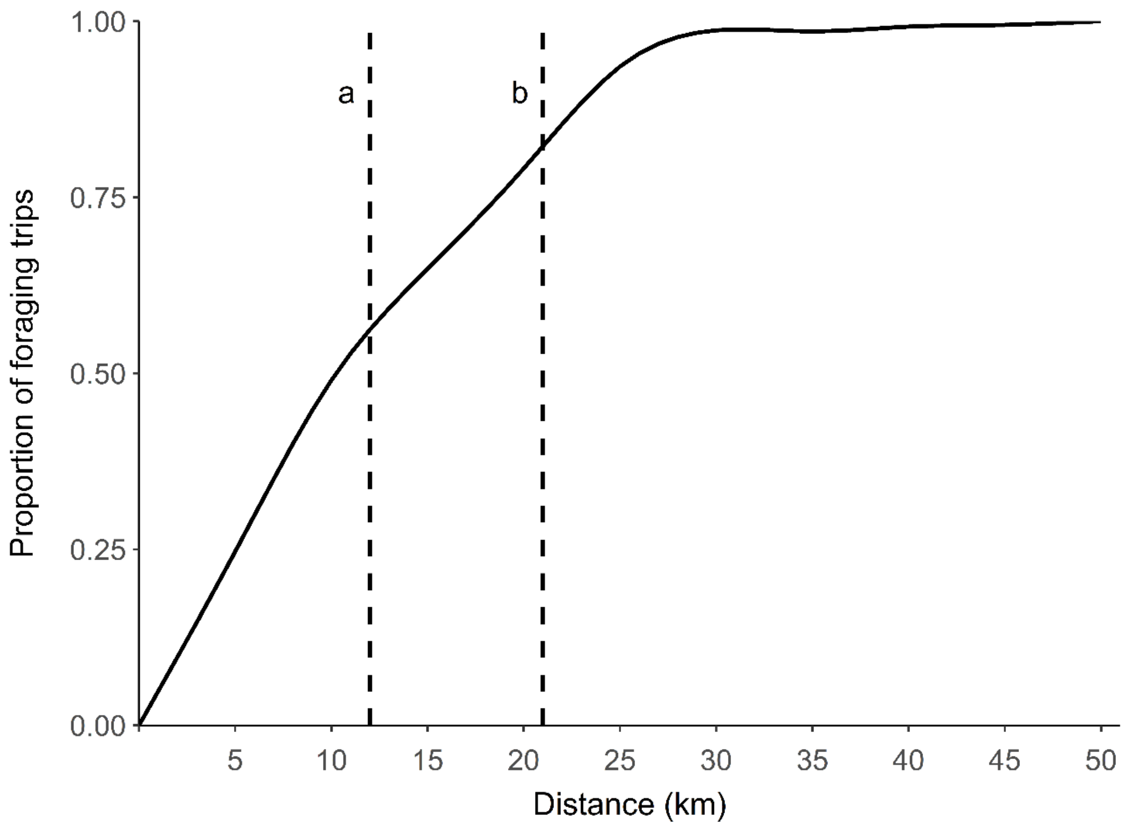

Figure 8. Modeled cumulative distribution of foraging distance for breeding double-crested cormorants at colony sites in the Columbia River estuary. Fitted curve derived from relocations of female and male double-crested cormorants radio-marked on East Sand Island, corrected to 50:50 sex ratio. Distance of colonies at (a) Astoria-Megler Bridge (12 km) (b) and East Sand Island (21 km) from former nesting colony on Rice Island (0 km) denoted by dashed lines. Data from Anderson et al. (2004).

Table 1. Location of known double-crested cormorant colony sites in the Columbia River estuary during 1979–2020.

ID	Colony name	Latitude (decimal degrees)	Longitude (decimal degrees)	Year first detected	Notes
C1	Estuary Navigation Aids RKM 0–22	46.261940	-124.013859	2015	Colony complex comprising 3 navigation markers along the stretch of river from the Columbia River mouth to the Astoria-Megler Bridge (ca. RKM 21.6). Each sub-colony is located on an individual navigation marker.
C2	Trestle Bay	46.220000	-123.990833	1980	Historic colony site located on abandoned trestle used for construction of south jetty at river mouth (CREST 1984)
C3	East Sand Island	46.262190	-123.982252	1988	Human-modified island located near site of historical natural island (USACE 2015).
C4	Desdemona Sands Pilings	46.209722	-123.876389	1990	Colony site located on pilings from historical Desdemona Sands Lighthouse. Probably unsuitable for nesting since at least early 2010s (Adam Peck-Richardson, Oregon State University, pers. comm.).
C5	Astoria-Megler Bridge	46.198015	-123.853266	2004	Nesting occurs along entire 6 km length, but most breeding associated with south-most 1.5 km segment of bridge, which supported 81% of nesting effort in 2020 (ODFW unpubl data).
C6	Rice Island	46.248694	-123.716442	1988	Artificial island composed of dredge material. Historical colony site at west end of island.
C7	Miller Sands Spit	46.246084	-123.679441	2001	Island largely created from dredge material. Historical colony site at west end of island.
C8	Estuary Navigation Aids RKM 22–51	46.244692	-123.635143	1980	Colony complex comprising 12 navigation markers along the stretch of river from

					Astoria-Megler Bridge upstream through river km 51. Each sub-colony is located on an individual navigation marker. This complex comprises all navigation aids from "Miller Sands Navigational Aids" and "Upper Estuary Navigational Aids" in Adkins and Roby (2010). At least 10–20 pairs present in 1980, when colony first observed (CREST 1984).
C9	Longview Bridge	46.104545	-122.961960	2014	Colony located on beams associated with two mid-channel piers supporting bridge.
C10	Troutdale Towers	45.567872	-122.412055	2013	Colony located on 5 power transmission towers 1 km downriver of mouth of Sandy River.
C11	Estuary Navigation Aids RKM 51–234	45.565447	-122.182918	2015	Colony complex comprising 6 navigation markers along stretch of river from river km 51 to Bonneville Dam (ca. river km 234). Each sub-colony is located on an individual navigation marker.

Table 2. Locations of constituent sub-colonies for double-crested colony complexes in the Columbia River estuary during 1979–2020.

Colony complex	ID	Sub-colony name	Latitude (decimal degrees)	Longitude (decimal degrees)	Year first detected	Notes
Estuary Navigation Aids RKM 0–22	S1	Jetty A Tower	46.265954	-124.037809	2015	
	S2	Sand Island Range Front Light	46.265832	-123.992948	2018	
	S3	Sand Island Range Rear Light	46.267293	-123.981073	2018	
Estuary Navigation Aids RKM 22–51	S4	Harrington Point Channel 52 Light	46.234162	-123.714198	≤2005	Upper Estuary Channel Marker (UECM) 1 in Roby et al. (2021)
	S5	Tongue Point Channel Range Front Light	46.232948	-123.713511	≤2005	UECM 2 in Roby et al. (2021)
	S6	Tongue Point Channel Range Rear Light	46.235152	-123.705794	≤2005	UECM 3 in Roby et al. (2021)
	S7	Harrington Point Range Front Light	46.255911	-123.677035	≤2005	UECM 4 in Roby et al. (2021)
	S8	Harrington Point Range Rear Light	46.256534	-123.668582	≤2005	UECM 5 in Roby et al. (2021)
	S9	Miller Sands Dike Light 5	46.261769	-123.665627	≤2005	UECM 6 in Roby et al. (2021)
	S10	Miller Sands Dike Light 11	46.261145	-123.641955	≤2005	UECM 7 in Roby et al. (2021)
	S11	Miller Sands Range Front Light	46.262415	-123.636661	≤2005	UECM 8 in Roby et al. (2021)
	S12	Pillar Rock Lower Range Front Light	46.252761	-123.543447	≤2009	UECM 9 in Roby et al. (2021)
	S13	Pillar Rock Lower Range Rear Light	46.251728	-123.529404	≤2009	UECM 10 in Roby et al. (2021)
	S14	Pillar Rock Upper Range Front Light	46.260721	-123.515554	≤2009	UECM 11 in Roby et al. (2021)
	S15	Pillar Rock Upper Range Rear Light	46.261706	-123.502956	≤2009	UECM 12 in Roby et al. (2021)
Estuary Navigation Aids RKM 51–234	S16	Martin Island Lower Range Front Light	45.957934	-122.808994	2020	
	S17	Martin Island Lower Range Rear Light	45.955872	-122.806304	2018	
	S18	Washougal Upper Range Rear Light	45.551788	-122.339813	2016	
	S19	Fashion Reef Lower Range Front Light	45.585095	-122.127023	2016	
	S20	Fashion Reef Lower Range Rear Light	45.586233	-122.119301	2015	
	S21	Warrendale Lower Range Rear Light	45.613594	-122.037611	2016	Apparently misnamed "Warrenton Lower Range Rear Light" in several instances in Anchor QEA (2017).

Table 3. Peak size of the East Sand Island double-crested cormorant colony during management phases associated with the management plan *Double-crested Cormorant Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (USACE 2015), and the period of peak abundance prior to management, 2004–2014.

Management phase	Year	Peak size	Average size
Pre-management	2004–2013	10,950–14,916	12,917
Peak abundance	2004–2014	10,950–14,916	12,982
Phase 1	2015	12,150	7,489
	2016	9,772	
	2017	544	
Phase 2	2018	5,999	2,116
	2019	350	
	2020	0	

Table 4. Adaptive management measures proposed in the federal management plan *Double-crested Cormorant Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (USACE 2015). These measures were proposed for the Astoria-Megler Bridge and other colony sites in case of dispersal associated with management of the East Sand Island double-crested cormorant colony. However, hazing has not been implemented on the Astoria-Megler Bridge, despite double-crested cormorant dispersal to this colony site. Table represents reproduced image of Table 5-2 from page 5-11 of USACE (2015).

Key Estuary Monitoring/Hazing Locations*	Monitoring and Hazing Protocols
Astoria-Megler Bridge	1) Begin surveys in April in areas where suitable nesting habitat is present to detect incipient nesting attempts before eggs are laid. Continue surveys through mid-June or until nest initiation has stopped. 2) Once DCCOs are detected on suitable nesting habitat, use binoculars and/or spotting scope to count the number of individuals and determine whether the birds are roosting or initiating nesting. 3) Coordinate with the landowner/state and federal resource agencies for access and regarding management activities where species of concern occur, such as occupied streaked horned lark critical habitat. 4) Use non-lethal methods to deter nesting, primarily by human hazing. 5) Collect eggs only under approved USFWS permit. Record and report any eggs collected.
Tongue Point Piers	
Trestle Bay	
Rice Island	
Miller Sands Spit	
Pillar Rock Island	
Lewis and Clark Bridge	
Troutdale Transmission Tower	
Willamette Falls/Oregon City	

*Additional locations for hazing would be determined from the results of surveys and monitoring.

Appendix A: Synopsis of Management, 2015–2020

A.1. Proposed Plan

Under the management plan entitled *Double-crested Cormorant Management Plan to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary* (ESI management plan; USACE 2015), the U.S. Army Corps of Engineers (Corps) proposed to reduce the size of the double-crested cormorant colony on East Sand Island to 5,380–5,939 breeding pairs. This compares with the colony's average annual abundance of 12,917 breeding pairs during 2004–2013, the baseline abundance period for the ESI management plan (USACE 2015). The reduction in colony size was to occur primarily by means of lethal take of individuals and nest contents over the course of four consecutive years (phase 1) followed by restricting available nesting habitat by permanently modifying a portion of the island to allow for periodic tidal flooding (phase 2). To meet reductions in double-crested cormorant abundance as modelled under the plan, Phase 1 was to consist of culling adult individuals during the early portion of the breeding season during 2015–2018, which would result in loss of both adult individuals and their associated nests; 13.5% of adults associated with the colony were to be culled annually, amounting to a total of 10,912 individual double-crested cormorants over the four-year period. Culling of adult individuals during the post-breeding period was also possible under the plan, though this option would not reduce nest abundance to the degree modelled in the ESI management plan. In addition, 46% of active nests were to be sprayed with corn oil annually, which would suffocate the eggs and thereby reduce the number of fledged chicks produced by the colony; egg-oiling was to occur during the first three years of the plan (2015–2017). The total number of nests to be oiled was 15,184. Lethal take was not planned as a primary tool during phase 2; however, managers anticipated the need to take a limited number of eggs during phase 2 if non-lethal hazing efforts could not completely preclude nesting on undesired areas of East Sand Island.

A.2. Management Synopsis

A brief synopsis of management is provided below. Unless otherwise stated, I obtained this information from unpublished documents provided by the Corps and personal communication with staff from the Corps, its contractors, and the U.S. Department of Agriculture–Wildlife Services. All lethal take described in this section was performed under permit with the U.S. Fish and Wildlife Service.

Phase 1 of the ESI management plan extended from 2015 until 2017. Overall, 5,576 adult double-crested cormorants were culled, with 2,346, 2,982, and 248 taken in 2015, 2016, and

2017 respectively. The total number of oiled nests was 6,181, with 5,089, 1,092, and 0 nests oiled in 2015, 2016, and 2017, respectively. During 2015, culling of adult double-crested cormorants began in late-May, but only occurred on a limited basis to avoid taking adults associated with nests containing chicks. During late May through early July, 158 adults were culled on the island by means of .22 caliber rifle. Nests were oiled on two dates, with 1,769 oiled during daylight on May 23, and 3,320 nests oiled at night on June 29. Most take of adult double-crested cormorants during 2015 occurred following the nesting season during the interval including the week of Sept. 8 through the week of Oct. 28, when 2,346 individuals were culled by boat-based shooting at locations at least 500 m away from the colony site. Managers did not report any major colony disturbances in 2015. Although colony productivity (i.e. the number of chicks fledged per nest) was apparently not monitored during 2015, it is presumed to have been low as a planned result of management activity. In contrast to 2015, large numbers of double-crested cormorants were culled in 2016 prior to nest initiation and during the early incubation period. During 2016, 1,125 double-crested cormorants were culled during April and 1,269 were culled during early to mid-May; all of these were culled at least 500 m away from the colony by workers operating from watercraft. Workers oiled nests on only one occasion in 2016, when 1,089 nests were oiled on the night of May 10, about 16% of the 6,644 nests present during the week of May 9 (Anchor QEA 2017). Thereafter, a complete colony abandonment occurred at some point during May 12–May 16. This was apparently the first complete colony abandonment recorded at the East Sand Island colony since monitoring began in 1997 (Dan Roby, Oregon State University, pers. comm.). There was no evidence that colony disturbance by bald eagles was a contributing factor, and no other obvious sources of disturbance are known. Double-crested cormorants resumed nesting on East Sand Island by late June–early July, but apparently only about 3,000 nests persisted into early August (Anchor QEA 2017), long enough to have advanced to the chick stage. Colony productivity was not determined in 2016 but was likely low considering the late date the colony reformed relative to the typical seasonal nesting period (Perrins 1970). Workers culled an additional 588 double-crested cormorants during October 2016 via boat-based shooting. During 2017, workers culled 248 adult double-crested cormorants, all during April, before management was halted because of a lack of colony formation associated with persistent bald eagle disturbances. The colony was only intermittently active during May and June, but was not continuously active until mid-July, when it supported a peak of only 544 pairs (Turecek et al. 2018). No further management occurred in 2017. Colony productivity was not determined in 2017 but was likely low (Turecek et al. 2018). In contrast to the delayed breeding effort on East Sand Island during 2017, nesting chronology appeared normal for the Astoria-Megler Bridge and other non-ESI colony sites in 2017, with nest initiation generally occurring during early-April–early May. Productivity at the Astoria-Megler Bridge colony was not formally quantified in 2017, but appeared to be high (J. Tennyson, Real Time Research, pers. comm.). Phase 1 of the management plan ended in 2017, one year earlier than planned, because the number of double-crested cormorants breeding on East Sand Island in 2017 was substantially lower than anticipated by the ESI management plan.

Phase 2 of the ESI management plan began in 2018. A visual barrier fence was maintained near the west end of the island during 2018–2020, and double-crested cormorants were dissuaded from nesting east of this fence; at the same time, nesting habitat was maintained west of the fence (sanctuary area). Dissuasion implemented east of the fence included passive techniques such as use of fladry, balloons, and predator effigies, and by active techniques including workers actively frightening individuals and destroying nests as needed. In early 2019, a large proportion of previously used nesting habitat on the island was rendered unusable for nesting by modifying it with heavy equipment to allow for tidal inundation; this habitat modification occurred when double-crested cormorants were absent from the colony during the winter season. Habitat modification was completed in March 2019, several weeks prior to the typical period of colony initiation. About 0.7 ha of nesting habitat remained available for double-crested cormorant nesting in the sanctuary area during 2018–2020. During 2018–2020, dissuasion was largely successful at preventing double-crested cormorant use of habitat east of the fence, despite frequent attempts by individuals to use this area for nesting during 2018 and 2019. Workers destroyed nests under construction east of the fence as needed, generally before eggs were laid. However, egg take was required for nests initiated outside the sanctuary area during both 2018 and 2019, when workers took 3 and 97 eggs under permit, respectively; no eggs were taken in 2020.

Double-crested cormorant attendance at the East Sand Island colony during 2018–2020 was delayed and erratic throughout much of the typical mid-April–mid-August nesting period, with disturbances by bald eagles a major contributing factor. Periods of sustained nesting activity did not occur until about mid-July during 2018 and 2019 (Turecek et al. 2019, USACE unpubl. data), and no sustained nesting apparently occurred in 2020. In 2018, researchers estimated 1.8 young were fledged per active nest on East Sand Island (95% CI = 0.98–2.66; Turecek et al. 2019); however, the uncertainty associated with this estimate is high, and the estimate does not appear to account for the large number of active nests on the colony prior to a major colony abandonment in late May/early June. In 2019, few chicks of advanced age were observed on the colony (USACE unpubl. data), and it is unknown whether any of these fledged. In contrast to East Sand Island, nesting chronology appeared normal for the Astoria-Megler Bridge and other non-ESI colony sites during 2018–2020, with nest initiation occurring during early-April–early May (Oregon Department of Fish and Wildlife unpubl. data). Productivity at the Astoria-Megler Bridge colony appeared high during 2018–2020, based on both qualitative and quantitative information, apparently as high or higher than at the ESI colony prior to management (Turecek et al. 2019; M. J. Lawonn, pers. obs.). In 2018, the only year when quantitative data were collected, nesting success on the Astoria-Megler Bridge was 2.7 young raised/active nest (95% CI=2.55-2.93), higher than average productivity on East Sand Island during 1999–2013 (1.86 young raised/breeding pair; Lawes et al. 2021).

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Appendix B: Summary of Double-crested Cormorant Survey Data, 1979–2020

B.1. Analysis

This appendix provides annual abundance of double-crested cormorant nesting pairs at colonies located within the Columbia River estuary (Tables B1, B2). I considered the Columbia River estuary to consist of the Columbia River mainstem from its mouth to river km 234, at Bonneville Dam (Simenstad et al. 2011). I obtained survey data by conducting a thorough literature search and compiling annual survey data for breeding colonies. Additional unpublished data were provided by regional biologists and managers. I assumed surveys were conducted during the typical peak period of breeding abundance, although timing was not reported for some surveys. In cases where the same colony was surveyed several times in a season, peak abundance of active nests was used as a measure of annual abundance (Pacific Flyway Council 2013). Survey data considered to be of marginal quality, including incomplete colony surveys, were not included in this dataset. I defined active nests as those containing at least one viable egg or chick, or with at least one adult in attendance (i.e. sitting or standing on nest materials). This definition of active nests was used to maintain consistency with established survey protocols for the Pacific Flyway (Pacific Flyway Council 2013) and summaries for the Columbia River estuary during 1979–2009 (Naughton et al. 2007, Adkins and Roby 2010). In cases where the peak number of active nests was unclear from available data, I used the peak count as recognized by the U.S. Fish and Wildlife Service (USFWS) and Pacific Flyway Council. For colony complexes occurring on navigation aids during 2010–2020, the annual peak aggregate abundance of constituent sub-colonies (generally surveyed on the same date or week) is reported here to maintain consistency with previously summarized data (Naughton et al. 2007, Adkins and Roby 2010). The timing of peak abundance for individual sub-colonies, therefore, does not necessarily correspond with the timing of peak abundance for the complex as a whole.

I assumed that surveys conducted in 1979, 1991, 1993, 1995, and 1999 were sufficiently comprehensive to approximate the breeding abundance of double-crested cormorants across the entire Columbia River estuary. However, it is unclear from available sources whether colonies besides East Sand Island and Rice Island may have been active within the lower estuary (river kilometers 0–51) during these years but were not surveyed. Thus, total abundance of double-crested cormorants in the Columbia River estuary may be underestimated for these years. However, based on the low number of nesting double-crested cormorants from locations besides East Sand Island and Rice Island prior to 2015, any associated error seems likely to be minimal. Likewise, I assumed that annual survey totals accurately reflected double-crested cormorant abundance across the entire upper estuary (river kilometer 51–234). Although surveys in this area apparently occurred only intermittently prior to 2015, it is likely that if colonies had been present they would have been detected, owing to the accessibility of the area and the extensive research and management effort associated with avian and pinniped predation downstream of Bonneville Dam in recent years.

Table B1. Abundance of double-crested cormorant nesting pairs at colony sites in the Columbia River estuary during 1979–2020. Counts reflect annual peak abundance of active nests (i.e. breeding pairs). Colony details listed in main document (Table 1, Table 2). Abbreviations defined in Table B2.

Year	Colony/colony complex	Nests	Source
1979	East Sand Island	0	Carter et al. 1995
1979	Desdemona Sands Pilings	0	Carter et al. 1995
1979	Rice Island	0	Carter et al. 1995
1979	Estuary Navigation Aids RKM 22–51	0	Carter et al. 1995
1980	Trestle Bay	131[1]	Carter et al. 1995
1988	East Sand Island	Present	Carter et al. 1995
1988	Rice Island	Present	Carter et al. 1995
1989	East Sand Island	91	Naughton et al. 2007
1990	Desdemona Sands Pilings	40	Carter et al. 1995
1990	Rice Island	761	Naughton et al. 2007
1990	Estuary Navigation Aids RKM 22–51	17	Carter et al. 1995
1991	East Sand Island	2,026	Carter et al. 1995
1991	Rice Island	1,211	Carter et al. 1995
1992	Trestle Bay	16	Carter et al. 1995
1992	Rice Island	1,444	A. Clark, USFWS Willapa NWR Complex, unpubl. data
1992	Estuary Navigation Aids RKM 22–51	54	Carter et al. 1995
1993	East Sand Island	2,400	A. Clark, USFWS Willapa NWR Complex, unpubl. data
1993	Rice Island	1,201	A. Clark, USFWS Willapa NWR Complex, unpubl. data
1995	East Sand Island	2,808	A. Clark, USFWS Willapa NWR Complex, unpubl. data
1995	Estuary Navigation Aids RKM 22–51	125	A. Clark, USFWS Willapa NWR Complex, unpubl. data
1995	Rice Island	1,428	A. Clark, USFWS Willapa NWR Complex, unpubl. data
1997	East Sand Island	5,023	Naughton et al. 2007
1997	Desdemona Sands Pilings	130	Naughton et al. 2007
1997	Rice Island	1,141	Naughton et al. 2007
1997	Miller Sands Spit	0	OSU unpubl. data

[1] Included as component of 1979 estimate of estuary-wide abundance following Carter et al. (1995).

Year	Colony/colony complex	Nests	Source
1997	Estuary Navigation Aids RKM 22–51	64	Naughton et al. 2007
1998	East Sand Island	6,285	Adkins and Roby 2010
1998	Desdemona Sands Pilings	120	Adkins and Roby 2010
1998	Rice Island	795	Adkins and Roby 2010
1998	Miller Sands Spit	0	Adkins and Roby 2010
1998	Estuary Navigation Aids RKM 22–51	70	Adkins and Roby 2010
1999	East Sand Island	6,561	Adkins and Roby 2010
1999	Astoria-Megler Bridge	0	Adkins and Roby 2010
1999	Rice Island	0	Adkins and Roby 2010
1999	Miller Sands Spit	0	Adkins and Roby 2010
2000	East Sand Island	7,162	Adkins and Roby 2010
2000	Desdemona Sands Pilings	61	Adkins and Roby 2010
2000	Astoria-Megler Bridge	0	Adkins and Roby 2010
2000	Rice Island	0	Adkins and Roby 2010
2000	Miller Sands Spit	0	Adkins and Roby 2010
2000	Estuary Navigation Aids RKM 22–51	150	Adkins and Roby 2010
2001	East Sand Island	8,120	Adkins and Roby 2010
2001	Desdemona Sands Pilings	0	Adkins and Roby 2010
2001	Astoria-Megler Bridge	0	Adkins and Roby 2010
2001	Rice Island	150	Adkins and Roby 2010
2001	Miller Sands Spit	12	Adkins and Roby 2010
2001	Estuary Navigation Aids RKM 22–51	75	Adkins and Roby 2010
2002	East Sand Island	10,230	Adkins and Roby 2010
2002	Desdemona Sands Pilings	0	Adkins and Roby 2010
2002	Astoria-Megler Bridge	0	Adkins and Roby 2010
2002	Rice Island	53	Adkins and Roby 2010
2002	Miller Sands Spit	0	Adkins and Roby 2010
2002	Estuary Navigation Aids RKM 22–51	129	Adkins and Roby 2010
2003	East Sand Island	10,646	Adkins and Roby 2010
2003	Desdemona Sands Pilings	0	Adkins and Roby 2010
2003	Astoria-Megler Bridge	0	Adkins and Roby 2010
2003	Rice Island	211	Adkins and Roby 2010
2003	Miller Sands Spit	0	Adkins and Roby 2010
2003	Estuary Navigation Aids RKM 22–51	183	Adkins and Roby 2010
2004	East Sand Island	12,480	Adkins and Roby 2010
2004	Desdemona Sands Pilings	0	Adkins and Roby 2010

Year	Colony/colony complex	Nests	Source
2004	Astoria-Megler Bridge	6	Adkins and Roby 2010
2004	Rice Island	0	Adkins and Roby 2010
2004	Miller Sands Spit	0	Adkins and Roby 2010
2004	Estuary Navigation Aids RKM 22–51	194	Adkins and Roby 2010
2005	East Sand Island	12,287	Adkins and Roby 2010
2005	Desdemona Sands Pilings	0	Adkins and Roby 2010
2005	Astoria-Megler Bridge	14	Adkins and Roby 2010
2005	Rice Island	0	Adkins and Roby 2010
2005	Miller Sands Spit	0	Adkins and Roby 2010
2005	Estuary Navigation Aids RKM 22–51	208	Adkins and Roby 2010
2006	East Sand Island	13,738	Adkins and Roby 2010
2006	Desdemona Sands Pilings	0	Adkins and Roby 2010
2006	Astoria-Megler Bridge	7	Adkins and Roby 2010
2006	Rice Island	35	Adkins and Roby 2010
2006	Miller Sands Spit	41	Adkins and Roby 2010
2006	Estuary Navigation Aids RKM 22–51	162	Adkins and Roby 2010
2007	East Sand Island	13,771	Adkins and Roby 2010
2007	Desdemona Sands Pilings	0	Adkins and Roby 2010
2007	Astoria-Megler Bridge	11	Adkins and Roby 2010
2007	Rice Island	0	Adkins and Roby 2010
2007	Miller Sands Spit	90	Adkins and Roby 2010
2007	Estuary Navigation Aids RKM 22–51	160	Adkins and Roby 2010
2008	East Sand Island	10,950	Adkins and Roby 2010
2008	Desdemona Sands Pilings	0	Adkins and Roby 2010
2008	Astoria-Megler Bridge	20	Adkins and Roby 2010
2008	Rice Island	0	Adkins and Roby 2010
2008	Miller Sands Spit	129	Adkins and Roby 2010
2008	Estuary Navigation Aids RKM 22–51	216	Adkins and Roby 2010
2009	East Sand Island	12,087	Adkins and Roby 2010
2009	Desdemona Sands Pilings	0	Adkins and Roby 2010
2009	Astoria-Megler Bridge	24	Adkins and Roby 2010
2009	Rice Island	0	Adkins and Roby 2010
2009	Miller Sands Spit	0	Adkins and Roby 2010
2009	Estuary Navigation Aids RKM 22–51	235	Adkins and Roby 2010
2010	Trestle Bay	0	Presumed based on Dan Roby, OSU, pers. comm.

Year	Colony/colony complex	Nests	Source
2010	East Sand Island	13,596	Roby et al. 2011
2010	Desdemona Sands Pilings	0	Dan Roby, OSU, pers. comm.
2010	Astoria-Megler Bridge	63	Roby et al. 2011
2010	Rice Island	0	Roby et al. 2011
2010	Miller Sands Spit	0	Roby et al. 2011
2010	Estuary Navigation Aids RKM 22–51	254	Roby et al. 2011
2011	Trestle Bay	0	Adam Peck-Richardson, OSU, pers. comm.
2011	East Sand Island	13,045	Roby et al. 2012
2011	Desdemona Sands Pilings	0	Dan Roby, OSU, pers. comm.
2011	Astoria-Megler Bridge	60	Roby et al. 2012
2011	Rice Island	0	Roby et al. 2012
2011	Miller Sands Spit	0	Roby et al. 2012
2011	Estuary Navigation Aids RKM 22–51	248	Roby et al. 2012
2012	Estuary Navigation Aids RKM 0–22	8	Roby et al. 2021
2012	Trestle Bay	0	Adam Peck-Richardson, OSU, pers. comm.
2012	East Sand Island	12,301	Roby et al. 2014
2012	Desdemona Sands Pilings	0	Dan Roby, OSU, pers. comm.
2012	Astoria-Megler Bridge	139	Roby et al. 2013
2012	Rice Island	0	Roby et al. 2013
2012	Miller Sands Spit	0	Roby et al. 2013
2012	Estuary Navigation Aids RKM 22–51	245	Roby et al. 2013
2013	Estuary Navigation Aids RKM 0–22	7	Roby et al. 2021
2013	Trestle Bay	0	Adam Peck-Richardson, OSU, pers. comm.
2013	East Sand Island	14,916	Roby et al. 2014
2013	Desdemona Sands Pilings	0	Dan Roby, OSU, pers. comm.
2013	Astoria-Megler Bridge	231	Roby et al. 2014
2013	Rice Island	0	Roby et al. 2014
2013	Miller Sands Spit	0	Roby et al. 2014
2013	Estuary Navigation Aids RKM 22–51	330	Roby et al. 2014
2013	Troutdale Towers	26[2]	Adam Peck-Richardson, OSU, unpubl. data

[2]Minimum estimate of nests active on 20 April.

Year	Colony/colony complex	Nests	Source
2014	Estuary Navigation Aids RKM 0–22	5	Roby et al. 2021
2014	Trestle Bay	0	Adam Peck-Richardson, OSU, pers. comm.
2014	East Sand Island	13,626	Roby et al. 2015
2014	Desdemona Sands Pilings	0	Dan Roby, OSU, pers. comm.
2014	Astoria-Megler Bridge	333	Roby et al. 2015
2014	Rice Island	0	Roby et al. 2015
2014	Miller Sands Spit	0	Roby et al. 2015
2014	Estuary Navigation Aids RKM 22–51	318	Roby et al. 2015
2014	Longview Bridge	11	Roby et al. 2015
2014	Troutdale Towers	80	Roby et al. 2015
2015	Estuary Navigation Aids RKM 0–22	5	DSA 2016
2015	Trestle Bay	0	DSA 2016
2015	East Sand Island	12,150	DSA 2016
2015	Desdemona Sands Pilings	0	DSA 2016
2015	Astoria-Megler Bridge	425	DSA 2016
2015	Rice Island	0	USACE unpubl. data
2015	Miller Sands Spit	0	USACE unpubl. data
2015	Estuary Navigation Aids RKM 22–51	319	DSA 2016
2015	Longview Bridge	122	DSA 2016
2015	Troutdale Towers	141	DSA 2016
2015	Estuary Navigation Aids RKM 51–234	9	DSA 2016
2016	Estuary Navigation Aids RKM 0–22	18	Anchor QEA 2017
2016	Trestle Bay	0	Anchor QEA 2017
2016	East Sand Island	9,772[3]	Anchor QEA 2017
2016	Desdemona Sands Pilings	0	Anchor QEA 2017
2016	Astoria-Megler Bridge	549[4]	Anchor QEA 2017
2016	Rice Island	0	USACE unpubl. data
2016	Miller Sands Spit	0	USACE unpubl. data
2016	Estuary Navigation Aids RKM 22–51	300	Anchor QEA 2017

[3]Represents abundance during the week of July 3, following return of adults to breeding colony following abandonment extending from week of May 17 through week of June 27. Most breeding pairs assumed to be initiating nests or in the early-incubation period at time of survey.

[4]The source for this estimate reports an “unconfirmed” peak of 622 breeding pairs.

Year	Colony/colony complex	Nests	Source
2016	Longview Bridge	102	Anchor QEA 2017
2016	Troutdale Towers	154	Anchor QEA 2017
2016	Estuary Navigation Aids RKM 51–234	32	Anchor QEA 2017
2017	Estuary Navigation Aids RKM 0–22	7	Roby et al. 2021
2017	Trestle Bay	0	USACE unpubl. data
2017	East Sand Island	544[5]	Turecek et al. 2018
2017	Desdemona Sands Pilings	0	Roby et al. 2021
2017	Astoria-Megler Bridge	834	Roby et al. 2021
2017	Rice Island	0	Roby et al. 2021
2017	Miller Sands Spit	0	Roby et al. 2021
2017	Estuary Navigation Aids RKM 22–51	204	Roby et al. 2021
2017	Longview Bridge	147	Roby et al. 2021
2017	Troutdale Towers	194	USACE unpubl. data
2017	Estuary Navigation Aids RKM 51–234	25	USACE unpubl. data
2018	Estuary Navigation Aids RKM 0–22	41	USACE unpubl. data
2018	Trestle Bay	0	Roby et al. 2021
2018	East Sand Island	5,999[6]	USFWS unpubl. data
2018	Desdemona Sands Pilings	0	Roby et al. 2021
2018	Astoria-Megler Bridge	1,737	Turecek et al. 2019
2018	Rice Island	0	Roby et al. 2021
2018	Miller Sands Spit	0	Roby et al. 2021
2018	Estuary Navigation Aids RKM 22–51	293	Roby et al. 2021
2018	Longview Bridge	201	Roby et al. 2021
2018	Troutdale Towers	200	USACE unpubl. data
2018	Estuary Navigation Aids RKM 51–234	55	USACE unpubl. data

[5]Estimated peak number of active nests during longest period of breeding activity uninterrupted by colony abandonments, observed on July 26. About 3,450 potentially active nests were present during the week of May 14 during a period of extensive colony disturbances by bald eagles, which preceded complete colony abandonments during the weeks of May 21–28 and the weeks of June 11–July 9.

[6]Estimated peak abundance of active nests, which occurred on May 30, prior to partial colony abandonment. Peak count according to the U.S. Army Corps of Engineers was 3,672 pairs, observed on July 27.

Year	Colony/colony complex	Nests	Source
2019	Estuary Navigation Aids RKM 0–22	73[7]	Roby et al. 2021
2019	Trestle Bay	0	USACE unpubl. data
2019	East Sand Island	350	Roby et al. 2021
2019	Desdemona Sands Pilings	0	Roby et al. 2021
2019	Astoria-Megler Bridge	3,542	Roby et al. 2021
2019	Rice Island	0	Roby et al. 2021
2019	Miller Sands Spit	0	Roby et al. 2021
2019	Estuary Navigation Aids RKM 22–51	219	Roby et al. 2021
2019	Longview Bridge	139	Roby et al. 2021
2019	Troutdale Towers	234	USACE unpubl. data
2019	Estuary Navigation Aids RKM 51–234	55	USACE unpubl. data
2020	Estuary Navigation Aids RKM 0–22	81	ODFW unpubl. data
2020	East Sand Island	0	ODFW unpubl. data
2020	Desdemona Sands Pilings	0	ODFW unpubl. data
2020	Astoria-Megler Bridge	5,081	ODFW unpubl. data
2020	Rice Island	0	ODFW unpubl. data
2020	Miller Sands Spit	0	ODFW unpubl. data
2020	Estuary Navigation Aids RKM 22–51	260	ODFW unpubl. data
2020	Longview Bridge	184	ODFW unpubl. data
2020	Troutdale Towers	229	ODFW unpubl. data
2020	Estuary Navigation Aids RKM 51–234	89	ODFW unpubl. data

[7] Represents sum of peak abundance for each of three constituent sub-colonies.

Table B2. Table of abbreviations used in Table B1.

NWR	National Wildlife Refuge
OSU	Oregon State University
ODFW	Oregon Department of Fish and Wildlife
RKM	River Kilometer
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service

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Appendix C: Analysis of Double-crested Cormorant Abundance Following Management-related Take

C.1. Analysis

Double-crested cormorants have nested inconsistently at the East Sand Island colony following the beginning of federal management in 2015 (USACE 2015). Consequently, it has been difficult to empirically verify the effects of management on the abundance of breeding pairs formerly associated with this colony. I used a deterministic, age-structured population model to estimate the annual abundance of double-crested cormorant breeding pairs associated with the East Sand Island colony during the first phase of management (USACE 2015). I used an extension of the same age-structured model used for population modeling for the management plan (USACE 2015, Appendix E-1 therein); I included parameters for productivity (a measure of number of fledged offspring per nest), nest failure, and culling to assess the effects of management and observed abnormal nesting patterns on the population:

1) $N_{SY(t+1)} = P * 2Nst_{(t)} * (a + bN_{ASY(t)})$
2) $Nst_{(t)} = N.obs_{(t)} – N.fail_{(t)}$
3) $N_{ASY(t+1)} = (N_{SY(t)} * S_{SY}) + S_{ASY}(N_{ASY(t)} – cull_{(t)})$

where;

$N_{SY(t)}$ = number of second-year (SY, assumed non-breeding) individuals in year t

$N_{ASY(t)}$ = number of after-second-year (ASY, breeding-age) individuals in year t

P = productivity coefficient, an index of productivity relative to the long-term average

a = annual recruitment rate (implicitly incorporates hatch-year survival rate)

b = density dependence parameter

$Nst_{(t)}$ = number of viable nests in year t

$N.obs_{(t)}$ = peak number active nests observed in year t

$N.fail_{(t)}$ = number nests presumed to have failed as a result of oiling in year t

S_{SY} = annual survival rate of second-year (SY) individuals

S_{ASY} = annual survival rate of after-second-year (ASY) individuals

$cull_{(t)}$ = number breeding-age individuals culled in year t

I incorporated the same values for demographic parameters as used in a previous modeling effort (USACE 2015) into the model, but I used derived or observed values for other parameters (Table C1). In addition, because of uncertainty related to colony productivity in 2016–2019 (Anchor QEA 2017, Turecek et al. 2018, 2019), I ran three iterations of the model using different estimates for P (productivity coefficient) during these years. For one iteration, I assumed 0% productivity for the East Sand Island colony (low nest survival scenario). For the other two iterations, I assumed 50% and 100% productivity (moderate and high nest survival scenarios, respectively). I did not incorporate random Monte Carlo-based variation into the model to simplify comparisons between modelled and observed double-crested cormorant abundance.

Model results revealed that that 9,705, 11,137 and 12,570 double-crested cormorant breeding pairs were associated with the East Sand Island colony in 2018 under the low, moderate, and high nest survival scenarios, respectively (Table C2). However, a peak of only 5,999 breeding pairs was observed on East Sand Island during 2018, the first year breeding habitat was restricted by managers. Thus, at least approximately 3,700 breeding pairs (low survival scenario) may have been precluded from breeding on East Sand Island in 2018; many of these individuals likely dispersed to different colony sites. For 2019 and 2020, model results suggest that 6,977–11,106 breeding pairs associated with the East Sand Island colony remained in the population (Table C2), although only 350 and 0 nesting pairs nested on East Sand Island during those two years, respectively. The rapid increase in size of the Astoria-Megler Bridge colony during 2018–2020 is consistent with dispersal associated with restriction of nesting habitat on the East Sand Island colony beginning in 2018.

Table C1. Parameters used in the population model.

Parameter	Parameter description	Parameter value	Source	Notes
N_{ASY}	Initial population size: breeding-age individuals	25,834	USACE 2015	
N_{SY}	Initial population size: second-year individuals	6,700	Table D2	
a	Recruitment parameter	0.471	USACE 2015	
b	Density dependence parameter	-8.1932E-06	USACE 2015	
S_{SY}	Annual survival of second year individuals	0.75	USACE 2015	
S_{ASY}	Annual survival of individuals ≥3 years old	0.85	USACE 2015	
N.obs	2015 active nests	12,150	DSA 2016	
N.obs	2016 active nests	9,772	Anchor QEA 2017	
N.obs	2017 active nests	544	Turecek et al. 2018	
N.obs	2018 active nests	5,999	U.S. Fish and Wildlife Service unpubl. data	
N.obs	2019 active nests	350	U.S. Army Corps of Engineers (USACE) unpubl. data	
N.obs	2020 active nests	0	USACE unpubl. data	
N.fail	2015 nests oiled	5,089	USACE unpubl. data	
N.fail	2016 nests oiled	1,092	USACE unpubl. data	Not used in this analysis because colony abandoned after egg oiling, before reinitiating later in the year.
N.fail	2017 nests oiled	0	USACE unpubl. data	
cull	2015 adult cull before/during breeding	158	USACE unpubl. data	
cull	2015 adult cull after breeding	2,188	USACE unpubl. data	
cull	2016 adult cull before/during breeding	2,394	USACE unpubl. data	
cull	2016 adult cull after breeding	588	USACE unpubl. data	
cull	2017 adult cull before/during breeding	248	USACE unpubl. data	
cull	2017 adult cull after breeding	0	USACE unpubl. data	

Table C2. Predicted abundance of double-crested cormorants associated with the East Sand Island colony under three productivity scenarios. Model assumes all colony growth is intrinsic (i.e. no immigration). Productivity reflects number of fledged offspring per nest relative to the long-term average. Number viable nests represents observed number of active nests minus number of oiled nests that persisted to the late-incubation period.

Productivity	Year	Viable nests (Nst)	Breeding-age (ASY)	Second year (SY, non-breeders)	Expected breeding pairs
0%	2015	7,061	25,834	6,700	12,917
	2016	9,772	24,990	5,834	12,495
	2017	544	23,082	0	11,541
	2018	5,999	19,409	0	9,705
	2019	350	16,498	0	8,249
	2020	0	14,023	0	7,012
50%	2015	7,061	25,834	6,700	12,917
	2016	9,772	24,990	5,834	12,495
	2017	544	23,082	3,820	11,541
	2018	5,999	22,274	254	11,137
	2019	350	19,124	2,531	9,562
	2020	0	18,153	164	9,077
100%	2015	7,061	25834	6,700	12,917
	2016	9,772	24990	5,834	12,495
	2017	544	23082	7,640	11,541
	2018	5,999	25139	508	12,570
	2019	350	21749	5,061	10,875
	2020	0	22283	328	11,141

C.2. Literature Cited

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Appendix D: Estimated Intrinsic Growth Rates for Selected Double-crested Cormorant Colonies

D.1. Analysis

I used a deterministic, age structured population model to determine credible maximum rates of intrinsic growth for double-crested cormorant colonies in the Columbia River estuary (in contrast to growth supplemented by dispersal). The model is a generalization of the population model presented in Appendix E-1 in USACE (2015), with parameters for nest/egg take and culling set to zero:

1) $N_{SY(t+1)} = N_{ASY(t)} * (a + bN_{ASY(t)})$
2) $N_{ASY(t+1)} = (N_{SY(t)} * S_{SY}) + (N_{ASY(t)} * S_{ASY})$

where;

$N_{SY(t)}$ = number of second-year (SY, non-breeding) individuals in year t

$N_{ASY(t)}$ = number of after-second-year (ASY, breeding-age) individuals in year t

a = annual recruitment rate (implicitly incorporates hatch-year survival rate)

b = density dependence parameter

S_{SY} = annual survival rate of second-year (SY) individuals

S_{ASY} = annual survival rate of after-second-year (ASY) individuals

I used a generalized equation for population growth at discrete intervals to obtain λ, the finite rate of population increase (Dinsmore and Johnson 2012):

3) $N_t = N_0 * \lambda^t$

where;

N_t = the future population at time t

N_0 = initial population size

λ = growth rate over discrete (annual) interval

t = number of discrete time steps into the future

I obtained parameter values for recruitment and survival from previous modelling work (USACE 2015, Table C1). However, for the model herein, I set the density dependent parameter to zero to determine the maximum credible rate of colony growth. The model assumed an arbitrary initial population size of 10 individuals (5 breeding pairs), with no assumed non-breeders in the population. Model results revealed that $\lambda = 1.156$ when the annual value of λ had stabilized at year 10 (Table C2), reflecting the equilibrium age structure of the modelled population (Dinsmore and Johnson 2012). To compare modelled growth with observed growth, I calculated λ from observed annual abundances for estuary colony sites other than East Sand Island (non-ESI colonies). During the interval 2010–2019, the most recent 10-year period for which survey data were available for all estuary colonies, $\lambda = 1.565$ and $\lambda = 1.335$ for the Astoria-Megler Bridge colony and non-ESI colony sites in aggregate, respectively.

I incorporated final model estimates for λ into equation 3 to estimate credible levels of intrinsic growth during 2010–2019 for non-ESI colonies. Under modelled intrinsic growth rates, the size of the Astoria-Megler Bridge colony would have grown from 63 breeding pairs (2010) to 232 breeding pairs (2019) if no immigration had taken place; in comparison, the actual size of this colony in 2019 was 3,542 pairs. Similarly, abundance at all non-ESI colonies combined would have grown from 254 pairs (2010) to 778 pairs (2019); in comparison, the actual aggregate abundance for these colonies was 4,262 pairs in 2019. The difference between the actual size of non-ESI colonies in 2019 and their expected size suggests their growth during 2010–2019 was overwhelmingly driven by immigration from East Sand Island.

Table D1. Parameters used in the population model.

Parameter	Description	Parameter value	Source
N_0	Initial population size	10	Assumed for this analysis
a	Recruitment parameter	0.471	USACE 2015
b	Density dependence parameter	0	Assumed for this analysis
S_{SY}	Annual survival of second-year individuals	0.75	USACE 2015
S_{ASY}	Annual survival of after-second-year individuals	0.85	USACE 2015

Table D2. Predicted abundance of two age classes of double-crested cormorants associated with a hypothetical modelled colony in the western population under empirically observed rates of recruitment. Model parameter values from Table D1. Model assumes all colony growth is intrinsic (i.e. no immigration). Annual growth rates indicated by λ_{annual}.

Year	Breeding-age (ASY)	Second-year (SY)	λ_{annual}
0	10	--	--
1	9	5	0.850
2	11	4	1.266
3	12	5	1.129
4	14	6	1.163
5	16	7	1.154
6	19	8	1.156
7	22	9	1.156
8	25	10	1.156
9	29	12	1.156
10	34	14	1.156

D.2. Literature Cited

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Appendix E: Surplus Recruits Associated with the Double-crested Cormorant Colony on East Sand Island

E.1. Analysis

The portion of a population that exceeds local habitat capacity represents a pool of so-called "surplus" individuals, which may disperse to new, unsaturated habitats (Pulliam 1988). I used a deterministic, age-structured population model to estimate the potential abundance of surplus individuals produced by the East Sand Island double-crested cormorant colony prior to and following management (USACE 2015; ESI management plan). The model is a generalization of the population model presented in Appendix E-1 in USACE (2015), with parameters for nest/egg take and culling set to zero:

1) $N_{SY(t+1)} = N_{ASY(t)} * (a + bN_{ASY(t)})$
2) $N_{ASY(t+1)} = (N_{SY(t)} * S_{SY}) + (N_{ASY(t)} * S_{ASY})$

where;

$N_{SY(t)}$ = number of second-year (SY, assumed non-breeding) individuals in year t

$N_{ASY(t)}$ = number of after-second-year (ASY, breeding-age) individuals in year t

a = annual recruitment rate (implicitly incorporates hatch-year survival rate)

b = density dependence parameter

S_{SY} = annual survival rate of second-year (SY) individuals

S_{ASY} = annual survival rate of after-second-year (ASY) individuals

I obtained parameter values for recruitment, density dependence, and survival from previous modelling work (USACE 2015, Table E1). I ran the model using both pre- and post-management values for abundance of breeding-age individuals at the East Sand Island colony (USACE 2015): 25,834 individuals (12,917 breeding pairs) and 11,320 individuals (5,660 breeding pairs), respectively. These values reflected estimates of the carrying capacity for the East Sand Island colony during 2004–2013, the pre-management period; and the mid-point of the expected range of breeding abundance following the first phase of management (5,380–5,939 breeding pairs, USACE 2015). For each modelled abundance scenario, the number of breeding pairs on East Sand Island was held constant across years to reflect a population at carrying capacity. The number of surplus individuals produced annually was determined at a time step corresponding to a stable age distribution for the modelled population.

According to the model, the East Sand Island colony produced an estimated 1,150 and 1,513 surplus breeding-age individuals annually during the pre-management and post-management scenarios, respectively (Tables E2, E3). Recent work suggests such surplus individuals would likely display fidelity to the Columbia River estuary (Peck-Richardson 2017), where they would presumably attempt to breed at other colony sites following habitat restriction on East Sand Island (USACE 2015). It appears likely the East Sand Island colony was a major source of breeding-age individuals to nearby colonies prior to its recent decline. Further, the post-management scenario suggests the East Sand Island colony would have produced large numbers of individuals available to disperse to nearby colony sites. Consequently, estuary colony sites may have grown as a result of immigration from East Sand Island even if the recent collapse of the East Sand Island colony had not occurred.

Table E1. Parameters used in the population model.

Parameter	Description	Parameter value	Source
a	Recruitment parameter	0.471	USACE 2015
b	Density dependence parameter	-8.1932 E-06	USACE 2015
S_{SY}	Annual survival of second-year individuals	0.75	USACE 2015
S_{ASY}	Annual survival of after-second-year individuals	0.85	USACE 2015

Table E2. Predicted abundance of age-classes of double-crested cormorants associated with the East Sand Island colony prior to management. Model assumes all colony growth is intrinsic (i.e. no immigration). Surplus ASY individuals are breeding-age birds that exceed local habitat capacity on East Sand Island and would thus disperse to new colony sites to breed.

Year	Breeding-age (ASY)	Second-year (SY)	Surplus ASY
0	25,834	--	--
1	21,959	6,700	--
2	23,690	6,392	1,471
3	24,930	6,560	1,054
4	25,834	6,650	1,045
5	25,834	6,700	1,112
6	25,834	6,700	1,150
7	25,834	6,700	1,150

Table E3. Predicted abundance of age classes of double-crested cormorants associated with the East Sand Island colony following the first phase of management. Model assumes all colony growth is intrinsic (i.e. no immigration). Surplus ASY individuals are breeding-age birds that exceed local habitat capacity on East Sand Island and would thus disperse to new colony sites to breed.

Year	Breeding-age (ASY)	Second-year (SY)	Surplus ASY
0	11,320	--	--
1	9,622	4,282	--
2	11,320	3,773	1,513
3	11,320	4,282	1,132
4	11,320	4,282	1,513
5	11,320	4,282	1,513

E.2. Literature Cited

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