AVIAN PREDATION IN THE COLUMBIA RIVER BASIN

2021 Final Annual Report

Title: Avian Predation in the Columbia River Basin: 2021 Final Annual Report

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Final Report Submitted: March 31, 2022

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EXECUTIVE SUMMARY

Three separate management plans have been developed and are now being implemented to reduce the impacts of avian predation on the survival of juvenile salmonids (*Oncorhynchus* spp.) in the Columbia River basin (CRB). The targets of management include the Caspian tern (*Hydroprogne caspia*; hereafter referred to as "tern" or "terns") and the double-crested cormorant (*Nannopterum auritum*; hereafter referred to as "cormorant" or "cormorants") breeding colonies on East Sand Island in the Columbia River estuary (CRE) and the tern colonies on Crescent Island (McNary Reservoir) and on Goose Island (Potholes Reservoir) in the Columbia Plateau region (CPR). The primary goal of these management initiatives is to reduce predation rates (percentage of available smolts consumed) by piscivorous waterbirds on Endangered Species Act (ESA)-listed anadromous salmonid populations (referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]) through reductions in the size of or elimination of breeding colonies at each of these four sites. Management initiatives implemented at these four colony sites have been primarily non-lethal strategies for terns (i.e. passive and active nest dissuasion) and a combination of lethal (i.e. culling and egg-oiling) and non-lethal (i.e. nesting habitat management) strategies for cormorants. As part of the management plans for terns, the U.S. Army Corps of Engineers (USACE) created or enhanced alternative nesting habitat for terns at locations outside the CRB (i.e. northeastern California, southern Oregon, and south San Francisco Bay) to compensate for reductions in tern nesting habitat at East Sand Island in the CRE and for near elimination of tern nesting habitat at Crescent and Goose islands in the CPR.

In 2021, implementation of management plans was conducted by the USACE for the tern and cormorant colonies on East Sand Island and by the Bureau of Reclamation (BOR) at the tern colony on Goose Island; active tern nest dissuasion was not conducted on Crescent Island in 2021. The primary objectives of this study were to evaluate the efficacy of management to reduce avian predation on ESA-listed juvenile salmonids in the CRB, and to assess the magnitude of predation on smolts by other piscivorous waterbirds nesting at unmanaged colonies, namely those of California gulls (*Larus californicus*) and ring-billed gulls (*L. delawarensis*; hereafter referred to as "gull" or "gulls") and American white pelicans (*Pelecanus erythrorhynchos*; hereafter referred to as "pelican" or "pelicans"). Specifically, we sought to (1) locate and estimate peak colony size for piscivorous waterbird species (terns, cormorants, gulls, pelicans) and colonies within foraging range of juvenile salmonids in the middle Columbia River, lower Snake River, lower Columbia River, and CRE; (2) estimate colony-specific predation rates on smolts by piscivorous colonial waterbirds; and (3) measure the cumulative effects of predation by piscivorous waterbirds from multiple breeding colonies on salmonid survival, including an investigation into the additive effects of avian predation on salmonid mortality. The continued monitoring of colony sizes, locations, and predation rates on salmonids by piscivorous waterbirds nesting at both managed and un-managed colonies in the basin will help ensure that the intended benefits of management efforts are achieved and sustained, and that the accrued benefits from colony management are not offset by the emergence of new and

growing colonies at other locations within the basin. In short, this research will guide managers in developing and monitoring long-term management initiatives for avian predators that are science-based, defensible, cost-effective, and have a high probability of success.

Size of Piscivorous Waterbird Colonies

Columbia Plateau Region: The primary objective of the eighth year of implementation of the *Inland Avian Predation Management Plan* (*IAPMP*) was to limit the numbers of Caspian terns breeding on Goose Island and other islands in Potholes Reservoir, and on Crescent Island in McNary Reservoir, to less than 40 breeding pairs at each site and to less than 200 breeding pairs within the CPR to reduce impacts from tern predation on ESA-listed juvenile salmonids. To accomplish this task, the suitable tern nesting habitat at these sites was nearly eliminated by installing a variety of passive nest dissuasion materials on Goose Island prior to the 2021 breeding season, and by planting vegetation (in 2016) on Crescent Island. On both Goose and Crescent islands, passive nest dissuasion materials and/or vegetation covered all areas where terns had previously nested, as well as all areas of open, sparsely vegetated habitat that might be used by ground-nesting terns or gulls. Once terns arrived to initiate nesting in 2021, active nest dissuasion (i.e. human hazing) was used to dissuade terns from nesting on Goose Island and on other islands in Potholes Reservoir; these activities are described in detail in a separate report prepared by the BOR's contractor (U.S. Department of Agriculture – Wildlife Services). Active hazing to prevent Caspian terns from nesting on Crescent Island was not necessary during 2015 to 2020; consequently, active nest dissuasion was not implemented at that site in 2021.

Despite the use of passive and active nest dissuasion techniques on Goose Island in each of the previous seven years, some terns continued to display high fidelity to the island as a nesting site in 2021. We estimated that 22 breeding pairs of terns were successful at raising young on Goose Island in 2021, up from 6 breeding pairs in 2020, and up from zero breeding pairs during 2016–2019. Successful tern nesting on Goose Island in 2021 was likely due, at least in part, to changes in the passive nest dissuasion array installed on the island prior to the 2021 nesting season. To create a passive nest dissuasion array that would require less in-season and between season maintenance, the braided rope that connected the stakes was replaced with wire prior to the 2021 breeding season. The flagging, a major component of the nest dissuasion array, was observed to slide down the wires during windstorms, leaving large areas of suitable tern nesting habitat without flagging during the 2021 breeding season. Efforts were made by the contractor to fix this problem during the 2021 field season; however, despite these efforts, terns were successful in establishing a colony and rearing young at a site on top of the island near the former tern colony site. The use of braided rope as part of the nest dissuasion array prior to 2021, although perhaps requiring more maintenance than wire, prevented the movement of flagging along the rope because the flagging was installed between the strands of rope. Similar to other years during the management period, terns were persistent in trying to nest along the shoreline of Goose Island in 2021, areas that became exposed during the nesting season as reservoir levels receded. These nesting attempts were not successful, however, likely due to active measures (i.e. human hazing) implemented by the contractor to prevent nesting in these areas.

A management effort that planted willows and other native vegetation on Crescent Island in 2016 eliminating most open, bare-ground habitat that terns prefer for nesting. Prior to management, there was also a large gull colony on Crescent Island. During the first five years of tern management on Crescent Island (2015–2019), nest dissuasion activities prevented both terns and gulls from nesting on the island. Beginning in 2020, and again in 2021, a gull colony became re-established on Crescent Island in and amongst the vegetation on the island. The recent die-back of vegetation on Crescent Island, perhaps caused by nesting gulls, beaver herbivory, and/or weather-related events (e.g., windstorms, drought), has created small patches of open, unvegetated habitat that is suitable for tern nesting. In 2021, one pair of Caspian terns was successful in rearing young on Crescent Island, the first year of successful tern nesting at that site since management commenced early in the 2015 nesting season. Without the implementation of mitigative measures to prevent further die-back of vegetation on Crescent Island, we expect that there will be an increasing number of terns nesting on Crescent Island in the future.

Aerial, ground, and boat-based surveys were conducted in the CPR to determine where terns displaced from the managed colonies in Potholes Reservoir and at Crescent Island might attempt to nest. In 2021, terns successfully nested at five different colony sites in the CPR: at both managed sites (Goose Island [22 breeding pairs] and Crescent Island [one breeding pair]) and at three unmanaged sites (Badger Island in McNary Reservoir [231 breeding pairs], Harper Island in Sprague Lake [85 breeding pairs], and Shoal Island in Lenore Lake [61 breeding pairs]). In 2021, a managed increase in the water level of John Day Reservoir during the 2021 Caspian tern breeding season eliminated all upland nesting habitat previously used by terns in the Blalock Islands. As a result, a tern colony was not established at that site in 2021, the first year that no tern colony formed in the Blalock Islands since the colony first became established in 2005. Most of the terns displaced from nesting at the Blalock Islands in 2021 apparently relocated to Badger Island to nest, where a total of 231 tern breeding pairs were successful in nesting at two sites on Badger Island, one near the shoreline and the other in the island's interior amongst nesting gulls and pelicans. Prior to 2021, terns had not successfully nested on Badger Island since 2018, when a small colony of 8 breeding pairs nested there. We expect that terns will continue to nest on Badger Island in the future if there is no suitable tern nesting habitat available in the Blalock Islands and suitable unvegetated nesting habitat persists on Badger Island.

A total of 400 breeding pairs of Caspian terns nested at five different colonies in the CPR in 2021, up from 209 breeding pairs in 2020. This is the first increase in the regional breeding population of terns since 2017. The cause(s) of the increase in the regional breeding population of terns is unknown but may be related to the poor nesting season for terns in the CRE and elsewhere within their breeding range.

In addition to Caspian terns, we also estimated the colony size of other piscivorous waterbirds breeding in the CPR during 2021. Because visits to these colonies were less frequent (1–2 times per month) as compared to our monitoring of tern colonies in the region (2–4 times per

month), there is more uncertainty in our estimates of peak colony size at these other piscivorous waterbird colonies (i.e. actual peak colony size could be somewhat higher). As was the case in 2020, there was a total of nine active gull colonies in the CPR during the 2021 breeding season, ranging in size from about 1,200 breeding individuals (Lenore Lake) to over 14,000 breeding individuals (Goose Island in Potholes Reservoir). The biggest change in gull colony size in 2021 relative to 2020 was at Crescent Island, where colony size grew more than 5-fold; a total of 400 breeding individuals were counted in 2020 and 2,084 breeding individuals in 2021. Double-crested cormorants were confirmed nesting at four colony sites in the CPR, with colony sizes ranging from 147 breeding pairs (Shoal Island in Lenore Lake) to 520 breeding pairs (Harper Island in Sprague Lake). Finally, the size of the Badger Island pelican colony was estimated to be 3,624 breeding individuals in 2021, larger than the estimated colony size in 2020 (3,165 breeding individuals) and one of the largest estimates since counts of the Badger Island pelican colony were first conducted in 2005.

Columbia River Estuary: In 2021, we estimated the size of the double-crested cormorant colony on the Astoria-Megler Bridge to be 4,151 breeding pairs, down from 5,081 breeding pairs in 2020 (J. Lawonn, ODFW, pers. comm.). Estimates of colony size at the Astoria-Megler Bridge should be considered a minimum estimate in that cormorants are now nesting in areas of the bridge (inside beams and girders) not visible from the boat-based and aerial vantages used to count nesting cormorants on the bridge. Prior to 2021, the cormorant colony on the Astoria-Megler Bridge had increased in size every year since 2011, with the recent exponential growth in size of the bridge colony corresponding with the precipitous decline in the size of the East Sand Island colony; no egg-laying by double-crested cormorants was observed on East Sand Island in either of the last two years (Brandtner and Tidwell 2021).

The USACE estimated that 2,050 breeding pairs of Caspian terns nested on the prepared 1-acre main colony site on East Sand Island in 2021 (Brandtner and Tidwell 2021). As was the case in 2020, the size of the Caspian tern colony on East Sand Island was significantly less than the target colony size of 3,125–4,375 breeding pairs stipulated in the *Caspian Tern Management Plan for the Columbia River Estuary*. Large numbers of terns (hundreds to thousands of adults per week) once again attempted to nest along the southeast and east beaches of East Sand Island in 2021, despite ongoing nest dissuasion efforts to prevent terns from nesting outside of the prepared 1-acre colony area. None of the tern eggs laid outside the 1-acre colony area apparently hatched, and none of the tern chicks hatched on the 1-acre colony area survived to fledging, with gull depredation of eggs and chicks being the biggest factor in tern nest failure on the East Sand Island colony in 2021. In 2020, the East Sand Island colony of Caspian terns produced very few, if any, young. As was the case in 2020, terns attempted to nest but were unsuccessful in establishing a colony on Rice Island or elsewhere in the upper Columbia River estuary in 2021 (K. Tidwell, USACE, pers. comm).

Predation Rates on Juvenile Salmonids
To investigate the impacts of predation by piscivorous colonial waterbirds on the survival of juvenile salmonids (smolts), and to determine the efficacy of on-going management actions to reduce predation, we estimated salmonid population (ESU/DPS)-specific predation rates based

on recoveries of smolt PIT tags on bird colonies following the 2021 nesting season. Estimates were generated using previously published, standardized methods, providing a means to compare predation rates across avian predator species and colonies, salmonid species and ESUs/DPSs, and years. A total of 7,236 steelhead smolts were captured, PIT-tagged, and released into the tailrace of Rock Island Dam (RIS) in the middle Columbia River as part of this study in order to ensure that a sufficient number of ESA-listed Upper Columbia River (UCR) steelhead, a population that is highly susceptible to avian predation and therefore a suitable population to evaluate the efficacy of management actions, were tagged and available for predation analyses in 2021. Tagging at RIS commenced in 2008, resulting in a long-term dataset (2008–2021) with which to evaluate the impacts of piscivorous waterbirds on the survival of ESA-listed steelhead and to evaluate relative changes in predation and survival associated with management actions.

Efficacy of Avian Management Plans: Two primary objectives of the *IAPMP* were to reduce predation rates by Caspian terns on ESA-listed juvenile salmonids to less than 2% per salmonid ESU/DPS, per colony, and to less than 5% per salmonid ESU/DPS by terns from all colonies in the CPR combined. Recoveries of smolt PIT tags in 2021 indicated that predation rates exceeded this threshold for UCR steelhead, whereby an estimated 3.9% (95% credible interval = 2.4–6.8%) of available smolts were consumed by terns on Goose Island in Potholes Reservoir and an estimated 6.3% (4.5–9.3%) were consumed by terns from all colonies in the CPR combined. Predation rates by terns that nested on Crescent Island in 2021 were < 0.1% per ESU/DPS due to the small size (1 nesting pair) of that colony. Predation rates at the unmanaged tern colony on Lenore Lake (North Rock) were < 1.0% per ESU/DPS, with the highest rate observed on UCR steelhead at 0.9% (0.6–1.5%). Due to higher reservoir levels, no terns nested on the Blalock Islands in 2021, formerly the largest colony in the CPR and a location where predation rates often exceeded the 2% threshold for several ESUs/DPSs in previous years (2015–2020). Coincident with the elimination of the Blalock Islands colony, terns nested at an unmanaged colony site on Badger Island in 2021 and predation rate estimates of 1.4% (1.0–2.2%) and 1.5% (0.8–2.9%) were observed on UCR steelhead and Snake River (SR) steelhead, respectively. Estimates of predation rates on salmon populations by terns nesting on Badger Island, however, were < 1.0% per ESU/DPS, with the highest predation rate observed on SR spring/summer Chinook salmon at 0.8% (0.2–2.4%).

Coincident with increases in the number of terns nesting at colonies in the CPR in 2021, tern predation rates on juvenile salmonids were greater than those observed in 2020 for most of the salmonid ESUs/DPSs evaluated. Over the course of the entire study period, however, comparisons of tern predation rates on juvenile salmonids prior to (2007–2013) and during (2014–2021) implementation of the *IAPMP* indicate that there have been survival benefits to several salmonid ESUs/DPSs, especially UCR steelhead, from management under the *IAPMP*. For example, average annual predation rates by terns nesting on Goose Island and elsewhere in Potholes Reservoir have been reduced from an estimated 15.7% (14.1–18.9%) prior to management (2007–2013) to 1.7% (1.2–2.5%) during the management period (2014–2021). There was also evidence that survival of UCR steelhead smolts, on average, has increased significantly in the river reach where most terns from the Goose Island and Crescent Island

colonies foraged (RIS to McNary Dam) following implementation of the *IAPMP*; increases in steelhead survival rates were commensurate with reductions in tern predation rates. In 2020, tern predation rates were at record low levels, coincident with record high survival rates for UCR steelhead smolts. In 2021, the increase in tern predation rates was commensurate with a decrease in steelhead survival (see also *Additive Effects of Predation below*). Due to increases in predation on juvenile salmonids by terns nesting on Goose Island and the re-establishment of tern colonies on Crescent Island and on Badger Island in 2021, adaptive management actions will likely be necessary to achieve the over-all goals of the *IAPMP* in the future.

Estimates of predation rates on multiple salmonid ESUs/DPSs by terns nesting on East Sand Island in the CRE during 2021 were not available, estimates that were generated with funding from the USACE in previous years. Smolt PIT tags, however, were recovered on the East Sand Island tern colony by the USACE following the 2021 breeding season and thus predation rates could be retrospectively analyzed in the future. In 2020, estimates of predation rates were the lowest recorded since terns began nesting on East Sand Island in 1999, with estimates ranging from 0.4% (0.2–0.9%) on UCR spring Chinook salmon to 5.9% (4.5–8.1%) on SR steelhead. Results from 2020 indicated that predation by terns nesting on East Sand Island had been reduced by 65% to 76% for steelhead DPSs, reductions that met or exceeded those anticipated in the *Caspian Tern Management Plan for the Columbia River Estuary.* Although predation rates by terns that nested on East Sand Island in 2021 are currently unknown, impacts may be greater than those in 2020 based on the number of smolt PIT tags recovered in 2021 (n = 5,081) compared with 2020 (n = 4,595). Large numbers of terns also continued to nest outside of the designated 1-acre nesting areas on East Sand Island in 2021, as well as nesting attempts by terns on Rice Island in the upper Columbia River estuary. Thus, continued implementation of nest dissuasion activities, continued monitoring of tern nesting sites throughout the estuary, and empirically derived estimates of ESU/DPS-specific predation rates will be necessary to ensure that predation impacts in the future do not exceed the levels stipulated in the management plan.

For the second consecutive year there was no evidence of an established (long-lived) nesting colony of cormorants on East Sand Island in the CRE during 2021. Several hundred pairs of cormorants did, however, attempt to nest on East Sand Island in 2021; however, no estimates of predation rates on juvenile salmonids by cormorants attempting to nest on East Sand Island are available, estimates that have been generated with funding from the USACE in previous years. Unlike the tern colony on East Sand Island, predation rates on salmonid smolts by cormorants that attempted but failed to nest on East Sand Island in 2021 were presumably low based on the small number of smolt PIT tags (n = 117) recovered following the breeding season.

Astoria-Megler Bridge Cormorant Predation Assessment: An unintended consequence of management actions at the double-crested cormorant colony on East Sand Island during 2015–2019 was the complete abandonment of the colony site and the subsequent rapid expansion of the cormorant colony on the Astoria-Megler Bridge (AMB). The AMB is located upstream of East Sand Island in the freshwater mixing zone of the CRE, an aquatic environment where fewer alternative prey fish are likely available and juvenile salmonids may be consumed in greater

proportion by cormorants relative to foraging areas around the former cormorant colony site on East Sand Island which is in the marine zone of the estuary. To estimate predation rates on salmonid smolts by cormorants breeding on the AMB in 2021, we recovered smolt PIT tags deposited by cormorants nesting in five discrete plots on a concrete footing of the bridge, areas where smolt PIT tags could be recovered by researchers following the breeding season. Estimates of per capita (per breeding pair) predation rates derived from cormorants nesting in plots (n = 159 breeding pairs) were then extrapolated to account for all double-crested cormorants nesting elsewhere on the bridge (n = 3,992 breeding pairs) to generate colony-wide estimates of predation rates. Results indicated that per capita predation rates of cormorants nesting on the AMB ranged from 0.0009% (95% credible interval = 0.0005–0.0016%) on yearling Chinook salmon smolts to 0.0040% (0.0028–0.0069%) on sub-yearling Chinook salmon smolts originating from populations upstream of Bonneville Dam. Per capita predation rates were 0.0013% (0.0004–0.0025%) on steelhead smolts and 0.0015% (0.0004–0.0031%) on coho salmon smolts originating from populations downstream of Bonneville Dam. Colony-wide estimates of predation rates ranged from 3.9% (2.0–6.8%) on yearling Chinook salmon to 16.8% (7.3–28.4%) on sub-yearling Chinook salmon and from 5.5% (1.8–10.5%) on steelhead to 6.1% (1.6–12.7%) on coho salmon. Estimates of per capita predation rates on salmonid smolts by cormorants nesting on the AMB in 2021 were approximately 2 to 4 times greater (depending on salmonid species/age-class) than average annual per capita predation rates by cormorants that nested on East Sand Island prior to implementation of management actions (2003–2014). Estimates of colony-wide predation probabilities by cormorants nesting on the AMB in 2021 were similar to those of cormorants breeding on East Sand Island during 2003–2014, despite the much smaller size of the cormorant colony on the AMB in 2021 (4,151 pairs) compared with the cormorant colony on East Sand Island (12,787 pairs on average during 2003–2014). Collectively, results suggest that colony location is closely related to the impact of cormorant predation on survival of salmonid smolts in the CRE, and that these impacts have likely remained unchanged since management actions on East Sand Island were first implemented in 2015.

Other Piscivorous Colonial Waterbirds: An investigation of predation by other, unmanaged piscivorous colonial waterbird species indicated that smolt consumption rates by California and ring-billed gulls nesting at certain colonies in the CPR were consistently higher than those of nearby managed and unmanaged Caspian tern colonies in 2021. Unlike terns, gulls are scavengers and are known to consume dead or moribund fish and to steal (kleptoparasitize) fish from other waterbirds, like terns. Consequently, we use the term "gull consumption rates" rather than "gull predation rates". Estimates of consumption rates as high as 12.4% (6.2–23.1%) and 13.9% (9.1–21.8%) on UCR and SR steelhead, respectively, were documented for gulls nesting at Miller Rocks in The Dalles Reservoir. Consumption rates as high as 2.8% (1.1–6.3%) on SR Fall Chinook salmon by gulls nesting at Miller Rocks were among the highest estimates of consumption rate observed for a salmon ESU in 2021. Estimates of consumption rates by gulls nesting at several other colonies in the CPR were in excess of 2% and as high as 8.6% (3.6–14.7%) for steelhead DPSs, including gulls nesting on Island 20 in the middle Columbia River, Badger and Crescent islands in McNary Reservoir, and the Blalock Islands in John Day Reservoir. Estimates of consumption rates by gulls nesting at colonies in the CPR during 2021 were similar

to or greater than those observed in previous years (2007–2020). Cormorant predation rates on salmonid smolts originating from Upper Columbia River ESUs/DPSs were low (< 0.2% per ESU/DPS) but were significantly higher on smolts from SR ESUs/DPSs, with predation on SR steelhead and SR Spring/Summer Chinook salmon at 3.0% (0.8–9.3%) and 4.4% (0.8–15.6%), respectively.

Despite the high levels of consumption/predation on smolts observed at some piscivorous waterbird colonies in 2021, impacts from birds nesting at other colonies, particularly impacts on salmon ESUs, were often low. For example, estimates of predation rates by cormorants nesting on Hanford Island in the middle Columbia River were < 0.4% per salmonid ESU/DPS, estimates of smolt predation rates for pelicans nesting at Badger Island in McNary Reservoir were < 1.0% per ESUs/DPSs, and estimates of predation rates by pelicans nesting on Miller Sands Spit in the Columbia River estuary were < 0.1% per ESU/DPS, among the lowest of any colony evaluated in 2021. Significantly higher predation impacts on non-ESA-listed juvenile salmonid stocks (e.g., subyearling Fall Chinook salmon from the Upper River Bright stock) and, possibly, adult salmonids (e.g., adult sockeye salmon) by pelicans have been documented in other studies and warrant additional investigation. Nevertheless, results of this study indicate that pelicans nesting on Badger Island in the CPR and, especially, at the Miller Sands Spit colony in the CRE, posed only a minor predation threat to survival of actively migrating, ESA-listed UCR and SR juvenile salmonids in 2021.

Cumulative Predation and Smolt Survival: To investigate the cumulative effects of avian predation (predation/consumption by all colonial waterbird species nesting at all colonies combined) and to determine what proportion of all sources of smolt mortality (1-survival) were associated with avian predation/consumption, we conducted a mark-recapture-recovery analysis on UCR steelhead smolts that were PIT-tagged and released at RIS in 2021. We used previously published methods to jointly estimate predation/consumption and survival probabilities during smolt passage through multiple river reaches and we compared results from 2021 to those from previous years (2008–2020). In 2021, the cumulative effects of avian predation/consumption on mortality of UCR steelhead during smolt passage from RIS to the Pacific Ocean were estimated at 42.4% (35.4–50.6%) of available smolts, with the highest levels of mortality associated with predation/consumption by gulls at 29.1% (22.8–36.3%), followed by predation by terns at 8.6% (6.7–11.8%), predation by cormorants at 2.9% (0.1–7.0%), and predation by pelicans at 0.4% (0.1–2.4%). Results indicate that despite reductions in average annual mortality rates of UCR steelhead smolts due to tern predation in the CPR and the CRE, the cumulative effects of avian predation/consumption remain a substantial source of steelhead smolt mortality during out-migration to the Pacific Ocean. Estimates of the cumulative effects of predation/consumption on mortality of UCR steelhead in 2021 were the third highest recorded since studies of system-wide predation begin in 2008, with estimates in 2021 only slightly lower than those observed in 2009 and 2015. Increases in tern predation probabilities in the CPR, gull consumption probabilities in the CPR, and the inclusion of predation probabilities from cormorants nesting on the AMB all contributed to higher estimates of cumulative predation/consumption of UCR steelhead smolts in 2021 relative to most, but not all, previous years.

Comparisons of total mortality (1-survival) to mortality associated with avian predation/consumption, indicated that avian predation/consumption in 2021 was associated with more mortality of UCR steelhead during smolt out-migration from RIS to Bonneville Dam than any other direct mortality factor, with avian predation/consumption associated with 61.6% (48.8–76.8%) of all smolt mortality in this river reach. Results from 2021 were consistent with those from most previous study years, with avian predation/consumption upstream of Bonneville Dam associated with more than 50% of all steelhead mortality during 11 of the previous 13 years (2008–2020). Even after passage through the impounded sections of the middle and lower Columbia River upstream of Bonneville Dam, the impact of predation by piscivorous colonial waterbirds on the survival of steelhead smolts in the CRE were substantial, with terns and cormorants breeding on East Sand Island and cormorants breeding on the AMB collectively depredating an estimated 13.5% (6.6–25.1%) of available UCR steelhead smolts in 2021.

Additive Effects of Predation: To investigate to what degree avian predation/consumption of UCR steelhead smolts limited fish survival, we used a previously published model to explicitly measure the strength, magnitude, and direction of the relationship between avian predation/consumption and fish survival. Data from 2021 were used to update a long-term dataset (2008–2020) to evaluate more recent trends in avian predation and smolt survival. Results indicated that the higher levels of tern predation on UCR steelhead in 2021 were associated with lower levels of smolt survival from RIS to Bonneville Dam compared with 2020, a year when record low levels of tern predation were associated with record high levels of UCR steelhead smolt survival. An investigation of weekly and annual estimates of tern predation and smolt survival probabilities suggested that a greater proportion of UCR steelhead smolts would have survived out-migration to Bonneville Dam in the absence of tern predation upstream of Bonneville Dam, with the estimated average annual difference in observed survival versus baseline survival (i.e. survival in the absence of tern predation) of 0.174 (0.106–0.234) during 2008–2021. Due to low levels of cormorant predation on UCR steelhead smolts upstream of Bonneville Dam (0.01 or 1%), only small increases in survival of UCR steelhead smolts to Bonneville Dam would be possible in the absence of cormorant predation upstream of Bonneville Dam. Although there was some evidence of a relationship between consumption probabilities by gulls and survival probabilities of UCR steelhead smolts, results were not statistically significant when considered across all years. The statistical power to accurately determine to what degree smolt consumption by gulls influenced smolt survival was limited by a truncated time series, coupled with high levels of uncertainty in both consumption and survival and a lack of weekly variation in estimates of consumption probabilities. Gulls are also known to consume dead fish and to kleptoparasitize fish from other piscivorous waterbirds, like terns, so it is likely that consumption of UCR steelhead smolts by gulls was a more compensatory source of mortality compared to predation by terns or cormorants.

There was evidence that higher levels of tern predation on UCR steelhead smolts in the CRE were associated with lower returns of adult steelhead to Bonneville Dam, with increases in tern predation probabilities associated with statistically significant decreases in adult survival

probabilities. Results suggest that in the absence of tern predation on UCR steelhead smolts, smolt-to-adult returns (SARs) for UCR steelhead would have nearly doubled, even though, in the absence of tern predation, the majority of smolts depredated by terns would have died from other causes before returning to Bonneville Dam as an adult (i.e., partial additivity). These results, along with those from a recently published, peer-reviewed study (Payton et al. 2020) indicate that tern predation on steelhead smolts is an additive source of mortality and a partially additive source of mortality to the adult life-stage. There was some evidence that higher levels of cormorant predation on UCR steelhead smolts in the estuary were associated with lower adult returns to Bonneville Dam; however, results were not statistically significant at an alpha level of 0.05 when considered across those years when adequate data were available (2008–2015). Collectively, our results indicate that efforts to reduce tern predation on UCR steelhead have enhanced steelhead smolt survival in the CRB, especially in those years when tern predation rates were dramatically reduced because of management actions at Goose and Crescent islands, as in 2020. More importantly from a steelhead conservation perspective, our results suggest that in the absence of tern predation on UCR steelhead smolts, significantly more adult steelhead would return to Bonneville Dam. Managers, regional stakeholders, and the public at-large must decide whether these potential increases in smolt survival and SARs for UCR steelhead warrant continued efforts to manage terns in the Columbia River basin. Additional research is needed, however, to evaluate to what degree cormorant predation and, especially, gull consumption of salmonids limits smolt survival and SARs in the CRB.

Biotic and Abiotic Factors: Previous research indicates that biotic and abiotic factors experienced by smolts during out-migration influence their survival, including, but not limited to, the probability of avian predation. As part of this study, and as recently recommended by the Northwest Power and Conservation Council's Independent Scientific Advisory Board, we are investigating the influence of various covariates on smolt survival during out-migration. Covariates under consideration include biotic factors such as fish size, rearing-type (hatchery, wild), abundance (density), and run-timing, plus abiotic factors such as spill, discharge, and measures of river flow speed (e.g., water transit time). The goal of this analysis is to describe those factors that best explain variation in smolt survival and to identify potential "management relevant" variables, variables that resource managers may be able to control to some degree. Analyses of covariates are on-going and herein we provide an update of results from analyses of acoustic-tagged UCR steelhead smolts and a discussion of next steps. A more complete analysis will be presented as part of our 2022 Annual Report.

Smolt Survival to Bonneville Dam

Recoveries of smolt PIT tags on breeding colonies of piscivorous colonial waterbirds can be used to increase the precision and accuracy of smolt survival estimates by increasing the sample sizes of tagged fish used in mark-recapture survival models. To provide information for use in smolt survival estimates to Bonneville Dam in 2021, we recovered smolt PIT tags from several piscivorous waterbird nesting, loafing, and roosting sites in the CRE, sites that were not included in the original scope of work for this study, but where we suspected PIT tags were being deposited by birds. In total, scanning associated with this additional effort detected 1,932 current migration year (2021) PIT tags from juvenile salmonids in the CRE. This information,

coupled with tags recovered by the USACE on the East Sand Island tern colony (n = 5,081) and tags of live fish detected at the National Marine Fisheries Service pair trawl net detection system (n = 4,537), were used by this project and other federally funded projects to estimate smolt survival to Bonneville Dam in 2021.

BACKGROUND

Avian predation on out-migrating juvenile salmonids (smolts) has been identified as a factor that can significantly limit the survival of some Endangered Species Act (ESA)-listed populations (referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]) of anadromous salmonids (*Oncorhynchus* spp.) in the Columbia River basin (CRB). Addressing the impact of avian predation on smolt survival is a component of Biological Opinions and Reasonable and Prudent Alternatives (RPAs) associated with management of the Federal Columbia River Power System (FCRPS). Over the last two decades, numerous research, monitoring, and evaluation (RM&E) studies of avian predation have been conducted to assess the impacts on smolt survival of consumption by Caspian terns (*Hydroprogne caspia*; hereafter referred to as "tern" or "terns"), double-crested cormorants (*Nannopterum auritum*; hereafter referred to as "cormorant" or "cormorants"), California and ring-billed gulls (*Larus californicus* and *L. delawarensis*; hereafter referred to as "gull" or "gulls"), and American white pelicans (*Pelecanus erythrorhynchos*; hereafter referred to as "pelican" or "pelicans") in the CRB.

To address concerns about the impact of avian predation on the survival of salmonid smolts originating from the CRB, the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps of Engineers (USACE), and their management partners developed and implemented three separate management plans to reduce predation rates on smolts by piscivorous waterbirds nesting at four breeding colonies in the CRB (USFWS 2005, USACE 2014, USACE 2015): the tern and cormorant breeding colonies on East Sand Island in the Columbia River estuary (CRE), formerly the largest known colonies for the respective species anywhere; and the two largest tern colonies in the Columbia Plateau region (CPR), those on Crescent Island in McNary Reservoir and on Goose Island in Potholes Reservoir. The primary goal of these management initiatives was to reduce predation rates (percentage of available smolts consumed) on ESA-listed salmonid ESUs/DPSs by reducing the number of birds breeding at each of these four colony sites. Management initiatives implemented at these four colony sites have involved primarily non-lethal strategies for terns (i.e. passive and active nest dissuasion) and a combination of lethal and non-lethal strategies for cormorants (i.e. culling and egg-oiling, plus reduction of nesting habitat). As part of the management plans for terns, the USACE created or enhanced alternative nesting habitat for terns outside the CRB (i.e. in northeastern California, southern Oregon, and south San Francisco Bay) to compensate for reductions in tern nesting habitat on East Sand Island in the CRE and for elimination of tern nesting habitat on Crescent and Goose islands in the CPR.

Despite some success in managing avian predators to increase smolt survival in the basin, several critical uncertainties remain, and recent developments suggest that predation impacts from both managed and unmanaged piscivorous waterbird colonies in the CRB may be increasing. First, the 1-acre of suitable tern nesting habitat on East Sand Island can accommodate in some years more than the target colony size of 3,125–4,375 breeding pairs that was specified in the Final EIS (2005) and Records of Decision (2006), and the area of suitable tern habitat provided will likely need to be reduced to ensure that the colony size does not exceed 4,375 breeding pairs. Second, terns are returning to nest on Rice Island and elsewhere on East Sand Island (outside designated colony area) in increasing numbers; Rice Island was the site of a large tern colony in the late 1990s, before the colony was relocated to East Sand Island to reduce its impact on smolt survival in the estuary. Third, cormorants dispersed from the East Sand Island colony site during 2016–2020, leaving the colony completely abandoned for extended periods. Concurrently, increasing numbers of cormorants nested on the Astoria-Megler Bridge. While impacts to survival of out-migrating smolts from cormorants nesting on the bridge were previously undetermined, there was evidence that the per capita predation rates on salmonid smolts of cormorants nesting on the bridge would be higher than for those nesting on East Sand Island (Cramer et al. 2021b). Fourth, some terns that have been dissuaded from nesting at Crescent and Goose islands in the CPR have remained in the region and are nesting at new sites where impacts of tern predation on smolt survival may be as high or higher than at the original managed colony sites. Finally, recent research indicates that smolt consumption rates by several unmanaged gull colonies in the CRB are as great as, and in some cases greater than, those of managed tern and cormorant colonies (Hostetter et al. 2015; Evans et al. 2019). Taken together, these developments indicate that continued monitoring and evaluation of avian predation in the CRB is warranted if adaptive management is to be successfully implemented and the intended benefits to smolt survival from management of avian predators are to be realized.

PROJECT OBJECTIVES

The primary objectives for ongoing RM&E of avian predation on juvenile salmonids in the CRB in 2021 were to (1) assess the distribution and size of piscivorous waterbird colonies and (2) estimate the colony-specific and cumulative system-wide impacts of predation/consumption by colonial waterbirds on juvenile salmonid survival in the CRB. As part of these objectives we (a) located and estimated the size of piscivorous waterbird colonies within foraging range of juvenile salmonids in CRB, including information on any new or incipient colonies; (b) estimated colony-specific predation/consumption rates of multiple salmonid ESUs/DPSs by piscivorous waterbirds; (c) measured reach-specific and cumulative predation/consumption probabilities by multiple avian predator species plus survival rates for UCR steelhead smolts, an indicator species in studies of avian predation; (d) estimated the additive effects of avian predation on smolt survival and smolt-to-adult survival; and (e) investigated the biotic and abiotic factors that influence steelhead smolt susceptibility to avian predation. In addition to these primary

objectives and tasks, we increased efforts to recover smolt PIT tags at piscivorous colonial waterbird nesting, loafing, and roosting sites in the CRE to better understand the systemwide impacts of piscivorous waterbirds on juvenile salmonid survival in the CRB. These PIT tag recoveries in the CRE can also be used to improve the precision of estimates of smolt survival to Bonneville Dam in 2021 (estimates generated by our team, by NOAA Fisheries, and other researchers).

METHODS & ANALYSES

This work is part of a comprehensive program to implement, monitor, and evaluate avian predation management plans (including adaptive management) to reduce the impacts of avian predators on the survival of ESA-listed juvenile salmonids in the CRB. Action effectiveness monitoring was conducted in the CPR at both managed (i.e. Crescent Island, Goose Island and elsewhere in northern Potholes Reservoir) and at unmanaged sites where Caspian terns might disperse to re-nest in response to management. As part of the adaptive management approach in 2021, the USACE planned to raise the water level elevation of John Day Reservoir during the peak of the tern breeding season to eliminate the nesting habitat that had been colonized by terns in previous years. The focus of the work reported here was to evaluate the efficacy of ongoing and new management initiatives to reduce avian predation on ESA-listed juvenile salmonids from the CRB and to help identify new and emerging avian predation impacts on smolt survival that may warrant future management consideration. The methods used in 2021 (see *below* for a brief description) were like those used in previous studies so that results were comparable across years, both before and during the implementation of avian predation management actions in the CRB.

NEST DISSUASION ACTIVITES

In 2021 nest dissuasion activities on Goose Island and elsewhere in Potholes Reservoir were conducted by the BOR and its contractor, U.S. Department of Agriculture-Wildlife Services (USDA-WS). These activities will be summarized in an upcoming annual report completed by the USDA-WS (S. Fesenmyer, BOR, pers. comm.). The Fisheries Field Unit (FFU) at the USACE carried out nest dissuasion efforts outside the designated tern colony area on East Sand Island and on Rice Island in 2021, a description of those activities is summarized in Brandtner and Tidwell (2021).

NESTING DISTRIBUTION & COLONY SIZE

Action effectiveness monitoring was conducted both at the colony-level and the system-level (region-wide). Colony monitoring was designed to evaluate the efficacy of nest dissuasion efforts and the need for adaptive management at Crescent Island, Goose Island, and elsewhere

in Potholes Reservoir in preventing terns from nesting at these sites (see upcoming report from USDA-WS for more details). Additionally, we monitored terns and other piscivorous colonial waterbirds (i.e. gulls, cormorants, and pelicans) nesting at unmanaged colonies to identify potential new and emerging threats to juvenile salmonid survival in the CRB (see section on *Avian Predation Rates* below).

A fixed-winged aerial survey of the CPR was flown early in the breeding season (4–5 May) to help identify all active nesting colonies of piscivorous waterbirds in 2021. The aerial survey was followed by periodic (weekly to monthly) ground- and boat-based surveys throughout the breeding season (April-July), as well as a second fixed-wing aerial survey late in the incubation period (25 May), to identify all active nesting colonies in the region and to assess nesting chronology and estimate colony size. Colony size was estimated by digitizing birds visible in high-resolution orthophotography acquired with drones during surveys corresponding with the peak in nesting activity at each colony. Colony size is reported as the number of birds on colony, and, in the case of terns and cormorants, the number of active breeding pairs. For those tern colonies that were not observable with field glasses and/or spotting scopes (i.e. the view of the colony was obstructed by vegetation or terrain), sitting terns were counted as a proxy for the number of active nests. To augment this criterion, we compared the number of sitting terns across multiple time scales (i.e. multiple drone flights within the same survey day, as well as multiple drone flights conducted on different days).

At Goose Island and other suitable nesting colony sites in northern Potholes Reservoir, we monitored the activities of terns and gulls weekly throughout the breeding season using at least two field crew members stationed in the CPR. Crescent Island was monitored twice monthly to determine whether active hazing and more frequent monitoring might be necessary (terns have not nested on Crescent Island since management commenced in 2015). Other piscivorous waterbird colonies (*Map 1*) in the CPR were monitored once or twice per month throughout the breeding season. Finally, the double-crested cormorant colony on the Astoria-Megler Bridge in the Columbia River estuary was monitored on three different occasions from late April through mid-June to determine nesting chronology and peak colony size.
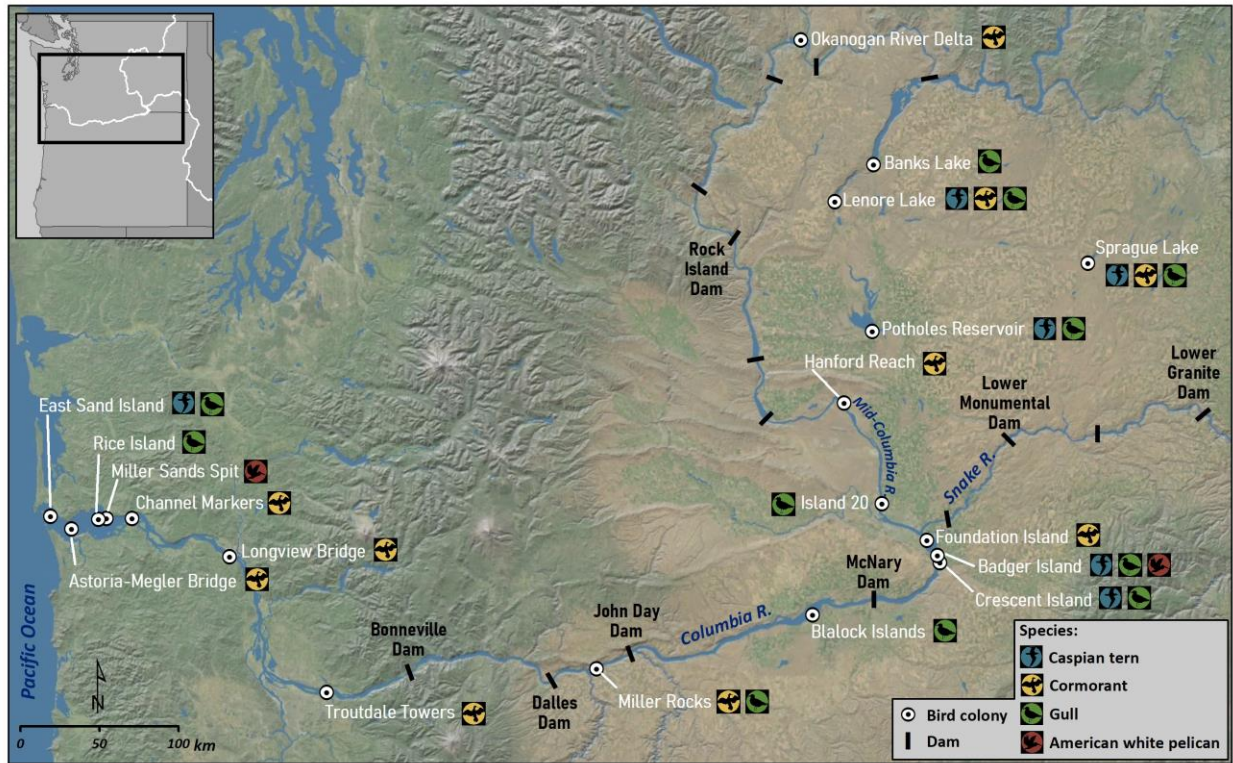
Map 1. Study area in the Columbia River basin in 2021.

Monitoring of piscivorous waterbird colonies was conducted from the air (fixed-wing aircraft or drone), a boat, and/or on foot, with precautions taken to minimize disturbance to actively nesting non-target species (e.g., gulls). Whenever possible, counts of piscivorous waterbirds at prospective nesting colony sites were differentiated by behavior (i.e. nesting vs. roosting), age (i.e. adult vs. juvenile), and location on the island. Each island was also closely monitored for the formation of new satellite colonies (i.e. away from the former colony site or in and around areas of passive nest dissuasion). Data collection methodologies followed established protocols such that the data collected in 2021 could be compared with analogous data collected in previous years and at other colonies (Antolos et al. 2004; Adkins et al. 2014; Roby et al. 2015; Collis et al. 2016, 2017, 2018, 2019, 2020, 2021a; Roby et al. 2021).

The size of the cormorant colony on the Astoria-Megler Bridge in 2021 was estimated as part of this study (see *Appendix A*), while the size of the tern colony and number of cormorants attempting to nest on East Sand Island in 2021 was estimated by the USACE (see Brandtner and Tidwell 2021).

AVIAN PREDATION RATES

We analyzed smolt PIT tags collected on bird colonies as part of this study to (1) estimate predation rates on ESA-listed salmonid ESUs/DPSs and to (2) assess relative differences in these predation rates prior to and during tern management actions associated with the *IAPMP*, with a focus on data collected in 2021. Comparisons between current (2021) and previous predation rates (2007–2020) were made in the context of management initiatives for terns nesting on Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir and relative to the management goal of achieving predation rates of less than 2% per salmonid ESU/DPS, per colony, per year (USACE 2014). We also estimated predation rates by terns at an unmanaged colony on Shoal Island in Lenore Lake, WA and on Badger Island in McNary Reservoir in 2021. PIT tag scanning during the pre-management and management periods were also conducted at these unmanaged tern colonies in years past. Although terns also nested on Harper Island in Sprague Lake, which is located 67 kilometers north of the lower Snake River, the island was not scanned for PIT tags following the breeding season due to a lack of permission from the landowner to access the site.

PIT-tagging of Upper Columbia River Steelhead

To ensure adequate numbers of ESA-listed Upper Columbia River (UCR) steelhead were available for predation rate analyses, smolts were intentionally captured, PIT-tagged, and released into the tailrace of Rock Island Dam (RIS) as part of this study in 2021. Previously research has demonstrated that the UCR steelhead population is highly susceptible to predation by terns and cormorants (Evans et al. 2012, Evans et al. 2019, Payton et al. 2020) and is therefore a suitable group to evaluate the efficacy management actions aimed at reducing avian predation. Efforts to tag steelhead smolts at RIS as part of avian predation studies have also been on-going since 2008, providing a long-term dataset in which to evaluate changes in predation rates associated with both managed and unmanaged piscivorous waterbird colonies (Evans et al. 2019, Collis et al. 2021b) and to investigate factors that influence smolt susceptibility to predation (Hostetter et al. 2021) and to what degree avian predation is an additive versus compensatory source of mortality (Payton et al. 2020; see *below*).

A detailed description of the sampling methods used to PIT tag steelhead at RIS are provided in Evans et al. (2014). In brief, steelhead were captured at the RIS juvenile fish trap, PIT-tagged (*Biomark* model APT12), measured (mm; fork-length), condition-scored (based the presence/absence of injuries, descaling, and disease), and released into the tailrace of RIS during the smolt out-migration period of April to June. Steelhead were randomly selected for tagging (i.e. tagged regardless of condition, origin, and size) and were tagged in-concert with, and in proportion to, the run-at-large to ensure that the tagged sample was representative of the steelhead population at-large (tagged and untagged fish at RIS). All juvenile steelhead captured at RIS were part of the ESA-listed UCR steelhead DPS, as all hatchery and wild steelhead originating from tributaries upstream of RIS are part of the ESA-listed population (NOAA 2021). The sampling approach used at RIS as part of this study ensures that (1)

steelhead from all natural spawning stocks of the ESA-listed DPS are included in the sample (Entiat, Methow, Okanogan, and Wenatchee rivers), (2) that smolts of all sizes, conditions, and rear-types are included in the sample in proportion to their relative abundance in-river, and (3) that smolts are tagged in-concert with the run at-large (tagging and untagged); criteria that allows us to make credible inference about the entire ESA-listed UCR steelhead population. The target sample size goal was to PIT-tag approximately 7,000 steelhead smolts for use in predation and survival analyses in 2021. This target sample size was selected because it was consistent with previous steelhead PIT-tagging efforts at RIS (Evans et al. 2014; Evans et al. 2019, Payton et al. 2020) and because it was estimated to result in a measurement of precision for predation rates of approximately ± 2% for those colonies that forage on smolts in the middle Columbia River immediately downstream of RIS, particularly managed tern colonies like those on Goose and Crescent islands.

For most other ESA-listed salmonid ESUs/DPSs, adequate numbers of PIT-tagged smolts were available for inclusion in predation rate analyses based on other regional tagging studies that occur upstream Lower Monumental Dam on the lower Snake River or upstream of McNary and Bonneville dams on the lower Columbia River (see *Predation Rate Estimates* section below for details). However, since fish from these other studies were generally not randomly selected for tagging (e.g., fish were culled based on size, condition, and rear-type) and were not tagged in proportion to and in-concert with the run at-large (tagged and untagged), predation and survival results may be biased to an unknown degree relative to fish tagged at RIS as part of this study.

Predation Rate Estimates

The previously published methods of Evans et al. (2012) and Hostetter et al. (2015) were used to recover smolt PIT tags from piscivorous waterbird colonies and to estimate colony- and salmonid ESU/DPS-specific predation rates. Detailed analytical methods are provided on the Monitoring Resources website (https://www.monitoringresources.org/) and in the recently completed Avian Predation Synthesis Report (see Roby et al. 2021). In brief, to recover (electronic detection) smolt PIT tags, PIT tag antennas were used to scan the entire area occupied by nesting birds following the breeding season, with a minimum of two complete sweeps or passes conducted at each colony. We used a hierarchical Bayesian model that integrated multiple factors of uncertainty in the tag recovery process, including imperfect detection of PIT tags on bird colonies, on-colony PIT-tag deposition probabilities, and temporal changes in smolt availability to birds nesting at each colony (see Roby et al. 2021, Appendix for details). Predation rates were modeled independently for each salmonid ESU/DPS and bird colony. The probability of recovering a PIT tag from a smolt on each colony was modelled as the product of the probability that (1) the fish was consumed (θ), (2) the PIT tag was deposited on-colony (ϕ), and (3) the PIT tag was detected on-colony after the breeding season (ψ_i):

$$k_i \sim \text{Binomial}(n_i, \theta_i * \phi * \psi_i)$$

where k_i is the number of smolt PIT tags recovered from the number available (n_i) in week *i*. The detection probabilities (ψ_i) and predation probabilities (θ_i) were each modeled as a function of time. The probability, ψ_i, that a tag, consumed in week *i* and deposited on the colony is detected, is assumed to be a logistic function of week. That is:

$$\text{logit}(\psi_i) = \beta_0 + \beta_1 * i$$

where β_0 and β_1 are both derived from non-informative priors (normal [0, 1000]). Predation rates nearer together in time are more similar than those further apart in time (Evans et al. 2016; Payton et al. 2019). To reflect this, variation in weekly predation probabilities, θ_i, was modeled as a random walk process with mean μ_θ and variance σ_θ^2, where:

$$\text{logit}(\theta_i) = \mu_\theta + \sum_{w \leq i} \varepsilon_w$$

and $\varepsilon_w \sim normal\left(0, \ \sigma_\theta^2\right) \forall w$. We placed non-informative priors on these two hyperparameters: $\text{logit}^{-1}(\mu_\theta) \sim uniform(0,1)$ and $\sigma_\theta^2 \sim uniform(0,20)$. This allows each week (*i*) to have a unique predation probability (θ_i), while still sharing information among weeks improving precision.

Informative Beta priors were used to model deposition probability(ϕ). The shape parameters (α, β) are dependent on the predator species (cormorant, tern, gull, pelican) and are assumed to be mutually independent from colony to colony. For terns, we assumed α = 16.20 and β = 6.55, for cormorants we assumed α = 15.98 and β = 15.29, for gulls we assumed α = 33.71 and β = 183.61, and for pelicans we assumed α = 6.70 and β = 7.37.

Weekly predation estimates were defined as the estimated number of PIT-tagged smolts consumed divided by the total number available each week. Annual predation rates were derived as the sum of the estimated number of PIT-tagged smolts consumed each week divided by the total number of PIT-tagged smolts available:

$$\sum_{i \in breeding \ season} (\theta_i * n_i) \Big/ \sum_{i \in breeding \ season} (n_i)$$

Summation of weekly consumption estimates is necessary to accurately reflect weekly variation and autocorrelation of predation rates and thus to create unbiased annual rates with accurate assessments of precision (Hamilton 1994).

Models were analyzed using the software STAN (SDT 2015), accessed through R version 3.6.3 (RDCT 2014), and using the rstan package (version 2.23.0; SDT 2020). Reported predation rate estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% Credible Interval [CRI]) calculated using the HDInterval package (version 0.2.0; Meredith and Kruschke 2016). Annual predation rates were calculated for salmonid ESUs/DPSs

where ≥ 500 PIT-tagged individuals were available to birds at each colony to avoid imprecise results that may occur from small sample sizes of available PIT-tagged smolts (Evans et al. 2012).

Efficacy of Avian Predation Management Plans: Predation rate estimates were used to compare and contrast smolt losses prior to and during implementation of management actions at tern and cormorant colonies in the CRB; data critical to evaluate the effectiveness of management plans aimed at reducing predation rates at managed bird colonies.

Inland Avian Predation Management Plan – A stated goal of the *IAPMP* is to reduce the impact of predation by terns on ESA-listed salmonids to less than 2.0% per salmonid ESU/DPS, per colony and to less than 5.0% per salmonid ESU/DPS, for all tern colonies combined (i.e. cumulative predation; USACE 2014). To help evaluate the efficacy of the *IAPMP* at reducing predation impacts to those levels, predation rates were compared and contrasted between the pre-management period (2007–2013) and the management period (2014–2021) at both managed and unmanaged tern colonies in the CPR.

East Sand Island Caspian Tern and Double-crested Cormorant Management Plans – The USACE and its contractors were responsible for recovering smolt PIT tags and estimating predation rates by terns and cormorant nesting on East Sand Island in the CRE in 2021. Although PIT tags were recovered from East Sand Island in 2021, no analysis of the data have occurred to-date.

As part of this study and as an independent analysis, we evaluated the cumulative, system-wide effects of predation on UCR steelhead tagged at RIS (see *above*), which includes recoveries of smolt PIT tags from tern and cormorant colonies on East Sand Island in 2021. Estimates of cumulative predation by multiple avian predator species (terns, cormorants, pelicans, and gulls) and colonies, however, rely on a different analytical framework (see *below*) than those of colony- and ESU/DPS-specific estimates of predation (see *above*), so the two estimates (ESU/DPS- and colony-specific versus cumulative) are not directly comparably to one another, although the two methods often generate similar estimates of predation at the colony-specific level (Evans et al. 2021a).

Astoria-Megler Bridge Cormorant Predation Assessment: See *Appendix A*.

Cumulative Predation and Survival: We evaluated the cumulative effects of avian predation (predation from all piscivorous colonial waterbird species and colonies combined) on ESA-listed UCR steelhead that were tagged at RIS in 2021. Salmonid smolts are also subject to numerous non-avian sources of mortality (e.g., hydroelectric dam passage, predation by piscivorous fish, disease, and other factors), and determining to what degree avian predation limits survival relative to these other sources of mortality may be critical for prioritizing recovery actions for ESA-listed salmonids (Evans et al. 2019). To investigate the cumulative effects of colonial waterbird predation and to estimate what proportion of all sources of smolt mortality (1-survival) were due to avian predation, we used the methods of Payton et al. (2019). UCR

steelhead smolts must migrate through the foraging areas of multiple avian predator species (terns, cormorants, gulls, and pelicans) from multiple breeding colonies. River reaches were defined by dams where PIT-tagged smolts were detected alive following tagging and release at RIS. Predation rates were based on the proportion of available smolts consumed by birds within each river reach, and survival rates were based on the proportion that survive out-migration through that reach. An estimate of the proportion of smolts that die from causes other than avian predation was also generated as part of these analyses, providing spatially explicit information on non-avian sources of smolt mortality.

The joint mortality and survival (JMS) estimation technique of Payton et al. (2019) was used to estimate reach-specific and cumulative predation and survival rates. In brief, this hierarchal Bayesian mark-recapture-recovery analytical framework incorporated both live and dead (Hostetter et al. 2018) detections of PIT tagged fish in space and time to simultaneously estimate predation and survival rates across *M* river segments, demarcated by recapture (detection) locations. Expressed as a state-space model, we let s_j represent the probability of a fish surviving through river segment *j*, and p_j represent the probability that the fish is then detected at the (passive) recapture site delimiting the downstream end of each river segment *j*. Using indicator variables $z_{i,j}$ and $y_{i,j}$ to represent, respectively, the continued survival and successful recapture of fish *i* through/after river segment *j*, and $z_{i,0}$ represents the release of a live fish (i.e. is always assumed to be 1), that is:

$$z_{i,j} \sim bernoulli(s_j * z_{i,(j-1)})$$
$$y_{i,j} \sim bernoulli(p_j * z_{i,j}).$$

As discussed previously (see), the inclusion of recoveries of tags from bird colonies requires the estimation of three additional processes: (1) the probability, $\theta_{j,c,w}$, that a PIT-tagged fish will be consumed by a bird from colony *c* in segment *j* during week *w*; (2) the probability, ϕ_c, that a PIT tag consumed by a bird from colony *c* will be subsequently deposited on its breeding colony; and (3) the probability, $\psi_{c,w}$, that the deposited PIT tag is recovered by researchers on colony *c* after the breeding season. We let $r_{i,c}$ indicate the recovery of the tag from fish *i* on colony *c*. Letting *w* refer to the week in which fish *i* was released, these processes can be expressed as:

$$r_{i,c} \sim bernoulli(\sum_{j=1}^{M}(z_{j-1} - z_j) * \theta_{j,c,w} * \phi_c * \psi_{c,w})$$
$$\phi_c \sim beta(\alpha_c, \beta_c)$$
$$logit(\psi_{c,w}) = a_c + b_c * w$$

where α_c and β_c are provided by previous research (Hostetter et al. 2015), and a_c and b_c are both determined from detection tags intentionally sown on colonies by researchers prior to, during, and after each breeding season.

Similarities among weeks in recapture rates and survival/predation rates were assumed to reflect autocorrelation. The recapture rates are modeled using a random walk process (on the logit scale). The multivariate version of the random walk process for use with the survival/predation rates requires special consideration. For each week, we first arrange the reach- and colony- specific mortality rates defined previously, $\theta_{j,c}$, as an M x C matrix, $\boldsymbol{\Theta}$. We then multiply this matrix by a diagonal matrix of cumulative survival rates to create a single *M* x *C* matrix such that

$$\boldsymbol{\Omega} = \mathrm{diag}(\begin{bmatrix} 1 \\ s_1 \\ s_1 s_2 \\ \vdots \\ s_1 s_2 \dots s_{K-1} \end{bmatrix})\boldsymbol{\Theta} \,,$$

where $\Omega_{j,c}$ represents the probability that a fish in the initial release survives $k-1$ segments before succumbing in segment j to predation associated with colony c. Adding a subscript w to denote the week of release and noting $\mathrm{vec}(\boldsymbol{\Omega}_w)$ is a simplex allows autocorrelation to be addressed with a logistic regression analogue. Using un-depredated survivor's past the estuary pair trawl/death in the ocean, $\Omega_{M,C}$, as the reference level, we assume:

$$\log\left(\frac{\Omega_{j,c,w+1}}{\Omega_{M,C,w+1}}\right) = \log\left(\frac{\Omega_{j,c,w}}{\Omega_{M,C,w}}\right) + \eta_{j,c,w} \cdot \,,$$

where $\eta_{j,c,w}$ is the normally distributed random walk component.

Weakly informative priors were assigned to most of the parameters of the model (Gelman et al. 2013; Payton et al. 2019). The prior for the initial week's detection probability in each year was defined to be uniform. Analogously, the prior distributions assigned for predation and, consequently, survival were effectively uniform. That is, we assume $\mathrm{vec}(\boldsymbol{\Omega}_1)$ to be Dirichlet(**1**), where **1** is the appropriately sized vector. Weakly-informative priors of half-normal(0,5) were also implemented for all variance parameters.

We simplify the state-space model into succinct likelihood equations and employ Hamiltonian Monte Carlo (HMC) simulations to construct approximations of the joint posterior distributions of the parameters in question. The simulated posterior distributions are the basis from which we infer survival and recovery estimates (posterior medians), along with 95% credible intervals (2.5 and 97.5 posterior percentiles; 95% CRI). The HMC simulations are performed using the software STAN accessed through R version 3.1.2 (RDCT 2014), using the rstan package (version 2.8.0; SDT 2015). Four parallel HMC simulations will be run for 2,000 adaptation iterations, followed by 2,000 posterior iterations. Chain convergence will be visually evaluated and verified using the Gelman-Rubin statistic. Any evidence of a lack of convergence will be corrected by repeatedly doubling the prescribed number of adaptation and simulation intervals until convergence can be presumed. A more detailed description of these models and their use can

be found in Monitoring Resources website (https://www.monitoringresources.org/) and in Payton et al. (2019).

Additive Effects of Predation: Weekly estimates of predation and survival on UCR steelhead tagged at RIS that were generated as part of cumulative predation analyses (see *above*) were also used to investigate to what degree avian predation was an additive source of steelhead mortality and, therefore, to what extent reductions in avian predation associated with management of bird colonies can potentially enhance smolt survival. In particular, we investigated if reductions in predation rates resulted in higher smolt survival (i.e. avian predation adds to total smolt mortality) or were most smolts consumed by birds destined to die regardless of avian predation (i.e. avian predation is compensated for by other mortality factors). The additive mortality hypothesis predicts that predation is directly related to survival in space and time. The compensatory mortality hypothesis predicts that predation and survival are unrelated in space and time, at least up to the point where the level of predation exceeds natural mortality, whereby the proportion of fish consumed by predators cannot be greater than the proportion that was alive at any given time (see also Payton et al. 2020). Our ability to address this question will depend on the magnitude (level) of predation and variation in predation rates across space and time. These analyses also depend on standardized datasets with a long time series (i.e. multiple years), where smolts are tagging in proportion to and in-concert with the run-at-large (tagging and untagged). Based on analyses conducted in previous years, predation on steelhead smolts best meet these criteria for use in this study (see also Payton et al. 2021a).

For UCR steelhead tagged and released at RIS, weekly and annual reach-specific and cumulative predation rate and survival rate estimates were used to investigate hypotheses of additive versus compensatory predation mortality. Predation by multiple avian predator species (terns, cormorants, and gulls) and from multiple breeding colonies were assessed. We use a mark-recapture-recovery model to assess the strength, magnitude, and direction of the relationship between predation and survival probabilities, including survival to adulthood based on the proportion of smolts that returned to Bonneville Dam as adults. Assessments include the relative contribution of avian predation rates on smolt survival across multiple spatial scales including regional (e.g., hydrosystem) and life cycle (e.g., adult returns) scales.

Not all active breeding colony sites were scanned for smolt PIT tags in all past years. As such, the time series available for additive analyses was truncated for some predator species. Specifically, gull colonies on Island 20 and in the Blalock Island Complex were active but not scanned for smolt PIT tags during 2008–2012, precluding estimates of gull predation in those years. The Foundation Island cormorant colony was also active but not scanned for smolt PIT tags during 2013 and 2015–2019, precluding estimation of predation in those years. Many, if not most, of the cormorants nesting on East Sand Island in the CRE dispersed to alternative colony sites in the upper estuary that were not accessible for PIT tag recovery during 2016–2020, preventing accurate estimates of predation by cormorants in the estuary in those years. Overall, tern predation probabilities were available during 2008–2020, gull predation

probabilities were available during 2013–2020, and cormorant predation probabilities were available during 2008–2015 and in 2020, depending on the colony (Foundation Island or East Sand Island).

The relationship between weekly variation in avian predation rates and weekly variation in smolt survival rates were investigated during smolt out-migration through the hydrosystem from RIS to Bonneville Dam (BON) and following hydrosystem passing from BON (as smolts) back to BON (as adults). We aggregate cumulative survival and predation across all segments prior to this recapture (detection) point (denoted as the set [REACH]) and across mortality sources associated with the genus of the avian predator (i.e. tern, cormorant, gull, or pelican) under consideration (referred to as the set [GENUS]). We therefore sharpen our focus on the survival rate to the specified downstream dam, a single genus-aggregated total mortality prior to this dam, and a single rate of other mortality due to all other mortality factors (referred to as the set [OTHER]). That is, we let

$$\theta_w^{\{GENUS\}} = \sum_{j \in \{REACH\}} \sum_{c \in \{GENUS\}} \theta_{j,c,w},$$
$$\theta_w^{\{OTHER\}} = \sum_{j \in \{REACH\}} \sum_{c \notin \{GENUS\}} \theta_{j,c,w},$$

and

$$s_w^* = 1 - \theta_w^{\{GENUS\}} - \theta_w^{\{OTHER\}}.$$

Therefore, within each year, we can construct simplified simplex weekly rates of survival and predation as,

$$\left[s_w^* \quad \theta_w^{\{GENUS\}} \quad \theta_w^{\{OTHER\}} \right]^T.$$

Following Sandercock et al. (2011), additive mortality rests on an assumption of annual "baseline survival" rate, s^0. s^0 can be interpreted as the hypothetical survival rate in the absence of bird predation. Therefore, in the absence of bird predation,

$$s^0 + \epsilon_w = 1 - \theta_w^{\{OTHER\}}$$

where s^0 is the yearly "baseline survival" rate across the segments of {REACH}, around which weekly survival rates were assumed to vary completely at random with the variation, denoted by ϵ_w.

Independent of the variation described above, any further decrease/increase in baseline survival is assumed to be relative to the level of bird genus-specific predation associated with that reach. This relationship is generally assumed to be proportional, unless $\theta_w^{\{GENUS\}}$ encompasses all unaccounted-for mortality, at which point the relationship becomes directly

inverse. The annual magnitude of this additive-mortality relationship will be measured with the parameter a. The measure of the resulting or "observed" weekly survival, s_w^*, can be expressed as

$$s_w^* = s^0 + \epsilon_w - a\theta_w^{\{\text{GENUS}\}}.$$

Generally, there is not enough precision in our estimates of s_w^*, $\theta_w^{\{GENUS\}}$, and $\theta_w^{\{OTHER\}}$ to get informative estimates of a. However, if we examine data across years, allowing for yearly variation in a around some mean level μ_a, we can provide unbiased and reasonably precise estimates of the effective extent to which avian predation reduces smolt survival across a reach.

The prior distribution assigned to $\mu_a \sim normal(\frac{1}{2}, 3)$ is based on the *a priori* assumption that predation is partially additive, with little prior credibility given to hypotheses of over-compensatory or depensatory mortality mechanisms. The prior distribution for each s^0 will be assumed to be uniform for all years.

Results were used to estimate smolt survival and smolt-to-adult survival based on the degree to which bird predation was determined to be an additive or a compensatory source of smolt mortality for each salmonid ESU/DPS where there are adequate sample sizes of tagged, released, and consumed smolts, and where variation in predation rate is large enough to identify a statistically significant relationship. Addition details regarding the modelling framework used to investigate the relationship between predation and survival probabilities can be found on Monitoring Resources website (https://www.monitoringresources.org/) and in Payton et al. (2020).

Biotic and Abiotic Factors: It is well documented that biotic and abiotic factors or conditions experienced by smolts during out-migration play an important role in their survival (Petrosky and Schaller 2010, Hostetter et al. 2011, Evans et al. 2019). While the magnitude of avian predation on some populations of juvenile salmonids has been high on average, there has been substantial intra- and inter-annual variability in avian predation rates (Lyons et al. 2014, Evans et al. 2016, Hostetter et al. 2021). For example, avian predation rates on the same salmonid population can vary significantly by week and by year. Furthermore, even within the same salmonid population, differences in predation probabilities based on a smolt's rear-type, size (fork-length, mass), condition (presence/absence of injuries), migration history, and run-timing have all been observed (Ryan et al. 2003, Hostetter et al. 2012, Lyons et al. 2014, Evans et al. 2016, Payton et al. 2016, Hostetter et al. 2021). Environmental factors experienced by smolts also vary within and across years, including hydroelectric dam operations (e.g., spill and discharge), river flows, and other measures of river speed (e.g., water transient times), factors that may also be directly or indirectly related to smolt survival. Understanding which factor or suite of factors best explain variation in survival may enhance our understanding of mechanisms that regulate survival during the smolt life stage and may elucidate ways to potentially reduce smolt susceptibility to bird predation.

To begin the investigation of which factors are most closely associated with variation in smolt survival probabilities, including avian predation, we can employ modified versions of the JMS models of Payton et al. (2019) and Payton et al. (2020; see also *above*). These models allow us to differentiate between direct impacts of specific mortality sources (e.g., predation) on survival versus more indirect or proximate causes of mortality as represented by covariates. That is, these models simultaneously measure the additive effects of predation, a necessarily linear relationship, and measure the extent to which variation in "baseline" survival can be explained by other combinations of biotic and abiotic factors identified as potential covariates, modelled using logistic regression techniques. Explicitly, the additive mortality model assumes the observed probability of survival in day d, s_d, can be expressed as

$$s_d = s^0 + \epsilon_d - a\Theta_d^{\{\text{predation}\}}.$$

This implies that the null model for baseline survival probability in day *d* is assumed to be

$$s_d^0 = s^0 + \epsilon_d.$$

It follows that we can introduce covariates to express the observed probability of survival though logistic regression

$$s_d = \text{logit}^{-1}(\mathbf{X}_d^T\vec{\beta} + \mathbf{Z}_d^T\vec{\epsilon}) - a\Theta_d^{\{\text{predation}\}}$$

where **X** and **Z** represent matrices of possible fixed and random covariates, respectively. Note, this is a simplified expression of the model which ignores situations in which $\Theta_d^{\{\text{predation}\}}$ and s_d sum to 1. These parameter space boundary issues are of significant concern and the piecewise equations they necessitate apply to these analyses as well (see Payton et al. 2020 for a more detailed description).

The investigation of covariates detailed herein focused on how to approach the question of whether predation by Caspian terns on steelhead smolts or abiotic conditions experienced by steelhead smolts during out-migration through the same river reach and years, best explains variation in steelhead survival within and across years. To provide the most accurate and precise characterization of tern predation and survival and therefore which abiotic variables are the most appropriate for covariate analyses, we analyzed acoustic-tagged (AT) steelhead that were migrating from Rock Island Dam to Priest Rapids Dam (Wanapum and Priest Rapids dams and reservoirs) during a four-year study period (2008–2010, 2014). This period of time represents years when there was a tern colony on Goose Island in Potholes Reservoir and releases of AT smolts. The network of acoustic arrays deployed in this river reach provided accurate estimates on where, spatially, mortality occurred on a daily basis in each year. Due to high in-river detection probabilities of AT smolts at each array (> 95%), precise estimates of survival were available. Detail methods regarding the capture, tagging, and release of AT smolts

and the network of arrays used to estimate survival are presented in Timko et al. (2011) and Hatch et al. (2016). Estimates of Caspian tern predation on AT steelhead were those of terns nesting on Goose Island, terns that foraged within the Wanapum-Priest Rapids Project and where variation in predation was documented within and across years. Three of the four study years (2008–2010) occurred prior to implementation of management actions on Goose Island and one year (2014) occurred during implementation of management actions. There are other studies that have investigated the impact of Goose Island terns on AT steelhead survival (Evans et al. 2016, Payton et al. 2016, Roby et al. 2016), however, these data have not previously been analyzed using the full JMS model and our investigation of abiotic and biotic effects begins with this step.

Models were implemented using the software STAN accessed through R version 4.1.0 (RCDT 2021) using the rstan package (version 2.21.1; SDT 2020). To simulate random draws from the joint posterior distribution we ran four Hamiltonian Monte Carlo (HMC) Markov Chain processes. Each chain contained 2,000 adaptation iterations, followed by 2,000 posterior iterations. Posterior iterations were then thinned by a factor of 2. Chain convergence was verified using the Gelman-Rubin statistic (Gelman et al. 2013) and all accepted chains reported zero divergent transitions.

RESULTS & DISCUSSION

NEST DISSUASION ACTIVITIES

USDA-WS carried out all nest dissuasion activities for Caspian terns on Goose Island and elsewhere in Potholes Reservoir in 2021, as was the case in 2020. A passive nest dissuasion array (stakes, wire, and flagging) was installed on all upland habitat that might be suitable for tern nesting on Goose Island prior to the onset of the 2021 breeding season. To create a passive nest dissuasion array that would require less in-season and between-season maintenance, the braided rope that was previously used to connect the stakes was replaced by wire and the flagging was affixed directly to the wires. During regular visits to Goose Island to conduct active hazing and to collect any tern eggs (under permit) that were discovered, the contractor made any necessary repairs to the passive nest dissuasion array as needed. USDA-WS also patrolled the islands located in northern Potholes Reservoir looking for prospecting terns and used passive and active nest dissuasion techniques to prevent tern nesting in these areas, if warranted. Further details on these nest dissuasion efforts will be provided in an upcoming report being prepared by BOR and its contractor (S. Fesenmyer, BOR, pers. comm.).

The growth of willows planted on Crescent Island in 2016 had mostly eliminated all upland habitat suitable for tern nesting on Crescent Island. As a consequence, the other passive nest dissuasion materials (i.e. fencing, stakes, and rope) that were previously installed on Crescent

Island were removed prior to the 2020 breeding season. Prior to 2020, passive nest dissuasion alone was sufficient to deter both terns and gulls from nesting on the island. In 2020, gulls reestablished a breeding colony in semi-vegetated habitat on Crescent Island (see *below* for details). Gulls were observed nesting in open areas where willows had died and/or where beaver herbivory on the willows had created patches of ground with minimal vegetative cover. Despite the presence of an active gull colony on Crescent Island, terns did not attempt to nest on the island in 2020. No active hazing of prospecting terns was conducted on Crescent Island in 2020-2021, or in any other year since 2015.

As was the case in 2019-2020 (USACE-FFU 2021), the USACE-FFU conducted tern monitoring and tern nest dissuasion efforts on East Sand Island in 2021 (K. Tidwell, USACE, pers. comm.). These efforts included the installation of passive nest dissuasion arrays (i.e. stakes, rope, and flagging) in areas where terns were observed to be prospecting for nest sites, the active hazing of terns, and the collection of any tern eggs laid (under permit) outside the designated 1-acre colony area. No active nest dissuasion activities were performed at the former cormorant colony on East Sand Island in 2021 (K. Tidwell, USACE-FFU, pers. comm.). An upcoming report prepared by the USACE-FFU will fully describe any nesting activity by terns and cormorants that was observed on East Sand Island in 2021, as well as any nest dissuasion efforts used to limit tern nesting outside the designated colony area (K. Tidwell, USACE, pers. comm.), as described in the management plan (USFWS 2005).

NESTING DISTRIBUTION & COLONY SIZE

Columbia Plateau Region

In 2021, terns successfully nested at five locations in the CPR: at both managed sites (Goose Island and Crescent Island) and at three unmanaged sites (Badger Island, Harper Island [Sprague Lake], and Lenore Lake [Shoal Island]; see *below* for more details). Nesting by other piscivorous colonial waterbirds in the CPR in 2021 occurred at nine locations for gulls (Miller Rocks, Blalock Islands, Crescent Island, Badger Island, Island 20, Goose Island/Potholes Reservoir, Lenore Lake, Harper Island/Sprague Lake, and Banks Lake), six locations for cormorants (Miller Rocks, Foundation Island, Hanford Reach, Okanogan River Delta, Lenore Lake, and Harper Island/Sprague Lake) and one location for pelicans (Badger Island; see *below* for details).

Managed Caspian Tern Colonies: As has been the case in previous years, terns continued to exhibit interest in the shoreline areas of Goose Island in Potholes Reservoir for roosting and nesting in 2021. As the breeding season progressed, shoreline areas became exposed as reservoir levels receded, opening up these mostly unvegetated areas without passive nest dissuasion (stakes, wire, and flagging) in place to deter terns from using them as nesting habitat. Weekly attendance by terns on Goose Island has been greatly reduced in the years since management was first implemented (2014–2021) compared to patterns of colony attendance prior to management (2010–2013; *Figure 1*). In 2021, tern use of shoreline areas on

Goose Island was highest during May and early June, similar to the tern attendance pattern in 2020 (Collis et al. 2021a, USDA-WS 2021). This contrasts with what was observed in previous management years (2014–2019), when the peak in numbers of terns attending the island was in late June through July, presumably due to an influx of post-breeding birds to the island (Collis et al. 2021a). Passive and active nest dissuasion, plus collection of any tern eggs discovered (96 tern eggs were collected under permit in 2021; S. Fesenmyer, BOR, pers. comm.) were effective in preventing successful tern nesting in these shoreline habitats on Goose Island.
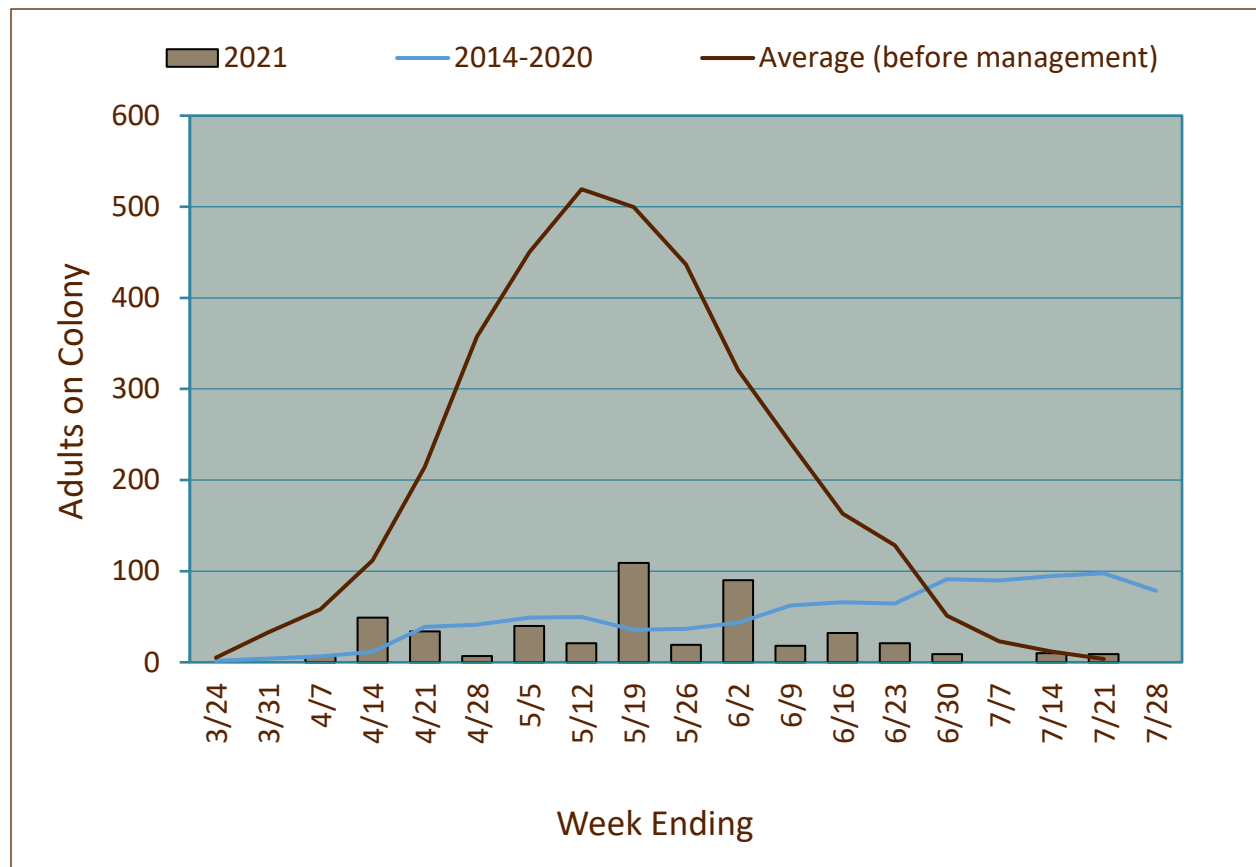
Figure 1. Weekly estimates from boat-based surveys showing the average number of adult Caspian terns on Goose Island and surrounding islets in Potholes Reservoir before tern management (2010–2013) and during tern management (2014–2020, 2021). During the management period, counts do not include terns present on the island but obscured from view from the boat (i.e. highest elevation of island near the historical colony site and where birds nested successfully in 2020 and 2021) as our island access was limited to boat-based observations during weekly monitoring visits to the island. As such, the management period counts represent tern use of shoreline habitats and do not include the nesting areas near the historical colony site.

In 2021, the flagging, a major component of the nest dissuasion array, moved down the wires during windstorms, leaving large areas of suitable tern nesting habitat without flagging during the 2021 breeding season. The use of braided rope as part of the nest dissuasion array prior to 2021, although likely requiring more maintenance than wire, prevented the movement of flagging along the rope because the flagging was installed between the strands of rope. Efforts were made by the contractor to fix this problem during the 2021 field season; however, despite these efforts, terns were successful in establishing a colony (22 breeding pairs; *Figure 2*) and rearing young at a site on top of the island near the former tern colony site. This was the second consecutive year that terns were successful in establishing a colony on top of the island; in 2020, 6 breeding pairs were successful in rearing young on Goose Island (*Figure 2*; Collis et al. 2021a).
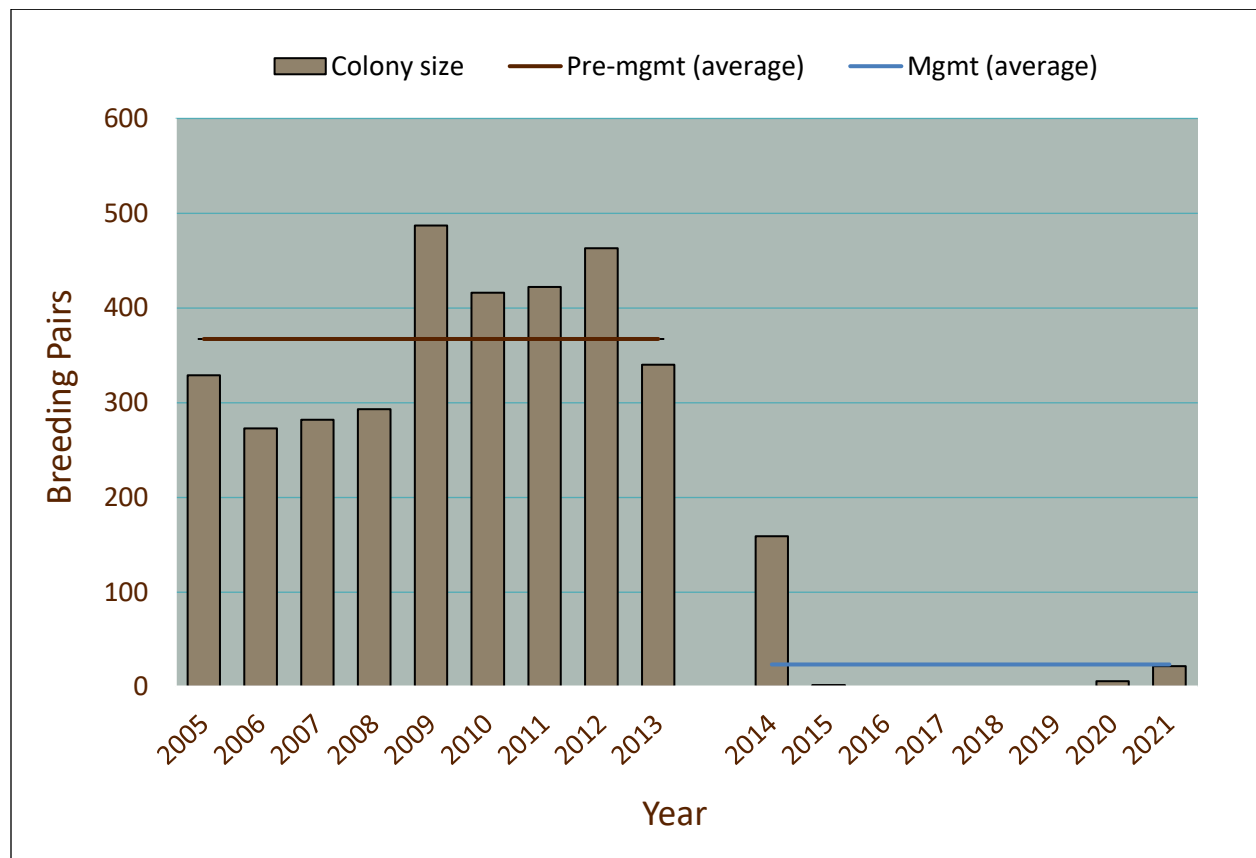
Figure 2. Annual colony size (number of breeding pairs) of Caspian terns at Goose Island and surrounding islets in Potholes Reservoir before tern management (2005–2013) and during tern management (2014–2021) in the region. No Caspian tern nesting occurred at Goose Island or surrounding islets during 2016–2019.

Successful nesting activity by terns on Goose Island in 2021 was likely due, in part, to problems with the newly deployed wire and flagging approach and insufficient active hazing and egg collection on top of the island where the tern colony had become re-established. The continued fidelity of terns to Potholes Reservoir despite ongoing nest dissuasion activities may be the result of a long history of tern nesting on the reservoir combined with the persistence of a large gull colony on Goose Island, both before and during management, which may function as social attraction to prospecting terns. The paucity of alternative tern colony sites in the CPR and vicinity may also be an additional factor that contributes to the strong fidelity of terns to the Potholes Reservoir area. Our research efforts indicate that continued adaptive management will be necessary to prevent terns from nesting on Goose Island in the future.

Prospecting terns were documented at 12 sites in northern Potholes Reservoir in 2021 (S. Fesenmyer, BOR, pers. comm.), down from 16 sites used by prospecting terns in 2020 (USDA-WS 2021). As was the case in 2020, targeted hazing alone was successful in preventing the formation of an incipient tern colony on islands in northern Potholes Reservoir in 2021. Egg laying by terns at these sites was not documented in 2020 (USDA-WS 2021) or in 2021 (S. Fesenmyer, BOR, pers. comm.). While these results are encouraging, some terns continue to show strong fidelity to Goose Island and other islands in Potholes Reservoir and may renest at sites in Potholes Reservoir in the future if management actions are not implemented to discourage tern nesting.

The recent reduction of vegetative cover on Crescent Island, perhaps caused by nesting gulls, beaver herbivory, and/or weather-related events (e.g., windstorms, drought), has created small patches of open, unvegetated habitat that are suitable for tern nesting. In 2021, one pair of Caspian terns was successful in raising young on Crescent Island, the first year of successful tern nesting at that site since 2014, the last breeding season before management commenced in 2015 (*Figure 3*). As was the case in 2020, the passive nest dissuasion that remained on Crescent Island (i.e. willows and other native vegetation) was not sufficient to prevent gulls from nesting on Crescent Island in 2021, which may have been a factor in terns re-establishing a colony on the island. Gulls, both California and ring-billed, were successful in re-establishing a breeding colony of approximately 400 breeding individuals in 2020 and ca. 2,000 breeding individuals in 2021 (see *below*). If growth in the size of the gull colony continues, we expect that the number of terns attempting to nest on Crescent Island will also increase in the future.
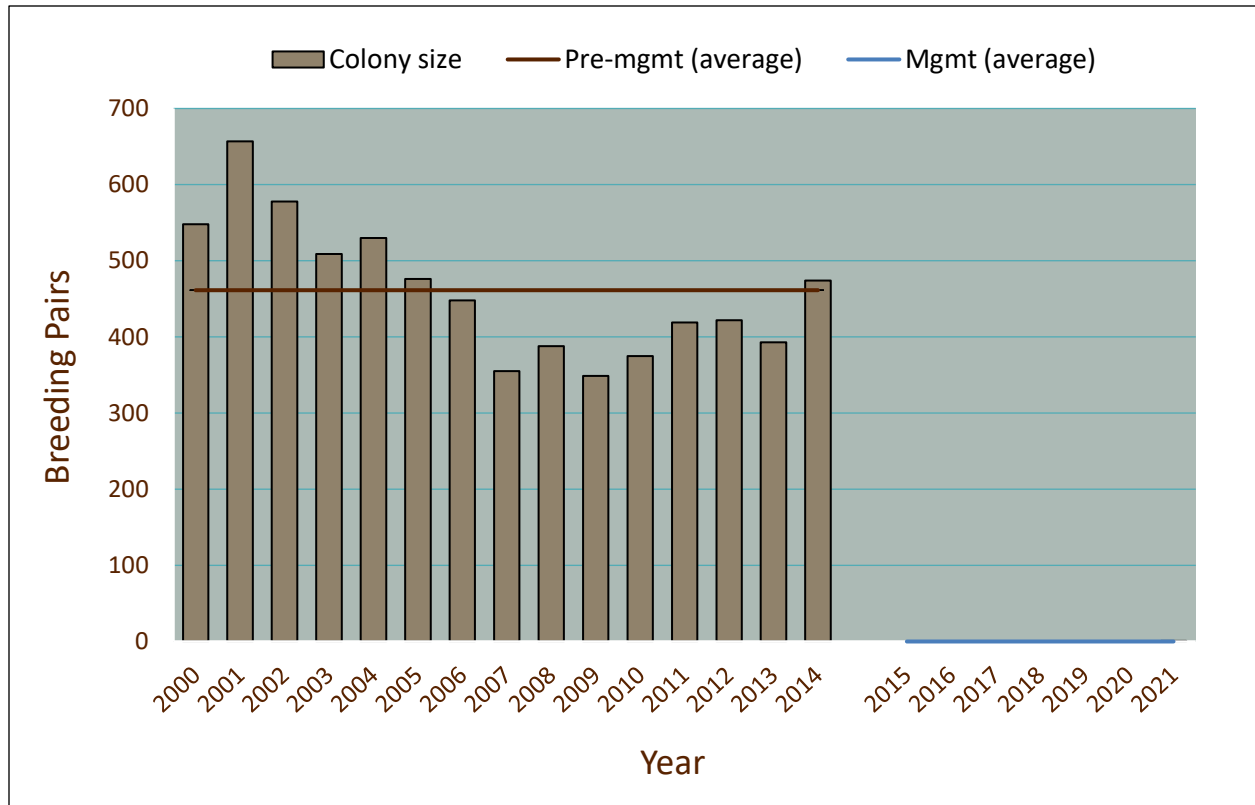
Figure 3. Annual colony size (number of breeding pairs) of Caspian terns at Crescent Island in the mid-Columbia River before tern management (2005–2014) and during tern management (2015–2021). No Caspian terns nested at Crescent Island during 2015–2020; one Caspian tern breeding pair successfully nested on Crescent Island in 2021.

Unmanaged Caspian Tern Colonies: Nesting by terns was confirmed at three unmanaged colony sites in the CPR in 2021: Badger Island in McNary Reservoir, Harper Island in Sprague Lake, and Shoal Island in Lenore Lake (*Table 1, Figure 4*). The former tern colony sites in the Blalock Islands in John Day Reservoir, on North Rock Island in Lenore Lake, and on Twinning Island in Banks Lake were not active in 2021 (*Table 1, Figure 4*). The largest tern colony in the CPR in 2021 was on Badger Island (see *below*), representing 58% of the total number of tern breeding pairs in the region during 2021 (*Table 1, Figure 4*).

Table 1. Annual colony size (number of breeding pairs) and region-wide total number of Caspian terns nesting at both managed and unmanaged colonies in the Columbia Plateau region prior to management (2005–2013) and during management (2014–2021).

Colony	Year																
	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Goose Is. (Potholes Reservoir)	325	273	282	293	487	416	422	463	340	159	2	0	0	0	0	6	22
Crescent Is. (Columbia River)	476	448	355	388	349	375	419	422	393	474	0	0	0	0	0	0	1
Blalock Islands (Columbia River)	6	110	43	104	79	136	20	6	26	45	677	483	449	313	379	150	0
Badger Is. (Columbia River)	0	0	0	0	0	0	33	60	0	0	0	0	41	8	0	0	231
Twinning Is. (Banks Lake)	13	23	31	27	61	34	19	22	13	67	64	6	0	0	0	0	0
Harper Is. (Sprague Lake)	7	7	0	11	4	4	4	30	1	8	10	3	92	79	18	0	85
North Rocks and Shoal Is. (Lenore Lake)	0	0	0	0	0	0	0	0	0	0	0	0	123	91	48	53	61
Total	**827**	**861**	**711**	**823**	**980**	**965**	**917**	**1003**	**773**	**755**	**769**	**675**	**705**	**491**	**445**	**209**	**400**
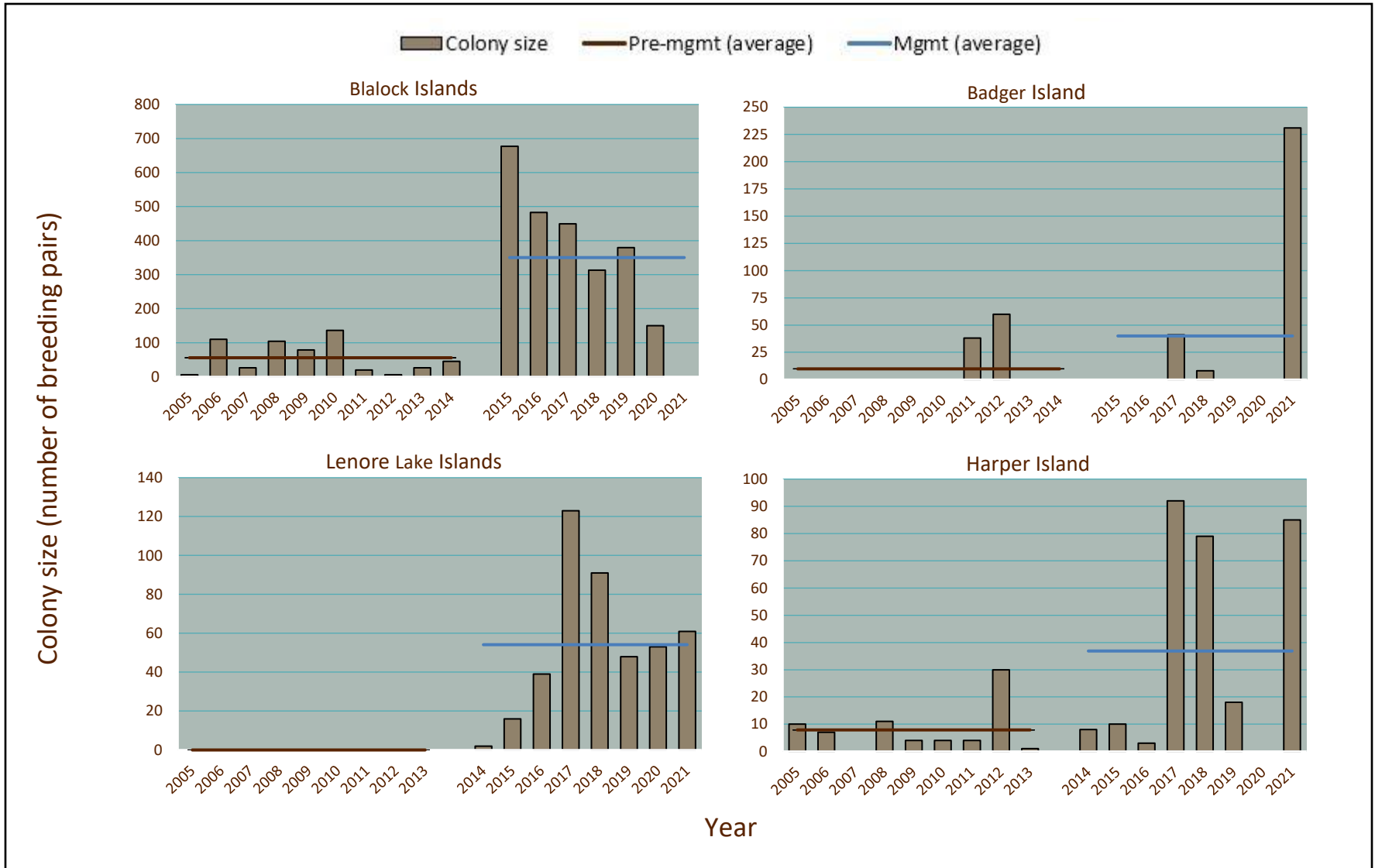
Figure 4. Annual colony size (number of breeding pairs) and average colony size at four unmanaged Caspian tern colonies before tern management (2005–2013) and during management (2014–2021) in the region, including the Blalock Islands complex colony, Badger Island colony, the North Rock and Shoal Island colonies on Lenore Lake, and the Harper Island colony on Sprague Lake. Note differences in the y-axis scales and differences in management period.

A managed increase in the water surface elevation of John Day Reservoir during the 2021 Caspian tern breeding season submerged and eliminated all tern nesting habitat previously used by terns in the Blalock Islands. As a result, a tern breeding colony did not form at the Blalock Islands in 2021, the first year without a tern colony there since the colony was first documented in 2005. It is likely that many of the terns displaced from the Blalock Islands in 2021, however, relocated to establish a new colony on Badger Island, where 231 breeding pairs nested in 2021, some of them successfully. One tern nesting area on Badger Island was located near the shoreline and another in the interior among nesting gulls and American white pelicans. Prior to 2021, terns had not nested successfully on Badger Island since 2018, when colony size was estimated to be 8 breeding pairs. Given that terns were successful in breeding at Badger Island in 2021, it is likely that a tern colony at that site will persist and grow, unless adaptive management to reduce tern nesting on Badger Island is implemented.

Region-wide Caspian Tern Breeding Population: A total of 400 Caspian tern breeding pairs nested at 5 colony sites in the CPR during 2021, double the maximum regional population size stipulated in the *IAPMP* (USACE 2014). The regional population size in 2021 represented a 54% reduction in the size of the regional tern breeding population compared to the pre-management average (873 breeding pairs; *Figure 5*), but a substantial increase compared to 2020 (209 breeding pairs). The general decline in the regional tern breeding population since the *IAPMP* was first implemented in 2014 suggests that some of the terns that formerly nested in the CPR have left the region, which was one of the objectives of the *IAPMP.*
While there is evidence that some terns displaced from the CPR have relocated to colonies outside of the basin (Lawes et al. 2021a), reductions in the number of terns nesting in the CPR have generally not resulted in an increased number of nesting terns in other regions of the Pacific Flyway (Peck-Richardson et al. 2019, Lawes et al. *in prep*.).
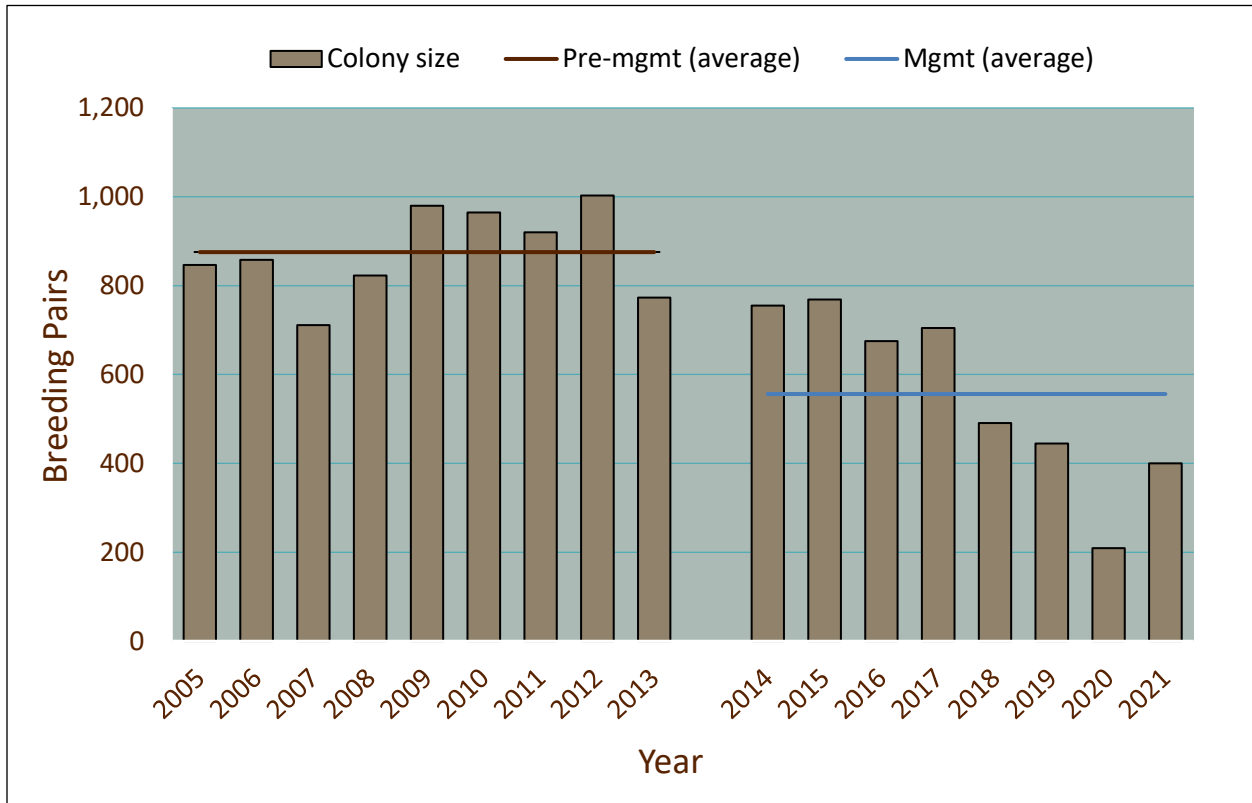
Figure 5. Annual and average number of Caspian tern breeding pairs at all known colonies in the Columbia Plateau region before (2005–2013) and during (2014–2021) tern management in the region.

Other Piscivorous Waterbird Colonies: In total, there were nine active California and ring-billed gull colonies in the CPR region in 2021, ranging in size from ca. 1,200 breeding individuals (Lenore Lake) to over 14,000 breeding individuals (Goose Island; *Table 2*). Double-crested cormorants were confirmed to be nesting at four colonies in the CPR in 2021, with colony size ranging from 147 breeding pairs (Lenore Lake) to 520 breeding pairs (Harper Island in Sprague Lake; *Table 2*). Finally, the size of the American white pelican colony on Badger Island was estimated to be 3,524 breeding individuals in 2021 (*Table 2*).

Table 2. Size of California/ring-billed gull (LAXX, double-crested cormorant (DCCO), and American white pelican (AWPE) nesting colonies in the Columbia Plateau region in 2021. "Active" denotes a colony that was active but not counted in 2021.

Colony	LAXX [1]	DCCO [2]	AWPE [1]
Miller Rocks (Columbia River)	5,491	1	
Blalock Islands (Columbia River)	2,340		
Crescent Is. (Columbia River)	2,084		
Badger Is. (Columbia River)	3,927		3,624
Foundation Is. (Columbia River)		307	
Island 20 (Columbia River)	9,369		
Hanford Reach (Columbia River)		25	
Goose Is. (Potholes Reservoir)	14,595		
North Rocks and Shoal Is. (Lenore Lake)	1,169	147	
Harper Is. (Sprague Lake)	active	520	
Twinning Is. and Goose Is. (Banks Lake)	active		

[1]Number of individuals
[2]Number of breeding pairs

Columbia River Estuary

The USACE estimated that 2,050 breeding pairs of Caspian terns nested on the prepared 1-acre main colony on East Sand Island in 2021 (Brandtner and Tidwell 2022), which is less than the target colony size for terns on East Sand Island specified in the management plan (i.e. 3,125–4,375 breeding pairs; USFWS 2005). However, large numbers of terns (hundreds to thousands of adults per week) attempted without success to nest along the southeast and east beaches of East Sand Island, despite ongoing nest dissuasion efforts to prevent terns from nesting outside of the prepared colony area. None of the tern eggs laid outside the 1-acre colony area are believed to have hatched and none of the tern chicks hatched on the 1-acre designated colony area survived to fledging, with gull depredation of eggs and chicks being the primary factor in tern nest failure on the main colony. This is the third breeding season out of the last five when terns nesting at the designated colony site on East Sand Island have completely failed to raise any young. As was the case in 2020, terns were unsuccessful in establishing a colony on Rice Island or elsewhere in the upper Columbia River estuary (Brandtner and Tidwell 2022).

As was the case in 2020, double-crested cormorants attempted but did not successfully nest on East Sand Island in 2021 (Brandtner and Tidwell 2022). Based on monitoring conducted as part of this study, the size of the cormorant colony on the Astoria-Megler Bridge was estimated to be 4,151 breeding pairs (see *Appendix A*), down from 5,081 breeding pairs in 2020 (J. Lawonn, ODFW, pers. comm.). The estimate of 4,151 breeding pairs should be considered a minimum estimate as some unknown number of cormorants were nesting in areas of the bridge that were not readily visible (i.e. inside beams and girders and within scaffolding) from aerial or boat-based vantage points. Prior to 2021, the cormorant colony on the Astoria-Megler Bridge had increased in size every year since 2011 (Roby et al. 2021), with the recent exponential growth of the cormorant colony on the bridge corresponding with the precipitous decline in the size of the cormorant colony on East Sand Island (Roby et al. 2021).

AVIAN PREDATION RATES

PIT-tagging of Upper Columbia River Steelhead

A total of 7,236 UCR steelhead smolts (5,578 hatchery-reared, 1,658 wild) were captured, PIT-tagged, measured (fork-length), scored based on external body condition, and released into the tailrace of RIS as part of this study in 2021. An additional 854 previously PIT-tagged (i.e. recaptured) steelhead smolts were also detected and sampled at the RIS trap, resulting in a total of 8,090 UCR steelhead available for predation and survival rate analyses in 2021. Steelhead were tagged and released at RIS from 9 April to 13 June 2021, a period which accounted for 98% of all steelhead encountered in the trap. Mean fork length was 188 mm (standard deviation [SD] = 23 mm; range = 80–299 mm) for all measured steelhead smolts. An evaluation of external fish condition indicated that most steelhead were in good over-all external condition in 2021, with just 4.3% of steelhead observed with severe body injuries (subcutaneous wounds/scars), disease (fungal or viral infections), severe descaling (>20% of scales missing), and/or major fin damage (>50% of fin tissue missing). The most common type of damage was body injuries, followed by severe descaling. For comparison, between 3.6% and 23.1% of steelhead tagged in previous years were in compromised conditions (Evans et al. 2014, Collis et al. 2021a).

PIT Tag Recovery

As part of this study, a total of 13 avian breeding colonies in the CPR were scanned for smolt PIT tags following the 2021 breeding season, including four tern colonies, six gull colonies, two cormorant colonies, and one pelican colony (*Table 3*). Scanning was also conducted at three known avian loafing/roosting sites in the CPR, areas where large numbers of piscivorous waterbirds were observed during the smolt out-migration period. A total of 24,284 PIT tags from 2021 migration-year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead combined) were recovered from both piscivorous waterbird breeding and loafing sites combined (*Table 3*), among the highest total number of smolt PIT tags recovered since scanning efforts at multiple colonies commenced in 2007 (Cramer et al. 2021a). Over 98% of

tags were recovered from the breeding colony sites, with the largest number of smolt PIT tags recovered on the gull, tern, and pelican colonies at Badger Island (n = 10,947), followed by the gull colony at Miller Rocks (n = 4,130), the cormorant colony at Foundation Island (n = 3,350), and gull colonies at the Blalock Islands (n = 1,640; *Table 3*). One breeding pair of Caspian terns nested at Crescent Island in 2021 and we recovered 14 smolt PIT tags at the location of the nest (*Table 3*). At Goose Island in Potholes Reservoir, where a colony of 22 tern breeding pairs nested, we recovered 893 smolt PIT tags. Additional terns attempted to nest unsuccessfully or roost at Goose Island, however, so not all recovered tags can be attributed to the colony of 22 breeding pairs. As many as 109 individual adult terns were observed during boat-based surveys of Goose Island during the peak smolt out-migration period in May, surveys that did not count terns nesting at the colony site near the top of the island, as it was not visible from the water.

Table 3. Number of recovered 2021 migration year PIT tags from salmonid smolts (Chinook salmon, sockeye salmon, coho salmon, and steelhead combined) and other age-classes and fish species at avian breeding colonies and loafing sites in the Columbia River basin. Piscivorous colonial waterbird species included Caspian terns (CATE), California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), Brandt's cormorants (BRAC), and American white pelicans (AWPE). Mixed loafing sites represent an unknown combination of the above and possibly other avian species. Recovered PIT tag numbers were not adjusted to account for tag loss due to on-colony PIT detection and deposition probabilities (see Table 4) and represent the minimum number of tagged fish consumed by birds at each site (see Methods & Analysis section).

Location	Rkm	Bird Species	Area Use	# Smolt PIT Tags	# Other PIT Tags[3]
Goose Island	Off-river	CATE	Breeding	893	
		Mixed	Loafing	39	
Lenore Lake	Off-river	CATE	Breeding	388	
Hanford Island	592	DCCO	Breeding	229	1
Island 20	549	LAXX	Breeding	1,664	
Foundation Island	518	DCCO	Breeding	3,350	
Badger Island	512	LAXX	Breeding	324	
		AWPE	Breeding	105	
		AWPE/LAXX	Breeding	8,424	55
		CATE	Breeding	2,094	
		Mixed	Loafing	189	
Burbank Slough	Off-river	CATE	Loafing	119	
Crescent	510	LAXX	Breeding	682	
		CATE	Breeding	14	
Blalock Islands	441-439	LAXX	Breeding	1,640	
Miller Rocks	331	LAXX	Breeding	4,130	
Miller Sands Island	38	AWPE	Breeding	53	
Astoria-Megler Bridge		DCCO	Breeding	1,344[1]	11[1]
		DCCO/BRAC	Loafing	416[1]	
East Sand Island	8	CATE	Breeding	4,230[2]	12[2]
			Breeding (Satellite)	851[2]	2[2]
		DCCO	Breeding	117	
		Mixed	Loafing	2	
Total				**31,297**	**71**

[1] *Tags recovered from a sub-sample of nests (see Appendix A)*

[2] *Tags recovered by the USACE*

[3] *Includes adult sockeye salmon (n = 49), sea-run cutthroat trout (n = 14; estuary colonies only), adult steelhead (n = 2), adult Chinook salmon (n = 2), white sturgeon (n = 3), and bull trout (n = 1).*

In addition to the CPR sites, we also scanned and recovered smolt PIT tags at avian breeding colony and loafing sites in the CRE downstream of Bonneville Dam, including the Astoria-Megler Bridge (n = 1,344 from breeding cormorants, n = 407 from loafing cormorants), East Sand Island (n = 117 from breeding cormorants, n = 2 from loafing cormorants), and Miller Sands Spit (n = 53 from breeding pelicans; see *Table 3* above) to estimate predation and to collect data to inform regional survival models. As part of USACE-funded efforts, scanning for smolt PIT tags also occurred at the East Sand Island tern colony in the CRE, where 4,230 and 851 PIT tags from 2021 migration year smolts were recovered from the prepared 1-acre colony area and from satellite breeding colonies located along the beach, respectively (Brandtner and Tidwell 2021). From all estuary locations combined, a total of 7,013 PIT tags from 2021 migration year smolts were recovered following the breeding season (see *Table 3* above). In comparison, a total of 4,537 PIT tags from 2021 migration year smolts were detected by the National Marine Fisheries Services pair trawl net detection system in the CRE (PSFMC 2021).

In addition to PIT tags from 2021 migration year smolts, a total of 55 tags from other salmonid species and age-classes and from non-salmonid species were recovered on the Badger Island pelican colony. This included PIT tags from 47 adult sockeye salmon, 2 adult steelhead, 2 adult Chinook salmon (both jacks), 3 juvenile white sturgeon (*Acipenser transmontanus*), and 1 bull trout (*Salvelinus confluentus;* see *Table 3* above). The 47 PIT tags from adult sockeye salmon (fish tagged as adults at the Bonneville Dam Adult Fish Facility in 2021) that were recovered on the Badger Island pelican colony in 2021 was the highest number recorded since scanning on the Badger Island colony commenced in 2007, with a previous record of 23 PIT tags from adult sockeye salmon in 2020. Tags from two adult sockeye salmon (both jacks) were also recovered on the Hanford Island cormorant colony (n = 1) and the Astoria-Megler Bridge colony (n = 1) in 2021 (see *Table 3* above). A total of 14 tags implanted in sea-run cutthroat trout (*O. clarkii clarkii*) were recovered on the East Sand Island Caspian tern colony in 2021 (see *Table 3* above).

PIT Tag Detection & Deposition Probabilities

Table 4 provides results for each piscivorous waterbird colony on the probability that a consumed PIT tag was deposited on-colony (i.e. deposition probability) and the probability that a deposited tag was detected by researchers following the breeding season (i.e. detection probability), data necessary to accurately estimate predation/consumption rates. Deposition probabilities were based on previous studies that empirically measured deposition rates for terns, cormorants, gulls, and pelicans at their breeding colonies (see also Hostetter et al. 2015 and Payton et al. 2021b). Detection probabilities were directly measured in 2021 based on the proportion of tags intentionally sown by researchers on each colony that were subsequently detected after the breeding season (see Hostetter et al. 2015 and Roby et al. 2021). Similar to results in previous years, detection probabilities were highly variable depending on the colony and when tags were sown during the breeding season (at the beginning vs. at the end; Table 4). The lowest detection probabilities were recorded at colonies where detection efficiency has gradually decreased over the years due to increasing rates of PIT tag collision as PIT tags

accumulate on-colony. PIT tag collision is caused by high densities of tags, a phenomenon that renders tag codes in close proximity to one another unreadable (see also Evans et al. 2019).

Table 4. Detection efficiency (range during nesting season) and deposition (95% credible interval) estimates (depicted as a proportion) for smolt PIT tags on bird colonies during 2021. Estimates were used to generate predation rates based on the number of smolt PIT tags recovered following the breeding season (see Table 3 above). Piscivorous colonial waterbird species include Caspian terns (CATE), California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), and American white pelicans (AWPE).

Location	Rkm	Bird Species	Detection (Range)	Deposition (95% CRI)[1]
Lenore Lake[2]	Off-river	CATE	0.54–0.99	0.71 (0.51–0.89)
Goose Island[2]	Off-river	CATE	0.15–0.86	0.71 (0.51–0.89)
Hanford Island	592	DCCO	0.52	0.51 (0.34–0.70)
Island 20	549	LAXX	0.60–0.90	0.15 (0.11–0.21)
Foundation Island	549	DCCO	0.16–0.30	0.51 (0.34–0.70)
Badger Island	512	AWPE	0.72–0.99	0.47 (0.24–0.73)
		LAXX	0.92–0.92	0.15 (0.11–0.21)
		CATE	0.72–0.88	0.71 (0.51–0.89)
Crescent Island	510	LAXX	0.68–0.96	0.15 (0.11–0.21)
		CATE	0.68–0.96	0.71 (0.51–0.89)
Blalock Islands	441-439	LAXX	0.95–0.98	0.15 (0.11–0.21)
Miller Rocks	331	LAXX	0.78–0.92	0.15 (0.11–0.21)
Miller Sands Spit	34	AWPE	0.96	0.47 (0.24–0.73)
Astoria-Megler Br.	21	DCCO	0.67–0.84	0.51 (0.34–0.70)
East Sand Island	8	CATE	0.58–0.86[3]	0.71 (0.51–0.89)

[1] Deposition estimates for CATE, LAXX, and DCCO are those of Hostetter et al. (2015); estimates for AWPE are those of Payton et al. (2021b).
[2] Variation in detection was partially inferred from other years (see Payton et al. 2019)
[3] Data courtesy of USACE

Efficacy of Avian Predation Management Plans

Currently there are three avian predation management plans being implemented to increase survival of ESA-listed juvenile salmonids in the CRB. Where data were available, we measured predation rates at both managed and unmanaged colonies in 2021 and compared those rates to those prior to implementation of management to evaluate the efficacy of these management plans.

Inland Avian Predation Management Plan: Estimates of tern predation rates were higher for most salmonid ESUs/DPSs evaluated in 2021, compared with those in 2020 (*Table 5* and *Table 6*). In 2020, estimates of predation by terns were amongst the lowest ever recorded in the CPR and the goal of reducing predation to less 2% per colony, per ESU/DPS, was achieved for nearly

all salmonid ESUs/DPSs evaluated (see *Appendix B, Table B1*). In 2021, estimates of tern predation rates were near or above the 2% threshold for several ESUs/DPSs and the overall or cumulative rate of tern predation was greater than 5% on ESA-listed UCR steelhead with an estimated 6.3% (95% credible interval = 4.5–9.3%) depredated by terns from all colonies in the CPR combined (see *below*).

Table 5. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESUs/DPSs, with runs of spring (Sp), summer (Su), and fall fish, by Caspian terns nesting on Goose Island in Potholes Reservoir, Shoal Island in Lenore Lake, Badger Island in McNary Reservoir, and Crescent Island in McNary Reservoir during 2021. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable predation estimates (see Methods). See Appendix B for estimates of predation by these colonies in previous years.

ESU/DPS	Goose Island	Lenore Lake	Badger Island	Crescent Island
SR Sockeye	NA	NA	NA	NA
SR Sp/Su Chinook	< 0.1%	< 0.1%	0.8% (0.2–2.4)	< 0.1%
UCR Sp Chinook	0.3% (0.1–1.5)	0.2% (0.1–0.8)	< 0.1%	< 0.1%
SR Fall Chinook	< 0.1%	< 0.1%	0.3% (0.1–0.8)	< 0.1%
SR Steelhead	< 0.1%	< 0.1%	1.5% (0.7–2.9)	< 0.1%
UCR Steelhead	3.9% (2.4–6.8)	0.9% (0.5–1.5)	1.4% (1.0–2.2)	< 0.1%

Table 6. Average annual predation rates (95% credible intervals) by Caspian terns at managed colonies during the pre-management period (2007–2013 for Goose Island and North Potholes Is.; 2007–2014 for Crescent Island) and during the management period (2014–2021 for Goose Island and North Potholes; 2015–2021 for Crescent Island). ESA-listed salmonid populations (ESUs/DPSs) from the Snake River (SR) and Upper Columbia River (UCR), with runs of spring (Sp), summer (Su), and fall fish evaluated. Time periods are denoted as the average of all years with data or data from the last three years of the study period (2019–2021). See Appendix B for annual estimates. NC denotes that no colony existed during that period.

ESU/DPS	Goose Is.			North Potholes Is.			Crescent Is.		
	Pre-Management 2007–2013	Management 2014–2021[1]	Last 3-years 2019–2021[1]	Pre-Management 2007–2013	Management 2016[2]	Last 3-years 2019–2021	Pre-Management 2007–2014	Management 2015–2021	Last 3-years 2018–2021[3]
SR Sockeye	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	NC	1.5% (1.2–2.0)	< 0.1%	< 0.1%
SR Sp/Su Chinook	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	NC	0.8% (0.7–1.0)	< 0.1%	< 0.1%
UCR Sp Chinook	2.5% (1.7–3.6)	< 0.1%	< 0.1%	NC	0.1% (0.1–0.3)	NC	0.5% (0.3–0.9)	< 0.1%	< 0.1%
SR Fall Chinook	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	NC	1.0% (0.9–1.2)	< 0.1%	< 0.1%
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	NC	4.5% (4.1–5.1)	< 0.1%	< 0.1%
UCR Steelhead	15.7% (14.1–18.9)	1.2% (0.7–1.9)	1.3% (0.7–2.0)	NC	4.1% (2.9–6.3)	NC	2.5% (2.2–2.9)	< 0.1%	< 0.1%

[1] *Caspian terns were observed during the peak smolt out-migration period in May of 2020 (USDA-WS 2021), but tags were not recovered that year*
[2] *Nesting colony formed in 2016 and was successfully managed in all subsequent years*
[3] *One pair of Caspian terns was observed nesting in 2021, the first year since management was first implemented on Crescent Island in 2015*

Increases in tern predation in 2021 were due to the re-establishment of the colony on Goose Island in Potholes Reservoir; the estimated predation rate on UCR steelhead was 3.9% (2.4–6.8%) and on UCR spring Chinook salmon was 0.3% (0.1–1.5%; see *Table 5* above). Weekly estimates of predation probability indicated that steelhead smolts were consumed by Goose Island terns from late-April to early-June, coincident with the run-timing of steelhead in the middle Columbia River. Most terns that attempted to nest on Goose Island were at the historical colony site (i.e. the site used prior to management), with 22 breeding pairs observed during the traditional peak nesting period in late May. The attendance of terns at Goose Island during the peak of the smolt out-migration period in May was amongst the highest (minimum of 109 adults) observed since 2015 and indicated that far more terns attempted to nest on Goose Island than implied by the 22 breeding pairs that nested together at the top of the island. Since 2015 and prior to 2020, only small numbers of terns (less than 50 adults) were observed on Goose Island during the peak breeding and smolt out-migration period (Collis et al. 2021a, b). Starting in 2020, however, larger numbers (upwards of 180; USDA-WS 2021) of terns have been observed during the peak smolt out-migration period and a successful colony has formed in both 2020 and 2021. Predation rates on UCR steelhead by Goose Island terns prior to implementation of management actions in 2014 were the highest of any tern colony in the region, averaging 15.7% (14.1–18.9%) during 2007-2013 (see *Table 6* above). Estimated average annual predation rates on UCR Spring Chinook salmon were 2.5% (1.7–3.6%) during 2007–2013 (see *Table 6* above). In 2016, a colony of 144 breeding pairs formed on an unnamed island in northern Potholes Reservoir. Recoveries of smolt PIT tags indicated that terns consumed an estimated 4.1% (2.9–6.3%) of UCR steelhead in 2016 (see *Appendix B, Table B1*), impacts that prompted adaptive management actions at this and other surrounding islands in northern Potholes Reservoir during 2017–2021. Since then, active and passive dissuasion techniques have been successful at preventing terns from nesting on islands in northern Potholes Reservoir, thereby eliminating impacts from tern predation. The estimate of predation probability on UCR steelhead by terns nesting on Goose Island in 2021 exceeded the 2% threshold identified in the *IAPMP*. Due to the re-established and continued growth in the size of the tern colony on Goose Island in 2020 and 2021, increased efforts to prevent terns from nesting using both passive and active dissuasion techniques will likely be necessary in 2022 and beyond to achieve the goals of the *IAPMP*.

For the first time in seven years, terns nested on Crescent Island in McNary Reservoir, but only one pair was confirmed during the peak nesting period in late May (see *Table 1* above). Based on the paucity of smolt PIT tags recovered near the nest (n = 14; see *Table 3* above), predation rates were < 0.1% for all ESUs/DPSs in 2021 (see *Table 5* above). During 2015–2020, no (zero) terns nested on Crescent Island, effectively reducing the impact of tern predation to zero in those years (Collis et al. 2021a, b). Prior to management actions in 2015, predation rates on salmonids by Crescent Island terns were highest on steelhead populations, with an estimated average annual predation rate of 2.5% (2.2–2.9%) and 4.5% (4.2–5.1%) on UCR steelhead and SR steelhead, respectively (see *Table 6* above). Average annual predation rates on salmon ESUs were less than 2% prior to management, with the highest predation rates observed on SR

sockeye salmon at 1.5% (1.2–2.0; see *Table 6* above). Comparisons of predation rates prior to and during management indicated that management actions at Crescent Island have been successful in reducing predation rates, with the greatest benefits to SR salmonids due to the greater susceptibility of SR salmonid ESUs/DPSs compared with UCR ESUs/DPSs at that colony (see *Table 6* above; see also Collis et al. 2021b). Due to the re-establishment and growth of a gull colony on Crescent Island in 2020-2021 and the presence of a nesting pair of terns in 2021, future management to prevent terns from nesting on Crescent Island may be necessary to achieve the goals of the *IAPMP*.

Estimates of predation rates on smolts at the unmanaged tern colony on Lenore Lake were below the 2% threshold for all salmonid ESUs/DPSs evaluated in 2021, with the highest rate of 0.9% (0.6–1.5%) on UCR steelhead (see *Table 5* above). Rates were 0.2% (0.1–0.8%) for UCR spring Chinook salmon and less than 0.1% for all other ESUs/DPSs (see *Table 5* above). Predation rate estimates for terns nesting on islands in Lenore Lake were available starting in 2015, the first year a colony was observed at Lenore Lake, and estimates have been consistently at or below 1.0% per ESU/DPS (see *Appendix B, Table B3*). These results suggest that at its current size the tern nesting colony on Lenore Lake poses a relatively minor risk to the survival of UCR steelhead and poses little to no risk to other salmonid populations. Based on average annual per-capita predation rates on UCR steelhead by terns nesting at Lenore Lake, the nesting colony would have to increase to about 150 or more breeding pairs to potentially reach the 2% predation threshold identified in the *IAPMP*.

Based on aerial imagery taken during the peak of the breeding season in May of 2021, there were no terns nesting at the unmanaged colony site on Twinning Island in Banks Lake in 2021. The last time a tern colony was observed on Twinning Island was in 2016 (Collis et al. 2021b). In 2014 and 2015, following implementation of management actions at the nearby tern colony on Goose Island in Potholes Reservoir, 66 pairs and 64 pairs nested on Twinning Island, respectively, and predation rates on UCR steelhead were 1.2% (0.3–6.4%) and 2.6% (1.8–3.9%), respectively. These results demonstrate that terns nesting in Banks Lake commute to the middle Columbia River to forage on smolts and that impacts can exceed the 2% threshold in some years (see *Appendix B, Table B1*). As such, continued monitoring of Twinning Island and other islands in Banks Lake to detect any re-establishment of a tern colony is warranted.

There was a tern colony on Harper Island in Sprague Lake (see *Map 1* above) in 2021, where an estimated 85 pairs nested, the largest number observed at this site since monitoring began in 2008 (Collis et al. 2021b). Harper Island has had a relatively small tern colony (< 40 breeding pairs) in most years (Collis et al. 2021b). Because the island is privately owned, however, we were not able to scan the island for PIT tags in 2021. Scanning for PIT tags at the Harper Island tern colony has occurred only once in the past (2012), the only year when island access was granted for PIT tag recovery by the private landowner (Collis et al. 2021b). Predation rate estimates in 2012 indicated that terns consumed less than 0.3% of available SR and UCR ESUs/DPSs, with the highest rates observed on SR steelhead at 0.2% (see Roby et al. 2021,

Appendix B). Predation rates on smolts were low, at least in part, due the small size of the colony (30 breeding pairs; Collis et al. 2021b) in 2012. Given the colony was substantially larger in 2021, future monitoring of predation rates at this colony may be warranted if a larger tern colony becomes established and if landowner permission is granted to access the island to scan for smolt PIT tags.

Estimates of predation rates by terns nesting on Badger Island, the largest tern colony in the CPR in 2021, ranged from 0.3% (0.1–0.8%) on SR fall Chinook salmon to 1.5% (0.7–2.9%) on SR steelhead (*Table 7*). Estimates of predation rates were the highest on SR steelhead and on UCR steelhead at 1.4% (1.0–2.2%), estimates that were consistent with, but slightly lower than, that expected based on a colony size of 231 nesting pairs in McNary Reservoir. For example, prior to management actions, predation rates by terns nesting on Crescent Island, located just 2 Rkm downstream of Badger Island, averaged 2.5% (2.2–2.9%) and 4.5% (4.1–5.1%) on UCR and SR steelhead, respectively, with an average colony size of 397 nesting pairs during 2007–2014 (see *Table 1* above and *Appendix B, Table B1*). Per capita predation rates may have been lower than expected due to the delayed formation of the colony on Badger Island in 2021, whereby terns were first documented nesting on Badger Island in early-May, roughly 2–3 weeks later than nest initiation at most other tern colonies in the CRB. This finding was further supported by weekly estimates of predation rates, whereby predation impacts by terns nesting on Badger Island in 2021 were first documented for smolts last detected alive passing Rock Island Dam or Lower Monumental Dam in early May, as compared to mid-April for other tern colonies in CPR.

Table 7. Average annual predation rates (95% credible intervals) for Caspian terns nesting at unmanaged colonies during the pre-management period (2007–2013) and during the management period (2014–2021). ESA-listed salmonid populations (ESUs/DPSs) from the Snake River (SR) and Upper Columbia River (UCR), with runs of spring (Sp), summer (Su), and fall fish evaluated. See Appendix B for annual estimates. Time periods represent the average of all years or data from the last three years of the management (2019–2021). NA denotes that predation estimates were not available. NC denotes that no colony existed during that period.

	Twinning Island			Badger Island			Blalock Islands		
	Pre-Management	Management	Last 3-years	Pre-Management	Management	Last 3-years	Pre-Management	Management	Last 3-years
ESU/DPS	2008–2013	2014–2017	2019–2021	2007–2013[1]	2017, 2021[2]	2021[2]	2007–2013	2014–2020[3]	2018–2020[3]
SR Sockeye	< 0.1%	0.1% (0.0–0.5)	NC	NA	NA	NA	0.2% (0.1–0.4)	1.6% (1.0–2.5)	1.8% (0.7–4.0)
SR Sp/Su Chinook	< 0.1%	< 0.1%	NC	NA	0.4% (0.1–1.1)	0.8% (0.2–2.4)	0.1% (0.1–0.2)	0.6% (0.5–0.8)	0.4% (0.2–0.6)
UCR Sp Chinook	< 0.1%	0.2% (0.0–0.7)	NC	NA	< 0.1%	< 0.1%	< 0.1%	0.6% (0.4–0.8)	0.5% (0.2–0.9)
SR Fall Chinook	< 0.1%	< 0.1%	NC	NA	0.2% (0–0.4)	0.3% (0.1–0.8)	< 0.1%	0.7% (0.5–0.9)	0.8% (0.5–1.2)
SR Steelhead	< 0.1%	< 0.1%	NC	NA	0.4% (0.1–1.1)	1.5% (0.8–2.9)	0.5% (0.4–0.9)	3.5% (2.9–4.3)	2.6% (1.8–3.8)
UCR Steelhead	0.1%	1.1% (0.8–1.6)	NC	NA	0.9% (0.6-1.3)	1.4% (1.0–2.2)	0.5% (0.3–0.7)	4.0% (3.2–5.0)	3.5% (2.3–5.2)

	Lenore Lake Islands			Harper Island		
	Pre-Management	Management	Last 3-years	Pre-Management	Management	Last 3-years
ESU/DPS	2007–2013	2014–2021	2019–2021	2007–2013	2014–2021	2019–2021
SR Sockeye	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
SR Sp/Su Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
UCR Sp Chinook	NC	0.1% (0.1–0.3)	0.1% (0.0–0.4)	< 0.1%	NA	NA
SR Fall Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
SR Steelhead	NC	< 0.1%	< 0.1%	0.2% (0.1–1.3)	NA	NA
UCR Steelhead	NC	0.7% (0.6–1.0)	0.9% (0.7–1.2)	< 0.1%	NA	NA

[1] *Colonies existed in 2011 and 2012 but no estimates of predation were available in these years*
[2] *Colonies and predation estimates were available in 2017 and 2021*
[3] *No established tern nesting colony was present in the Blalock Islands during 2021*

For the first time since studies of avian predation in the CPR commenced in 2007, there was no Caspian tern colony on the Blalock Islands in John Day Reservoir, the site of the largest tern colony in the CPR during 2015–2020 (see *Table 1* above). The managed increase in water levels in the John Day Reservoir in 2021 inundated (flooded) the former colony sites on low-lying islands in the Blalock Islands. Prior to 2021, predation rates by terns nesting in the Blalock Islands had been, on average, significantly higher since management actions on the Crescent Island tern colony were implemented in 2015 (see *Appendix B, Table B2*). During 2007–2020, increases in predation rates were commensurate with the increase in the size of the Blalock Islands tern colony, with the colony increasing from an average of 57 breeding pairs (range = 6 to 136) during 2007–2014 to an average of 409 breeding pairs (range = 150 to 677) during 2015–2020 (see *below*; also see Collis et al. 2021a, b). During the management period, the estimated average annual predation rates by terns nesting on the Blalock Islands was comparable to or higher than that of terns nesting on Crescent Island during the pre-management period for most of the ESUs/DSPs evaluated, particularly for ESUs/DPSs originating from the Snake River (see *Table 7* above). For example, predation rates on SR steelhead by tern nesting on the Blalock Islands increased from an average of 0.5% (0.4–0.9%) prior to management to an average of 4.0% (3.3–4.9%) during implementation of management at Goose and Crescent islands during 2015–2020 (see *Table 7* above). Consequently, as described by Collis et al. (2021b), increases in predation rates on salmonid smolts by terns nesting on the Blalock Islands had offset the benefits achieved by the elimination of the tern colonies on Crescent and Goose islands due to management. Although average annual predation rates have been significantly higher during much of the management period, the impacts of predation by terns from this colony were greatly reduced (if not eliminated) in 2021 due the inundation of the former colony site on the Blalock Islands. Results indicated that the elimination of nesting habitat on the Blalock Islands successfully prevented the formation of a tern colony on the Blalock Islands and therefore greatly reduced predation by terns in 2021.

In summary, predation rates by terns nesting at colonies in the CPR in 2021 were generally higher than those observed in 2020, commensurate with increases in the size of several tern colonies in the CPR, as well as the overall tern breeding population in the CPR in 2021. Throughout the course of the management period, however, reductions in tern colony sizes at both Goose Island and Crescent Island have, on average, reduced predation by terns breeding at these sites (see also *Cumulative Predation and Survival* section below). Management actions that raised water levels in the John Day Reservoir also successfully preventing tern nesting on the Blalock Islands in 2021. Some proportion of terns that formerly nested at the Blalock Islands presumably renested at Badger Island in McNary Reservoir in 2021. Predation rates by terns nesting at Badger Island were below the 2% threshold for all ESU/DPS evaluated in 2021. The late arrival of terns on Badger Island in 2021, however, likely contributed to the lower than anticipated levels of predation (based on the size of the Badger Island colony, 231 breeding pairs), thus continued adaptive management at the Blalock Islands and Badger Island tern nesting sites will likely be needed in 2022 and beyond to achieve the full benefits of the tern management actions associated with the *IAPMP* in the CPR.

East Sand Island Caspian Tern & Double-crested Cormorant Management Plans: Population specific (ESU/DPS) estimates of predation rates on juvenile salmonids by Caspian terns nesting on East Sand Island in 2021 were not available at the time this report was written, although smolt PIT tags were recovered on East Sand Island by the USACE following the 2021 breeding season (see *Table 3* above). In 2020, predation estimates were the lowest recorded since terns resumed nesting on East Sand Island in 1999, with estimates ranging from 0.4% (0.2–0.9%) on UCR spring Chinook salmon to 5.9% (4.5–8.1%) on SR steelhead (Evans et al. 2021b). Results from 2020 indicated that predation by terns nesting on East Sand Island have been reduced by 65% to 76% on steelhead DPSs (Evans et al. 2021b), reductions that meet or exceed those targeted by the *Caspian Tern Management Plan for the Columbia River Estuary.* Large numbers of terns (several hundred to several thousand), however, continued to nest outside of the designated 1-acre nesting area on East Sand Island in 2021 (K. Tidwell, USACE pers. comm.), as well as nesting attempts by terns on Rice Island in the upper CRE. Again, predation rates by terns that nested on East Sand Island in 2021 are currently unknown, but those rates could be greater than those of 2020 based on the number of smolt PIT tags recovered in 2021 (n = 5,081) compared with 2020 (n = 4,595; Evans et al. 2021b; also see *Table 3* above). Thus, continued implementation of nest dissuasion activities and continued monitoring of tern nesting sites throughout the estuary will be necessary to ensure that predation rates do not exceed the levels stipulated in the *Caspian Tern Management Plan for the Columbia River Estuary*.

For the second consecutive year there was no evidence of an established nesting colony of double-crested cormorants on East Sand Island in 2021. Similar to the tern colony on East Sand Island, monitoring to estimate predation rates on salmonid smolts by cormorants nesting on East Sand Island has traditionally been funded by USACE and are currently unavailable, but predation rates were presumably low in 2021 based on the small number of smolt PIT tags (n = 117) recovered on-colony following the breeding season (see *Table 3* above).

Historical colony-specific and ESU/DPS-specific predation rates by terns and cormorants nesting on East Sand Island dating back to 2000 for terns and back to 2003 for cormorants are provided in the *Appendix B, Tables B3* and *B4*, respectively (see also Roby et al. 2021).

Astoria-Megler Bridge Cormorant Predation Assessment: See *Appendix A.*

Predation Rates by Other Piscivorous Colonial Waterbirds

Predation/consumption rates of juvenile salmonids by gulls, cormorants, and pelicans nesting at several colonies in the CPR were equal to or often greater than those of terns nesting at colonies in the CPR, with rates being more than 2% and as high as 13% for several ESA-listed ESUs/DPSs of salmonids that were evaluated in 2021 (see *below*). Estimated consumption rates of smolts were particularly high for some gull colonies in 2021. Gulls, however, are known to consume dead or moribund fish and to kleptoparasitize fish from other piscivorous waterbirds, such as terns. Consequently, smolt PIT tag recoveries on gull colonies may be more indicative of consumption rates, rather than predation rates (Cramer et al. 2021a). Unlike gulls, terns, cormorants, and pelicans are strictly piscivorous and are known to rarely consume dead fish. As

such, we refer to gull estimates of fish loss as "consumption rates" and losses to terns, cormorants, and pelicans as "predation rates". Below is a summary of colony- and ESU/DPS-specific predation/consumption rates by gulls, cormorants, and pelicans nesting at unmanaged colonies in the CPR in 2021, along with comparisons to estimates of predation/consumption rates from previous years.

It should also be noted that small sample sizes of PIT-tagged smolts (less than 2,000 available tagged fish) resulted in imprecise estimates of predation/consumption, as indicated by the width of 95% credible intervals, for most but not all of the ESUs/DPSs evaluated in 2021 (see *Appendix B, Table B1, B5,* and *B7*). Large confidence intervals limit what can be inferred about inter-annual differences in estimates of predation/consumption rates for birds nesting at colonies upstream of McNary Dam, including gulls nesting at Island 20, cormorants nesting at Foundation Island, gulls, cormorants, terns, and pelicans nesting at Badger Island, and gulls nesting at Crescent Island (see *below*). Of those salmonid ESUs/DPSs last detected alive passing Rock Island Dam and Lower Monumental Dam, UCR steelhead was the only ESA-listed DPSs with a relatively large sample size of available tagged fish in 2021 (n = 8,090; see also *Appendix B* for ESU/DPS-specific sample sizes of available tagged smolts in previous years).

Gulls - Island 20: Consumption rates of salmonid smolts by gulls nesting on Island 20 varied by salmonid ESU/DPS, with rates ranging from 0.1% (<0.1–1.1%) for SR Fall Chinook salmon to 6.1% (4.1–9.3%) for UCR steelhead (*Table 8*). As observed in previous years, estimates of consumption were significantly higher for steelhead DPSs compared with salmon ESUs. Previous studies suggest that higher gull consumption rates of steelhead smolts compared with salmon smolts are associated with the larger average size of steelhead smolts and the surface orientation of steelhead smolts, factors that increase the susceptibility of steelhead to surface feeding predators like gulls and terns (Evans et al. 2016, Evans et al. 2019, Cramer et al. 2021a, Hostetter et al. 2021). Estimates of smolt consumption by gulls nesting on Island 20 in 2021 were similar to or higher than those observed in previous years, with the highest estimates observed for SR steelhead at 4.8% (2.1–9.5%) and UCR steelhead at 6.1% (4.1–9.3%; *Table 8*). Estimates of predation rates on steelhead were also elevated in 2015, 2016, and 2019 (see *Appendix B, Table B5*). In most other years, however, consumption rates of steelhead DPSs by gulls nesting on Island 20 were less than 3.0% of available smolts and were less than 1.0% of available salmon ESUs (see *Appendix B, Table B5*). Increased rates of consumption by Island 20 gulls in some years, like 2015 and 2021, may be associated with increases in colony size and/or changes in environmental factors (e.g., river flow, water transit time, turbidity) that increase smolt exposure times to gull consumption during out-migration (Hostetter et al. 2012, Roby et al. 2016, Payton et al. 2016, Hostetter et al. 2021).

Table 8. Estimated predation/consumption rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESU/DPS), with runs of spring (Sp), summer (Su), and fall fish, by California and ring-billed gulls nesting on Island 20, Badger Island, and Crescent Island in McNary Reservoir, Blalock Islands in John Day Reservoir, and Miller Rocks in The Dalles Reservoir during 2021. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods). See Appendix B for estimates of predation/consumption by these colonies in previous years.

ESU/DPS	Island 20	Badger Is.	Crescent Is.	Blalock Is.	Miller Rocks Is.
SR Sockeye	NA	NA	NA	NA	NA
SR Sp/Su Chinook	1.4% (0.2–6.2)	0.6% (0.2–3.8)	1.9% (0.2–8.6)	0.5% (0.2–1.0)	1.2% (0.7–2.2)
UCR Sp Chinook	0.5% (0.1–2.1)	0.2% (<0.1–1.3)	0.7% (0.1–3.0)	0.9% (0.2–2.7)	0.9% (0.2–2.8)
SR Fall Chinook	0.1% (<0.1–1.1)	1.4% (0.1–6.8)	0.2% (<0.1–2.2)	0.2% (<0.1–0.3)	2.8% (1.1–6.3)
SR Steelhead	4.8% (2.1–9.5)	6.6% (1.7–17.9)	2.8% (0.7–8.6)	5.7% (3.3–9.6)	13.9% (9.1–21.8)
UCR Steelhead	6.1% (4.1–9.3)	8.6% (3.6–14.7)	2.9% (1.7–5.0)	7.9% (3.6–15.8)	12.4% (6.2–23.1)

Gulls - Badger Island: Consumption/predation rates on salmonid smolts for gulls nesting at Badger Island ranged from 0.2% (<0.1–1.3%) for UCR spring Chinook salmon to 8.6% (3.6–14.7%) for UCR steelhead. Consumption rates by gulls nesting at Badger Island were also high for SR steelhead in 2021 with an estimated 4.9% (1.7–9.3%) of available fish consumed (see *Table 8* above). As noted above, however, smaller sample sizes of PIT-tagged smolts in 2021 resulted in imprecise estimates of consumption rates for several ESUs/DPSs (see *Table 8;* above; see *Appendix B, Table B6* for sample sizes of available tagged smolts in years past). Similar to consumption estimates for gulls from the nearby colony on Island 20 in the middle Columbia River (see *above*), estimates of consumption rates by gulls nesting at Badger Island in 2021 were similar to or higher than those observed in previous years (*Appendix B, Table B5*). Estimates of consumption rates were also elevated in 2015, 2019, and 2020. Unlike the gull colony at Island 20, terns were co-nesting with gulls on Badger Island in 2021, so an unknown number of smolt PIT tags recovered on the Badger Island gull colony were from smolts initially captured by terns and subsequently stolen by gulls. There was no tern colony on Badger Island during five of the last six years (2015–2020), however, so presumably no smolts included in estimates of gull consumption rates were depredated by terns in these years.

Gulls - Blalock Islands: Since at least 2012, there have been gull colonies on two separate islands (i.e. Anvil Island and Straight Six Island) within the Blalock Islands complex in John Day Reservoir. Of the two gull colonies in the Blalock Islands complex, smolt consumption rates have been significantly higher for gulls nesting on Anvil Island compared to gulls nesting on Straight Six Island (see also Roby et al. 2016). Differences between colonies in smolt consumption rates can be attributed to a difference in the size of the two colonies (with substantially more gulls nesting on Anvil Island) and a difference in gull species composition at the two colonies (Anvil Island was dominated by nesting California gulls and Straight Six Island was dominated by nesting ring-billed gulls; the former are known to consume a higher proportion of juvenile salmonids; Collis et al. (2002). Also, data from Hostetter et al. (2015) and Cramer et al. (2021a) indicated that per capita (per bird) consumption of juvenile salmonids was consistently greater for gull colonies dominated by California gulls compared to those

dominated by ring-billed gulls. This difference in smolt consumption rates between the gull species is likely due to differences in body size and energy requirements (Winkler 1996), as well as the proportion of the diet that consists of fish (Collis et al. 2002), both of which are greater for California gulls compared with ring-billed gulls (see also Cramer et al. 2021a).

For the purpose of this and other studies of smolt predation/consumption rates (Evans et al. 2019, Cramer et al. 2021a), we have combined estimates of gulls nesting on Anvil and Straight Six islands (see also Evans et al. 2019). Results from both gull colonies (Anvil and Straight Six) combined indicate that consumption rates were less than 1% for all ESA-listed salmon ESUs but were significantly higher for steelhead DPSs with estimates of 5.7% (3.3–9.6%) and 7.9% (3.6–15.8%) for SR steelhead and UCR steelhead, respectively (see *Table 8* above). Estimates of consumption rates for gulls nesting at the Blalock Islands in 2021 were similar to or higher than those observed in previous years (see *Appendix B, Table B6*). Analogous to the average smolt consumption rates by gulls nesting at Island 20 and Badger Island in 2021 (see *above*), increases in smolt consumption rates by gulls nesting at the Blalock Islands in 2021 may be associated with environmental factors such as river flows, water transit time, turbidity, and other factors that influence smolt exposure times to predators during out-migration (Hostetter et al. 2012, Roby et al. 2016, Payton et al. 2016, Hostetter et al. 2021).

Gulls - Miller Rocks: Estimates of smolt consumption rates by gulls nesting at Miller Rocks in The Dalles Reservoir were the highest of any colony evaluated in 2021 in the CPR. For example, consumption rates by gulls nesting at other colonies were consistently < 1% for salmon ESUs and < 9% for steelhead DPSs, but consumption rates by gulls at Miller Rocks were as high as 2.8% (1.1–6.1%) for SR Fall Chinook salmon and 13.9% (9.1–21.8%) for SR steelhead (see *Table 8* above). Estimates of smolt consumption rates by gulls nesting at Miller Rocks in 2021 were similar to or higher than those observed at that colony in previous years, particularly for consumption of steelhead DPSs, with an estimated 12.4% (6.2–23.1%) of UCR steelhead and 13.9% (9.1–21.8%) of SR steelhead consumed, among the highest estimated consumption rates for a particular piscivorous waterbird colony since studies began in 2007 (see *Appendix B, Table B7*). Although the sample size of PIT-tagged SR sockeye salmon was too small in 2021 to generate reliable estimates of consumption rate, results from previous years indicate that, among available salmon ESUs, SR sockeye salmon smolts are particularly susceptible to consumption by gulls nesting at Miller Rocks, with estimates often in excess of 5% and upwards of 9.1% (5.3–15.1%) observed in some years (see *Appendix B, Table B7*).

Miller Rocks is in close proximity to John Day and The Dalles dams (18 Rkm and 23 Rkm, respectively). Evans et al. (2016) observed that gulls disproportionately consumed smolts near dams and hypothesized that smolts may be more vulnerable near dams as a result of (1) increased smolt travel times or delayed migration in the forebay of dams, (2) smolt morbidity or mortality associated with dam passage, or (3) smolts being temporarily stunned or disoriented by hydraulic conditions in the tailrace of dams. Gull consumption of smolts, however, is not limited to foraging near dams, with gulls consuming substantial numbers of smolts in apparent good-condition in open reservoirs and free-flowing sections of the river as well (see Evans et al. 2016 for a detailed discussion). There were no terns nesting on Miller Rocks, so salmonid smolts

consumed by gulls were presumably not depredated by terns. Nevertheless, the proportion of smolts consumed by gulls that were dead, moribund, or otherwise compromised when consumed, rather than depredated in an alive and healthy state, is currently unknown. Addressing this question is paramount to understanding what proportion of salmonid smolts consumed by gulls were depredated, and to what degree the mortality of smolts that were depredated by gulls was additive mortality, and therefore limits smolt survival in the CRB (see also *below* and Cramer et al. 2021a)*.* Finally, coincident with higher estimates of smolt consumption rates, the numbers of gulls nesting on Miller Rocks in 2021 were also higher than those observed in most previous years; an estimated 5,491 adult gulls were counted during the peak nesting period in May of 2021 (see *Table 2* above). Estimates of colony size during 2007–2020 averaged 4,446 adult gulls (range = 3,223–6,016 individuals; Cramer et al. 2021a).

Cormorants - Hanford Island: The double-crested cormorant colony on Hanford Island or Locke Island in Hanford Reach on the Columbia River has periodically been scanned for smolt PIT tags since nesting cormorants were first confirmed in 2007 (Cramer et al. 2021a). Similar to most previous years, estimates of predations in 2021 were low, with less than 0.4% of available fish depredated per ESU/DPS; the highest estimated predation rate was 0.3% (0.1–0.7%) on UCR steelhead. The small size of this cormorant colony (less than 20 pairs) is presumably responsible for the low predation rates on juvenile salmonids, especially given the colony's location on the middle Columbia River. It should be noted that the island is regularly submerged during the early part of the nesting season, flooding which may have washed away or buried some unknown proportion of smolt PIT tags that were deposited on the island. No estimate of pre-season detection efficiency, however, was available at this colony site in 2021 (see *Table 4* above) and thus estimates should be considered minimum estimates of predation rates.

Cormorants - Foundation Island: Estimates of predation rates for double-crested cormorants nesting at the colony on Foundation Island, the largest cormorant colony located on the Columbia River upstream of Bonneville Dam, ranged from < 0.1% on UCR spring Chinook salmon to 4.4% (0.8–15.6%) on SR spring/summer Chinook salmon in 2021 (*Table 9*). Predation rates on SR steelhead were also substantial at 3.0% (0.8–9.3%) of available tagged smolts. Small sample sizes of tagged smolts last detected passing Lower Monumental Dam, however, resulted in imprecise estimates of predation rates by cormorants nesting at Foundation Island and for other birds nesting at colonies upstream of McNary Dam in 2021. Similar to previous years, predation rates on UCR steelhead and UCR spring Chinook salmon smolts were low at less than 0.1% per ESU/DPS (*Table 9*). Higher predation rates on SR smolts compared with UCR smolts have been attributed to the cormorants nesting at Foundation Island disproportionately foraging in the lower Snake River compared with the middle Columbia River (Evans et al. 2016). Higher river turbidity and the greater abundance of salmonids in the lower Snake River compared with the middle Columbia River are factors that may explain the increased susceptibility of SR salmonids to predation by cormorants nesting at Foundation Island (Hostetter et al. 2012). Predation rates for cormorants nesting at the Foundation Island colony in 2021 were similar to those of previous years (see *Appendix b, Table B7*), as were estimates of colony size (307 breeding pairs in 2021; see *Table 2* above).

Table 9. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESUs/DPSs), with runs of spring (Sp), summer (Su), and fall fish, by double-crested cormorants nesting at Hanford Island in the middle Columbia River and Foundation Island in McNary Reservoir during 2021. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods). See Appendix B for estimates of predation by these colonies in years past.

ESU/DPS	Hanford Island	Foundation Island
SR Sockeye	NA	NA
SR Sp/Su Chinook	0.1% (<0.1–1.6)	4.4% (0.8–15.6)
UCR Sp Chinook	0.1% (<0.1–0.5)	< 0.1%
SR Fall Chinook	<0.1%	0.1% (<0.1–0.6)
SR Steelhead	0.1% (<0.1–0.8)	3.0% (0.8–9.3)
UCR Steelhead	0.3% (0.1–0.7)	0.1% (<0.1–0.3)

Predation rates for cormorants nesting at the Foundation Island colony on SR smolts have been similar to or greater than those of nearby tern colonies (e.g., Crescent Island and the Blalock Islands). For example, predation rates for Foundation Island cormorants on SR steelhead and SR sockeye salmon have average 4.0% (3.4–4.7%) and 3.4% (2.4–4.5%), respectively, compared with 4.5% (4.1–5.1%) and 1.5% (1.2–2.0%), respectively, by terns nesting at Crescent Island during the same time period (2007–2014; Cramer et al. 2021a; see also *Appendix B, Tables B1 and B7*). Despite similar levels of predation, however, the Foundation Island cormorant colony was not included in management plans associated with the *IAPMP* because at the time the management plan was written only minimum estimates of predation rates were available due to a lack of information on PIT tag deposition probabilities for cormorants (see also Cramer et al. 2021a).

Pelicans - Badger Island: Estimates of on-colony PIT tag deposition probabilities for American white pelicans breeding on Badger Island were available for the first time in 2020 (Payton et al. 2021b), providing the necessary input data to generate accurate estimates of predation rates for the first time since 2007, the first year the pelican colony was scanned for smolt PIT tags. The Badger Island pelican colony was again scanned for PIT tags in 2021 and, similar to results from 2020, estimates of predation/consumption rates were consistently lower than those for terns, gulls, and cormorants nesting at nearby colonies, ranging from 0.2% (<0.1–1.8%) for SR spring/summer Chinook salmon to 1.0% (0.1–7.7%) for SR steelhead (*Table 10*). Analogous to other colonies located upstream of McNary Dam, small sample sizes of PIT-tagged smolts in 2021, especially SR smolts, resulted in imprecise estimates of predation rates as indicated by the width of 95% credible intervals. Despite this, results from both 2020 (where sample sizes of tagged smolts were greater) and 2021, suggest that pelicans breeding on Badger Island pose little threat to ESA-listed UCR and SR salmonid smolts. Pelicans generally forage in shallow

water less than three meters deep by dipping their bill into the water and scooping prey items (Knopf and Evans 2004). Pelicans have also been observed congregating and foraging near diversion structures such as Horn Rapids Dam on the Yakima River and in the tailrace of hydroelectric dams such as John Day Dam and The Dalles Dam, particularly during June and July when large numbers of American shad (*Alosa sapidissima*) and sub-yearling Fall Chinook salmon are migrating (Stinson 2016; Payton et al. 2021b).

Table 10. Estimated predation rates (95% credible interval) on Snake River (SR), Upper Columbia River (UCR), and Middle Columbia River (MCR) salmonid populations (ESU/DPS), with runs of spring (Sp), summer (Su), and fall fish, by American white pelicans nesting on Badger Island in McNary Reservoir and Miller Sands Spit Island in the Columbia River Estuary during 2021. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods). A dash (-) denotes that no estimates of predation for that ESU/DPS were available due to the colony's location relative to the out-migration corridor of all smolts.

ESU/DPS	Badger Island	Miller Sands Spit
SR Sockeye	NA	< 0.1%
SR Sp/Su Chinook	0.2% (<0.1–1.4)	< 0.1%
UCR Sp Chinook	< 0.1%	< 0.1%
SR Fall Chinook	0.9% (<0.1–2.9)	< 0.1%
UWR Sp Chinook	-	NA
SR Steelhead	1.0% (0.1–7.7)	< 0.1%
UCR Steelhead	0.4% (0.1–2.4)	< 0.1%
MCR Steelhead	-	< 0.1%

In a study of non-ESA-listed sub-yearling Chinook salmon from the Upriver Bright stock in the Hanford Reach of the middle Columbia River, Payton et al. (2021b) estimated that predation rates by pelicans nesting on Badger Island were substantial in some, but not all years, with estimates greater than 10% of available smolts during 2015–2019. Estimates of pelican predation rates were significantly higher on wild subyearling Chinook salmon, fish that rear and reside in the middle Columbia River prior to out-migration, than their hatchery counterparts. Given evidence that particular salmon stocks may be more susceptible to pelican predation than others (e.g., sub-yearling versus yearling), additional research regarding predation by pelicans nesting at Badger Island is warranted (Payton et al. 2021b). Furthermore, unlike gulls and terns, pelicans are also capable of consuming adult-sized salmonids, with PIT tags implanted in adult sockeye salmon, adult steelhead (both pre- and post-spawned), and jack Chinook salmon recovered on the Badger Island pelican colony. Adult salmonids ranging in size from 325 mm fork length (jack sockeye salmon) to 770 mm fork length (steelhead) have been consumed by Badger Island pelicans (Roby et al. 2017). In 2021, 47 tags from adult sockeye salmon that were tagged at the Bonneville Dam adult fishway were recovered on the Badger Island pelican colony (see *Table 3* above). Given that relatively small numbers of adult sockeye salmon are annually tagged at Bonneville Dam (e.g., 1,534 adults in 2021; PSFMC 2021),

predation rates by pelicans on adult sockeye salmon may be greater than that on juvenile sockeye salmon. Kock et al. (2021) observed that pelicans consumed upwards of 40% of adult sockeye salmon that were radio-tagged on the Yakima River, a location were larger number of pelicans are known to congregate and forage in the summer when sockeye salmon are migrating. Thus, future research to estimate predation rates by pelicans on adult sockeye salmon is warranted given the relatively larger number of adult sockeye salmon tags recovered on the Badger Island pelican colony in 2020 and 2021 and the increasing size of the Badger Island pelican colony, with the count from 2021 (n = 3,624 individuals) being the highest observed since counts were first conducted in 2005 (annual range = 1,057–3,330 individuals; Cramer et al. 2021a). In addition to adult sockeye salmon, tags from other fish species have also been recovered on Badger Island, including white sturgeon, bull trout, and pikeminnow (see *Table 3* above; see also Cramer et al. 2021a). The diverse diet and presence of adult-sized fishes highlight the differences in diet composition between pelicans and other piscivorous colonial waterbirds nesting in the CRB.

Pelicans - Miller Sands Spit: Estimates of predation rates by American white pelicans breeding on Miller Sands Spit in the CRE were the lowest of all colonies evaluated in 2021. Despite an estimated colony size of 845 individuals, only 53 tags from juvenile salmonids were recovered on the colony following the breeding season (see *Table 3* above), resulting in estimates of predation rates of <0.1% per salmonid ESU/DPS (see *Table 10* above). Similar to the pelicans breeding on Badger Island in 2021 (see *above*), results suggest that pelicans breeding on Miller Sands Spit pose little threat to the survival of ESA-listed juvenile salmonids from UCR and SR ESUs/DPSs, particularly predation on actively migrating, yearling smolts. Of the 53 smolt PIT tags recovered on the Miller Sands Spit pelican colony, 34 or 64% were from sub-yearling Chinook salmon, fish that were potentially still rearing and/or that congregated in shallow water habitats in the lower estuary (Sebring et al. 2013, Lawes et al. 2021b).

Cumulative Predation & Smolt Survival

An investigation of cumulative (all predator species and colonies combined) effects of avian predation/consumptions on the survival of ESA-listed UCR steelhead indicated that predation/consumption rates in 2021 were among the highest recorded since system-wide estimates were first documented in 2008. Of the avian predator species capable of foraging between RIS and McNary Dam, the highest predation/consumption rates on UCR steelhead were those by gulls at 17.4% (95% CRI = 11.8–23.8%), followed by terns at 6.2% (4.3–8.9%), pelicans at Badger Island (the lone pelican colony in this river reach) at 0.4% (0.1–1.4%), and cormorants at Foundation Island (the lone cormorant colony in this river reach) at 0.1% (<0.1–0.3%). As noted *above*, previous research indicates that cormorants nesting on Foundation Island in McNary Reservoir disproportionally forage on smolts in the lower Snake River (Evans et al. 2016), which presumably explains why predation rates on UCR steelhead were significantly lower than those on other salmonid ESUs/DPSs. Cumulative predation/consumption probabilities for UCR steelhead smolts during passage from RIS to McNary Dam were estimated at 24.2% (18.1–31.1) of available smolts, the third highest estimate for this river reach since system-wide studies of avian predation were initiated in 2008 (*Figure 6*).
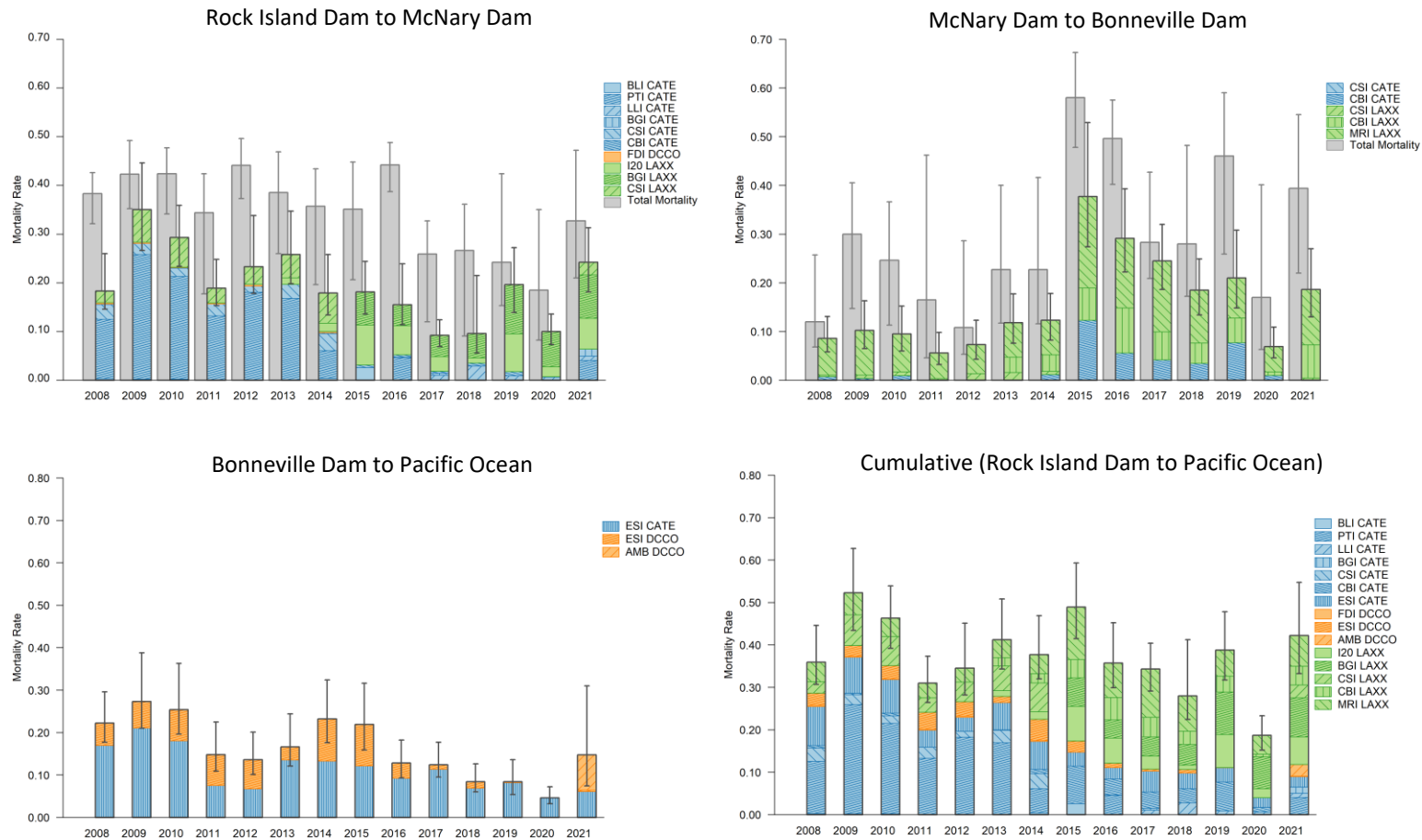
Figure 6. Estimated total mortality (grey bars) and mortality associated with colonial waterbird predation/consumption (colored bars) for steelhead smolts during passage from Rock Island Dam to McNary Dam, McNary Dam to Bonneville Dam, Bonneville Dam to the Pacific Ocean, and cumulative predation from Rock Island Dam to Pacific Ocean. Colony locations include Banks Lake islands (BLI), Potholes Reservoir islands (PTI), Lenore Lake islands (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks (MRI), East Sand Island (ESI), and the Astoria-Megler Bridge (AMB). Avian predator species include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation probabilities. Large numbers of cormorants nested on the AMB bridge during 2016–2020 but no estimates of predation probabilities were available in those years.

Of those gull colonies in the CPR capable of foraging on steelhead smolts between McNary Dam and Bonneville Dam, consumption by gulls was again the highest of those avian predator species evaluated, with an estimated 17.2% (11.5–24.5%) of available UCR steelhead smolts consumed (see *Figure 6* above). There was no evidence from PIT tag recoveries that terns, cormorants, and pelicans foraged on UCR steelhead in this river reach during 2021, although terns and pelicans nesting on Crescent and Badger islands are known to have consumed other salmonid ESUs/DPSs in this river reach in 2021 and also UCR steelhead in previous years (Evans et al. 2016, Collis et al. 2021a, b). Also, the elimination of the tern colony on the Blalock Islands, terns that predominately foraged between McNary Dam and John Day Dam, contributed to a notable lack of avian predation by terns in this river reach during 2021.

Quantifying the effects of predation by birds foraging on multiple juvenile salmonid species and ESUs/DPSs between Bonneville Dam and the Pacific Ocean was limited by a lack of funding in 2021 (see *above*), whereby analyses typically funded by the USACE for terns and cormorants nesting on East Sand Island were not conducted in 2021. Estimates of the cumulative, system-wide effects of avian predation on UCR steelhead, however, were analyzed as part of this study and include predation probabilities for four piscivorous waterbird colonies in the CRE: (1) terns on East Sand Island, (2) cormorants on East Sand Island, (3) cormorants on the Astoria-Megler Bridge (see *Appendix A*), and (4) pelicans on Miller Sands Spit. Results indicated that cumulative effects of predation by birds foraging on UCR steelhead between Bonneville Dam and Pacific Ocean were estimated at 13.5% (6.6–25.1%) of available fish (*Figure 6* above). Predation rates were highest for cormorants nesting on East Sand Island and on the Astoria-Megler Bridge at 7.5% (1.0–18.1%), followed by terns nesting on East Sand Island at 5.8% (3.8–9.1%), and pelicans nesting on Miller Sands Spit at < 0.1% (*Figure 6* above).

The cumulative effects of avian predation/consumption (predation/consumption by all avian predator species and colonies combined) on UCR steelhead during smolt passage from RIS to the Pacific Ocean in 2021 were substantial at an estimated 42.4% (35.4–50.6%; see *Figure 6* above). Of the avian predator species evaluated, estimated consumption of UCR steelhead by gulls was the greatest at 29.1% (22.8–36.3%), followed by tern predation at 8.6% (6.7–11.8%), cormorant predation at 2.9% (0.6–7.2%), and pelican predation at 0.4% (0.1–2.4%). It is important to note that the colony-specific and cumulative effects of avian predation/consumption differ significantly by ESA-listed salmonid ESU/DPS, particularly predation/consumption of salmon ESUs, where predation/consumption rates by colonial waterbirds are often, but not always, significantly lower than those of steelhead DPSs (Evans et al. 2021a). An analysis of colony-specific avian predation/consumption rates and the cumulative effects of avian predation/consumption on the survival of other ESA-listed salmonid ESUs/DPSs, including UCR spring Chinook salmon, SR steelhead, SR spring and fall Chinook salmon, and SR sockeye salmon during 2008–2018 is presented in Evans et al. (2021a).

Estimates of UCR steelhead smolt survival from RIS to McNary Dam in 2021 were similar to those from previous years, with an estimated annual survival probability in 2021 of 61.6% (48.8–76.8%). In 2020, commensurate with record low estimates of avian predation/consumption rates, UCR steelhead survival between RIS and McNary Dam was the

highest observed since smolt tagging efforts at RIS commenced in 2008, with an estimated annual survival probability of 82.6% (66.6–92.3%; see *Figure 6* above). Smolt survival from McNary Dam to Bonneville Dam in 2021 was below average at 59.9% (45.7–77.7%; see *Figure 6* above). Cumulative survival from RIS to Bonneville Dam for UCR steelhead smolts in 2021 was also below average, with an estimated 40% (32.0–48.7%) of UCR steelhead smolts surviving out-migration through the hydrosystem, based on availability in the tailrace of RIS. Annual estimates of UCR steelhead smolt survival from RIS to Bonneville Dam ranged from a minimum of 27.2% (23.3–32.3%) to a maximum of 60.0% (44.4–74.0%) during 2008–2020, with the highest estimate observed in 2020, commensurate with the lowest levels of avian predation/consumption (Payton et al. 2020, Collis et al. 2021a, see also below).

Comparisons of total UCR steelhead smolt mortality (1-survival) to predation/consumption by piscivorous colonial waterbirds indicated that predation/consumption in 2021 accounted for 78.2% (50.2–98.1%) and 46.1% (25.2–92.9%) of all sources of mortality during smolt passage from RIS to McNary Dam and from McNary Dam to Bonneville Dam, respectively. Estimates of total smolt mortality and avian predation/consumption rates from RIS to Bonneville Dam indicated that predation/consumption by colonial waterbirds in 2021 was associated with 61.6% (48.8–76.%) of all smolt mortality. Due to the lack of PIT tag detection sites downstream of the bird colonies on East Sand Island in the estuary, estimates of total smolt mortality from Bonneville Dam to the Pacific Ocean are not available (see *Figure 6* above). Similar to results from 2021, comparisons of total UCR steelhead smolt mortality to predation/consumption rates by colonial waterbirds in previous years (2008–2020) indicated that avian predation/consumption was the one of the greatest sources, and in many years the single greatest source, of mortality for UCR steelhead smolts during out-migration to Bonneville Dam, with avian predation/consumption accounting for an estimated 42% (95% CRI = 30–56%) to 70% (95% CRI = 53–87%) of all UCR steelhead smolt mortality on average during 2008–2020 (see *Figure 6* above; see also Evans al 2019, Collis et al. 2021a, b). Results from this and several other studies (e.g., Evans et al. 2016, Evans et al. 2019, Payton et al. 2019, Payton et al. 2020, Collis et al. 2021a, b) indicate that mortality of UCR steelhead smolts associated with predation/consumption by piscivorous colonial waterbirds was greater than from all other mortality sources combined in 12 of the last 14 years (2008–2021). For example, UCR steelhead smolt losses associated with predation/consumption by piscivorous colonial waterbirds upstream of Bonneville Dam were greater than the combined direct losses associated with passage through six hydroelectric dams (Wanapum, Priest Rapids, McNary, John Day, The Dalles, and Bonneville dams), predation from piscivorous fish, predation by piscivorous waterbirds from colonies that were not included in the study, mortality from disease, and all other remaining mortality factors. Our results suggest that avian predation/consumption, although not the original cause of salmonid declines in the CRB, in now a factor limiting the survival of some salmonid populations that are listed under the U.S. Endangered Species Act, particularly steelhead populations (see also Evans et al. 2021a and Payton et al. 2021a).

In 2020, record low (since 2008) estimates of cumulative avian predation/consumption rates on UCR steelhead were coincident with record high estimates of smolt survival to both McNary Dam and Bonneville Dam. The greatest reductions in predation/consumption rates in 2020

were associated with the elimination of tern colonies on Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir, the two tern colonies managed as part of the *IAPMP* (USACE 2014, Collis et al. 2021a). In 2021, increases in predation rates by terns and increases in consumption rates by gulls were both coincident with lower estimates of steelhead survival in this river reach compared with 2020. Terns nesting at the colonies on Goose and Crescent islands foraged on smolts in the Columbia River between RIS and McNary Dam, and an increase in predation rates by terns nesting on Goose Island in 2021 was coincide with lower UCR steelhead smolt survival relative to 2020. On average over the course of the entire management period (2014–2021), however, tern predation rates on UCR steelhead smolts have significantly declined following management actions at these two tern colonies (*Figure 7*). Our results suggest that management efforts to reduce tern predation rates have increased smolt survival rates within the area where management actions have occurred, but that if tern predation rates increase in this river reach, as they did in 2021, or in other river reaches, smolt survival will again decrease, perhaps to levels observed prior to management. For instance, during 2015–2020, the opposing trends in estimated avian predation rates and UCR steelhead smolt survival rates were observed in the river reach between McNary Dam and Bonneville Dam, where tern predation rates on steelhead smolts increased significantly and steelhead survival rates decreased significantly; these changes were attributable to the elimination of the tern colonies on Goose and Crescent islands and the subsequent dispersal of terns to the Blalock Islands between McNary Dam and Bonneville Dam (*Figure 7*). In 2020 and in 2021, however, predation rates on UCR steelhead by terns breeding on the Blalock Islands were the lowest recorded since management commenced in 2014 (*Figure 7*). It should be noted that the estimate of no (zero) tern predation between McNary and Bonneville dams in 2021 is likely inaccurate, however, because terns were observed on islands in the Blalock archipelago early in the 2021 breeding season (April), but since a colony did not form on a specific site, smolt PIT tags could not be collected. As such, tern predation rates on UCR steelhead and other salmonid ESUs/DPSs were presumably greater than zero in the river reach between McNary and Bonneville Dam but could not be quantified in 2021. Comparisons of tern predation rates on steelhead smolts and survival rates of steelhead smolts downstream of Bonneville Dam were not available due to the lack of smolt survival estimates from Bonneville Dam to the Pacific Ocean (see also above).
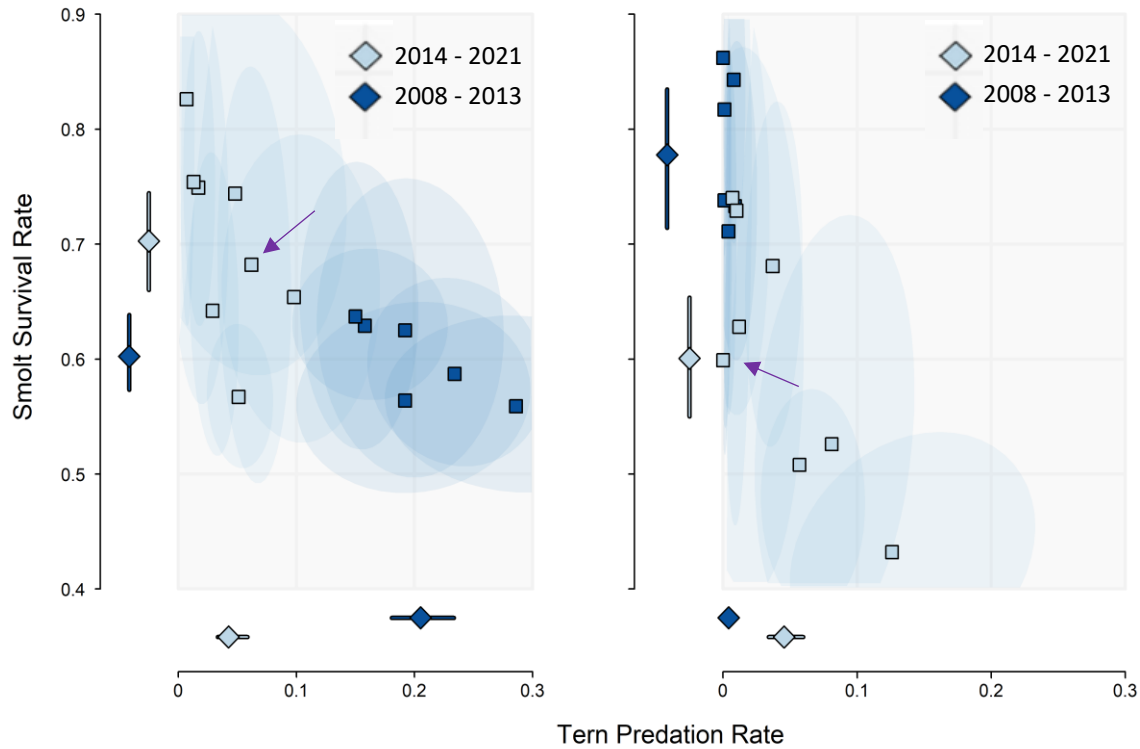
Figure 7. Caspian tern predation rates and survival rates of Upper Columbia River steelhead smolts during passage from Rock Island Dam to McNary Dam (left panel) and from McNary Dam to Bonneville Dam (right panel) prior to (dark blue) and following (light blue) management actions that reduced the size of tern colonies at Goose Island and Crescent Island; terns nesting at these two colonies forage between RIS and McNary Dam. Median annual rates and average annual rates for the entire study period (diamonds) are shown. Error bars represent 95% credible intervals for annual averages and shaded ellipses represent 95% credible regions for the joint estimation of survival rate and predation rate (see also Payton et al. 2020). Arrows indicates estimates from 2021. Results are those of Collis et al. (2021a) updated with estimates from 2021.

In summary, the cumulative effects of avian predation/consumption on survival of UCR steelhead smolts in 2021 were substantial and continued to be one of the greatest sources of smolt mortality during out-migration. Avian predation/consumption rates in 2021 were significantly higher than those observed in 2020, a year when predation/consumption rates were the lowest recorded since system-wide studies of avian predation/consumption began in 2008, particularly predation by terns. Increases in tern predation rates on UCR steelhead in 2021 were largely due to the reformation of a tern colony on Goose Island, along with predation by terns nesting at the new tern colony on the Badger Island. Consumption rates by gulls nesting at colonies in the CPR were the greatest of the four avian predator species evaluated. Because gulls are known to consume dead, moribund, and injured smolts and are known to kleptoparasitize smolts that have been depredated by other piscivorous waterbirds, like terns, the actual impact of gull consumption on smolt survival is unknown (see *Additive Effects of Predation* section below). Despite increases in avian predation/consumption and the subsequent decrease in survival rate of UCR steelhead smolts in 2021 compared with 2020,

reductions in cumulative avian predation/consumption rates, in particular tern predation rates, were coincident with increases in UCR steelhead smolt survival upstream of Bonneville Dam during the management period (2014–2021) as a whole.

Additive Effects of Predation

A long-term (2008–2021) investigation of the additive effects of avian predation on UCR steelhead smolts indicated that increases in tern predation on UCR steelhead smolts above Bonneville Dam (BON) were associated with statistically significant decreases in smolt survival to BON (*Figure 8* and *Table 11*). In all years of this 14-year study, results suggest that significantly more steelhead smolts would have survived out-migration to BON in the absence of tern predation, with an estimated difference between observed survival and baseline survival (i.e. survival in the absence of tern predation) of 0.174 (0.106–0.234) across all years (*Table 11*)*.* For example, average annual steelhead smolt survival from RIS to BON was 0.446 (or 44.6%; 0.423–0.469) and results suggest that survival would have been 0.620 (or 62.0%; 0.514–0.854) in the absence of tern predation. Results also suggest that tern predation was a super-additive source (**a** > 1.0) of steelhead smolt mortality upstream of BON (*Table 11*).

Our results indicated that cormorants nesting on Foundation Island and on Hanford Island, the only two cormorant colonies upstream of BON that were included in the study, depredated a negligible proportion of available UCR steelhead smolts, with predation probabilities averaging less than 0.01 (1%) annually in those years when the cormorant colony was scanned for smolt PIT tags (*Table 11*). As such, there was no evidence of a statistically significant relationship between cormorant predation probabilities on UCR steelhead smolts and UCR steelhead survival probabilities to BON (*Table 11*). Similar to terns, gulls consumed a substantial proportion of UCR steelhead smolts in most, but not all years, with an average annual consumption probability of 0.21 (0.19–0.23) of available smolts during 2013–2021 (*Table 11*). Despite such high levels of consumption by gulls, however, there was no evidence of a statistically significant relationship between gull consumption of UCR steelhead smolts and smolt survival to BON. The average annually point estimate of additive smolt mortality associated with gull consumption was 0.18, indicating partial additivity, but estimates of gull consumption ranged from super-additive in some years to over-compensatory (**a** < 0) in other years, with no consistent trend identified across all years. It should also be noted that weekly probabilities of smolt consumption by gulls were less variable and less precise (based on the width of the 95% credible intervals) compared with those by terns, factors that make it difficult to determine to what degree gull consumption limits smolt survival (see also Payton et al. 2021a).

Table 11. Average annual (2008–2021) predation/consumption probabilities for Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX) and survival probabilities for Upper Columbia River steelhead smolts during out-migration from Rock Island Dam to Bonneville Dam and smolt-to-adult survival probabilities from Bonneville Dam (as smolt) to Bonneville Dam (as adult). Estimates of the magnitude of the association between predation probabilities and survival probabilities (a, additivity) and the difference in survival probabilities from estimated baseline survival probabilities (Φ^Δ) are also provided. Values are reported as medians with 95% credible intervals. Statistically significant ($prob[a > 0] > 0.95$) relationships between predation and survival are in bold, red font. Estimates of additivity in other salmonid species (e.g., Chinook and sockeye salmon) and ESA-listed populations (e.g., those from the Snake River) are available in Payton et al. 2021a.

Predator	Reach (life-stage)	Year(s)	Survival	Predation	a	Φ^Δ
CATE	RIS to BON (smolt)	2008-2021	0.446 (0.423–0.469)	0.13 (0.12–0.15)	**1.47 (0.97–1.93)**	**0.174 (0.106–0.234)**
	BON to BON (SAR)	2008-2018	0.025 (0.023–0.026)	0.18 (0.17–0.20)	**0.12 (0.06–0.20)**	**0.015 (0.007–0.024)**
DCCO	RIS to BON (smolt)	2008-2012, 2014, 2020-2021	0.477 (0.444–0.507)	< 0.01	0.50 (-1.57–2.56)	0.065 (-0.088–0.157)
	BON to BON (SAR)	2008-2015 [2]	0.030 (0.028–0.032)	0.07 (0.06–0.08)	0.05 (-0.03–0.13)	0.004 (-0.002–0.010)
LAXX [1]	RIS to BON (smolt)	2013-2021	0.433 (0.402–0.463)	0.21 (0.19–0.23)	0.22 (-0.69–0.90)	0.182 (-0.008–0.331)

[1] No information on smolt consumption rates by gulls nesting at colonies downstream of Bonneville Dam were available as these colonies were not included in the study.

[2] No information on smolt predations rates by cormorants nesting in the Columbia River estuary were available during 2016–2020 due to partial or total abandonment of colony on East Sand Island and dispersal to the colony on the Astoria-Megler Bridge, where predation rates were not measured until 2021 (see *Appendix A*). Although estimates of predation rates on smolts by cormorants nesting on the AMB were estimated in 2021, complete adult returns will not be available until 2024.
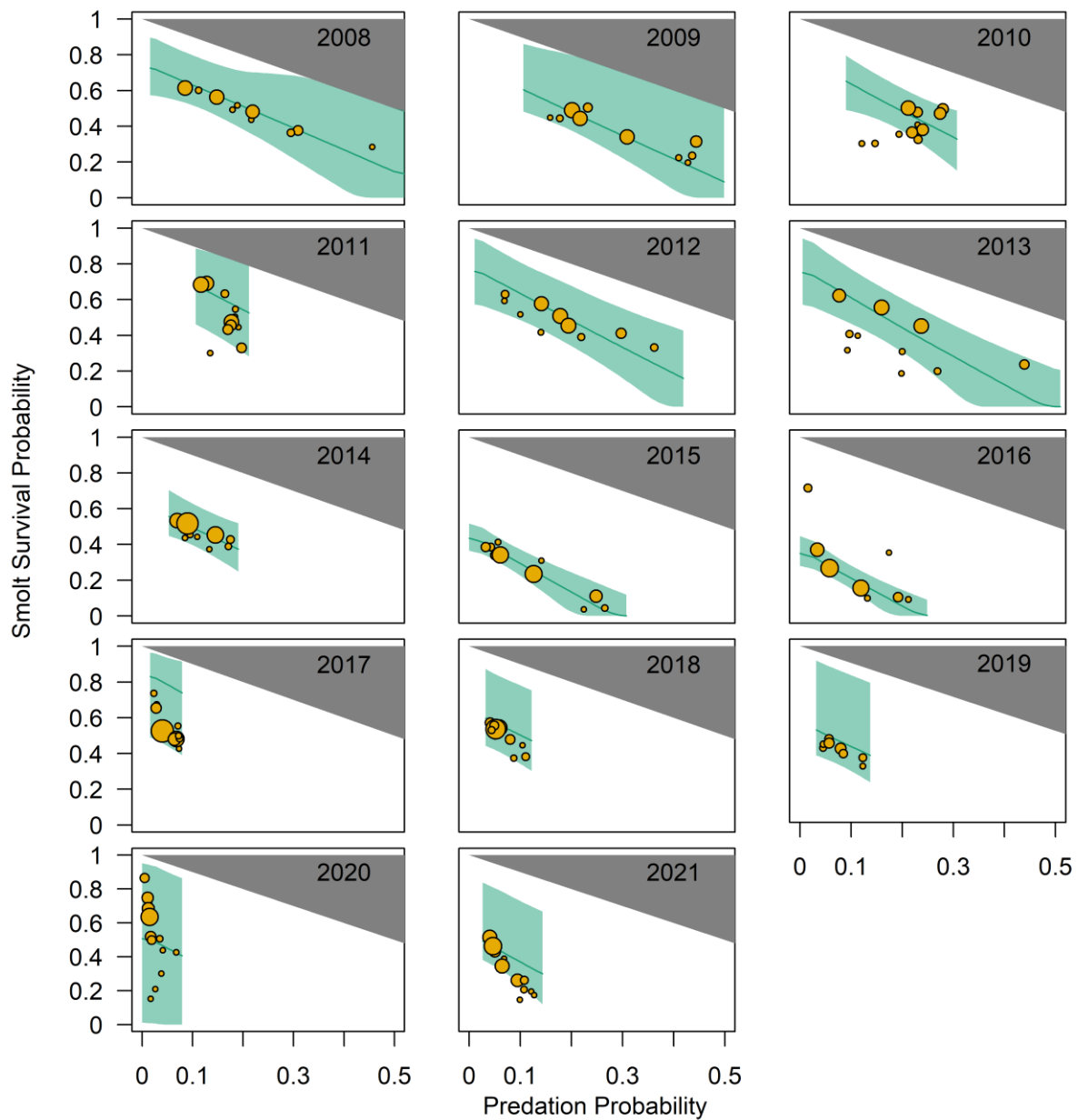
Figure 8. Weekly probability estimates of steelhead smolt survival and Caspian tern predation, and the estimated annual relationship between smolt survival and tern predation during out-migration from Rock Island Dam to Bonneville Dam during the 14-year study period. The size of circles depicts relative numbers of steelhead smolts tagged and released each week at Rock Island Dam. Dashed lines represent the best fit estimate of the linear relationship between survival and predation and shading denotes 95% credible intervals around the best fit. Results are those from Payton et al. (2021), updated with estimates from 2019 to 2021.

There was strong evidence that tern predation on UCR steelhead smolts was a partially additive source of mortality to the adult life-stage, with increases in tern predation probabilities on smolts associated with statistically significant decreases in steelhead adult returns to BON (*Figure 9* and see *Table 11* above). For example, our results suggest that in the absence of tern predation on smolts in the CRE, average annual smolt-to-adult return (SAR) probabilities could have been 0.040 (or 4.0%; 0.031–0.052), instead of 0.025 (or 2.5%; 0.023–0.026), in the presence of tern predation in the estuary (see *Table 11* above). Although our results indicate that the vast majority of UCR steelhead smolts at BON would have died before returning as adults to BON with or without tern predation, on average SARs to BON would have nearly doubled in the absence of tern predation. Estimated increases in smolt survival and smolt-to-adult survival presented herein are depicted in the complete absence of tern predation within a given river reach, but management plans are not attempting to eliminate all predation by terns, nor is the complete elimination of tern predation likely to occur in the CRB. If the complete elimination of tern predation in the CRB were to occur due to management, this effort would threaten the viability of this protected migratory bird species in the Pacific Flyway, especially if adequate, alternative nesting sites at other locations were unavailable (Lawes et al. 2021a). Modelled results presented herein assume a linear relationship between UCR steelhead survival and tern predation in each year and across all years, so managers can use these results to predict potential increases in smolt survival and smolt-to-adult survival with varying levels of tern predation. Although results presented herein apply specifically to UCR steelhead, the same statistically significant, additive relationship between tern predation on smolts and adult steelhead returns to BON was observed for Snake River steelhead (see Payton et al. 2021a), indicating tern predation limits the survival of multiple steelhead DPSs during both the smolt and smolt-to-adult life-stages. Collectively, results indicate that the best available science and most prudent conclusion from this report and a recently published peer-reviewed article (Payton et al. 2020) is that tern predation on steelhead smolts is an additive source of mortality during the smolt life-stage and a partially additive source of mortality to the smolt-to-adult life-stage (see also ISAB 2021).
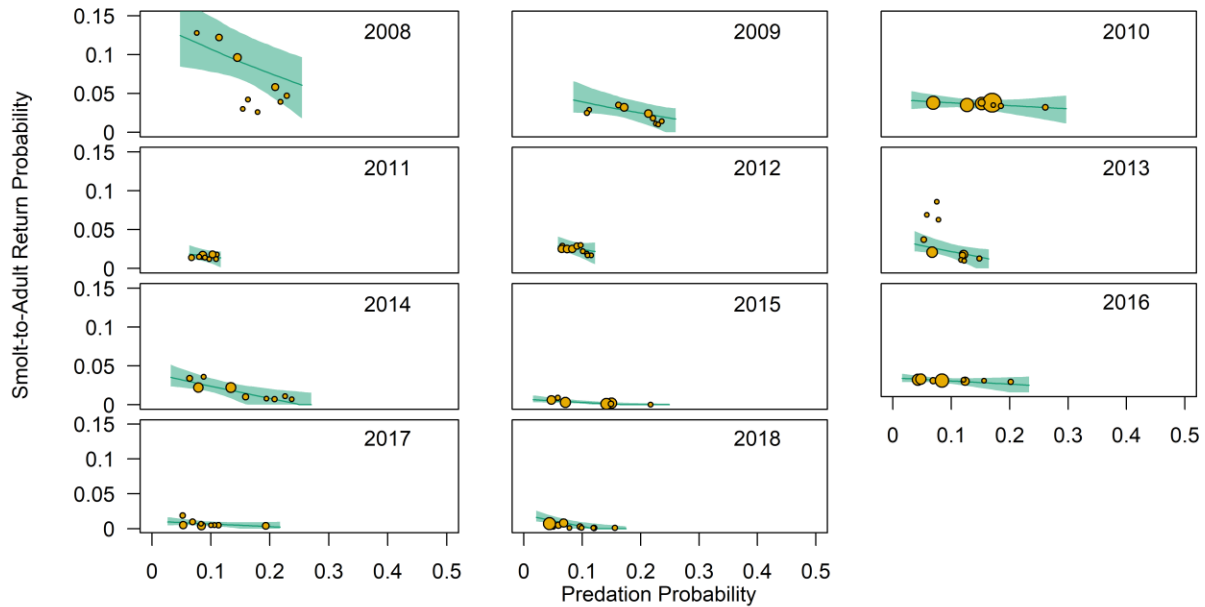
Figure 9. Weekly smolt-to-adult survival probabilities for Upper Columbia River steelhead as a function of Caspian tern predation probabilities during out-migration from Bonneville Dam as smolts to return to Bonneville Dam as adults in each year from 2008 to 2018. The size of circles depicts the relative number of PIT-tagged smolts detected at Bonneville Dam each week. Lines represent the estimate of the best linear fit to the data and shading denotes 95% credible intervals around the best fit. Results are those of Payton et al. (2021a) updated with estimates from 2017 and 2018, the most recent years with complete adult returns.

As discussed in Payton et al. (2020), tern predation upstream of BON was estimated to be a super-additive source of smolt mortality (**a** > 1.0), whereby a particular probability of tern predation was associated with an even greater probability of smolt mortality. Several factors may be responsible for the apparent super-additivity of mortality from tern predation. First, super-additivity may be related to the foraging behavior of terns, where unsuccessful predation attempts resulted in some proportion of smolts that escaped predation being injured (lethally- or sub-lethally), losses that are akin to latent mortality or crippling losses in harvest management (Williams et al. 2002, Schaub and Lebreton 2004, Payton et al. 2020). In additional to crippling losses, some proportion of smolts depredated by terns were kleptoparasitized (i.e. stolen) by gulls prior to the smolt being consumed by the foraging tern, its mate, or its young. For example, an unknown, but possibly substantial, proportion of tagged steelhead smolts captured by terns are stolen by gulls (Garcia et al. 2010, Patterson 2012, Adkins et al. 2014). These smolt losses, however, were not fully incorporated into tern predation probabilities (Payton et al. 2020, 2021a) because tags from these steelhead smolts would not have been deposited on the tern nesting colony. Other factors, like terns disproportionally consuming larger-sized steelhead smolts that may be more likely to survive out-migration or factors that may covary with tern predation in space and time may also exist (Payton et al. 2021a). As such, estimates of tern predation probabilities reported herein may underestimate the full impact of

all tern predation on UCR steelhead smolts during out-migration, and thus produce estimates of super-additive mortality from tern predation during the smolt life stage (Payton et al. 2020). The difficulties in calculating unbiased estimates of additivity and the ambiguity associated with the interpretation of this parameter are the primary motivation for instead focusing on the differences in survival probabilities with and without predation (*i.e.* Φ^A). Unlike estimates of additivity, this metric provides management-relevant information with which to quantify the degree to which predation limits both smolt survival and SARs.

The average annual predation probability on UCR steelhead smolts by cormorants nesting on East Sand Island in the CRE was 0.07 (0.06–0.08) of available fish during 2008–2015 (years with accurate estimates of cormorant predation probabilities and complete adult returns; see *Table 11* above) and ranged annually from 0.04 (0.02–0.06) to 0.12 (0.07–0.18). There was some evidence that cormorant predation on UCR steelhead smolts was a partially additive source of mortality to the smolt-to-adult life-stage, with increasing cormorant predation probabilities associated with decreasing adult returns to BON in some, but not all, of the years evaluated. When data from all years were considered, however, there was not a statistically significant relationship at a > 0.95 criteria level (see *Table 11* above); results would, however, be significant at the > 0.90 level. Lower levels of cormorant predation, less variability in levels of cormorant predation, and high levels of total mortality (1-survival) for the smolt-to-adult life-stage contributed to uncertainty in estimates of additivity for mortality caused by cormorant predation in the CRE relative to tern predation in the CRE. Although an estimate of predation by cormorants breeding on the Astoria-Megler Bridge was available in 2021 (ca. 7.2%), SARs are currently unavailable but will be included in future studies. Additional research, including the inclusion of covariates that account for density dependent relationships (Hostetter et al. 2021) and the effects of smolt arrival times in the estuary on adult returns (Payton et al. 2021a), are warranted to better understand to what degree cormorant predation in the estuary limits adult salmonid returns (see Covariate Analysis *below*).

Although sufficient data to understand to what degree consumption by gulls limited UCR steelhead smolt survival were generally lacking in the present study, some proportion of gull consumption of juvenile salmonids is clearly compensatory mortality. Unlike terns and cormorants, gulls acquire much of their food energy by scavenging, and are known to consume dead fish and to kleptoparasitize dead fish from other piscivorous waterbirds, like terns (Antolos et al. 2005, Winkler 1996; see *above*). Relatively large smolts, such as steelhead, are also disproportionately kleptoparasitized by gulls compared with smaller-sized juvenile salmonids, such as Chinook and sockeye salmon (Adkins et al. 2011). Previous research also indicates that smolts are especially susceptible to gull predation in the tailrace of hydroelectric dams, areas where smolts may be injured or stunned because of dam passage or where turbulent hydraulic conditions may disorient fish, thereby increasing their risk of being depredated by gulls (Ruggerone 1986, Zorich et al. 2011, Evans et al. 2016). The proportion of smolts consumed by gulls in the current study that were healthy fish capable of surviving out-migration versus dead or moribund fish, however, is unknown and warrants additional research (see also Cramer et al. 2021a).

The ability to accurately assess the relationship between predation and survival probabilities depends on several criteria, including the level or magnitude of predation, intra-annual (e.g., weekly) variation in estimates of predation, the salmonid life-stage (smolt or smolt-to-adult) evaluated, and sufficient sample sizes of tagged smolts to generate precise estimates of predation and survival probabilities (see also Payton et al. 2021a). Accounting for annual variation in predation and survival probabilities and the use of weekly cohorts within years as replicates are also critical components of the modelling framework (Payton et al. 2020, ISAB 2021). In cases where predation probabilities were low or where baseline mortality was high, larger sample sizes of tagged fish may still be insufficient to identify the relationship between predation and survival probabilities (Payton et al. 2021a). However, in these circumstances, where predation probabilities were low, reductions in avian predation presumably would not dramatically increase fish survival. For example, in 2020, when tern predation probabilities in the CPR were estimated to be 0.02, reductions in tern predation would do little to increase smolt or smolt-to-adult survival.

In summary, results of this study, which updates the results of Payton et al. (2020) using data collected during 2019–2021, provide evidence that predation by terns, and to a lesser extent cormorants, limited the survival of UCR steelhead. Results indicate that efforts to reduce predation on smolts by terns nesting at colonies in the CPR (those upstream of BON) have, on average, increased UCR steelhead smolt survival to Bonneville Dam. More important from a conservation perspective, results indicate that reductions in tern predation on UCR steelhead smolts would increase the number that survive to adulthood, a key finding for those concerned with the restoration of ESA-listed steelhead in the CRB (ISAB 2021). Additional research, however, is needed to better understand how predation/consumption by cormorants and, especially, gulls limits smolt survival. The potential influence of biotic and abiotic factors, like river flows, fish abundance (density), size, condition, run-timing, ocean conditions, and other factors that potentially regulate smolt and smolt-to-adult survival should be investigated and could provide valuable information regarding the degree to which avian predation limits salmonid smolt survival in the CRB (see also *Biotic & Abiotic Factors* section below).

Finally, a more detailed discussion of the modelling approach used in this study is provided in Payton et al. (2020, 2021a) and can also be found in a recently completed report by the Independent Scientific Advisory Board (ISAB 2021). Results of the additive effects of avian predation on other ESA-listed salmonid species (i.e. Chinook and sockeye salmon) and populations (e.g., Snake River salmonid ESUs/DPSs) are provided in Payton et al. (2021a). It is important to note that the magnitude of avian predation rates differs significantly amongst salmonid ESUs/DPSs, as does the potential benefits of managing avian predators to increase salmonid smolt survival in the CRB.

Biotic & Abiotic Factors

Our covariate analyses are on-going and herein we provide preliminary results of an analysis that focused on the survival of acoustic-tagged (AT) steelhead smolts during out-migration through the Wanapum-Priest Rapids Project (Wanapum and Priest Rapids dams and reservoirs). A more complete and comprehensive analysis of covariates, including biotic factors (e.g., smolt

size, condition, and abundance) and an analysis of smolt survival during out-migration through larger river reaches, over longer time periods, are currently in preparation and will be included in our 2022 Annual Report.

Of the four years when acoustic-tagged steelhead were released and terns were still nesting on Goose Island in Potholes Reservoir (2008–2010 and 2014), the JMS model estimated that Goose Island terns were responsible for the vast majority of smolt mortality during passage through the Priest Rapids Project. Estimates of the percentage of total smolt mortality due to tern predation in this river reach were 73.2% (95% credible interval = 39.4–99.8%), 94% (80.6–99.9%), 89.4% (64.7–100.0%), and 78.2% (35.7–99.9%) for 2008, 2009, 2010, and 2014, respectively. In 2014, the first year when management actions to reduce the size of the tern colony on Goose Island were implemented and when the colony was reduced from an average size of 399 breeding pairs during 2008–2010 to 159 breeding pairs in 2014, predation rates were significantly reduced relative to other years, but there low levels of intra-annual variation (*Figure 10*). Decreases in tern predation rates in 2014 were commensurate with increases in smolt survival, as demonstrated in other groups of PIT-tagged steelhead (see *Figure 8* above; also see Payton et al. 2020). The upper bounds of estimates of tern predation imply credible evidence that the terns were responsible for the vast majority of the mortality experienced by AT steelhead smolts during passage through the Priest Rapids Project (*Figure 10*). This is in some part related to the uncertainty inherent in estimates of avian predation rates. The use of AT resulted in high levels of precision for the estimates of survival; however, this added precision is not as impactful for estimates of tern predation. The level of uncertainty remaining in the estimates of tern predation reflects the uncertainty associated with estimates of tag deposition and detection probabilities on tern colonies, uncertainty that is essentially unavoidable if unbiased estimates of predation are calculated (Hostetter et al. 2015, Payton et al. 2019).
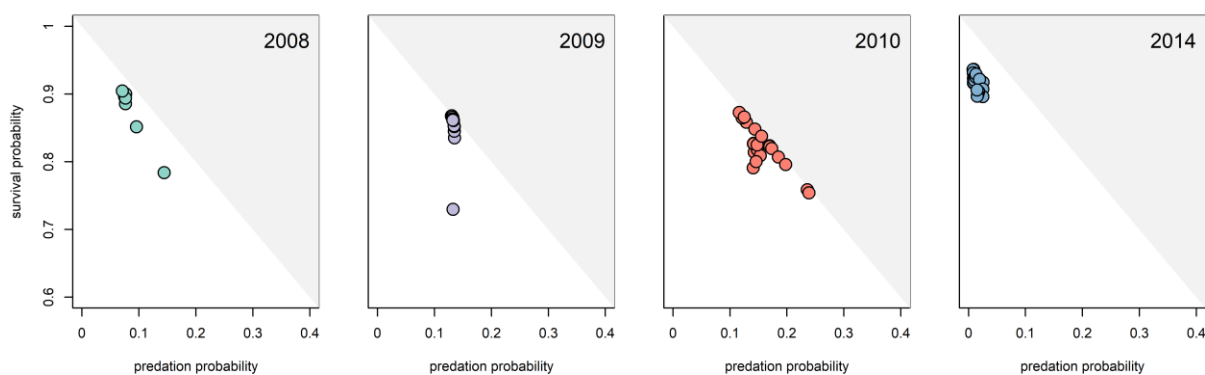
Figure 10. Weekly probability estimates of steelhead smolt survival and predation by Caspian terns nesting on Goose Island in Potholes Reservoir for acoustic-tagged smolts during out-migration from Rock Island Dam to Priest Rapids Dam during 2008–2010 and in 2014. Tern management at the Goose Island colony was first implemented in 2014. Shaded areas representative the boundary or parameter space where predation is necessarily additive (see also Payton et al. 2020).

Estimates of baseline survival and additivity were essentially not identifiable in the AT steelhead dataset. When tern predation represents nearly all smolt mortality in a particular river reach, then predation is necessarily completely additive. It is worth noting that a completely additive effect is likely in cases where cause-specific mortality is high and when measuring the probability of survival immediately after exposure to the specified cause of mortality. However, evidence of "necessarily additive" effects is not informative as to the additive "nature" of predation otherwise. That is, the equation for survival is a piecewise function, where, in the presence of significant sources of mortality not attributable to the predator in question, the relationship between observed estimates of survival (s_d) and predation ($\Theta_d^{\{\text{PRED}\}}$) is approximately defined as

$$s_d = s^0 - a\Theta_d^{\{\text{PRED}\}},$$

and, with repeated measures of survival and predation, we can estimate the parameters representing baseline survival (s^0) and the level of additivity (a). However, in the presence of little to no extraneous sources of mortality (i.e. joint-estimates of survival and predation lie along the parameter space boundary; see *Figure 10* above), the "piece" of the equation that defines the relationship between survival and predation is simply

$$s_d = 1 - \Theta_d^{\{\text{PRED}\}},$$

an equation that does not involve the baseline survival or additivity parameters. Consequently, joint measures of survival and predation at (or very near) this boundary provide little to no information to estimate (identify) these parameters. For many of the daily observations, significant portions of the associated joint posterior distribution abutted the parameter space boundary, confounding the ability to effectively employ the additive-effects JMS model to analyze this particular dataset.

In addition to the increased precision in estimates of survival, there are other advantages to using AT smolts in covariate analyses, such as the ability to more accurately characterized abiotic conditions experienced by smolts during out-migration. However, there were also significant limitations to using these acoustic-tagged fish. Specifically, AT smolts were selected prior to tagging, whereby only smolts in good condition and larger-sized smolts were tagged (i.e. not a random sample of fish); similar numbers of AT smolts were released each day rather than a sample that was in proportion to and in-concert with the run at-large (tagged and untagged); and releases occurred during a short time-period (1 to 3 weeks) relative to the run at-large (8 to 12 weeks, depending on the year).

On-going Analyses: PIT-tagged only smolts (i.e. single tagged fish, like those sampled as part of this project at RIS) will provide a more representative group of tagged fish across larger spatial- and temporal-scales in which to evaluate covariates. For example, in 2009, there was little variation in tern predation probabilities on AT smolts during the 20 days of releases that year,

but tern predation occurred throughout the entire 70 day + (10-week) out-migration period in 2009 (see *Figure 8* above). Furthermore, detailed information on individual fish characteristics (size, rear-type, condition) are available for PIT-tagged fish from RIS and other PIT-tagging projects in the CRB, providing data to evaluate the relationship between individual-level variables and survival. As such, our on-going covariate analysis will focus on abiotic and biotic factors and survival across larger river reaches, time periods, and life-stages (smolt, smolt-to-adult) based on groups of PIT-tagged fish. Our ability to address the effects of abiotic factors on fish survival using PIT-tagged smolts, however, may be limited due to the ability to accurately measure environmental conditions experienced by those smolts during out-migration through larger spatial- and temporal-scales (see also Payton et al. 2016).

The use of AT smolts in this investigation was beneficial and it allowed us to identify where, spatially, tern predation was occurring, information that can then be used to identify and quantify which abiotic factors to include in our future, on-going covariate analyses. For instance, we found that between 52.7% (19.7-84.5%) and 68.5% (56.3-79.4%) of Goose Island tern predation occurred upstream of Priest Rapids Dam, depending on the year, and that over 95% of all predation from this colony was located upstream of Rkm 581 in the Hanford Reach. It follows that the abiotic covariates to be considered in the next steps of the analysis should reflect the location of this predation as, again, it represented the dominant source of mortality for these smolts within this stretch of river in these years. Therefore, this initial investigation has helped us identify the most appropriate abiotic covariates for on-going analyses, which include measures of (1) release-date, (2) water temperature (°C), (3) spill percentage, (4) discharge, (5) water transit times, and (6) fish travel times as modelled through observations of PIT-tagged only steelhead. Water transit time (WTT) indices can be calculated as the ratio between discharge and reservoir elevation. Values of WTT were generated with the COMPASS water velocity model. Data relating to temperature, discharge, and spill percentage will be obtained from the Data Access in Real Time website. These covariates, along with estimates of predation and survival on groups of PIT-tagged steelhead over larger spatial scales (Rock Island to McNary) and temporal scales (weekly estimates over the entire out-migration period) are currently in preparation and will be presented as part of our 2022 Annual Report.

MANAGEMENT RECOMMENDATIONS

Based on our results from 2021 and those from previous years (Collis et al. 2021a, Roby et al. 2021), we offer the following management recommendations to maximize the benefits to ESA-listed juvenile salmonids associated with management of piscivorous colonial waterbirds in the CRB and to minimize the impacts of management on protected populations of migratory birds:

1. Caspian terns continued to show fidelity to managed colony sites in the Columbia Plateau region during 2021 (i.e. Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir). Continued efforts to dissuade terns from nesting on Goose Island

in Potholes Reservoir and Crescent Island in McNary Reservoir will likely be required to prevent re-establishment of tern colonies larger than 40 breeding pairs at these two sites, as stipulated by the *Inland Avian Predation Management Plan*. Management strategies to dissuade nesting by terns should include multiple techniques, both passive and active, and should focus on implementing these strategies early in the breeding season, particularly during the peak of the smolt out-migration period (i.e. April–June). The use of braided rope for passive nest dissuasion is recommended at Goose Island and both passive and active tern nest dissuasion efforts may once again be necessary to prevent tern colony growth at Crescent Island in 2022 and beyond.

2. The size of the East Sand Island colony of Caspian terns dropped below the target colony size specified in the *Caspian Tern Management Plan for the Columbia River Estuary* (3,125–4,375 breeding pairs) for the first time in 2020, when colony size was 2,387 breeding pairs, and did so again in 2021, when colony size was 2,050 breeding pairs. Few, if any, young terns were raised at the East Sand Island colony in 2020, and no young were raised on the colony in 2021. Nesting conditions for terns on East Sand Island have apparently declined to such an extent that the colony is at risk of being abandoned. High rates of tern egg and chick predation by gulls nesting on or near the 1-acre tern colony area, especially during disturbances to the colony by bald eagles, combined with the poor quality of tern nesting substrate on the 1-acre designated colony site (partially vegetated and does not drain properly during heavy rains) are responsible for increasingly frequent nesting failures and declining colony size. If the tern colony on the designated 1-acre colony site on East Sand Island were abandoned, it would place much greater pressure from terns prospecting for alternative colony sites on Rice Island, other dredged material disposal sites in the upper estuary, and satellite colony sites on East Sand Island. Also, the Caspian tern breeding population in the Pacific Flyway is currently in steep decline and population modeling indicates that stabilizing the population will require sustaining a productive tern colony on East Sand Island or similar secure coastal colony site. We recommend that limited lethal control of predatory gulls and improvements to tern nesting substrate be conducted at the 1-acre designated colony site on East Sand Island in 2022 and beyond so that tern nesting success is sufficient to ensure that the colony is not abandoned, and colony size is prevented from declining any further.

3. One unintended consequence of Caspian tern management actions on East Sand Island has been the large number of terns (several hundred to several thousand) that have attempted to nest outside of the prepared 1-acre nesting area on East Sand Island, as well as nesting attempts by terns on Rice Island and other sites in the upper estuary. In order to continue to meet the objectives of the *Caspian Tern Management Plan for the Columbia River Estuary,* we recommend (1) continued use of Caspian tern nest dissuasion techniques to prevent tern nesting outside the designated colony area on East Sand Island and on Rice Island and at other sites in the upper estuary; (2) resumption of close monitoring of tern colonies throughout the estuary using previously established methods to ensure that colony size at the designated colony site on East

Sand Island is within the range specified in the *Plan* and no tern colonies become established outside the designated colony site; and (3) continued recovery and analysis of smolt PIT tags on all tern colony sites in the estuary to estimate tern predation rates on juvenile salmonids using previously established, robust methodology to ensure that impacts to salmonid ESUs/DPSs from tern predation do not exceed acceptable levels.

4. We recommend that water levels in John Day Reservoir be raised during the early stages of the Caspian tern nesting season in 2022 and beyond to prevent tern colony formation in the Blalock Islands, as was successfully accomplished in 2021. A managed increase in the elevation of John Day Reservoir during the 2021 Caspian tern breeding season eliminated all habitat previously used by nesting Caspian terns at the Blalock Islands. As a result, a tern colony did not form in the Blalock Islands during 2021, the first year since 2004 that no tern colony has formed on the Blalock Islands. This recommendation assumes that are there are no indirect, negative effects of higher reservoir levels on salmonid smolt survival in John Day Reservoir and that active and passive tern dissuasion techniques, like those used at the Crescent Island and Goose Island tern colony sites, are not permitted at the Blalock Islands in 2022 and beyond.

5. A sizeable nesting colony of Caspian terns (231 breeding pairs) formed for the first time on Badger Island in 2021. This new colony supported more than half of all the Caspian terns nesting in the Columbia Plateau region in 2021 and, by itself, was responsible for the regional tern breeding population exceeding the level stipulated in the *Inland Avian Predation Management Plan*. Adaptive management by way of tern nest dissuasion techniques (e.g., stakes, rope, and flagging; planted vegetation; placement of woody debris) on Badger Island will likely be needed to prevent further growth of the Badger Island tern colony and the associated impact on smolt survival that predation by terns from a large Badger Island colony would have. American white pelicans are listed as "Threatened" by the State of Washington, so efforts to dissuade terns from nesting on Badger Island will need to be designed not to disturb pelicans nesting nearby on Badger Island.

6. Continued monitoring of unmanaged tern colonies in the CPR is recommended to identify those colony sites where tern predation rates remain high (i.e. above the 2% trigger level established by the *IAPMP*) and to help identify colony sites in the CPR where tern predation rates on juvenile salmonids are low and acceptable (i.e. places where management might be implemented to encourage tern nesting).

7. An investigation of the cumulative effects of avian predation/consumption on salmonid smolt survival suggests that smolt consumption by gulls, primarily California gulls, nesting at several unmanaged colonies is a source of substantial mortality for smolts during out-migration to Bonneville Dam. Research is needed to better understand factors that influence the susceptibility of smolts to gull consumption and to quantify the degree to which consumption by gulls limits smolt survival (i.e. is an additive source of mortality). Management actions to reduce the size of specific gull colonies in the CPR

is currently planned for the Miller Rocks gull colony and may be warranted for the Badger Island gull colony, but currently there are few data to suggest that managing gull colonies in the CPR would enhance smolt survival. Consumption of smolts by gulls from other colonies in the region, particularly those on islands at considerable distance from the Columbia River (e.g., Goose Island in Potholes Reservoir or islands in Lake Lenore), pose little threat to smolt survival and may be sites where gulls from managed colonies can be relocated non-lethally.

8. As an unintended consequence of implementation of the *Double-crested Cormorant Management Plan for the Columbia River Estuary*, cormorants have abandoned East Sand Island as a nesting site and many are now nesting further upriver on the Astoria-Megler Bridge. The colony on the Astoria-Megler Bridge is currently the largest double-crested cormorant breeding colony anywhere in the Pacific Flyway of North America. In 2021, PIT tag recoveries from the cormorant colony on the Astoria-Megler Bridge indicated that per capita (per breeding pair) smolt predation rates were 2 to 4 times higher (depending on salmonid species) than for cormorants nesting on East Sand Island. In terms of colony-wide predation probabilities, impacts on smolt survival by double-crested cormorants nesting on the bridge in 2021 were similar to those observed at the East Sand Island cormorant colony prior to management, even though the size of the bridge colony is much smaller than the average size of the colony on East Sand Island prior to management. These results indicate that management of cormorants in the Columbia River estuary has not improved smolt survival, as intended, and that adaptive management to dissuade cormorants from nesting on the bridge and to restore a cormorant colony on East Sand Island that is less than 5,380–5,939 breeding pairs are needed to meet management objectives.

9. Irrespective of the need for additional management to reduce avian predation on juvenile salmonids in the CRB, accounting for factors that limit smolt survival to the degree observed in this and other studies may be paramount in interpreting the results of, and measuring the efficacy of, other management actions aimed at restoring ESA-listed salmonids in the basin. Conversely, by not considering avian predation when evaluating the efficacy of other management actions to restore ESA-listed salmonids, the benefits of such actions would likely be confounded and could otherwise be masked due to unaccounted for fluctuations in avian predation rates.

ACKNOWLEDGMENTS

LITERATURE CITED

Adkins, J.Y., D.D. Roby, D.E. Lyons, Y. Suzuki, P.J. Loschl, K. Collis, and A.F. Evans. 2011. Colony size, nesting success, and limiting factors of piscivorous colonial waterbirds in the Columbia Plateau region, 2004-2009 *in* D.D. Roby, K. Collis, D.E. Lyons, and A.F. Evans (editors). Impacts of Avian Predation on Salmonid Smolts from the Columbia and Snake Rivers: 2004–2009 Synthesis Report. Submitted to the U.S. Army Corps of Engineers – Walla Walla District, Walla Walla, Washington.

Adkins, J.Y., D.E. Lyons, P.J. Loschl, D.D. Roby, K. Collis, A.F. Evans, and N.J. Hostetter. 2014. Demographics of piscivorous colonial waterbirds and management implications for ESA-listed salmonids on the Columbia Plateau. Northwest Science 88:344-359.

Antolos, M., D.D. Roby, and K. Collis. 2004. Breeding ecology of Caspian terns at colonies on the Columbia Plateau. Northwest Science 78:303-312.

Antolos, M., D.D. Roby, D.E. Lyons, K. Collis, A.F. Evans, M. Hawbecker, and B.A. Ryan. 2005. Caspian tern predation on juvenile salmonids in the mid-Columbia River. Transactions of the American Fisheries Society 134:466-480.

Brandtner, C.M. and K.S. Tidwell. 2021. Distribution and dissuasion efforts of Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auratus*) on East Sand Island of the Columbia River: 2021 Season Summary Report. Available from the U.S. Army Corps of Engineers, Fisheries Field Unit, Bonneville Lock and Dam, Cascade Locks, Oregon.

Collis, K., D.D. Roby, D.P. Craig, S. Adamany, J. Adkins, and D.E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: Implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Collis, K., D.D. Roby, P.J. Loschl, Y. Suzuki, A. Munes, J. Mulligan, E. Schniedermeyer, A.F. Evans, B. Cramer, A. Turecek, and Q. Payton. 2016. Implementation of the Inland Avian Predation Management Plan. 2015 Final Annual Report. Submitted to the U.S. Army Corps of Engineers – Walla Walla District (Walla Walla , WA) and the U.S. Bureau of Reclamation (Boise, Idaho).

Collis, K., D.D. Roby, P.J. Loschl, Y. Suzuki, K. Kelly, E. Schniedermeyer, A.F. Evans, B. Cramer, A. Turecek, and Q. Payton. 2017. Implementation of the Inland Avian Predation Management Plan. 2016 Final Annual Report. Submitted to the U.S. Army Corps of Engineers – Walla Walla District (Walla Walla , WA) and the U.S. Bureau of Reclamation (Boise, Idaho).

Collis, K., A.F. Evans, B. Cramer, A. Turecek, Q. Payton, K. Kelly, F. Stetler, S. Fitzmaurice, and P.J. Loschl. 2018. Implementation of the Inland Avian Predation Management Plan. 2017 Final Annual Report. Submitted to the U.S. Army Corps of Engineers – Walla Walla District (Walla Walla , WA) and the U.S. Bureau of Reclamation (Boise, Idaho).

Collis, K., A.F. Evans, B. Cramer, A. Turecek, Q. Payton, R. Bhatt, T. Kaufman, A. Gibson, and T. Lawes. 2019. Implementation of the Inland Avian Predation Management Plan. 2018 Final Annual Report. Submitted to the U.S. Army Corps of Engineers – Walla Walla District (Walla Walla , WA) and the U.S. Bureau of Reclamation (Boise, Idaho).

Collis, K., A.F. Evans, J. Tennyson, A. Turecek, Q. Payton, and R. Bhatt. 2020. Avian predation in the Columbia Plateau region: Management, monitoring, and evaluation. 2019 Final Annual Report. Submitted to Grant County Public Utility District and the Priest Rapids Coordinating Committee, Ephrata, Washington.

Collis, K., A.F. Evans, D.D. Roby, J. Tennyson, A. Turecek, Q. Payton, and T.J. Lawes. 2021a. Avian predation in the Columbia River Basin: 2020 Final Annual Report. Submitted to the Bonneville Power Administration, Portland, Oregon and the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington.

Collis, K., D.D. Roby, A.F. Evans, D.E. Lyons, T.J. Lawes, Q. Payton, B. Cramer, A. Turecek, and
    A.G. Patterson. 2021b. Chapter 2: Caspian tern management in the Columbia Plateau region
    in D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the
    Columbia River Basin: Synopsis of Ecology and Management. A synthesis report submitted
    to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power
    Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids
    Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and
    Wildlife, Salem, Oregon. 789 pp.

Cramer, B., K. Collis, A.F. Evans, D.D. Roby, D.E. Lyons, T.J. Lawes, Q. Payton, and A. Turecek.
    2021a. Chapter 6: Predation on juvenile salmonids by colonial waterbirds nesting at
    unmanaged colonies in the Columbia River basin in D.D. Roby, A.F. Evans, and K. Collis
    (editors). Avian Predation on Salmonids in the Columbia River Basin: Synopsis of Ecology
    and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla
    Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant
    County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington;
    and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Cramer, B., A.F. Evans, Q. Payton, K. Collis, and D.D Roby. 2021b. Chapter 5: Relative impacts of
    double-crested cormorants and Caspian terns on juvenile salmonids in the Columbia River
    estuary: A retrospective analysis of PIT tag data in D.D. Roby, A.F. Evans, and K. Collis
    (editors). Avian Predation on Salmonids in the Columbia River Basin: Synopsis of Ecology
    and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla
    Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant
    County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington;
    and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Evans, A.F., N.J. Hostetter, D.D. Roby, K. Collis, D.E. Lyons, B.P. Sandford, R.D. Ledgerwood, and
    S. Sebring. 2012. Systemwide evaluation of avian predation on juvenile salmonids from the
    Columbia River based on recoveries of passive integrated transponder tags. Transactions of
    American Fisheries Society, 141:975-989.

Evans, A.F., N.J. Hostetter, K. Collis, D.D. Roby, and F.J. Loge. 2014. Relationship between
    juvenile fish condition and survival to adulthood in steelhead. Transactions of the American
    Fisheries Society 143:899-909.

Evans, A.F., Q. Payton, A. Turecek, B.M. Cramer, K. Collis, D.D. Roby, P.J. Loschl, L. Sullivan, J.
    Skalski, M. Weiland, and C. Dotson. 2016. Avian predation on juvenile salmonids: Spatial
    and temporal analysis based on acoustic and Passive Integrated Transponder
    tags. Transactions of the American Fisheries Society 145:860-877.

Evans, A.F., Q. Payton, B.M. Cramer, K. Collis, N.J. Hostetter, D.D. Roby, and C. Dotson. 2019.
    Cumulative effects of avian predation on Upper Columbia River Steelhead. Transactions of
    the American Fisheries Society 148:896-913.

Evans, A.F., Q. Payton, K. Collis, B. Cramer, A. Turecek, N.J. Hostetter, and D.D. Roby. 2021a. Chapter 7: Cumulative effects of avian predation on juvenile salmonids in the Columbia River basin in D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 789 pp.

Evans, A.F, Q. Payton, B. Cramer, and K. Collis. 2021b. 2020 East Sand Island Avian Predation Rate Analysis. Draft Technical Report submitted to the U.S. Army Corps of Engineers in partial fulfillment of Contract No. W9127N19D0006, Order No. W9127N221F0041.

Garcia, G.O., M. Favero, and A.I. Vassallo. 2010. Factors affecting kleptoparasitism by gulls in a multi-species seabird colony. Condor 112:521-529.

Gelman, A., J.B. Carlin, H.S. Stern, D.B. Dunson, A. Vehtari, and D.B. Rubin. 2013. Bayesian Data Analysis (3rd ed.). Chapman and Hall/CRC, Boca Raton, Florida, USA.

Hamilton, J.D. 1994. Time Series Analysis. Princeton University Press, Princeton, NJ.

Hatch, K.B., L.S. Sullivan, M.A. Timko, J.R. Skalski, R.L. Townsend, and C. Dotson. 2016. Behavior and survival analysis of juvenile steelhead and sockeye salmon through the Priest Rapids Hydroelectric Project in 2015. Draft Report prepared for Public Utility District No. 2 of Grant County by Blue Leaf Environmental, Inc. in Ellensburg, Washington and Columbia Basin Research in Seattle, Washington.

Hostetter, N.J., A.F. Evans, D.D. Roby, K. Collis, M. Hawbecker, B.P. Sandford, D.E. Thompson, and F.J. Loge. 2011. Relationship of external fish condition to pathogen prevalence and out-migration survival in juvenile steelhead. Transactions of the American Fisheries Society 140(5):1158-1171.

Hostetter, N.J., A.F. Evans, D.D. Roby, and K. Collis. 2012. Susceptibility of juvenile steelhead to avian predation: the influence of individual fish characteristics and river conditions. Transactions of the American Fisheries Society 141:1586-1599.

Hostetter, N.J., A.F. Evans, B.M. Cramer, K. Collis, D.E. Lyons, and D.D. Roby. 2015. Quantifying avian predation on fish populations: Integrating predator-specific deposition probabilities in tag–recovery studies. Transactions of the American Fisheries Society 144:410-422.

Hostetter N.J., B. Gardner, A.F. Evans, B.M. Cramer, Q. Payton, K. Collis, and D.D. Roby. 2018. Wanted dead or alive: a state-space mark-recapture-recovery model incorporating multiple recovery types and state uncertainty. Canadian Journal of Fisheries and Aquatic Sciences 75:1117-1127.

Hostetter, N.J., A.F. Evans, Q. Payton, D.D. Roby, D.E. Lyons, and K. Collis. 2021. Chapter 10: Factors affecting the susceptibility of juvenile salmonids to avian predation in D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

ISAB (Independent Scientific Advisory Board). 2021. Comparison of Research Findings on Avian Predation Impacts on Salmon Survival. Report (No. 2021-2) for the Northwest Power and Conservation Council, Columbia River Basin Indian Tribes, and National Marine Fisheries Services. Available on-line at https://www.nwcouncil.org/sites/default/files/ISAB%202021-2%20AvianPredationComparison23April.pdf

Knopf, F.L., and R.M. Evans. 2004. American White Pelican (*Pelecanus erythrorhynchos*), version 2.0. *In* The Birds of North America (A.F. Poole, editor). Cornell Lab of Ornithology, Ithaca, New York.

Kock, T.J., Hansen, A.C., Evans, S.D., Visser, R., Saluskin, B., Matala, A., and Hoffarth, P., 2021, Evaluation of factors affecting migration success of adult sockeye salmon (*Oncorhynchus nerka*) in the Yakima River, Washington, 2020: U.S. Geological Survey Open-File Report 2021–1075, 30 p., https://doi.org/ 10.3133/ ofr20211075.

Lawes, T.J., D.D. Roby, K.S. Bixler, D.E. Lyons, K. Collis, A.F. Evans, and A.G. Patterson. 2021a. Chapter 3: Caspian tern management at alternative colony sites outside of the Columbia River Basin *in* D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Lawes, T.J., K.S. Bixler, D.D. Roby, D.E. Lyons, K. Collis, A.F. Evans, A. Peck-Richardson, B. Cramer, Y.Suzuki, J.Y. Adkins, K. Courtot, and Q. Payton. 2021b. Chapter 4: Double-crested cormorant management in the Columbia River estuary *in* D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Lawes, T.J., D.D. Roby, and D.E. Lyons. In prep. Pacific Flyway Caspian tern population monitoring, 2021 Final Annual Report. Department of Fisheries and Wildlife, Oregon State University, Corvallis, Oregon. Draft report submitted to the U.S. Fish and Wildlife Service, Migratory Birds and Habitat Programs. 29 pp.

Lyons, D.E., D.D. Roby, A.F. Evans, N.J. Hostetter, and K. Collis. 2014. Benefits to Columbia River anadromous salmonids from potential reductions in predation by double-crested cormorants nesting at the East Sand Island colony in the Columbia River estuary. Final Report. Submitted to U.S. Army Corps of Engineers – Portland District, Portland, Oregon.

Meredith, M., and J. Kruschke. 2016. HDInterval: highest (posterior) density intervals. R package version 0.1, 3. URL https://CRAN.R-project.org/package=HDInterval.

NOAA (National Oceanic and Atmospheric Administration). 2021. Updated status of federally listed ESUs of West Coast salmon and steelhead. Available from NOAA Fisheries, Seattle, Washington.

Patterson, A. 2012. Breeding and foraging ecology of Caspian terns nesting on artificial islands in the Upper Klamath Basin, California. Unpubl. M.Sc. Thesis, Oregon State University, Corvallis, Oregon, USA.

Payton, Q., A.F. Evans, and B. Cramer. 2016. Effects of biotic and abiotic factors on juvenile steelhead survival in the Middle Columbia River, 2008-2015. Final Report submitted to the Grant County Public Utility District and the Priest Rapids Coordinating Committee.

Payton, Q., N.J. Hostetter, and A.F. Evans. 2019. Jointly estimating survival and mortality: Integrating recapture and recovery data from complex multiple predator systems. Environmental and Ecological Statistics 26:107-125.

Payton, Q., A.F. Evans, N.J. Hostetter, D.D. Roby, B. Cramer, and K. Collis. 2020. Measuring the additive effects of predation on prey survival across spatial scales. Ecological Applications 30:e02193. 10.1002/eap.2193 https://doi.org/10.1002/eap.2193.

Payton, Q., A.F. Evans, N.J. Hostetter, B. Cramer, K. Collis, and D.D. Roby. 2021a. Chapter 8: Additive effects of avian predation on the survival of juvenile salmonids in the Columbia River basin in D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 789 pp.

Payton, Q., A. Evans, J. Fryer, and T. Garrison. 2021b. Factors Influencing Avian Predation of Juvenile Upriver Bright Chinook Salmon. Final Technical Report submitted to the Pacific Salmon Commission Chinook Technical Committee in partial fulfillment of Contract No. 048-001. Available at: www.birdresearchnw.org

Peck-Richardson, A., D.E. Lyons, D.D. Roby, and T. Lawes. 2019. Pacific Flyway Caspian tern population monitoring, 2018 Final Annual Report. Department of Fisheries and Wildlife, Oregon State University, Corvallis, Oregon. Final report submitted to the U.S. Fish and Wildlife Service, Migratory Birds and Habitat Programs. 25 pp.

Petrosky, C.E., and H.A. Schaller. 2010. Influence of river conditions during seaward migration and ocean conditions on survival rates of Snake River Chinook Salmon and steelhead. Ecology of Freshwater Fish 19:520–536.

PSMFC (Pacific States Marine Fisheries Commission). 2021. PIT Tag Information System (PTAGIS): A regional database. Available online at www.ptagis.org.

RDCT (R Development Core Team). 2014. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org.

RDCT (R Development Core Team). 2021. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org.

Roby, D.D., K. Collis, P.J. Loschl, J. Tennyson, Y. Suzuki, A. Munes, S. Toomey, A.F. Evans, B. Cramer, A. Turecek, and Q. Payton. 2015. Implementation of the Inland Avian Predation Management Plan. 2014 Final Annual Report. Submitted to the U.S. Army Corps of Engineers – Walla Walla District and the U.S. Bureau of Reclamation.

Roby, D.D, K. Collis, D.E. Lyons, T. Lawes, Y. Suzuki, P.J. Loschl, K. Bixler, E. Hanwacker, J. Mulligan, E. Schniedermeyer, A.F. Evans, B. Cramer, A. Turecek, Q. Payton, and M. Hawbecker. 2016. Evaluation of Foraging Behavior, Dispersal, and Predation on ESA-listed Salmonids by Caspian Terns Displaced from Managed Colonies in the Columbia Plateau Region. 2015 Final Annual Report. Submitted to the Grant County Public Utility District and the Priest Rapids Coordinating Committee.

Roby, D.D, K. Collis, D.E. Lyons, T. Lawes, Y. Suzuki, P.J. Loschl, K. Bixler, K. Kelly, E. Schniedermeyer, A.F. Evans, B. Cramer, J. Morella, A. Turecek, Q. Payton, and M. Hawbecker. 2017. Evaluation of Foraging Behavior, Dispersal, and Predation on ESA-listed Salmonids by Caspian Terns Displaced from Managed Colonies in the Columbia Plateau Region. 2016 Final Annual Report. Submitted to the Grant County Public Utility District and the Priest Rapids Coordinating Committee.

Roby D.D, A.F. Evans, and K. Collis (editors). 2021. Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S. Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Ruggerone, G.T. 1986. Consumption of migrating juvenile salmonids by gulls foraging below a Columbia River dam. Transactions of the American Fisheries Society 115:736-742.

Ryan, B.A., S.G. Smith, J.M. Butzerin, and J.W. Ferguson. 2003. Relative vulnerability to avian predation of juvenile salmonids tagged with Passive Integrated Transponders in the Columbia River estuary, 1998–2000. Transactions of the American Fisheries Society 132:275-288.

Sandercock, B.K, E.B. Nilsen, H. Broseth, and H.C. Pedersen. 2011. Is hunting mortality additive or compensatory to natural harvest? Effects of experimental harvest on the survival and cause-specific mortality of willow ptarmigan. Journal of Animal Ecology 80:244-258.

Schaub, M., and J.D. Lebreton. 2004. Testing the additive versus the compensatory hypothesis of mortality from ring recovery data using a random effects model. Animal Biodiversity and Conservation 27:73-85.

Sebring, S.H., M.C. Carper, R.D. Ledgerwood, B.P. Sandford, G.M. Matthews, and A.F. Evans. 2013. Relative vulnerability of PIT-tagged subyearling fall Chinook salmon to predation by Caspian terns and double-crested cormorants in the Columbia River estuary. Transactions of the American Fisheries Society 142:1321-1334.

SDT (Stan Development Team). 2015. Stan: A C++ Library for Probability and Sampling, Version 2.8.0. URL http://mc-stan.org/.

SDT (Stan Development Team). 2020. RStan: the R Interface to Stan. R package version 2.21.1.

Stinson, D.W. 2016. Periodic status review for the American White Pelican in Washington. Washington Department of Fish and Wildlife, Olympia, Washington.

Timko, M.A., L.S. Sullivan, S.E. Rizor, R.R. O'Connor, C.D. Wright, J.L. Hannity, C.A. Fitzgerald, M.M. Meagher, J.D. Stephenson, and J.R. Skalski, R.L. Townsend. 2011. Behavior and survival analysis of juvenile steelhead and sockeye salmon through the Priest Rapids Project in 2010. Report prepared for Public Utility District No. 2 of Grant County, Washington by Blue Leaf Environmental, Inc., Ellensburg, Washington.

USACE (U.S. Army Corps of Engineers). 2014. Inland Avian Predation Management Plan Environmental Assessment. U.S. Army Corps of Engineers, Walla Walla District, Northwestern Division. January 2014. Available from USACE-Walla Walla District, Walla Walla, Washington.

USACE (U.S. Army Corps of Engineers). 2015. Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary: Final Environmental Impact Statement, February 2015. Available from USACE-Portland District, Portland, Oregon.

USACE-FFU (U.S. Army Corps of Engineers-Fisheries Field Unit). 2021. Distribution and Dissuasion Efforts of Caspian Terns (*Hydroprogne caspia*) on East Sand Island of the Columbia River: 2020 Season Summary Report. Available from USACE-FFU, Cascade Locks, Oregon.

USDA-WS (United State Department of Agriculture-Wildlife Services). 2021. Tern Deterrence at Goose Island. 2020 Annual Report to the Bureau of Reclamation. Available from USDA-WS, Olympia, Washington.

USFWS (U.S. Fish and Wildlife Service). 2005. Caspian tern management to reduce predation on juvenile salmonids in the Columbia River estuary, Final Environmental Impact Statement. Portland, OR. Available from USFWS, Portland, Oregon.

Williams, B K., J.D. Nichols, and M J. Conroy. 2002. Analysis and Management of Animal Populations. Academic Press, San Diego, California, USA.

Winkler, D.W. 1996. California gull (*Larus californicus*), version 2.0. *In* The Birds of North America. *Edited by* A. Poole. Cornell Lab of Ornithology, Ithaca, New York.

Zorich, N.A., M.R. Jonas, and P.L. Madson. 2011. Avian predation at John Day and The Dalles dams 2010: Estimating fish consumption using direct observations and diet analysis. Report to the U.S. Army Corps of Engineers, Portland District, Portland, Oregon.

APPENDIX A: Predation on Juvenile Salmonids by Double-crested Cormorants Nesting on the Astoria-Megler Bridge

ABSTRACT

An unintended consequence of management actions at the double-crested cormorant colony on East Sand Island during 2015–2019 was the complete abandonment of the colony site and the subsequent rapid expansion of the cormorant colony on the Astoria-Megler Bridge (AMB). The AMB is located upstream of East Sand Island in the freshwater mixing zone of the CRE, an aquatic environment where fewer alternative prey fish are likely available and juvenile salmonids may be consumed in greater proportion by cormorants relative to foraging areas around the former cormorant colony site on East Sand Island which is in the marine zone of the estuary. To estimate predation rates on salmonid smolts by cormorants breeding on the AMB in 2021, we recovered smolt PIT tags deposited by cormorants nesting in five discrete plots on a concrete footing of the bridge, areas where smolt PIT tags could be recovered by researchers following the breeding season. Estimates of per capita (per breeding pair) predation rates derived from cormorants nesting in plots (n = 159 breeding pairs) were then extrapolated to account for all double-crested cormorants nesting elsewhere on the bridge (n = 3,992 breeding pairs) to generate colony-wide estimates of predation rates. Results indicated that per capita predation rates of cormorants nesting on the AMB ranged from 0.0009% (95% credible interval = 0.0005–0.0016%) on yearling Chinook salmon smolts to 0.0040% (0.0028–0.0069%) on sub-yearling Chinook salmon smolts originating from populations upstream of Bonneville Dam. Per capita predation rates were 0.0013% (0.0004–0.0025%) on steelhead smolts and 0.0015% (0.0004–0.0031%) on coho salmon smolts originating from populations downstream of Bonneville Dam. Colony-wide estimates of predation rates ranged from 3.9% (2.0–6.8%) on yearling Chinook salmon to 16.8% (7.3–28.4%) on sub-yearling Chinook salmon and from 5.5% (1.8–10.5%) on steelhead to 6.1% (1.6–12.7%) on coho salmon. Estimates of per capita predation rates on salmonid smolts by cormorants nesting on the AMB in 2021 were approximately 2 to 4 times greater (depending on salmonid species/age-class) than average annual per capita predation rates by cormorants that nested on East Sand Island prior to implementation of management actions (2003–2014). Estimates of colony-wide predation probabilities by cormorants nesting on the AMB in 2021 were similar to those of cormorants breeding on East Sand Island during 2003–2014, despite the much smaller size of the cormorant colony on the AMB in 2021 (4,151 pairs) compared with the cormorant colony on East Sand Island (12,787 pairs on average during 2003–2014). Collectively, results suggest that colony location is closely related to the impact of cormorant predation on survival of salmonid smolts in the CRE, and that these impacts have likely remained unchanged since management actions on East Sand Island were first implemented in 2015.

INTRODUCTION

Predation by double-crested cormorants (DCCO; *Nannopterum auritum*) in the Columbia River basin (CRB) and elsewhere in North America has been identified as a factor limiting the recovery of some fish populations of conservation concern (e.g., Pacific salmon and trout [*Oncorhynchus* spp.], Atlantic salmon and trout [*Salmo* spp.], and suckers [Catostomidae]; Weitkamp et al. 2014, Scoppettone et al. 2014, Evans et al. 2016a, Lawes et al. 2021, Roby et al. 2021). Following the ban of the organochloride pesticides in the United States, DCCO, a species native to North America and protected under the Migratory Bird Treaty Act, established what would become the largest breeding colony in the Western North America on East Sand Island (ESI) in the Columbia River estuary (CRE). This colony was first established in 1989 with less than 100 breeding pairs and grew to an average size of ca. 12,800 breeding pairs during 2003 – 2014 (Naughton et al. 2007, Lawes et al. 2021, Roby et al. 2021). This large DCCO colony was supported by an abundance of suitable ground nesting habitat and the reliable availability in both marine and freshwater fishes in the CRE (Adkins et al. 2014, Lawes et al. 2021). Among the forage fish available to piscivorous waterbirds nesting in the CRE are 13 anadromous salmonid populations protected under the U.S. Endangered Species Act (ESA; NOAA 2021). The breeding season for DCCO in the CRE (March – August) overlaps with the peak outmigration period for juvenile salmonids from the CRB (April – August, Evans et al. 2012, Adkins et al. 2014, Lawes et al. 2021). There has been substantial concern by resource managers that DCCO nesting in the CRE may be impacting the survival of out-migrating smolts and the population recoveries of these imperiled salmonid populations.

Millions of juvenile salmonids can be consumed annually by DCCO nesting at large colonies in the CRE; the relative proportion of salmonids in the diet composition is, however, associated with the location of the colony within the CRE (Collis et al. 2002, Anderson et al. 2004, Cramer et al. 2021). Analysis of the diet composition of DCCO nesting concurrently on Rice Island, located in the freshwater zone of the CRE at river km (Rkm) 34, and ESI, located in the marine zone of the CRE at Rkm 8 (*Figure 1*), demonstrated that juvenile salmonids were 2 – 3 times more prevalent in the diet of DCCO nesting at the colony in the freshwater zone as compared to those nesting at the colony in the marine zone (Collis et al. 2002, Anderson et al. 2004). Further, the number of marked (tagged) juvenile salmonids consumed by DCCO nesting in the CRE has been approximately 5 – 18 times higher for DCCO nesting in the freshwater zone compared to DCCO nesting in the marine zone of the CRE (Cramer et al. 2021). While differences in consumption rates based on colony location are presumably due to differences in the relative availability of different prey types within the CRE (Anderson et al. 2004, Lyons 2010), the drivers of this pattern are not fully understood, as DCCO nesting on ESI have been observed foraging over broad geographic areas and throughout all salinity zones of the CRE (Peck-Richardson et al. 2018).

Figure 1. Locations of historical and current double-crested cormorant colonies in the CRE, including East Sand Island in the marine salinity zone, the Astoria-Megler Bridge in the mixing salinity zone, and Miller Sands Island and Rice Island in the freshwater salinity zone (derived from Simenstad et al. [1990], as modified by Anderson et al. [2004]).

As the DCCO colony on ESI increased from 5,023 breeding pairs in 1997 to 14,916 breeding pairs at its peak in 2013, management of the DCCO colony was implemented to reduce consumption of juvenile salmonids in the CRE (USACE 2015). The primary goal of the management plan was to reduce the size of the ESI colony to less than 5,380 – 5,939 breeding pairs and to minimize dispersal to new colony sites where predation impacts on smolts could be greater than at ESI, including at locations in the upper CRE (USACE 2015). As the management plan was implemented, the DCCO colony on ESI rapidly declined from 12,150 breeding pairs in 2015, to partial colony abandonment during 2016 – 2018, to complete colony abandonment during 2019 – 2020 (Lawes et al. 2021, Lawonn in prep.). While the management plan succeeded in reducing the size of the DCCO colony on ESI to levels specified in the management plan, there were unintended outcomes including (1) the total abandonment of the ESI DCCO colony, which presumably led to approximately 20,000 DCCO prospecting for new nesting sites and (2) a substantial increase in the number of DCCO nesting at other colony sites in the upper CRE and elsewhere on the Columbia River. These colonies were located on artificial structures such as channel markers and bridges, which had been used for DCCO nesting previously, but only in small numbers (Evans et al. 2019a, Lawes et al. 2021). One such structure, the Astoria-Megler Bridge (AMB), was of particular concern, where, coinciding with management actions on ESI, the DCCO colony increased from 425 breeding pairs in 2015 to 5,081 breeding pairs in 2020 (*Figure 2*) and replaced ESI as the largest DCCO colony in Western North American (Lawes et al. 2021). Fisheries managers are concerned with the rapid growth of DCCO colony on AMB and its location in the mixing salinity zone of the CRE where out-migrating juvenile salmonids may be more susceptible to DCCO predation on a per capita (per breeding pair) basis as

compared to the former colony site on ESI in the marine zone of the CRE (see *Figure 1* above). Although the DCCO nesting colony on the AMB has yet to reach a size comparable to that of the ESI colony prior to management, additional suitable nesting habitat for DCCO exists on the AMB with the potential to support upwards of 10,000 breeding pairs (Lawonn in prep.). Studies that provide estimates of consumption and predation rates (percentage of available prey consumed) for out-migrating juvenile salmonids (smolts) by DCCO nesting on the AMB have not been conducted to-date, and the impacts of DCCO nesting in the mixing salinity zone of the CRE remains a critical uncertainty to the recovery of ESA-listed salmonid populations. Results of this investigation also help provide an assessment of the efficacy of ongoing management at ESI to reduce predation by DCCO on juvenile salmonids throughout the CRE.
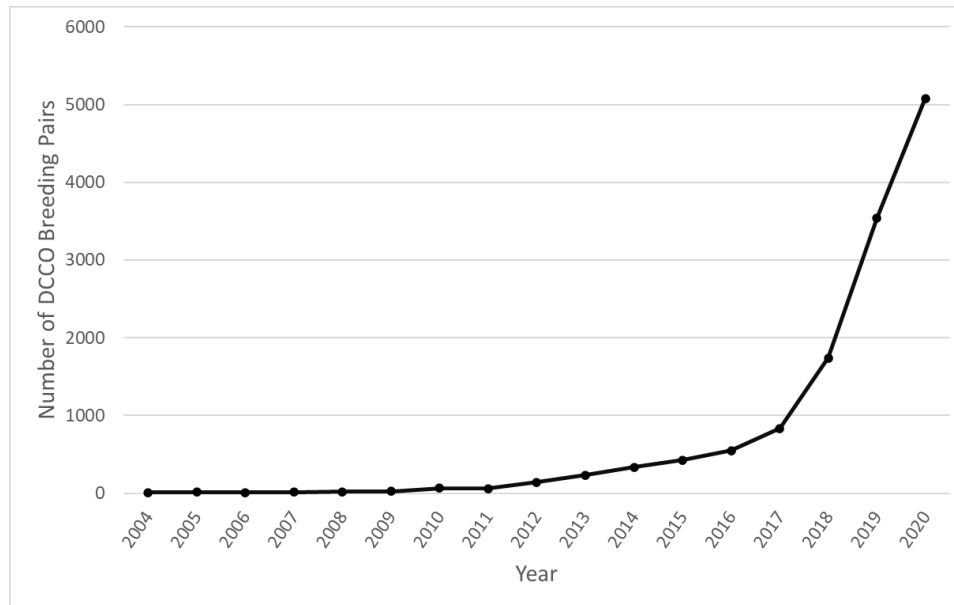
Figure 2. Size (number of breeding pairs) of the double-crested cormorant (DCCO) colony on the Astoria-Megler Bridge in the Columbia River estuary during 2004-2020. Data for 2015 provided by David Smith and Associates, Inc.; data for 2016 provided by Anchor QEA; data for 2019 provided by the U.S. Army Corps of Engineers; data for 2020 provided by the Oregon Department of Fish and Wildlife.

Detection of passive integrated transponder (PIT) tags that have been inserted into salmonid smolts, consumed by foraging piscivorous colonial waterbirds, and subsequently deposited on breeding colonies have been used as the primary method to assess predation impacts of DCCO in the CRE and elsewhere in the CRB (Collis et al. 2001, Ryan et al. 2003, Evans et al. 2012, Sebring et al. 2013, Evans et al. 2019b). PIT-tags recovered from piscivorous waterbird colonies can provide information specific to individual fish (e.g., species, population, date, and location of release and recapture) and have been used to estimate predation probabilities or rates. Methods to estimate avian predation rates, however, have been developed exclusively for ground-nesting colonies, where PIT-tag deposition probabilities (i.e., the proportion of

consumed tags deposited on-colony) and detection probabilities (i.e., proportion of deposited tags subsequently detected by researchers following the breeding season) can be readily quantified (see also Hostetter et al. 2015). PIT-tag detection probabilities for colonies of ground-nesting DCCO in the CRE (i.e., ESI, Rice Island, and Miller Sands Island), however, may differ considerably from those of DCCO colonies located on structures over water, whereby a large proportion of egested tags may be deposited in the water where they cannot be detected by researchers following the breeding season. Due to challenges in accessing DCCO colonies situated over water to recover PIT-tags and a lack of information on detection probabilities at those colonies, predation estimates are currently lacking for DCCO nesting on the AMB.

The primary objectives of this study were to (1) estimate the size of the DCCO colony on the AMB, (2) recover PIT-tags from a known number of DCCO nests located at the bridge colony, (3) quantify PIT-tag detection probabilities in those nests, and (4) estimate per capita (per breeding pair) and colony-wide (all breeding pairs) predation rates on juvenile salmonids by DCCO breeding on the AMB in 2021. Results from this work build upon previous tag recovery studies conducted at many different piscivorous waterbird colonies located in the CRE and elsewhere CRB (see Roby et al. 2021) and yield several important contributions to our understanding of predator-prey interactions in the CRE, namely mark-recapture-recovery methods used to estimate predation rates by piscivorous birds nesting on bridges and other elevated artificial structures.

METHODS

Study Area and Design

This study was conducted in 2021 at a DCCO breeding colony on the AMB, which contains the longest single truss span of any bridge in North America (Eskerod and Ang 2017; *Figure 3*). The entire 6.5 km of the bridge, connecting Oregon and Washington, was surveyed to determine colony attendance and colony size (i.e., number of breeding pairs) during the 2021 breeding season. While the CRE extends from the Columbia River mouth at Rkm 0 to the uppermost extent of tidal influence at Bonneville Dam at Rkm 234 (Simenstad et el. 2011), for the purpose of this study, we define the CRE in terms of three salinity zones where DCCO colonies have nested in the past, the (1) marine zone (Rkm 0 to Rkm 12), (2) mixing zone (Rkm 12 to Rkm 29), and (3) freshwater zone (Rkm 29 to Rkm 38). The AMB is located entirely over water in the mixing zone of the CRE (Rkm 23, see *Figure 1* above).
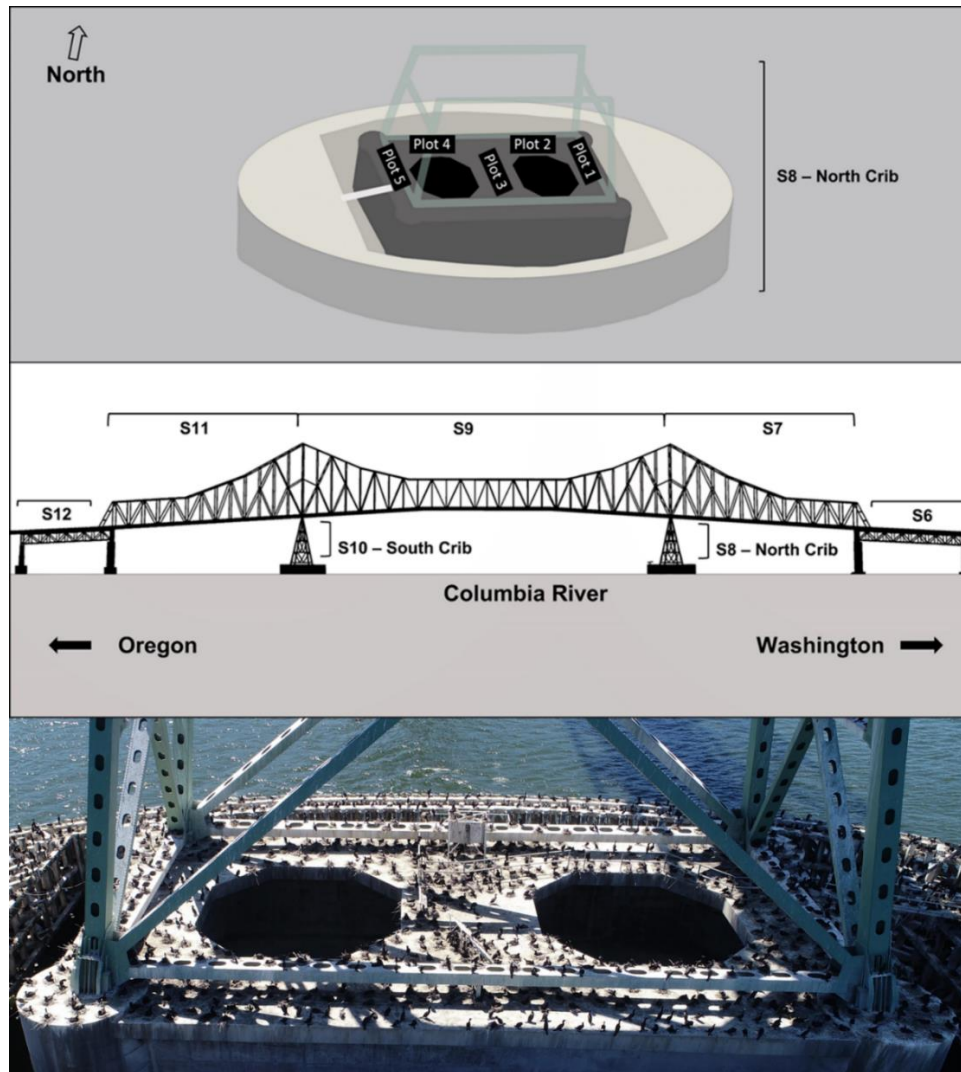
Figure 3. Locations of experimental plots (1-5) on the Astoria-Megler Bridge concrete platform (Top panel). Plots were used to enumerate double-crested cormorant nests during the breeding season and to recover smolt PIT-tags deposited by double-crested cormorants after the breeding season. The concrete platform was located on section S8 – North Crib (Middle panel) below the main truss of the Astoria-Megler Bridge. Aerial photo (Bottom panel) shows section S8 – North Crib during the peak double-crested cormorant nesting period on June 2, 2021.

Prior to the 2021 breeding season, we accessed a sub-section of the AMB, Section S8 (hereafter referred to as the "North Crib"; see *Figure 3* above), a protective structure around one of the support trestles where large numbers of DCCO have nested in previous years. Within the North Crib, we placed five discrete plots on the concrete platform previously used for DCCO nesting where birds could be enumerated during the breeding season and smolt PIT-tags deposited by DCCO could be recovered by researchers following the breeding season (see *Figure 3* above). PIT-tags were also intentionally sown by researchers prior to and following the breeding season

within each plot so that tag detection probabilities could be measured (see *Predation Rates* section below). Each plot measured 3 x 6 m and was located approximately 1 m from the water's edge of the concrete platform to maximize the probability of recovering tags deposited by DCCO nesting within each plot. Multiple plots were used as replicates to assess variation in per capita predation rates. Elsewhere on the AMB but not on the North Crib, Brandt's cormorants (*Phalacrocorax penicillatus;* BRAC) and pelagic cormorants (*Phalacrocorax pelagicus,* PECO) also nested in 2021; all cormorant species were separately enumerated during bridge-wide counts, but predation rates were only generated for DCCOs.

Colony Size

We used a combination of boat-based, fixed-wing aerial, and unmanned aircraft system (UAS) surveys to estimate colony size (i.e., number of breeding pairs) for all three cormorant species (DCCO, BRAC, and PECO) nesting on the AMB in 2021. We considered each nest with an attending adult cormorant to represent one breeding pair.

Boat-based surveys: During the 2021 breeding season, we conducted nest surveys by boat between low tide and high tide on April 20, June 2, and June 15-16. Two observers used binoculars to count cormorant nests in 22 discrete bridge sections (*Figure 4*) while moving parallel along the east and west sides of the bridge to ensure that all nesting cormorants present on the bridge were counted. For each section, we used the average of the two independent nest counts, by species (DCCO, BRAC, and PECO). If the two counts were not within 10 percent of each other for a section, the reason(s) for the discrepancy were discussed, and the count for that section was repeated.
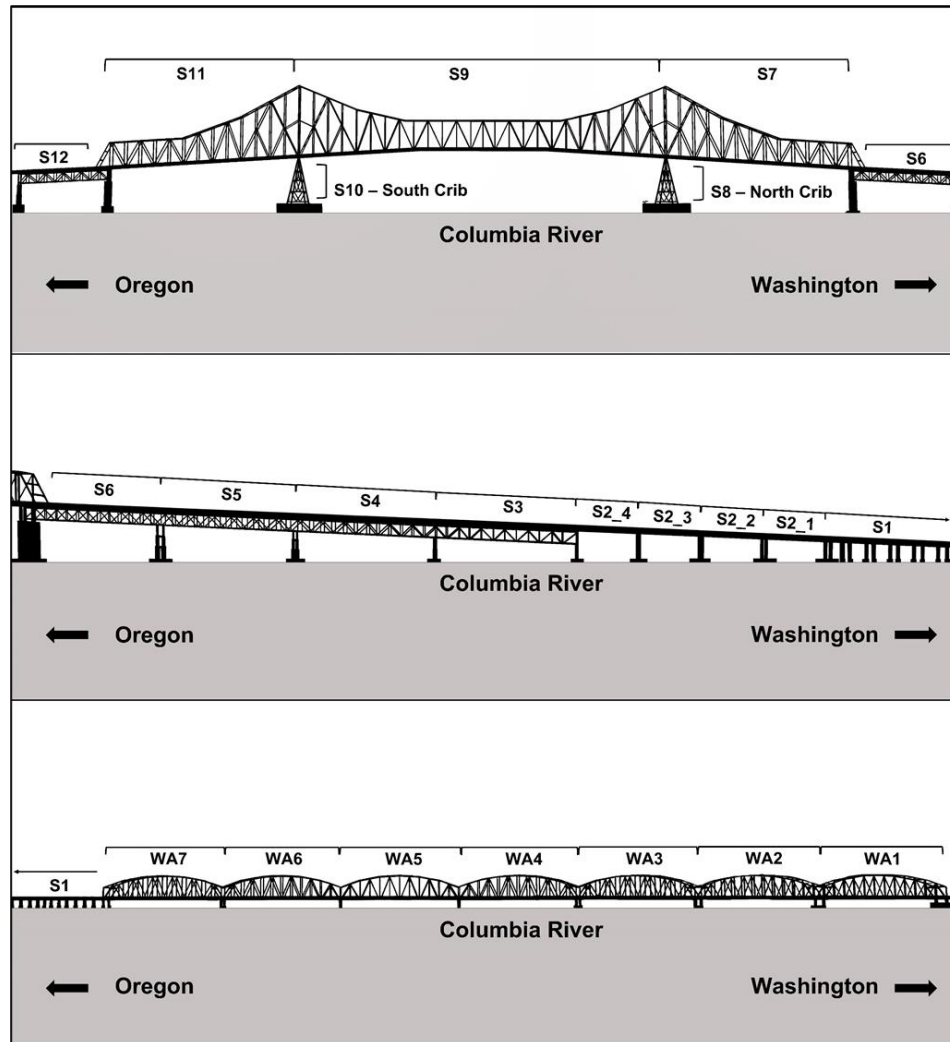
Figure 4. Sub-sections of the Astoria-Megler Bridge that were used to count double-crested cormorants, Brandt's cormorants, and pelagic cormorants across the entire span of the bridge in 2021. In total, the bridge spans 6.5 km, and the panels are not displayed to scale.

Aerial surveys: To supplement boat-based surveys and provide imagery for nest counts on sections of the bridge that were obstructed from our view from a boat, we conducted three fixed-wing and three UAS aerial surveys of the AMB during the breeding season. Fixed-wing surveys were carried out on May 18, June 4, and June 16 to obtain oblique imagery of nesting and roosting cormorants across the entire span of the AMB. During these flights, we used a 24-megapixel APSC DSLR camera with a 28-105 mm or 70-200 mm zoom lens while flying at low altitude (i.e., 150250 m above ground level [AGL]). UAS surveys were conducted on April 21, June 2, and June 15 to photograph nests on the North Crib to count nests within and adjacent to the experimental plots (see *above*). Oblique, UAS-derived imagery was acquired from between 20 and 30 m AGL.

Imagery from fix-wing and UAS aerial surveys were used to validate nest counts from boat surveys and to count nests of cormorant species on North and South cribs of the bridge (see *Figure 4* above). The UAS-derived imagery was also used to count nests within each of the five experimental plots on the North Crib. Nest counts for each plot were manually derived by inspecting the estimated plot location from multiple viewing angles (i.e., multiple photos) because the plots were in the interior areas of the concrete pad, and bridge infrastructure partially obscured portions of the plots from any single vantage point.

Peak colony counts: For each survey, nest counts from each section of the bridge were summed for a total nest count on the AMB by cormorant species (DCCO, BRAC, and PECO). The peak number of breeding pairs on the AMB in 2021 was determined from the highest bridge-wide count from three survey periods that had both a boat and aerial survey (fix-wing or UAS) on the same or consecutive days (April 19-20, June 2, and June 14-16). Only DCCO nested in the plots on the North Crib. Peak nest counts within the five plots and across the entire bridge colony occurred during the same survey period on June 2. The survey period identified as the peak in the plots on the North Crib was thus assumed to be the peak in nesting of all DCCOs on the AMB in 2021, which corresponded with the peak observed in previous years (Roby et al. 2021).

Predation Rates

Availability of PIT-tagged Smolts: The availability of PIT-tagged juvenile salmonids to predation by DCCO nesting on the AMB in 2021 was based on the number of tagged fish last detected alive while passing Bonneville Dam at Rkm 234 or the number of PIT-tagged fish released into tributaries downstream of Bonneville Dam, but upstream of the AMB in 2021. Availability was based on PIT-tagged smolts detected/released from March 1 to August 31, the period of overlap between active smolt out-migration and the breeding season for DCCO on the AMB and elsewhere in the CRE (Evans et al. 2012). PIT-tagged fish were grouped by species (Chinook salmon [*O. tshawytscha*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*], and steelhead trout [*O. mykiss*]) and age-class (sub-yearling [0] and yearling [1] for Chinook salmon only). Only groups with ≥ 4,000 PIT-tagged smolts were included in the predation analyses to increase the probability that the tags from those groups would be deposited in one of the five experimental plots and to avoid imprecision resulting from small sample sizes (Evans et al. 2012). Previous predation rate studies of smolts released in tributaries downstream of Bonneville Dam have shown that mortality occurs following release but prior to the fish reaching the mainstem Columbia River, which can affect the number of PIT-tagged smolts available to avian predators that nest at colonies in the CRE (Roby et al. 2021). As such, predation rates on smolts released into tributaries downstream of Bonneville Dam may represent minimum estimates of predation if large numbers of smolts did not survive out-migration to the mainstem Columbia River (see *below*). Release and recapture information for each PIT-tagged fish included in the study was retrieved from the PIT Tag Information System (PTAGIS), a regional, mark, recapture, and recovery database in the CRB (PSMFC 2021).

PIT-tag Recovery: The methods of Evans et al. (2012) were used to recover smolt PIT-tags that were deposited by DCCO nesting within each of the five plots located on the North Crib of the AMB immediately following the 2021 breeding season. To ensure equal effort within and across

each plot, we systematically scanned the entire area within each plot twice with hand-held PIT-tag antennas (Biomark, Boise, ID). All nests constructed on bridge support beams directly above each plot were also scanned and those nests were included in nest counts for the plot. PIT-tag codes recovered from each plot were uploaded to a central storage drive at the completion of scanning, along with metadata regarding the plot number, recovery date, and the scan number (1 or 2).

PIT-tag Deposition and Detection Probabilities: Not all PIT-tags that are ingested by colonial waterbirds are subsequently deposited on the nesting colony (Hostetter et al. 2015). For instance, a portion of PIT-tags consumed by DCCO are damaged and rendered unreadable following digestion or are regurgitated off-colony at loafing, staging, or other areas used by birds during the breeding season. Deposition probability (i.e., probability that a tag consumed by a nesting bird was deposited on its breeding colony) has been previously estimated by intentionally feeding PIT-tagged fish to ground-nesting DCCO on ESI with the proportion of known ingested tags subsequently recovered following the breeding season used to estimate deposition probabilities (see also Hostetter et al. 2015). The distribution of the median deposition probability derived from those studies was 0.51 (95% credible interval [CRI]= 0.34–0.70). Results from DCCO deposition studies indicated that deposition probabilities did not vary significantly within or between years, so deposition probabilities from these past studies were used to represent deposition probabilities of DCCO nesting within each of five plots located on the North Crib.

Not all PIT-tags deposited by DCCO on their nesting colony are subsequently detected by researchers after the nesting season (Ryan et al. 2003, Evans et al. 2012, Hostetter et al. 2015). For example, tags can be blown off the colony during windstorms, washed away during flooding events, or otherwise damaged or lost during the breeding season. Furthermore, the detection methods used to find PIT-tags on bird colonies are not absolute, as some proportion of detectable tags are missed by researchers during the scanning process due to tag collision (i.e., inability of the receivers to read tags in close proximity to one another) and perhaps other factors, like interference from ferrous material. Unlike deposition probabilities, detection probabilities can vary significantly within and between nesting seasons. This variability in detection probabilities necessitates a direct measure of detection probabilities for each colony, in each year (Evans et al. 2012, Hostetter et al. 2015, Evans et al. 2016b, Payton et al. 2019). To address this, PIT-tags with known tag codes were randomly sown (hereafter referred to as "control tags") within each of the five experimental plots prior to (March 25) and immediately following (September 11) the breeding season to quantify PIT-tag detection probability within each plot. Control tags were the same size and type as those used to mark juvenile salmonids from the CRB (12-mm [length] × 2-mm [width], full duplex). We used logistic regression to derive the probability of detecting control tags that are deposited at different times during the breeding season. Equal numbers of control tags were distributed before and after breeding (n = 125) and within each plot (n = 25) for a total sample size of 250 control tags to allow for direct comparisons of detection probabilities, with similar precision between plots and across time periods (see also Payton et al. 2019).

Predation Rate Estimates: We used the hierarchical Bayesian model of Hostetter et al. (2015) to estimate predation rates based on recoveries of smolt PIT-tags within each plot. Plot-specific predation rate estimates were calculated using the proportion of smolt PIT-tags found in each plot from the available number last detected passing Bonneville Dam or released downstream of Bonneville Dam (i.e., smolt availability), and then adjusting that rate by the probability that a consumed PIT-tag was subsequently deposited in a plot (i.e. deposition probability) and then detected by researchers in that plot following the breeding season (i.e. detection probability). Following the methodology of Hostetter et al. (2015), predation rates were modeled independently for each salmonid species (Chinook salmon, coho salmon, steelhead trout, and sockeye salmon), age-class (yearling, sub-yearling; for Chinook salmon only), and detection/release location (detected at Bonneville Dam, released downstream of Bonneville Dam). For each plot, the probability of recovering a PIT-tag from a smolt was modelled as the product of the three probabilities described above, the probability that (1) the fish was consumed (θ), (2) the PIT-tag was deposited on-colony (ϕ), and (3) the PIT-tag was detected on-colony after the breeding season (ψ_i):

$$k_i \sim \text{Binomial}(n_i, \theta_i * \phi * \psi_i)$$

where k_i is the number of smolt PIT-tags recovered from the number available (n_i) in week *i*. The detection probabilities (ψ_i) and predation probabilities (θ_i) were each modeled as a function of time. The probability, ψ_i, that a tag, consumed in week *i* and deposited on the colony is detected, is assumed to be a logistic function of week. That is:

$$\text{logit}(\psi_i) = \beta_0 + \beta_1 * i$$

where β_0 and β_1 are both derived from non-informative priors (normal [0, 1000]). Predation rates nearer together in time are more similar than those further apart in time (Payton et al. 2019). To reflect this, variation in weekly predation probabilities, θ_i, was modeled as a random walk process with mean μ_θ and variance σ_θ^2, where:

$$\text{logit}(\theta_i) = \mu_\theta + \sum_{w \leq i} \varepsilon_w$$

and $\varepsilon_w \sim normal\left(0, \ \sigma_\theta^2\right) \forall \ w$. We placed non-informative priors on these two hyperparameters: $\text{logit}^{-1}(\mu_\theta) \sim uniform(0,1)$ and $\sigma_\theta^2 \sim uniform(0,20)$. This allows each week (*i*) to have a unique predation probability (θ_i), while still sharing information among weeks to improve precision of the estimates.

Informative Beta priors were used to model deposition probability(ϕ). The shape parameters (α, β) are dependent on the predator species, and for DCCO, we assumed α = 15.98 and β = 15.29 (see also Hostetter et al. 2015).

Weekly plot-specific predation estimates were defined as the estimated number of PIT-tagged smolts consumed divided by the total number available each week. Annual plot-specific predation rates ($\dot{\theta}$) were derived as the sum of the estimated number of PIT-tagged smolts consumed each week divided by the total number of PIT-tagged smolts available:

$$\dot{\theta} = \frac{\Sigma_i\,(\theta_i * n_i)}{\Sigma_i(n_i)}.$$

Summation of weekly consumption estimates is necessary to accurately reflect weekly variation and autocorrelation of predation rates and thus to create unbiased annual rates with accurate assessments of precision (Payton et al. 2019).

For each plot (p), per capita (defined as per breeding pair) predation rates ($\dot{\theta}_p$) were calculated by dividing the species- and age-class specific predation rate by the peak measure of colony size (see *above*) within each plot:

$$\dot{\theta}_p = \frac{\Sigma_w\,(\theta_{wp} * n_{wp})/\Sigma_w\,(n_{wp})}{C_p}$$

where C_p is the peak colony size for plot p during the year. Colony-wide estimates of predation probabilities (θ_{AMB}) were calculated by multiplying the sum of the plot specific per capita predation rates by the peak measure of colony size for DCCO on the AMB (C_{AMB}).

$$\hat{\dot{\theta}}_{AMB} = C_{AMB} * \sum_p \dot{\theta}_p$$

Models were implemented using the software STAN accessed through R version 4.1.0 (RCDT 2021) using the rstan package (version 2.21.1; SDT 2020). To simulate random draws from the joint posterior distribution, we ran four Hamiltonian Monte Carlo Markov Chain processes. Each chain contained 4,000 warm-up iterations followed by 4,000 posterior iterations thinned by a factor of 4. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013); only chains with zero reported divergent transitions were accepted. Posterior predictive checks compared simulated and observed annual aggregate raw recapture and recovery numbers to ensure model estimates reflected the observed data. Reported estimates represent simulated posterior medians along with the 95% highest (posterior) density intervals (95% credible interval [CRI]).

Modelling Assumptions: Results of predation rate analyses were based on the following assumptions (see also Hostetter at al. 2015):

A1. Information from PTAGIS on releases and in-river interrogations (recapture histories) of PIT-tagged fish were complete and accurate.

A2. PIT-tagged fish detected/released in-river were available to birds nesting downstream (i.e., fish were actively migrating and available as prey).

A3. Predation, deposition, and detection probabilities for PIT-tagged fish were independent.

A4. Detection probabilities for PIT-tags sown by researchers in nesting plots were equal to those of PIT-tags naturally deposited by birds nesting in plots.

A5. Deposition probabilities for PIT-tags measured in previous years were equal to those of PIT-tags consumed by birds breeding on the AMB in 2021.

A6. PIT-tags from consumed fish were egested by DCCO on the AMB within a relatively short time period (one week) from when the PIT-tagged fish was last detected/release alive.

A7. Per capita predation rates for DCCO nesting within plots were representative of DCCO nesting elsewhere on the AMB.

To help meet the first assumption (A1), irregular entries in PTAGIS were either verified by the respective coordinator of the PIT-tagging effort or were censored from the analysis. Detections of PIT-tagged salmonids at recapture sites upstream of bird colonies is deemed the most appropriate measure of fish availability given the downstream movement of smolts, the ability to standardize data across sites, and the ability to define unique groups of salmonids by a known location and passage date (Assumption A2). Assumption A2 assumes all PIT-tagged fish last detected (recaptured) at Bonneville Dam or captured and released downstream of Bonneville Dam but upstream of the AMB were alive and thus available as prey to DCCO nesting on the AMB. If large numbers of fish halted their out-migration or died prior to reaching the foraging range of DCCO on the AMB, predation rates would be biased low. Smolts detected passing Bonneville Dam were actively migrating fish originating from populations further upstream. Some unknown proportion of coho salmon and steelhead that were captured, tagged, and released into tributaries downstream of Bonneville Dam, however, may not have been actively migrating in 2021 and/or did not survive migration to the mainstem Columbia River downstream of Bonneville Dam. As such, predation rates on coho salmon and steelhead released into tributaries of the lower Columbia River were potentially biased low to an unknown degree relative to actively migrating smolts last detected passing Bonneville Dam. The fate of each PIT-tag implanted in a smolt is assumed to be independent (A3). Lack of independence among PIT-tagged fish could potentially bias predation probabilities and overinflate measures of precision (i.e., credibility intervals). Detection probability estimates (A4) were generally high with little variation between plots (see *Results*) suggesting that any possible violations of assumption A4 would have little effect on estimates of predation rates. Data on deposition rates collected in previous years (when multiple estimates of deposition rates were measured over the course of different time periods both within and across years) showed no evidence of inter- or intra-annual trends in deposition probabilities (Assumption

A5). Assumption A6 relates to the use of the last date of live fish recapture as a proxy for the date a PIT-tag was deposited on a bird colony. This assumption needs to be only roughly true because detection efficiency did not change dramatically on a weekly basis. Finally, significant differences in per capita predation rates between birds nesting in plots on the North Crib compared with birds nesting elsewhere on the bridge could bias colony-wide estimates of predation probabilities to an unknown degree (A7). There was no evidence that per capita predation rates differed significantly by plot on the North Crib (see *Results*), nor did the nesting chronology of birds in the plots differ from birds nesting elsewhere on the bridge, suggesting extrapolation of per capita predation rate estimates from plots to birds nesting elsewhere on the bridge was reasonable.

Per capita predation rates at the species level for DCCO nesting on ESI from 2003 to 2014 and the AMB in 2021 were compared to investigate predation rates in relation to colony location in the marine and mixing zones, respectively. Colonies at these locations in these years were unmanaged, and per capita predation rates were annually adjusted for availability of PIT-tagged smolts and detection probabilities. Predation rates on ESI have been conducted over multiple years with averages presented in this report. Numbers of available smolts used in previous predation rate analyses for ESI are presented in Roby et al. (2021), Appendix B.

RESULTS

Colony Sizes

We estimated the peak colony size of DCCO on the AMB at 4,151 breeding pairs on June 2, 2021. Of the 4,151 breeding pairs, the peak count of DCCO nesting in plots on the North Crib was 159 breeding pairs, with the number varying by plot (see *below*). We estimated the peak colony size of BRAC nesting on the AMB at 955 breeding pairs on June 2. We estimated the peak colony size of PECO nesting on the AMB at 82 breeding pairs on June 16.

Predation Rates

PIT-tag Recovery: A total of 346 smolt PIT-tags were recovered from the five plots on the North Crib following the 2021 breeding season (*Table 1*). Of the salmonid species, most tags were from Chinook salmon (n = 224, yearling and sub-yearling combined), followed by steelhead trout (n = 95), coho salmon (n = 17), and sockeye salmon (n = 9). The number of PIT-tagged smolts recovered varied by plot (range = 23–40 breeding pairs/plot; *Table 1*). Numbers of PIT-tags from smolts recovered in plots were commensurate with the number of available tagged smolts, whereby larger sample sizes of available tagged smolts corresponded with larger numbers of tags recovered on the AMB.

Table 1. Nest count of double-crested cormorants, number of smolt PIT-tags recovered by salmonid species and age class (sub-yearling [0] and yearling [1], Chinook salmon only), and average PIT-tag detection probability (range during the breeding season) by plot on the North Crib of the Astoria-Megler Bridge in 2021.

Plot	Nest Count	Chinook 1	Chinook 0	Steelhead	Coho	Sockeye	Detection Probability
1	35	45	13	17	2	1	0.86 (0.72–0.99)
2	40	18	8	11	1	2	0.60 (0.56–0.64)
3	23	34	12	25	3	4	0.80 (0.80–0.80)
4	31	35	18	21	6	2	0.66 (0.48–0.84)
5	30	31	10	22	5	0	0.86 (0.80–0.92)
All	**159**	**163**	**61**	**96**	**17**	**9**	**0.76 (0.67–0.84)**

PIT-tag Detection and Deposition Probabilities: The detection probabilities varied both by plot and when the tag was deposited/sown (i.e., at the beginning or end of the breeding season; *Table 1*). Tags deposited at the beginning of breeding season were less likely to be recovered relative to those deposited at the end of breeding season, except for plot 3, where detection probabilities were consistent across the breeding season. We used a deposition probability for DCCO nesting in all five plots of 0.51 (95% credible interval [CRI] = 0.34–0.70) based on previously published estimates of deposition probabilities for DCCO nesting on ESI (see Hostetter et al. 2015 for additional details).

The number of PIT-tagged smolts that were available was adequate to generate predation rates (see *below*) for Chinook salmon (n = 60,071 for yearling and sub-yearling combined), steelhead trout (n = 45,039), and sockeye salmon (n = 4,841) based on fish that were last detected (recaptured) passing Bonneville Dam. There was also an adequate sample of available fish for steelhead trout (n = 9,971) and coho salmon (n = 6,953) released downstream of Bonneville Dam in 2021 (see *below*). Smolts detected at Bonneville Dam were a combination of fish from ESA-listed and non-listed populations and a combination of hatchery- and wild-origin smolts. All smolts released downstream of Bonneville Dam, however, were wild and ESA-listed (NOAA 2021). The sample size of PIT-tagged Chinook salmon (n = 892 for yearling and sub-yearling combined) released downstream of Bonneville Dam, however, was inadequate for predation rate analysis.

Per Capita Predation Rates: For smolts last detected at Bonneville Dam, results indicated that there was some variation in the estimated per capita predation rates by plot, but 95% CRIs overlapped indicated results were not statistically significant (*Figure 5*). Per capita predation rates were highest for sub-yearling Chinook salmon at 0.0040% (0.0028–0.0069) and lowest for sockeye salmon at 0.0013% (0.0001–0.0031, *Table 2*). Due to smaller sample sizes of available PIT-tagged sub-yearling Chinook salmon and sockeye salmon smolts (< 10,000 tagged smolts), per capita predation rates were less precise compared to those from other stocks. For

steelhead and coho salmon released below Bonneville Dam, per capita predation rates were estimated at 0.0013% (0.0004–0.0025) and 0.0015% (0.0004–0.0031), respectively (*Table 2*).
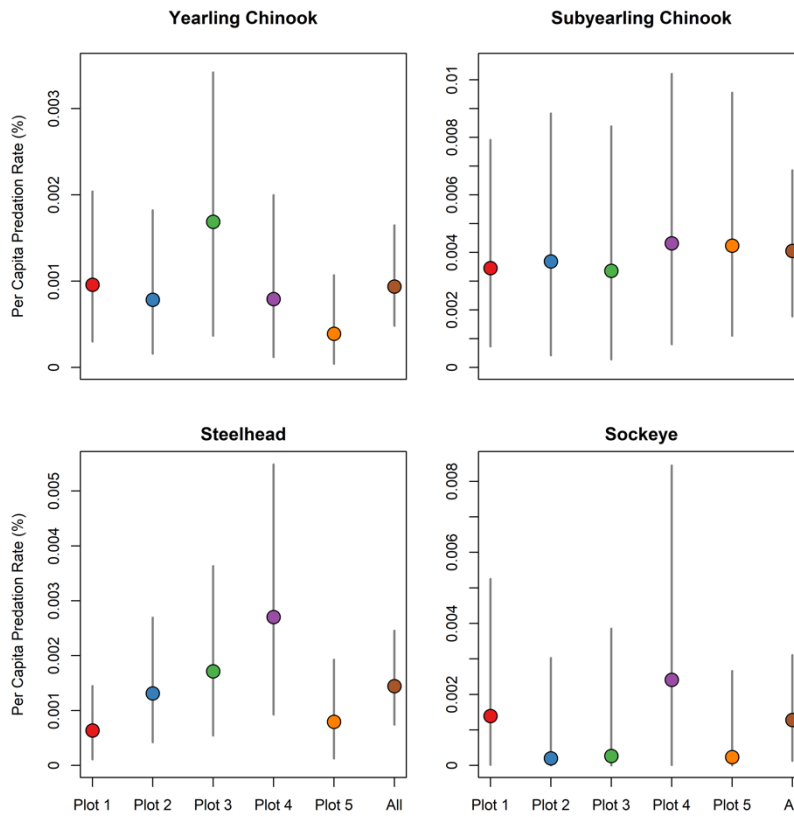
Figure 5. Estimated per capita predation rates by plot (1-5) and across all plots (All) by salmonid species and age class (Chinook salmon only).

Colony-wide Predation Rates: For smolts last detected at Bonneville Dam, colony-wide estimates of predation ranged from 3.9% (2.0–6.8) for yearling Chinook salmon to 16.8% (7.3–28.4) for sub-yearling Chinook salmon (*Table 2*). For smolts released downstream of Bonneville Dam, colony-wide predation rates ranged from 5.5% (1.8–10.5) for steelhead to 6.1% (1.6–12.7) for coho salmon (*Table 2*).

Table 2. Estimates of per capita (per breeding pair) and colony-wide predation rates (95% credible intervals) by double-crested cormorants nesting on the Astoria-Megler Bridge (AMB) in 2021 and on East Sand Island (ESI) during 2003-2014 on yearling Chinook salmon (1), sub-yearling Chinook salmon (0), all Chinook salmon (yearling and sub-yearling combined), sockeye salmon, coho salmon, and steelhead. Smolts were last detected (recaptured) passing Bonneville Dam or were released downstream of Bonneville Dam. ESI predation rates only include ESA-listed salmonid species, while AMB predation rates include ESA-listed and non-listed species (see Methods). [NA = not available]. Estimate of per capita and colony-wide predation of double-crested cormorants nesting on ESI are those of Lawes et al. (2021).

Detection/Release Location	Salmonid Species	Per Capita Predation Rate		Colony-wide Predation Rate	
		AMB (2021)	ESI (2003-2014)	AMB (2021)	ESI (2003-2014)
At Bonneville Dam	Chinook 1	0.0009% (0.0005–0.0016)	NA	3.9% (2.0–6.8)	NA
	Chinook 0	0.0040% (0.0028–0.0069)	NA	16.8% (7.3–28.4)	NA
	All Chinook	0.0014% (0.0008–0.0024)	0.0003% (0.0003–0.0004)	5.8% (3.1–9.9)	3.7% (3.2-4.4)
	Steelhead	0.0014% (0.0007–0.0025)	0.0006% (0.0005–0.0007)	6.0% (3.1–10.2)	6.9% (6.0–8.3)
	Sockeye	0.0013% (0.0001–0.0031)	0.0003% (0.0002–0.0004)	5.3% (0.5–12.9)	4.2% (3.3–5.3)
Downstream of Bonneville Dam	Chinook 1	NA	NA	NA	NA
	Chinook 0	NA	NA	NA	NA
	All Chinook	NA	0.0029% (0.0026–0.0033)	NA	27.5% (24.3–30.7)
	Steelhead	0.0013% (0.0004–0.0025)	0.0007% (0.0006–0.0008)	5.5% (1.8-10.5)	5.4% (4.5–6.3)
	Coho	0.0015% (0.0004–0.0031)	0.0010% (0.0008–0.0012)	6.1% (1.6-12.7)	15.0% (12.2–18.1)

AMB and ESI Predation Rate Comparisons: We estimated that, for smolts last detected at Bonneville Dam, per capita predation rates were approximately 2 to 4 times greater (depending on the species) for DCCO nesting on the AMB in 2021 compared to average annual per capita predation rates by DCCO nesting on ESI during 2003-2014 (Lawes et al. 2021, see *Table 2* above). Whereas, for smolts released below Bonneville Dam, we estimated that the per capita predation rates were approximately 2 times greater for DCCO nesting on the AMB in 2021 compared to average annual rates by DCCO nesting on ESI during 2003-2014 (Lawes et al. 2021). Predation rate estimates for smolts released downstream of Bonneville Dam by DCCO on the AMB in 2021, however, may be biased low to an unknown degree relative to DCCO on ESI. This is because estimates from the AMB were not adjusted to account for smolt survival to the mainstem Columbia River, and estimates were restricted to wild-origin smolts only (see *Methods*). Although we recovered PIT-tags from Chinook smolts released downstream of Bonneville Dam on the AMB in 2021, we were unable to make the relative comparisons of predation between cormorants breeding on the AMB and ESI for due to an inadequate sample size of tagged fish from this stock (see *Table 2* above).

Colony-wide predation rate estimates were similar between DCCO nesting on the AMB in 2021 compared with DCCO nesting on ESI during 2003-2014, with the except of coho salmon released below Bonneville Dam, where estimates were higher by DCCO nesting on ESI (see *Table 2* above). Colony size at the AMB in 2021 (n = 4,151 breeding pairs) was approximately one-third the size of the average colony size on ESI prior to management actions on ESI (n = 12,787 breeding pairs during 2003 – 2014; Lawes et al. 2021). Results suggest that despite the smaller size of the AMB colony, predation rate estimates by DCCO on juvenile salmonids in the CRE have remained similar before and after management due to the higher per capita predation rates of DCCO nesting on AMB compared to DCCO nesting on ESI.

DISCUSSION

Piscivorous colonial waterbird species nest in a variety of habitat types in the CRE and elsewhere in North America (Lawes et al. 2021). Of these species, DCCO are particularly flexible in the type of nesting habitat that they can successfully exploit, capable of forming ground, cliff, and arboreal nesting colonies, as well as colonies on artificial structures such as bridges, navigational markers, and transmission towers (Hobson et al. 1989, Carter et al. 1995, Atkins et al. 2014). The methods developed in this study used estimates of per capita (per breeding pair) predation rates from a sub-sample of DCCO breeding pairs on the AMB to evaluate the colony-wide predation impacts of DCCO nesting on the AMB in 2021. Through a combination of colony monitoring techniques (boat, fixed-wing, and UAS), peak nest counts were obtained within nesting plots and across the entire bridge nesting colony. Despite most of the breeding habitat for DCCO being inaccessible to researchers, our sampling plots targeted DCCOs nesting on a subsection of the bridge where all three processes associated with fish tag recovery on avian colonies (i.e. consumption, deposition, and detection probabilities) could be measured to generate unbiased estimates of predation (Hostetter et al. 2015). Cause-specific mortality due to DCCOs breeding on the AMB, the largest DCCO colony in the western population in recent

years (Lawes et al. 2021), provide relevant information to (1) evaluate the efficacy of ongoing management plans and (2) adaptively manage those plans to maximize the survival benefits to ESA-listed juvenile salmonids derived from managing DCCO in the CRE.

Previously studies have demonstrated that the diet composition of DCCO nesting in the CRE is related to colony location, with DCCO at colony sites located in the freshwater zone of the CRE consuming more juvenile salmonids on a per capita basis than those at colonies sites in the marine zone of the CRE (Collis et al. 2002, Cramer et al. 2021). While the linear distance between the AMB and ESI is only 15 Rkm, this study presents evidence that DCCO breeding at the AMB in 2021, in the mixing zone of the CRE, consume more juvenile salmonids on a per capita basis than those breeding at ESI in the marine zone of the CRE during 2003-2014. Colony-wide predation impacts were relatively similar between ESI and the AMB, although colony size at the AMB in 2021 was approximately one-third the size of the average colony size on ESI prior to management (Lawes et al. 2021); a finding consistent with results and conclusions drawn from other related studies in the CRE. For example, previous studies investigating the diet composition and foraging behavior of piscivorous colonial waterbirds nesting concurrently on both Rice Island and ESI found that juvenile salmonids were significantly more prevalent in the diet of DCCO, Caspian terns (*Hydroprogne caspia*), and gulls (*Larus* spp.) nesting at colonies in the freshwater zone versus the marine zone of the CRE (Collis et al. 2002, Roby et al. 2002, Lyons et al. 2005). As the data presented here are based on a one-year study at the AMB and lack comparisons of per capita predation rates at sites within different salinity zones within the same year, conclusions from this study should be interpreted within this more limited scope.

In addition to colony location, colony size contributes to the impact of predation by DCCO on the survival of out-migrating juvenile salmonids in the CRE (Collis et al. 2002, Evans et al. 2012, Hostetter et al. 2021). While ongoing management actions as part of the Cormorant Management Plan in the CRE have succeeded at reducing the size of the DCCO breeding colony on ESI to less than 5,939 breeding pairs, the number of breeding DCCO on the AMB in 2021 was the second highest to date for the site at 4,151 breeding pairs. Many, if not most, of the DCCO that abandoned the ESI colony are believed to have renested on the AMB. Reasons for the decline in colony size on the AMB in 2021 compared to 2020 are uncertain but could be due to maintenance activities on the bridge in April and June in 2021 that displaced some birds, and/or greater interest by DCCO in 2021 to nest in areas on the bridge that were not visible from the water or air (i.e., inside beams and girders).

Minimizing dispersal to new colony sites in the CRE and elsewhere in the CRB was an important component of the Cormorant Management Plan in the CRE (USACE 2015), but commensurate with the precipitous decline in the size of the ESI DCCO colony during management, colony size for DCCO at other colonies have increased, namely DCCO nesting on artificial structures such as the AMB, navigational aids in the upper CRE, the Lewis and Clark Bridge near Longview, WA, and transmission towers near Troutdale, OR. The factors limiting and contributing to the size of the DCCO colonies on these artificial structures has not received much research attention, and in the absence of a colony on ESI, whether these colonies will continue to increase in future years remains to be seen. The formation of DCCO nesting colonies on bridges in the Strait of

Georgia, British Columbia, Canada has been hypothesized to provide refuge from bald eagle (*Haliaeetus leucocephalus*) disturbances that were associated with colony declines at less protected nesting locations (Carter et al. 2018); bald eagle disturbances have also been noted in recent years at the ESI DCCO colony (Adkins et al. 2014, Lawes et al. 2021). Based on the results from this study, monitoring of DCCO colonies on artificial structures in the CRB should help to assess whether the benefits of management of the ESI DCCO colony have been offset (or surpassed) by the impacts of DCCO at other incipient or growing historical colony locations within the CRB.

Although we relied on a sample-based approach to estimate colony-wide predation rates, results were consistent with other published studies of DCCO predation that indicate PIT-tag consumption estimates vary by salmonid species and age-class, even after accounting for differences in colony location (Collis et al. 2002, Evans et al. 2012, Sebring et al. 2013, Cramer et al. 2021). We found estimates of predation to be the greatest for sub-yearling Chinook salmon relative to yearling Chinook salmon, coho salmon, sockeye salmon, and steelhead in the CRE during our 2021 study period. Chinook salmon are also the most abundant species of salmonid smolt in the CRE, and therefore the most heavily depredated salmonid species by generalist predators such as DCCO (Sebring et al. 2013, Dorr et al. 2014). The higher predation rate for sub-yearling Chinook salmon in the CRE may also be due, in part, to a more prolonged residency of sub-yearling Chinook salmon in the estuary compared to other age-classes (i.e., yearling) and species (Bottom et al. 2011, Roegner et al. 2012, Sebring et al. 2013, McNatt et al. 2016). Chinook salmon that originate from tributaries downstream of Bonneville Dam have also been shown to be particularly vulnerable to DCCO predation in the CRE (Sebring et al. 2013, Lawes et al. 2021), possibly as a result of their extended use of wetland side-channels as rearing habitat, with colony-wide predation rate estimates on lower Columbia River Chinook salmon by DCCO nesting at ESI in excess of 40% of available fish in some years (Sebring et al. 2013, Lawes et al. 2021). In this study, however, there were inadequate sample sizes of PIT-tagged lower Columbia River Chinook salmon for inclusion in predation rate analyses involving the AMB. As such, this data gap remains a critical uncertainty as millions of both hatchery-reared and wild ESA-listed Chinook salmon originate from the lower Columbia River and its tributaries annually, few of which are PIT-tagged (PSFMC 2021).

The efficacy of the Cormorant Management Plan in the CRE (USACE 2015) must account for the per capita and colony wide predation impact for DCCO nesting on the AMB, currently the largest colony in the Western population. While differences in per capita predation by DCCO nesting in the freshwater, mixing, and marine zones of the CRE are interesting from an ecological perspective, fisheries managers need to understand whether any significant reductions in predation by DCCO on out-migrating smolts were achieved following implementation of management on ESI beginning in 2015. Estimates of colony-wide predation probabilities by DCCO breeding on the AMB in 2021 were similar to those of DCCO nesting on ESI prior to management, despite the smaller size of the bridge colony (4,151 breeding pairs) as compared to ESI (average 12,787 breeding pairs 2003 – 2014, Lawes et al. 2021). Given this, impacts of DCCO on smolt survival throughout the CRE may be unchanged since management actions on ESI were first implemented in 2015. It should be noted, however, that annual

estimates of DCCO predation on juvenile salmonids in the CRE are highly variable (Evans et al. 2012, Evans et al. 2019b, Lawes et al. 2021), so estimates from this study for the AMB (i.e. from one colony, in one year) should be viewed cautiously.

This study highlights the importance of predation by DCCO breeding on artificial structures to evaluate avian predation in the CRE and to better understand the system-wide, cumulative effects of predation by multiple piscivorous colonial waterbirds in the CRB. As DCCO colonies have expanded and new artificial structures have been colonized in the CRE, other opportunities exist to further examine predation impacts in response to the Cormorant Management Plan with methods modelled from this study. For instance, while there are approximately 15 channel markers in the CRE (Rkm 0-40) historically used by nesting DCCO, future predation studies could focus on several markers for detection efficiency and per capita estimates in the freshwater, mixing, and marine zones and then extrapolate results to all channel markers that are colonized in each respective zone. Outside of the CRB, DCCO in the Western population have established nesting colonies on bridges and other anthropogenic structures in other Oregon estuaries, the San Francisco Bay, and the Strait of Georgia British Columbia, Canada (Rauzon et al. 2001, Atkins et al. 2014, Carter et al. 2018, Rauzon et al. 2019). To our knowledge, predation estimates by DCCO nesting at these locations have not been conducted, and thus, their impact on fish populations of conservation concern are unknown. Limitations on avian predation analyses outside of the CRB could also be the result of inadequate sample sizes of PIT-tagged fish. The growing colony of BRAC on the AMB (955 breeding pairs in 2021) is likely another consequence of recent management actions on ESI, with BRAC that formerly nested on ESI moving to the AMB to renest. While predation probabilities on smolts by BRAC on ESI were consistently less than 1% of available smolts (Evans et al. 2021), it is unknown if that is the case for BRAC nesting on AMB.

In most years, except for 2020, the National Marine Fisheries Services has operated a pair trawl PIT-tag detection system in the lower CRE downstream of Bonneville Dam (between Rkms 75-85) to collect data regarding the number of PIT-tagged smolts that survive out-migration to Bonneville Dam (Rkm 234; Holcombe et al. 2019). Each year, however, thousands of PIT-tags are also recovered on avian colonies in the CRE within 40 km of the mouth of the Columbia River and primarily used for predation assessments (Lawes et al. 2021, Cramer et al. 2021). Tag recoveries on avian colonies in the CRE, however, can increase the accuracy and precision in mark-recapture survival models for smolt survival in the CRB by providing an additional or alternative source of data (Hostetter et al. 2018), as indicated by > 6,000 current year migration PIT-tags recovered at nesting and roosting sites throughout the CRE in 2020 (Collis et al. 2021). Using detections of smolt PIT-tags on bird colonies in the CRE, Payton et al. (2019) demonstrated that these tags can also be used to determine how many smolts survived outmigration to Bonneville Dam to jointly estimate predation and survival, providing a better understanding of predation and survival across space and time. Based on the amount of potentially suitable nesting habitat, it is estimated that the AMB can support a colony of approximately 10,000 DCCO breeding pairs (Lawonn in prep.). Discussions amongst resource managers are underway about future efforts to dissuade DCCO from nesting on the AMB to reduce predation impacts on out-migrating salmonid smolts and protect the bridge structures

from corrosion caused by guano. In addition, bridge maintenance activities are planned in 2022 (J. Lawonn, ODFW, pers. comm.), which could impact birds attempting to use the bridge for nesting. Ongoing monitoring efforts should help clarify the effects of construction disturbances on the DCCO colony size at the AMB and provide insights into the potential effectiveness of any future dissuasion efforts at that colony.

Enumerating DCCO nests within our plots on the North Crib was difficult and time consuming because of visual obstructions caused by bridge structures located above the crib, a methodological issue that we did not adequately account for while establishing the plots prior to the breeding season. We recommend that future monitoring consider using larger plots capable of containing a larger number of nesting DCCO and selecting plot placements that are less obscured by bridge supports. Larger plots may also provide the opportunity to recover a greater number of PIT-tags from the accessible areas of the AMB, which would help improve the accuracy and precision of per capita predation estimates. While we only used nests that we could visibly confirm for peak colony counts and predation estimates, we also observed some unknown number of DCCO that appeared to be nesting inside the beams of the bridge which we could not count accurately. As noted above, this may have biased our colony-wide counts low. We suggest more frequent surveys during the suspected peak breeding period and additional attention to beam-nesting DCCO to increase the accuracy of the colony-wide counts. Finally, continued monitoring efforts should help clarify the effects of construction disturbances on the DCCO colony size at the AMB and provide insights into the potential effectiveness of any future dissuasion efforts at that colony.

Conclusions

Results from this study suggest that colony-wide predation rates by the DCCO colony on the AMB in 2021 were similar to those of DCCO nesting on ESI prior management, despite the AMB colony being approximately one-third the size of the average pre-management ESI colony. As such, the Cormorant Management Plan has not reduced predation by DCCO on juvenile salmonids throughout the entire CRE as planned. Further adaptive management to address the recent shift of breeding DCCO from ESI to AMB are necessary if the management objectives are to be met. As a first step, relocating nesting DCCO from the AMB back to ESI, where their per capita predation impacts are less, would greatly reduce the impacts of DCCO on smolt survival in the CRE. Colony relocation can likely be achieved by dissuasion of DCCO nesting on the AMB combined with nesting substrate enhancement and social attraction at the former colony site on ESI (Lawes et al. 2021). If the resulting DCCO colony at ESI surpasses the target colony size stipulated in the management plan, further adaptive management may be necessary at that site.

LITERATURE CITED

Adkins, J.Y., D.E. Lyons, P.J. Loschl, D.D. Roby, K. Collis, A.F. Evans, and N.J. Hostetter. 2014. Demographics of piscivorous colonial waterbirds and management implications for ESA-listed salmonids on the Columbia Plateau. Northwest Science 88:344-359.

Anderson, C.D., D.D. Roby, and K. Collis. 2004. Foraging patterns of male and female double-crested cormorants in the Columbia River estuary. Canadian Journal of Zoology 82:541-554.

Bottom, D.L., A.M. Baptista, J. Burke, L.A. Campbell, E. Casillas, S.A. Hinton, D.A. Jay, M. Lott, G.T. McCabe Jr., R.A. McNatt, M. Ramirez, G.C. Roegner, C.A. Simenstad, S. Spilseth, L. Stamatiou, D.J. Teel, and J.E. Zamon. 2011. Estuarine habitat and juvenile salmon: current and historical linkages in the lower Columbia River and estuary. Final Report by the National Marine Fisheries Service to the U.S. Army Corps of Engineers – Portland District, Portland, Oregon.

Carter, H.R., A.L. Sowls, M.S. Rodway, U.W. Wilson, R.W. Lowe, G.J. McChesney, F. Gress, and D.W. Anderson. 1995. Population size, trends, and conservation problems of the double-crested cormorant on the Pacific coast of North America. Colonial Waterbirds 18:189-215.

Carter, H.R., T.A. Chatwin, and M.C. Drever. 2018. Breeding population sizes, distributions, and trends of pelagic, double-crested, and Brandt's cormorants in the Strait of Georgia, British Columbia, 1955–2015. Northwestern Naturalist 99(1):31-48.

Collis, K., D.D. Roby, D.P. Craig, B.A. Ryan, and R.D. Ledgerwood. 2001. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary: Vulnerability of different salmonid species, stocks, and rearing types. Transactions of American Fisheries Society 130:385-396.

Collis, K., D.D. Roby, D.P. Craig, S.K. Adamany, J.Y. Adkins, and D.E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Collis, K., A.F. Evans, D.D. Roby, J. Tennyson, A. Turecek, Q. Payton, and T. Lawes. 2021. Avian Predation in the Columbia River Basin: 2020 Final Annual Report. Annual Report to Bonneville Power Administration (Contract No. 60846, Project No. 1997-024-00) and Grant County Public Utility District and the Priest Rapids Coordinating Committee (Agreement No. HFA 601-30H).

Cramer, B., A.F. Evans, Q. Payton, K. Collis, and D.D. Roby. 2021. Chapter 5: Relative impacts of double-crested cormorants and Caspian terns on juvenile salmonids in the Columbia River estuary: a retrospective analysis of the PIT-tag data *in* D. D. Roby, A. F. Evans, and K. Collis, editors. Avian predation on salmonids in the Columbia River basin: a synopsis of ecology and management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Dorr, B.S., J.J. Hatch, and D.V. Weseloh. 2014. Double-crested cormorant (*Phalacrocorax auritus*), version 2.0. *In* The Birds of North America (A. F. Poole, editor). Cornell Lab of Ornithology, Ithaca, NY.

Eskerod, P. and K. Ang. 2017. Stakeholder value constructs in megaprojects: A long-term assessment case study. Project Management Journal 48(6): 60-75.

Evans, A.F., N.J. Hostetter, D.D. Roby, K. Collis, D.E. Lyons, B.P. Sandford, R.D. Ledgerwood, and S. Sebring. 2012. Systemwide evaluation of avian predation on juvenile salmonids from the Columbia River based on recoveries of passive integrated transponder tags. Transactions of the American Fisheries Society 141:975-989. doi: 10.1080/00028487.2012.676809.

Evans, A.F., D.A. Hewitt, Q. Payton, B.M. Cramer, K. Collis, and D.D. Roby. 2016a. Colonial waterbird predation on Lost River and Shortnose suckers in the Upper Klamath Basin. North American Journal of Fisheries Management 36:1254-1268.

Evans, A.F., Q. Payton, A. Turecek, B. Cramer, K. Collis, D.D. Roby, P.J. Loschl, L. Sullivan, J. Skalski, M. Weiland, C. Dotson. 2016b. Avian predation on juvenile salmonids: spatial and temporal analysis based on acoustic and passive integrated transponder tags. Transactions of the American Fisheries Society 145: 860–877.

Evans, A.F., Q. Payton, B.M. Cramer, K. Collis, and J. Tennyson. 2019a. East Sand Island passive integrated transponder tag recovery and avian predation rates analysis, 2018. Final Technical Report submitted to the U.S. Army Corps of Engineers in partial fulfillment of Contract No. W912EF-14-D-0004, Order No. W9127N18F0120, U.S. Army Corps of Engineers, Portland, Oregon. 50 pp.

Evans, A.F., Q. Payton, B.M. Cramer, K. Collis, N.J. Hostetter, D.D. Roby, and C. Dotson. 2019b. Cumulative effects of avian predation on Upper Columbia River steelhead. Transactions of the American Fisheries Society 148:896-913.

Evans, A.F., Q. Payton, K. Collis, B. Cramer, A. Turecek, N.J. Hostetter, and D.D. Roby. 2021. Chapter 7: Cumulative effects of avian predation on juvenile salmonids in the Columbia River basin in D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 789 pp.

Gelman, A., J.B. Carlin, H.S. Stern, D.B. Dunson, A. Vehtari, and D.B. Rubin. 2013. Bayesian Data Analysis (3rd ed.). Chapman and Hall/CRC, Boca Raton, Florida.

Hobson, K.A., R.W. Knapton, and W. Lysack. 1989. Population, diet and reproductive success of double-crested cormorants breeding on Lake Winnipegosis, Manitoba, in 1987. Colonial Waterbirds:191-197.

Holcombe, E.F., A.J. Borsky, J.M. Biron, P.J. Bentley, and B.P. Sandford, 2019. Detection of PIT-Tagged Juvenile Salmonids Migrating in the Columbia River Estuary, 2018. Report submitted to Bonneville Power Administration in partial fulfillment of Contract 46273 RL58, Project 1993-029-00, Portland, Oregon. 55 pp.

Hostetter, N.J., A.F. Evans, B.M. Cramer, K. Collis, D.E. Lyons, and D.D. Roby. 2015. Quantifying avian predation on fish populations: Integrating predator-specific deposition probabilities in tag–recovery studies. Transactions of the American Fisheries Society 144:410-422.

Hostetter, N.J., B. Gardner, A.F. Evans, B.M. Cramer, Q. Payton, K. Collis, and D.D. Roby. 2018. Wanted dead or alive: a state-space mark-recapture-recovery model incorporating multiple recovery types and state uncertainty. Canadian Journal of Fisheries and Aquatic Sciences 75:1117-1127.

Hostetter, N.J., Q. Payton, A.F. Evans, D.D. Roby, and K. Collis. 2021. Chapter 9: Functional responses across predator species, space, and time: how piscivorous waterbirds respond to changes in juvenile steelhead abundance *in* D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Lawes, T.J., K.S. Bixler, D.D. Roby, D.E. Lyons, K. Collis, A.F. Evans, A. Peck-Richardson, B. Cramer, Y.Suzuki, J.Y. Adkins, K. Courtot, and Q. Payton. 2021. Chapter 4: Double-crested cormorant management in the Columbia River estuary *in* D.D. Roby, A.F. Evans, and K. Collis (editors). Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Lawonn, M.J. In preparation. Summary of double-crested cormorant colony monitoring in the Columbia River estuary, 2020 and 2021 Draft report. Oregon Department of Fish and Wildlife, Salem, Oregon.

Lyons, D.E., D.D. Roby, and K. Collis. 2005. Foraging ecology of Caspian terns in the Columbia River estuary, USA. Waterbirds 28:280-291.

Lyons, D.E. 2010. Bioenergetics-based predator-prey relationships between piscivorous birds and juvenile salmonids in the Columbia River estuary. Unpublished Ph.D. dissertation, Oregon State University, Corvallis, Oregon. 330 pp.

McNatt, R.A., D.L. Bottom, and S.A. Hinton. 2016. Residency and movement of juvenile Chinook salmon at multiple spatial scales in a tidal marsh of the Columbia River estuary. Transactions of the American Fisheries Society 145:774-785.

Naughton, M.B., D. S. Pitkin, R.W. Lowe, K.J. So, and C.S. Strong. 2007. Catalog of Oregon seabird colonies. U.S. Department of Interior, Fish and Wildlife Service, Biological Technical Publication FWS/BTP-R1009-2007, Washington, D.C.

NOAA (National Oceanic and Atmospheric Administration). 2021. Updated status of federally listed ESUs of West Coast salmon and steelhead. Available from NOAA Fisheries, Seattle, Washington.

PSMFC (Pacific States Marine Fisheries Commission). 2021. PTAGIS (Columbia Basin PIT Tag Information System) [online database]. Pacific States Marine Fisheries Commission, Portland, Oregon. Available: www.ptagis.org.

Payton, Q., N.J. Hostetter, and A.F. Evans. 2019. Jointly estimating survival and mortality: integrating recapture and recovery data from complex multiple predator systems. Environmental and Ecological Statistics 26: 107-125.

Peck-Richardson, A.G., D.E. Lyons, D.D. Roby, D.A. Cushing, and J.A. Lerczak. 2018. Three-dimensional foraging habitat use and niche partitioning in two sympatric seabird species, *Phalacrocorax auritus* and *P. penicillatus*. Marine Ecology Progress Series 586: 251-264.

RDCT (R Development Core Team). 2021. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org.

Rauzon, M.J., W.J. Sydeman, M.L. Elliott, and J. Roth. 2001. Population size and breeding success of double-crested cormorants (*Phalacrocorax auritus*) on the Richmond-San Rafael Bridge, California, in 2000. Unpublished Report, San Francisco State University, Department of Biology, San Francisco, California, and Point Reyes Bird Observatory, Stinson Beach, California.

Rauzon, M.J., M.L. Elliott, P.J. Capitolo, L.M. Tarjan, G.J. McChesney, J.P. Kelly, and H.R. Carter. 2019. Changes in abundance and distribution of nesting Double-crested Cormorants *Phalacrocorax auritus* in the San Francisco Bay area, 1975–2017. Marine Ornithology 47: 127-138.

Roby, D.D., K. Collis, D.E. Lyons, D.P. Craig, J.Y. Adkins, A.M. Myers, and R.M. Suryan. 2002. Effects of colony relocation on diet and productivity of Caspian terns. Journal of Wildlife Management 66: 662-673.

Roby D.D, A.F. Evans, and K. Collis (editors). 2021. Avian Predation on Salmonids in the Columbia River Basin: A Synopsis of Ecology and Management. A synthesis report submitted to the U.S. Army Corps of Engineers, Walla Walla, Washington; the Bonneville Power Administration, Portland, Oregon; the Grant County Public Utility District/Priest Rapids Coordinating Committee, Ephrata, Washington; and the Oregon Department of Fish and Wildlife, Salem, Oregon. 788 pp.

Roegner, G.C., R.A. McNatt, D.J. Teel, and D.L. Bottom. 2012. Distribution, size, and origin of juvenile Chinook Salmon in shallow-water habitats of the lower Columbia River and estuary, 2002–2007. Marine and Coastal Fisheries 4(1): 450-472.

Ryan, B.A., S.G. Smith, J.M. Butzerin, and J.W. Ferguson. 2003. Relative vulnerability to avian predation of juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary, 1998–2000. Transactions of the American Fisheries Society 132: 275-288.

Sebring, S.H., M.C. Carper, R.D. Ledgerwood, B. . Sandford, G.M. Matthews, and A.F. Evans. 2013. Relative vulnerability of PIT-tagged subyearling fall Chinook salmon to predation by Caspian terns and double-crested cormorants in the Columbia River estuary. Transactions of the American Fisheries Society 142: 1321-1334.

Scoppettone, G.G., P.H. Rissler, M.C. Fabes, and D. Withers. 2014. American white pelican predation on Cui-ui in Pyramid Lake, Nevada. North American Journal of Fisheries Management 34(1): 57-67.

Simenstad, C. A., L.F. Small, C.D. McIntire, D.A. Jay, and C. Sherwood. 1990. Columbia River estuary studies: an introduction to the estuary, a brief history, and prior studies. Progress in Oceanography 25 (1-4): 1-13.

Simenstad, C.A., J.L. Burke, J.E. O'Connor, C. Cannon, D.W. Heatwole, M.F. Ramirez, I.R. Waite, T.D. Counihan, and K.L. Jones. 2011. Columbia River estuary ecosystem classification—concept and application. U.S. Geological Survey Open-File Report 2011-1228, 54 p.

SDT (Stan Development Team). 2020. RStan: the R Interface to Stan. R package version 2.21.1.

USACE (U.S. Army Corps of Engineers). 2015. Double-crested cormorant management plan to reduce predation on juvenile salmonids in the Columbia River estuary: final environmental impact statement, February 2015. U.S. Army Corps of Engineers, Portland District, Portland, Oregon.

Weitkamp, L.A., G. Goulette,, J.Hawkes, M. O'Malley, and C. Lipsky. 2014. Juvenile salmon in estuaries: comparisons between North American Atlantic and Pacific salmon populations. Reviews in Fish Biology and Fisheries 24(3), 713-736.

APPENDIX B: Summary Tables

Table B1. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on Crescent Island and Badger Island in McNary Reservoir, Goose Island in Potholes Reservoir, an unnamed island in Potholes Reservoir, and islands in Lenore Lake on ESA-listed salmonid populations originating from the Snake River (SR; based on detections at Lower Monumental Dam) and Upper Columbia River (UCR; based on detections at Rock Island Dam) during 2007-2021. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimates from 2007-2020 are those previously reported by Roby et al. (2021) and Collis et al. (2021a).

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
Crescent Island, McNary Reservoir						
2007	0.4% (0.3-0.6)	0.9% (0.4-1.7)	-	-	3.9% (3.1-5.6)	2.5% (1.7-3.8)
N	22,730	2,147		254	17,122	3,782
2008	0.9% (0.7-1.3)	1.6% (1.2-2.3)	-	1.7% (0.6-3.7)	5.9% (4.7-8.5)	2.9% (2.1-4.3)
N	30,142	22,968		767	28,653	8,403
2009	1.5% (1.1-2.2)	1.1% (0.8-1.6)	0.2% (<0.1-1.2)	1.0% (0.5-1.7)	4.6% (3.7-6.6)	2.3% (1.7-3.5)
N	20,679	26,567	738	2,651	52,102	8,025
2010	0.6% (0.4-1.1)	1.3% (1.0-1.9)	0.9% (0.3-2.3)	1.5% (0.5-3.4)	5.5% (4.2-7.9)	1.8% (1.3-2.7)
N	5,790	28,067	929	566	7,913	8,382
2011	0.8% (0.6-1.2)	0.6% (0.5-0.9)	0.5% (0.1-1.2)	0.9% (0.7-1.4)	3.0% (2.3-4.3)	2.4% (1.8-3.6)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	0.7% (0.5-1.0)	0.6% (0.5-1.0)	0.2% (0.1-0.8)	2.4% (1.5-3.8)	3.1% (2.3-4.5)	1.2% (0.8-2.0)
N	41,258	24,772	1,812	2,884	25,841	6,845
2013	0.7% (0.5-1.1)	0.9% (0.6-1.5)	0.4% (<0.1-1.2)	1.2% (0.5-2.7)	3.5% (2.7-5.1)	2.9% (2.1-4.3)
N	14,859	4,773	992	848	9,696	6,019
2014	0.8% (0.6-1.1)	0.6% (0.4-1.0)	0.7% (0.2-2.1)	1.5% (0.8-2.8)	6.1% (4.8-8.9)	3.4% (2.5-4.8)
N	22,195	6,043	641	1,414	16,599	7,757
2021	<0.1%	<0.1%	<0.1%	-	<0.1%	<0.1%
N	562	1,574	1,546		1,289	8,090
Badger Island, McNary Reservoir						

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2017	<0.1%	<0.1%	<0.1%	-	0.4% (0.2-0.6)	0.5% (0.3-0.8)
N	27,977	9,769	2,681		24,247	7,644
2021	0.8% (0.2-2.4)	0.3% (0.1-0.8)	<0.1%	-	1.5% (0.8-2.9)	1.4% (1.0-2.2)
N	562	1,574	1,546		1,289	8,090
Goose Island, Potholes Reservoir						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	<0.1%	0.3% (<0.1-1.1)	-	-	0.1% (<0.1-0.2)	15.3% (9.8-27.7)
N	22,730	2,147			17,122	3,782
2008	<0.1%	<0.1%	-	0.4% (<0.1-1.6)	<0.1%	11.1% (8.6-16.4)
N	30,142	22,968		767	28,653	8,403
2009	<0.1%	<0.1%	5.5% (2.7-10.7)	0.1% (<0.1-0.4)	0.1% (<0.1-0.1)	22.6% (17.2-33.7)
N	20,679	26,567	738	2,651	52,102	8,025
2010	<0.1%	<0.1%	2.0% (0.7-4.4)	0.3% (<0.1-1.9)	<0.1%	14.6% (11.0-21.8)
N	5,790	28,067	929	566	7,913	8,382
2011	<0.1%	<0.1%	0.6% (0.1-1.9)	<0.1%	<0.1%	12.9% (9.6-19.6)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	<0.1%	<0.1%	2.6% (1.2-5.4)	0.2% (<0.1-0.6)	0.2% (0.1-0.4)	18.4% (13.5-28.5)
N	41,258	24,772	1,812	2,884	25,841	6,845
2013	<0.1%	0.1% (<0.1-0.4)	2.5% (1.1-5.2)	0.1% (<0.1-1.1)	0.1% (0.1-0.4)	14.8% (11.4-21.6)
N	14,859	4,773	992	848	9,696	6,019
2014	<0.1%	0.1% (<0.1-0.8)	0.6% (0.1-2.2)	0.2% (<0.1-1.1)	<0.1%	2.9% (1.9-5.1)
N	22,195	6,043	641	1,414	16,599	7,757
2021	<0.1%	<0.1%	0.3% (0.1-1.5)	-	<0.1%	3.9% (2.4-6.3)
N	562	1,574	1,546		1,289	8,090
Unnamed Island, Northern Potholes Reservoir						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2016	<0.1%	<0.1%	0.1% (<0.1-0.3)	<0.1%	<0.1%	4.1% (2.9-6.3)
N	38,633	5,461	1,956	522	20,729	7,003

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
Lenore Lake Islands, Lenore Lake						
2015	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	4,471	1,393	766	1,262	2,400	7,222
2016	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	38,633	5,461	1,956	522	20,729	7,003
2017	<0.1%	<0.1%	0.3% (0.1-0.8)	-	<0.1%	1.0% (0.6-2.0)
N	27,977	9,769	2,681		24,247	7,644
2018	<0.1%	<0.1%	0.1% (0.1-0.8)	<0.1%	<0.1%	0.8% (0.4-1.7)
N	19,986	8,753	2,090	1,443	19,632	7,511
2019	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)	<0.1%	1.0% (0.6-1.7)
N	18,757	7,501	1,885	1,675	28,813	4,401
2020	<0.1%	<0.1%	0.3% (0.1-0.8)	-	<0.1%	1.0% (0.6-1.5)
N	2,931	1,607	947		1,130	6,843
2021	<0.1%	<0.1%	0.1% (0.1-0.8)	-	<0.1%	0.9% (0.6-1.5)
N	562	1,574	1,546		1,289	8,090
Twinning Island, Banks Lake						
2008	<0.1%	<0.1%	-	<0.1%	<0.1%	<0.1%
N	30,142	22,968		767	28,653	8,403
2009	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)
N	20,679	26,567	738	2,651	52,102	8,025
2010	<0.1%	<0.1%	<0.1%	0.2% (<0.1-1.2)	<0.1%	0.1% (<0.1-0.3)
N	5,790	28,067	929	566	7,913	8,382
2012	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)
N	41,258	24,772	1,812	2,884	25,841	6,845
2014	<0.1%	<0.1%	0.5% (<0.1-7.9)	0.1% (<0.1-0.6)	<0.1%	1.2% (0.3-6.4)
N	22,195	6,043	641	1,414	16,599	7,757
2015	<0.1%	<0.1%	0.2% (<0.1-0.9)	0.1% (0-0.5)	<0.1%	2.6% (1.8-3.9)
N	7,706	3,449	766	1,262	3,601	7,222

2016	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.2)
N	38,633	5,461	1,956	522	20,729	7,003

Table B2. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on the Blalock Islands on ESA-listed salmonid populations originating from the Snake River (SR) and Upper Columbia River (UCR) based on detections at McNary Dam during 2007-2020. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimated are those previously reported by Roby et al. (2021) and Collis et al. (2021).

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
Blalock Islands, John Day Reservoir						
2007	<0.1%	0.1% (<0.1-0.2)	<0.1%	-	0.9% (0.6-1.4)	1.0% (0.6-1.7)
N	74,905	7,374	6,764		7,683	3,111
2008	0.1% (0.1-0.2)	<0.1%	0.1% (<0.1-0.2)	-	0.8% (0.6-1.2)	0.7% (0.4-1.2)
N	27,288	36,857	4,713		15,449	3,399
2009	0.3% (0.2-0.4)	<0.1%	0.2% (0.1-0.5)	<0.1%	0.6% (0.4-0.9)	0.5% (0.3-1.0)
N	60,155	43,461	3,982	2,088	29,877	3,663
2010	0.1% (<0.1-0.1)	<0.1%	0.1% (<0.1-0.1)	0.2% (<0.1-0.6)	0.9% (0.7-1.4)	0.9% (0.6-1.6)
N	52,129	29,587	10,456	1,327	17,806	4,161
2011	0.1% (<0.1-0.1)	0.1% (0.1-0.2)	<0.1%	0.3% (0.1-0.8)	0.1% (0.1-0.2)	0.1% (<0.1-0.3)
N	38,629	41,007	3,981	2,769	16,759	5,155
2013	<0.1%	0.1% (<0.1-0.1)	<0.1%	<0.1%	0.1% (<0.1-0.2)	0.2% (<0.1-0.5)
N	47,685	14,398	6,778	1,213	9,391	2,621
2014	0.1% (0.1-0.2)	0.3% (0.2-0.5)	0.2% (0.1-0.4)	0.4% (0.1-1.1)	0.4% (0.2-0.7)	0.6% (0.3-1.2)
N	41,109	10,293	4,611	1,922	10,389	2,686
2015	1.4% (1.1-2.2)	0.4% (0.4-0.8)	0.9% (0.5-1.5)	1.3% (0.7-2.5)	8.0% (6.0-11.4)	8.2% (5.9-12.4)
N	31,474	4,390	4,921	1,712	6,824	2,056
2016	0.3% (0.2-0.5)	0.6% (0.4-1.1)	0.2% (0.1-0.4)	2.3% (1.2-4.1)	3.9% (3.9-5.7)	3.1% (2.3-4.6)
N	47,573	6,726	11,320	1,095	14,332	7,414
2017	0.9% (0.6-1.3)	0.6% (0.4-1.1)	1.1% (0.7-1.8)	-	3.4% (2.4-5.1)	4.2% (2.7-6.5)
N	17,215	9,230	6,517		5,795	2,536
2018	0.5% (0.3-0.9)	0.7% (0.4-1.4)	0.3% (0.1-0.8)	2.0% (0.4-6.1)	2.5% (1.4-4.5)	2.9% (1.5-5.2)
N	17,963	8,450	5,228	514	3,585	2,228
2019	0.4% (0.2-0.8)	1.3% (0.6-2.5)	0.9% (0.2-2.1)	1.4% (0.4-3.7)	3.0% (1.9-4.7)	5.9% (3.4-10.0)
N	11,225	3,395	2,838	1,167	5,878	1,671
2020	0.1% (<0.1-0.4)	0.2% (0.1-0.8)	0.1% (<0.1-0.5)	-	2.2% (0.7-5.4)	1.3% (0.3-4.2)
N	9,618	3,522	2,422		1,448	707

Table B3. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on East Sand Island on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Bonneville Dam), Upper Columbia River (UCR; based on detections at Bonneville Dam), middle Columbia River (MCR; based on detection at Bonneville Dam), and Upper Willamette River (WR; based on detections at Sullivan Dam) during 1999-2020. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Accurate and comparable estimates of predation were not available (NA) in 2019. No estimates of predation were available in 2021, although smolt PIT tags were recovered following the breeding season (Table 3).Estimated are those previously reported by Roby et al. (2021) and Evans et al. (2021).

	East Sand Island, Columbia River Estuary							
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2000	4.6% (3.6-6.6)	3.3% (2.1-5.3)	2.2% (1.2-3.8)	-	-	-	10.5% (8.4-15.0)	16.3% (12.8-22.9)
N	11,810	1,323	1,123				10,356	3,100
2001	14.0% (11.1-20.0)	6.4% (4.2-10.0)	13.2% (9.9-19.5)	-	-	15.0% (11.1-21.9)	33.9% (26.3-49.1)	-
N	8,845	807	1,230			872	774	
2002	2.9% (2.3-4.1)	1.7% (1.2-2.6)	2.5% (1.9-3.5)	-	-	-	21.9% (17.6-31.0)	14.2% (10.1-21.3)
N	30,617	4,899	20,493				7,331	561
2003	4.7% (3.7-6.9)	2.7% (2.0-4.0)	3.7% (2.9-5.3)	-	-	-	26.0% (21.0-36.2)	19.0% (15.4-26.9)
N	28,150	6,234	30,723				8,553	27,918
2004	4.8% (3.6-7.0)	1.3% (0.6-2.6)	3.7% (2.9-5.4)	-	-	-	25.8% (19.7-37.3)	14.1% (11.3-19.8)
N	4,816	929	9,533				803	6,040
2005	3.0% (2.2-4.4)	1.3% (0.6-2.6)	2.4% (1.6-3.8)	-	-	-	28.3% (21.6-40.6)	15.1% (11.9-21.6)
N	5,935	1,121	2,518				753	5,610
2006	3.3% (2.4-5.0)	2.5% (1.7-3.9)	3.6% (1.8-6.6)	-	-	-	27.5% (21.0-39.1)	23.4% (18.1-34.1)
N	5,570	4,057	731				1,100	2,064
2007	3.1% (2.5-4.4)	3.4% (2.3-5.3)	1.9% (1.2-3.2)	1.4% (0.8-2.5)	-	18.7% (14.6-26.8)	22.6% (18.2-32.4)	15.7% (12.4-22.6)
N	23,830	2,005	2,268	1,505		2,234	6,391	3,042
2008	2.5% (1.9-3.6)	1.9% (1.5-2.7)	1.7% (1.0-2.9)	4.4% (3.2-6.7)	-	13.5% (10.6-19.2)	14.2% (11.5-19.9)	16.7% (13.1-24.2)
N	11,425	24,136	1,662	2,509		2,291	19,572	2,513
2009	4.7% (3.7-6.9)	2.0% (1.5-2.9)	3.7% (2.5-5.6)	1.7% (1.2-2.7)	1.3% (0.7-2.2)	14.1% (11.1-20.0)	14.5% (11.9-20.1)	20.0% (15.6-29.3)
N	17,396	16,314	2,064	5,573	1,845	2,700	23,311	2,265
2010	3.4% (2.7-4.8)	0.7% (0.5-1.1)	2.9% (2.2-4.3)	1.8% (0.6-4.4)	1.6% (0.8-2.9)	11.9% (9.4-17.4)	14.3% (11.3-20.4)	13.7% (11.0-19.3)
N	*38,441*	*17,974*	*5,972*	*510*	*1,382*	*8,515*	*40,024*	*12,284*
2011	2.5% (1.8-3.6)	0.7% (0.5-1.1)	2.9% (1.4-5.3)	0.9% (0.3-2.0)	0.4% (0.1-1.3)	9.6% (6.6-14.7)	12.0% (9.4-17.3)	9.1% (6.9-13.4)

N	6,557	12,327	704	1,119	826	865	7,028	2,419
2012	2.2% (1.7-3.3)	0.7% (0.5-1.1)	1.2% (0.7-2.1)	0.7% (0.4-1.3)	2.1% (1.2-3.7)	9.4% (6.5-14.4)	10.2% (7.7-14.9)	7.5% (5.6-11.3)
N	17,929	10,742	3,227	3,731	1,457	1,084	4,768	3,357
2013	1.2% (0.8-1.8)	0.9% (0.5-1.6)	0.7% (0.3-1.4)	1.0% (0.5-1.8)	0.8% (0.3-2.0)	9.9% (7.0-15.3)	12.7% (9.6-18.5)	8.9% (6.6-13.4)
N	16,167	4,465	3,112	2,629	1,454	1,865	8,516	4,473
2014	1.1% (0.8-1.7)	1.0% (0.5-1.9)	1.4% (0.7-2.5)	1.2% (0.5-2.5)	1.6% (0.8-3.0)	9.5% (6.5-14.5)	8.6% (6.7-12.5)	11.4% (8.5-16.8)
N	14,828	2,800	2,297	1,587	1,739	1,119	8,812	3,841
2015	2.0% (1.5-2.9)	0.8% (0.4-1.5)	1.9% (1.3-2.9	0.4% (0.1-1.5)	1.6% (1.0-2.6)	7.8% (5.9-11.4)	10.2% (8.2-14.6)	10.5% (8.2-15.0)
N	20,245	2,629	5,943	768	3,311	3,927	16,451	6,004
2016	0.8% (0.6-1.2)	0.7% (0.3-1.3)	1.4% (0.9-2.1)	1.2% (0.4-3.2)	-	8.8% (6.4-13.0)	6.1% (4.8-8.8)	7.5% (5.8-10.7)
N	21,874	2,887	5,939	604		2,086	14,473	8,123
2017	0.8% (0.5-1.2)	0.2% (0.1-0.5)	1.4% (0.9-2.3)	-	-	8.4% (5.6-13.1)	5.3% (3.9-7.7)	6.5% (4.7-9.6)
N	13,151	4,635	4,622			1,069	6,497	3,275
2018	1.4% (1.0-2.1)	1.3% (0.7-2.1)	1.4% (0.9-2.3)	-	4.2% (2.9-6.4)	5.3% (3.8-8.0)	6.9% (5.3-10.2)	6.5% (4.8-9.7)
N	11,174	5,981	3,370		2,546	3,209	9,572	5,322
2019 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2020	0.7% (0.5-1.1)	0.3% (0.1-0.7)	0.4% (0.2-0.9)	-	1.1% (0.6-2.2)	5.4% (3.8-7.9)	5.9% (4.5-8.1)	4.5% (3.3-6.4)
N	20,246	3,389	4,895		2,122	3,157	11,868	5,894
2021 *N*	NA	NA	NA	NA	NA	NA	NA	NA

Table B4. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting on East Sand Island on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Bonneville Dam), Upper Columbia River (UCR; based on detections at Bonneville Dam), middle Columbia River (MCR; based on detection at Bonneville Dam), and Upper Willamette River (UWR; based on detections at Sullivan Dam) during 2003-2021. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates. Accurate and comparable estimates of predation were not available (NA) in 2019. There was no established double-crested cormorant colony (NC) on East Sand Island in 2020 and no estimates were available in 2021. Estimated are those previously reported by Roby et al. (2021).

| Year | East Sand Island, Columbia River Estuary | | | | | | | |
	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2003	1.7% (1.2-2.7)	1.1% (0.7-2.0)	1.4% (0.9-2.1)	-	-	-	1.9% (1.2-3.0)	1.5% (1.0-2.4)
N	28,150	6,234	30,723				8,553	27,918
2004	5.1% (3.3-8.5)	1.9% (0.6-4.7)	4.7% (3.2-7.6)	-	-	-	3.6% (1.4-8.0)	7.4% (5.1-11.8)
N	4,816	929	9,533				803	6,040
2005	4.8% (3.2-7.9)	3.6% (1.8-6.9)	4.5% (2.8-7.8)	-	-	-	4.3% (2.0-8.6)	5.5% (3.7-8.8)
N	5,935	1,121	2,518				753	5,610
2006	5.2% (3.5-8.5)	2.7% (1.6-4.6)	4.7% (2.2-9.5)	-	-	-	13.1% (8.2-22.7)	4.7% (2.8-8.2)
N	5,570	4,057	731				1,100	2,064
2007	1.7% (1.1-2.7)	1.6% (0.7-3.3)	2.7% (1.5-5.1)	1.0% (0.3-2.6)	-	2.8% (1.5-5.2)	3.5% (2.3-5.8)	3.4% (2.1-6.1)
N	23,830	2,005	2,268	1,505		2,234	6,391	3,042
2008	3.5% (2.4-5.5)	2.6% (1.9-4.2)	3.6% (2.0-6.6)	3.3% (1.9-5.8)	-	14.0% (9.5-23.2)	14.7% (10.6-23.2)	6.2% (4.0-10.4)
N	11,425	24,136	1,662	2,509		2,291	19,572	2,513
2009	6.8% (4.9-10.7)	4.5% (3.2-7.1)	2.7% (1.5-4.9)	1.4% (0.8-2.4)	5.7% (3.5-9.8)	14.9% (10.3-23.8)	16.6% (12.0-25.7)	7.2% (4.7-12.0)
N	17,396	16,314	2,064	5,573	1,845	2,700	23,311	2,265
2010	5.3% (3.9-8.4)	3.9% (2.7-6.1)	3.3% (2.3-5.4)	4.2% (1.6-9.2)	2.6% (1.3-4.9)	8.2% (5.8-13.1)	7.5% (5.5-12.0)	6.8% (4.9-10.6)
N	38,441	17,974	5,972	510	1,382	8,515	40,024	12,284
2011	4.3% (2.9-6.9)	1.9% (1.3-3.1)	5.6% (2.9-10.8)	0.4% (0.1-1.5)	4.8% (2.4-9.1)	7.8% (4.6-14.0)	5.3% (3.7-8.5)	11.4% (7.8-18.6)
N	6,557	12,327	704	1,119	826	865	7,028	2,419
2012	3.7% (2.6-6.0)	2.6% (1.8-4.2)	2.1% (1.2-3.7)	0.6% (0.3-1.3)	3.7% (2.0-6.9)	3.3% (1.7-6.4)	4.9% (3.2-8.1)	6.5% (4.3-10.8)
N	17,929	10,742	3,227	3,731	1,457	1,084	4,768	3,357
2013	3.6% (2.5-5.7)	2.2% (1.3-3.7)	3.0% (1.8-5.3)	1.0% (0.4-2.0)	3.3% (1.8-6.2)	2.1% (1.0-4.1)	2.5% (1.7-4.0)	3.4% (2.2-5.7)
N	16,167	4,465	3,112	2,629	1,454	1,865	8,516	4,473

Year								
2014	8.5% (6.1-13.2)	2.4% (1.5-4.2)	6.1% (3.9-10.1)	1.8% (0.9-3.6)	4.5% (2.7-7.7)	6.4% (3.7-10.7)	7.8% (5.6-12.0)	10.4% (7.3-16.3)
N	14,828	2,800	2,297	1,587	1,739	1,119	8,812	3,841
2015	14.5% (10.5-22.4)	8.7% (6.0-14.0)	8.3% (5.9-12.9)	2.4% (0.9-5.2)	2.4% (1.5-4.1)	12.4% (8.8-19.2)	12.8% (9.3-19.6)	10.5% (7.6-16.2)
N	20,245	2,629	5,943	768	3,311	3,927	16,451	6,004
2016[a]	5.1% (3.7-8.1)	2.1% (1.1-3.9)	3.5% (2.3-5.7)	0.4% (0-2.1)	-	2.7% (1.4-4.9)	6.8% (4.8-10.7)	5.1% (3.6-8.2)
N	21,874	2,887	5,939	604		2,086	14,473	8,123
2017[a]	0.7% (0.4-1.1)	0.1% (0-0.2)	0.4% (0.1-0.8)	-	-	0.7% (0.1-2.1)	0.4% (0.2-0.8)	1.4% (0.8-2.7)
N	13,151	4,635	4,622			1,069	6,497	3,275
2018[a]	0.5% (0.3-0.8)	0.9% (0.5-1.6)	0.6% (0.3-1.2)	-	0.9% (0.5-1.9)	0.4% (0.1-1.0)	0.5% (0.3-0.9)	0.7% (0.4-1.4)
N	11,174	5,981	3,370		2,546	3,209	9,572	5,322
2019	NA	NA	NA	NA	NA	NA	NA	NA
N								
2020	NA	NA	NA	NA	NA	NA	NA	NA
N								
2021	NA	NA	NA	NA	NA	NA	NA	NA
N								

Table B5. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by California and ring billed gulls nesting at Crescent and Badger islands in McNary Reservoir, Island 20 in the middle Columbia River, and Goose Island in Potholes Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detections at Lower Monumental Dam) and Upper Columbia River (UCR; based on detections at Rock Island Dam) during 2007-2021. A dashed line indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates. Estimated from 2000-2020 are those previously reported by Roby et al. (2021) and Collis et al. (2021).

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
Crescent Island, McNary Reservoir						
2007	0.6% (0.3-1.1)	0.6% (0.1-2.3)	-	-	4.1% (2.7-6.5)	5.9% (3.5-10.1)
N	22,730	2,147			17,122	3,782
2008	0.9% (0.5-1.4)	0.6% (0.3-1.1)	-	1.7% (0.2-6.1)	4.0% (2.8-5.9)	3.0% (1.8-4.8)
N	30,142	22,968		767	28,653	8,403
2009	0.9% (0.5-1.6)	0.7% (0.4-1.1)	0.8% (<0.1-4.7)	2.6% (1.1-5.4)	4.7% (3.4-6.8)	7.5% (5.0-11.4)
N	20,679	26,567	738	2,651	52,102	8,025
2010	1.4% (0.7-2.8)	0.3% (0.2-0.6)	0.5% (<0.1-3.3)	0.9% (<0.1-5.4)	5.1% (3.3-7.9)	7.8% (5.3-11.7)
N	5,790	28,067	929	566	7,913	8,382
2011	1.1% (0.7-1.7)	0.7% (0.4-1.2)	0.4% (<0.1-2.4)	1.6% (0.9-2.8)	3.4% (2.4-5.1)	3.6% (2.2-5.9)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	1.1% (0.7-1.7)	0.6% (0.3-1.2)	1.3% (0.3-4.0)	1.3% (0.4-3.2)	5.2% (3.5-8.0)	4.7% (2.9-7.8)
N	41,258	24,772	1,812	2,884	25,841	6,845
2013	1.0% (0.5-1.8)	0.8% (0.2-2.0)	0.6% (<0.1-3.6)	2.8% (0.6-8.3)	5.8% (3.8-8.9)	6.1% (3.8-9.7)
N	14,859	4,773	992	848	9,696	6,019
2014	1.1% (0.6-1.8)	0.3% (<0.1-0.9)	2.1% (<0.3-7.5)	3.1% (1.1-7.0)	5.5% (3.8-8.2)	6.8% (4.6-10.4)
N	22,195	6,043	641	1,414	16,599	7,757
2021	1.9% (0.2-8.6)	0.2% (<0.1-2.2)	0.7% (0.1-3.0)	-	2.8% (0.7-8.6)	2.9% (1.7-5.0)
N	562	1,574	1,546		1,289	8,090
Badger Island, McNary Reservoir						
2015	0.1% (<0.1-0.5)	0.1% (0-0.9)	0.5% (<0.1-3.3)	1.1% (0.2-4.8)	2.9% (1.3-6.4)	5.2% (3.2-9.2)
N	7,706	3,449	766	1,262	3,601	7,222

Year						
2016	0.2% (0.1-0.4)	<0.1%	0.9% (0.1-3.3)	1.2% (0.1-7.7)	1.1% (0.6-1.9)	4.3% (2-13.8)
N	38,633	5,461	1,956	522	20,729	7,003
2017	0.2% (<0.1-0.4)	0.4% (0.1-1.0)	0.6% (0.1-2.1)	-	1.0% (0.6-1.8)	1.3% (0.6-2.6)
N	27,977	9,769	2,681		24,247	7,644
2018	1.0% (0.5-1.8)	1.0% (0.4-2.1)	1.1% (0.2-4.1)	4.0% (1.1-9.5)	4.3% (3-6.7)	4.8% (2.8-8.0)
N	19,986	8,753	2,090	1,443	19,632	7,511
2019	1.2% (0.6-2.1)	1.9% (0.8-8.4)	3.6% (1.1-8.8)	3.1% (0.9-8.3)	5.6% (3.9-8.4)	10.9% (6.7-17.7)
N	18,757	7,501	1,885	1,675	28,813	4,401
2020	1.0% (0.2-3.1)	1.0% (0.3-6.1)	0.6% (0.1-4.0)	-	9.2% (2.0-18.4)	4.9% (1.7-9.3)
N	2,931	1,607	947		1,130	6,843
2021	0.7% (0-3.4)	1.8% (0.1-5.4)	0.5% (0-1.8)	-	9.7% (1.3-20.9)	8.6% (3.6-14.7)
N	562	1,574	1,546		1,289	8,090

Island 20, Middle Columbia River						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2013	0.3% (0.1-0.7)	0.1% (<0.1-0.7)	0.5% (<0.1-3.2)	0.6% (<0.1-3.6)	0.7% (0.3-1.4)	1.4% (0.6-2.7)
N	14,859	4,773	992	848	9,696	6,023
2014	0.2% (0.1-0.5)	0.2% (<0.1-0.9)	0.8% (<0.1-5.1)	0.4% (<0.1-2.3)	0.6% (0.3-1.1)	1.6% (0.8-3.1)
N	22,195	6,043	641	1,414	16,599	7,757
2015	0.3% (0.1-0.8)	0.1% (0-0.8)	0.6% (<0.1-3.7)	NA	2.4% (1.2-4.5)	7.9% (5.3-12.0)
N	7,706	3,449	766	1,262	3,601	7,222
2016	0.2% (0.1-0.4)	<0.1%	0.2% (<0.1-3.8)	0.9% (<0.1-5.8)	1.2% (0.7-2.0)	5.7% (3.7-8.9)
N	38,633	5,461	1,956	522	20,792	7,003
2017	0.2% (<0.1-0.4)	0.2% (<0.1-0.6)	0.1% (<0.1-1.0)	-	1.7% (1.1-2.6)	3.0% (1.8-4.9)
N	27,977	9,769	2,681		24,247	7,644
2018	0.1% (<0.1-0.3)	0.3% (0.1-0.8)	0.2% (<0.1-1.3)	0.8% (0.1-2.8)	1.3% (0.8-2.1)	1.1% (0.5-2.0)
N	19,986	8,793	2,090	1,443	19,632	7,511
2019	0.2% (0.1-0.4)	0.2% (0.1-0.7)	0.2% (0.1-1.5)	0.2% (<0.1-1.5)	2.8% (1.9-4.2)	7.4% (4.8-11.7)
N	18,757	7,501	1,885	1,675	28,813	4,401
2020	0.3% (<0.1-1.3)	0.1% (<0.1-1.4)	0.1% (<0.1-1.8)	-	1.6% (0.3-4.9)	2.1% (1.1-3.6)
N	2,931	1,607	947		1,130	6,843
2021	1.4% (0.2-6.2)	0.1% (<0.1-1.1)	0.5% (0.1-2.1)	-	4.8% (2.1-9.5)	6.1% (4.1-9.3)

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
N	562	1,574	1,546		1,289	8,090
Goose Island, Potholes Reservoir						
2012	<0.1%	<0.1%	1.3% (0.3-4.0)	<0.1%	0.1% (<0.1-0.3)	2.8% (1.1-5.6)
N	41,258	24,772	1,812	2,884	25,841	6,845
2020	<0.1%	<0.1%	<0.1%	-	<0.1%	<0.1%
N	2,931	1,607	947		1,130	6,843

Table B6. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by California and ring billed gulls nesting Miller Rocks Island in The Dalles Reservoir and the Blalock Islands in the John Day Reservoirs on ESA-listed salmonid populations originating from the Snake River (SR) and Upper Columbia River (UCR) based on detections at McNary Dam during 2007-2021. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates. Estimates from 2000-2020 are those previously reported by Roby et al. (2021) and Collis et al. (2021).

	Miller Rocks Island, The Dalles Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	1.2% (0.8-1.7)	3.3% (2.1-5.4)	2.2% (1.3-3.8)	-	9.9% (6.9-14.6)	8.7% (5.5-13.9)
N	74,905	7,374	6,764		7,683	3,111
2008	2.3% (1.5-3.4)	2.7% (1.9-4.0)	1.6% (0.8-3.2)	-	9.2% (6.6-13.4)	7.2% (4.5-11.6)
N	27,288	36,857	4,713		15,449	3,399
2009	2.2% (1.5-3.2)	3.7% (2.6-5.4)	2.5% (1.2-4.6)	9.1% (5.3-15.1)	9.8% (7.1-14.3)	7.4% (4.6-11.8)
N	60,155	43,461	3,982	2,088	29,877	3,663
2010	1.8% (1.2-2.7)	1.0% (0.6-1.5)	1.7% (1.0-2.9)	4.1% (1.6-8.8)	9.1% (6.5-13.2)	7.0% (4.4-11.2)
N	52,129	29,587	10,456	1,327	17,806	4,161
2011	0.8% (0.5-1.3)	0.6% (0.4-1.0)	1.0% (0.3-2.3)	2.0% (0.8-4.2)	5.0% (3.5-7.6)	3.5% (2.0-5.9)
N	38,629	41,007	3,981	2,769	16,759	5,155
2012	0.6% (0.4-1.0)	0.7% (0.4-1.3)	1.5% (0.8-2.9)	6.3% (3.6-10.9)	4.6% (3.0-7.2)	7.2% (4.4-11.6)
N	40,168	25,017	6,800	2,492	8,840	3,804
2013	1.1% (0.7-1.7)	2.4% (1.6-3.8)	1.9% (1.0-3.4)	6.4% (3.0-12.5)	6.4% (4.4-9.7)	11.7% (7.4-18.4)
N	47,685	14,398	6,778	1,213	9,391	2,621
2014	1.0% (0.7-1.6)	1.8% (1.0-3.0)	1.3% (0.6-2.7)	4.4% (2.1-8.3)	5.3% (3.6-8.0)	6.1% (3.5-10.4)
N	41,109	10,293	4,611	1,922	10,389	2,686
2015	1.7% (1.1-2.6)	2.6% (1.4-4.6)	3.5% (2.1-6.0)	7.4% (4.1-13.1)	9.7% (6.6-14.6)	13.2% (8.3-21.1)
N	31,474	4,390	4,921	1,712	6,824	2,056
2016	1.2% (0.8-1.9)	1.0% (0.4-2.1)	2.5% (1.6-4.0)	6.4% (2.9-12.8)	6.7% (4.6-9.9)	10.1% (7.0-15.2)
N	47,573	6,726	11,320	1,095	14,332	7,414
2017	0.6% (0.3-1.1)	0.8% (0.4-1.7)	2.2% (1.2-3.9)	-	7.0% (4.3-11.0)	6.9% (3.8-12.0)
N	17,215	9,230	6,517		5,795	2,536
2018	0.4% (0.2-0.9)	2.1% (1.2-3.5)	1.2% (0.5-2.4)	7.0% (2.3-16.4)	3.3% (1.8-6.0)	8.3% (4.8-13.9)
N	17,963	8,450	5,228	514	3,585	2,228

Year					
2019	0.9% (0.5-1.7)	3.2% (1.7-5.8)	2.4% (1.1-4.9)	5.9% (2.6-11.8)	5.1% (3.2-8.1)	6.5% (3.4-11.8)
N	11,225	3,395	2,838	1,167	5,878	1,671
2020	1.1% (0.5-2.0)	1.9% (0.8-3.9)	3.1% (1.4-6.2)	-	4.4% (1.8-9.2)	2.5% (0.5-8.0)
N	9,618	3,522	2,422		1,448	707
2021	1.2% (0.7-2.2)	2.8% (1.1-6.3)	0.9% (0.2-2.8)	-	13.9% (9.1-21.8)	12.4% (6.2-23.1)
N	9,816	1,711	1,822		2,739	799

			Blalock Islands, John Day Reservoir			
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2013	0.1% (<0.1-0.2)	0.3% (0.1-0.8)	0.4% (0.1-1.2)	1.9% (0.4-5.6)	1.2% (0.6-2.3)	1.2% (0.3-3.1)
N	47,685	14,398	6,778	1,213	9,391	2,621
2014	0.2% (0.1-0.4)	0.4% (0.1-0.9)	0.4% (0.1-1.1)	1.0% (0.2-2.7)	1.5% (0.9-2.5)	2.5% (1.2-4.9)
N	41,109	10,293	4,611	1,922	10,389	2,686
2015	0.2% (0.1-0.4)	0.7% (0.2-1.7)	0.6% (0.2-1.5)	1.4% (0.4-3.7)	2.6% (1.6-4.2)	6.8% (3.9-11.4)
N	31,474	4,390	4,921	1,712	6,824	2,056
2016	0.1% (0.1-0.3)	0.5% (0.2-1.2)	0.2% (0-0.5)	3.7% (1.4-8.2)	3.5% (2.4-5.2)	6.3% (4.4-9.2)
N	47,573	6,726	11,320	1,095	14,332	7,414
2017	0.2% (0.1-0.5)	0.2% (<0.1-0.5)	0.2% (0-0.7)	-	2.5% (1.5-4.2)	5.2% (3-8.8)
N	17,215	9,230	6,517		5,795	2,536
2018	0.2% (0.1-0.4)	0.3% (0.1-0.8)	0.6% (0.2-1.5)	2.2% (0.3-8)	3.3% (1.8-5.8)	3.1% (1.5-6)
N	17,963	8,450	5,228	514	3,585	2,228
2019	02% (0.1-0.5)	0.3% (0.1-1.1)	0.3% (0.1-1.2)	1.4% (0.3-4.2)	1.6% (0.8-2.8)	1.5% (0.4-3.8)
N	11,225	3,395	2,838	1,167	5,878	1,671
2020	0.1% (<0.1-0.4)	0.1% (<0.1-0.7)	0.1% (<0.1-0.9)	-	2.6% (0.7-6.8)	3.2% (0.7-9.6)
N	9,618	3,522	2,422		1,448	707
2021	0.5% (0.2-1.0)	0.2% (<0.1-0.3)	0.9% (0.2-2.7)	-	5.7% (3.3-9.6)	7.9% (3.6-15.8)
N	9,816	1,711	1,822		2,739	799

Table B7. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting at Foundation Island in McNary Reservoir, Hanford Island in the middle Columbia River, and islands in Lenore Lake, and in Northern Potholes Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detections at Lower Monumental Dam) and Upper Columbia River (UCR; based on detections at Rock Island Dam) during 2007-2021. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates. The Foundation Island colony was active in 2013 and during 2015-2019 but was not scanned for smolt PIT tags so estimate in those years are unavailable. Estimated from 2000-2020 are those previously reported by Roby et al. (2021) and Collis et al. (2021).

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
Foundation Island, McNary Reservoir						
2007	1.5% (1.0-2.4)	1.9% (1.0-3.6)	-	-	4.7% (3.4-7.5)	< 0.1%
N	22,730	2,147			17,122	3,782
2008	2.0% (1.4-3.1)	0.9% (0.6-1.4)	-	2.4% (1.0-5.3)	4.7% (3.4-7.3)	0.3% (0.1-0.6)
N	30,142	22,968		767	28,653	8,403
2009	1.8% (1.3-2.9)	1.1% (0.7-1.7)	0.2% (<0.1-1.4)	4.3% (2.7-7.1)	3.6% (2.6-5.6)	0.2% (0.1-0.5)
N	20,679	26,567	738	2,651	52,102	8,025
2010	2.4% (1.5-3.9)	1.1% (0.7-1.7)	0.2% (<0.1-1.2)	3.8% (1.5-8.3)	3.6% (2.5-5.9)	0.2% (0.1-0.4)
N	5,790	28,067	929	566	7,913	8,382
2011	1.4% (1.0-2.4)	1.2% (0.8-1.9)	0.5% (0.1-1.8)	1.6% (1.0-2.8)	4.3% (2.9-7.0)	0.3% (0.1-0.6)
N	54,944	46,593	1,567	12,445	53,565	8,002
2012	0.9% (0.6-1.5)	0.6% (0.3-1.1)	0.5% (0.1-1.9)	4.1% (2.3-7.6)	2.4% (1.5-3.9)	0.5% (0.2-1.1)
N	41,258	24,772	1,812	2,884	25,841	6,845
2014	1.1% (0.5-2.1)	0.6% (0.2-1.9)	1.0% (<0.1-6.2)	2.8% (0.7-8.0)	1.8% (0.9-3.4)	0.2% (<0.1-0.8)
N	22,195	6,043	641	1,414	16,599	7,757
2020	2.5% (1.0-5.8)	0.8% (0.1-3.6)	0.1% (<0.1-2.4)	-	4.0% (1.2-10.7)	0.1% (<0.1-0.3)
N	2,931	1,607	947		1,130	6,843
2021	4.4% (0.8-15.6)	0.1% (<0.1-0.6)	<0.1%	-	3.0% (0.8-9.3)	<0.1%
N	562	1,574	1,546		1,289	8,090
Hanford Island, Middle Columbia River						
2018	<0.1%	<0.1%	0.2% (0.1-0.7)	<0.1%	<0.1%	0.2% (0.1-0.4)

N	19,986	8,753	2,090	1,443	19,632	7,511
2020	<0.1%	<0.1%	0.1% (<0.1-0.9)	-	<0.1%	<0.1%
N	2,931	1,607	947		1,130	6,843
2021	0.1% (<0.1-1.6)	<0.1%	0.1% (<0-0.5)	-	0.1% (<0.1-0.8)	0.3% (0.1-0.7)
N	562	1,574	1,546		1,289	8,090
Lenore Lake Islands, Lenore Lake						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2017	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	27,977	9,769	2,681	304	24,247	7,644
2019	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	18,757	7,501	1,885	1,675	28,813	4,401
2020	<0.1%	<0.1%	<0.1%	-	<0.1%	<0.1%
N	2,931	1,607	947		1,130	6,843