AVIAN PREDATION IN THE COLUMBIA RIVER BASIN

2022 Final Annual Report

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EXECUTIVE SUMMARY

Currently, there are three different management plans underway to reduce the impacts of predation by piscivorous colonial waterbirds on the survival of Endangered Species Act (ESA)-listed juvenile salmonids (smolts; *Oncorhynchus* spp.) in the Columbia River Basin (CRB). Two separate management plans target the breeding colonies of Caspian terns (*Hydroprogne caspia;* hereafter referred to as "tern" or "terns") and double-crested cormorants (*Nannopterum auritum;* hereafter referred to as "cormorant" or "cormorants") on East Sand Island (ESI) in the Columbia River estuary (CRE) and one plan targets the tern colonies on Crescent Island (McNary Reservoir) and Goose Island (Potholes Reservoir) in the Columbia Plateau region (CPR). As part of the management plans for terns both in the CRE and CPR, adaptive management actions have been conducted at various other colony locations where terns displaced from the managed colonies have relocated to nest. The primary objectives of this study were to evaluate the efficacy of management actions to reduce predation by terns and cormorants on juvenile salmonids in the CRB, and to assess the magnitude of predation on smolts by other piscivorous colonial waterbirds, namely California gulls (*Larus californicus*) and ring-billed gulls (*L. delawarensis*; hereafter referred to collectively 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 colonial waterbird species (terns, cormorants, gulls, and pelicans) at colonies within foraging range of juvenile salmonids out-migrating 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) estimate the cumulative effects of predation by piscivorous waterbirds from multiple breeding colonies, including an investigation into the additive effects of avian predation on salmonid mortality.

There was a total of 34 active piscivorous waterbird colonies detected in the CRB in 2022. Of those, cormorant and gull colonies were the most prevalent (13 and 11 colonies, respectively), followed by tern colonies (8), and pelican colonies (2). Most of these colonies were in the CPR (24 colonies), with 8 and 2 colonies being in the CRE and the lower Columbia River, respectively. Tern and cormorant colonies continue to be managed as part of ongoing management plans and management has resulted in a shift in the nesting distribution of these birds, prompting adaptive management at several previously unmanaged colony sites. As has been the case in the past, gulls were the most numerous (ca. 48,000 individuals) of all the piscivorous colonial waterbirds in the CRB, followed by cormorants (ca. 6,200 breeding pairs), pelicans (ca. 4,500 individuals), and terns (ca. 2,500 breeding pairs). Recent data suggest that the numbers of terns and cormorants nesting in the CRB have declined, the objective of management, but in the case of both terns and cormorants has apparently resulted in significant declines in the Pacific Flyway breeding populations, raising concerns about their conservation status, especially for the rapidly declining Pacific Flyway population of Caspian terns.

In the CRE, efforts to prevent large numbers of terns from nesting outside the designated 1-acre colony area on ESI were successful. However, the designated tern colony on ESI once again failed to produce any young, with complete nesting failure being observed at that colony in 5 out of the last 7 years. The reason(s) for the nesting failure at the ESI tern colony in 2022 is unknown but is likely due to gull predation on tern eggs, especially during disturbances of the colony by bald eagles (*Haliaeetus leucocephalus)*. During the period when terns mostly abandoned the ESI colony in late May, terns were counted in the thousands on Rice Island in the upper CRE. Adaptive management conducted to prevent nesting by terns on Rice Island were eventually successful in dispersing the terns from that site, but only after 2.5 weeks coinciding with the peak of the steelhead outmigration period when thousands of terns were present on Rice Island.

As was the case in the CRE, management implemented at the tern colonies in the CPR (Goose and Crescent islands) has resulted in a shift in the nesting distribution of terns in the region. Efforts to dissuade terns from nesting on Goose Island and elsewhere in Potholes Reservoir were largely successful; only 16 breeding pairs were observed on Goose Island in 2022. Management to dissuade terns from nesting on Crescent Island in McNary Reservoir ceased in 2021, however, and terns re-established a nesting colony of 149 breeding pairs on the island in 2022. Previously, most of the terns that were dissuaded from nesting on Crescent Island starting in 2015 had relocated to nest on the Blalock Islands in John Day Reservoir. Beginning in 2021, adaptive management to raise the elevation of John Day Reservoir was implemented to inundate all tern nesting habitat in the Blalock Islands and in 2021-2022 no nesting by terns was observed in the Blalock Islands. This adaptive management apparently resulted in a shift of terns back to the former colony site on Crescent Island, where tern dissuasion had ceased, and to Badger Island, located on the Columbia River just upstream of Crescent Island. Despite adaptive management implemented in 2022 to reduce tern nesting habitat on Badger Island (via placement of large woody debris on one of two nesting areas used by terns in 2021), the largest tern colony in the CPR during 2022 was on Badger Island (267 breeding pairs on a site without woody debris located on the northeast shoreline). A total of 511 breeding pairs of terns nested in the CPR in 2022, the highest regional population of terns observed since 2017. These trends indicate that monitoring and adaptive management will be necessary in the future if the goals and objectives of the management plans for terns in the CRE and the CPR are to be realized.

Ongoing management at the ESI cormorant colony has also resulted in the dispersal of birds away from ESI, with most nesting cormorants now located further upstream in the CRE on the Astoria-Megler Bridge. This year marks the fourth year in a row when there has been little to no cormorant nesting on ESI, with any nesting attempts on ESI failing early in incubation. Meanwhile, cormorant colony size on the Astoria-Megler Bridge increased exponentially during implementation of the cormorant management plan during 2015-2019, culminating in over 5,000 breeding pairs nesting on the bridge in 2020. This is a concern for fisheries managers because piscivorous waterbirds nesting in the freshwater or mixing zones of the CRE can consume a far greater number of juvenile salmonids compared to conspecifics nesting at sites in the marine zone of the CRE, such as ESI (see *below*).

To investigate the effects of predation by piscivorous colonial waterbirds on the survival of smolts, and to determine the efficacy of ongoing management actions to reduce avian predation, we estimated population (Evolutionarily Significant Unit [ESU] or Distinct Population Segment [DPS])-specific predation rates based on recoveries of smolt PIT tags on bird colonies following the 2022 nesting season. One of the primary objectives of the *Inland Avian Predation Management Plan* (*IAPMP*) was to reduce predation rates by terns in the CPR to less than 2% per salmonid ESU/DPS, per colony, per year. In 2022, predation rates by terns nesting on Shoal Island in Lenore Lake, WA, exceeded this threshold for Upper Columbia River (UCR) steelhead, with an estimated 2.1% (95% credible interval = 1.4–3.4%) of available smolts depredated by terns. Also, predation rates by terns nesting on Badger Island in McNary Reservoir exceeded this threshold for Snake River (SR) steelhead with an estimated 2.8% (2.0–4.4%) of available smolts depredated by terns. Predation rates on all other ESA-listed ESUs/DPSs were less than 2% per tern colony, meeting the objective of the *IAPMP* for those salmonid populations. Although predation rates by terns nesting on Crescent Island in 2022 were < 2% per salmonid ESU/DPS, predation rates were substantially higher compared to 2021, commensurate with the increase in size of the Crescent Island tern colony from just 1 pair in 2021 to 149 pairs in 2022; the highest estimated predation rate by Crescent Island terns was on SR steelhead at 1.3% (0.9–2.1%). Due to adaptive management actions that resulted in higher reservoir elevations in John Day Reservoir, no terns nested on the Blalock Islands in 2021-2022, a nesting site where predation rates exceeded the 2% threshold for several ESUs/DPSs in previous years (2015-2020).

Average annual predation rates on UCR steelhead 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 1.3% (0.8–2.3%) during the management period (2014-2022). There was also evidence that survival of UCR steelhead smolts has increased significantly in the river reach where terns from Goose Island and Crescent Island forage following implementation of the *IAPMP*. Average annual predation rates on SR steelhead and other SR ESUs/DPSs, however, have remained largely unchanged since management actions at Crescent Island commenced in 2015. This is due to terns relocating from Crescent Island in McNary Reservoir downstream to the Blalock Islands in John Day Reservoir during 2015-2020 and then from the Blalock Islands back upstream to Crescent Island and Badger Island during 2021-2022. Due to increases in predation on smolts by terns nesting on Crescent Island, Badger Island, and Lenore Lake in 2022, adaptive management actions will likely be necessary in the future to achieve the goals and objectives of the *IAPMP* for reducing tern predation rates.

Comparable estimates of predation rates on ESA-listed salmonid ESUs/DPSs by terns nesting on ESI in the CRE during the 2021-2022 nesting seasons were not available, estimates that were generated with support from the U.S. Army Corps of Engineers in previous years. In 2020, estimates of predation rates by terns nesting on ESI were the lowest recorded since 1999, with estimates ranging from 0.4% (0.2–0.9%) for UCR spring Chinook salmon to 5.9% (4.5–8.1%) for SR steelhead. Results from 2020 indicated that predation by terns nesting on ESI 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.* In 2022, terns attempted to nest on Rice Island in the upper CRE, in addition to nesting at the designated colony site on ESI. Estimated predation rates by terns that attempted to nest on Rice Island ranged from 0.1% (<0.1–0.5%) on SR sockeye salmon to 2.9% (1.6–5.1%) on Middle Columbia River steelhead. Our results indicate that continued implementation of nest dissuasion activities, continued monitoring of tern nesting sites throughout the CRE, 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 estuary tern management plan.

For the third consecutive year there was no evidence of persistent nesting by double-crested cormorants on ESI in 2022. An unintended consequence of management actions at the cormorant colony on ESI was the abandonment of that colony and the subsequent rapid expansion of the cormorant colony on the Astoria-Megler Bridge (AMB). The AMB is located upstream of ESI in the freshwater mixing zone of the CRE, an aquatic environment where fewer alternative prey fish are available and juvenile salmonids are consumed in greater proportion on a per capita basis relative to cormorants nesting on ESI in the marine zone of the CRE. Estimates of per capita (per breeding pair) predation rates on salmonid smolts by cormorants nesting on the AMB in 2022 were 2–4 times greater (depending on salmonid ESU/DPS) than average annual per capita predation rate by cormorants that nested on ESI prior to implementation of management actions in 2015. Estimates of colony-wide (all breeding pairs) predation rates by cormorants nesting on the AMB in 2022 ranged from 3.1% (2.1–7.9%) for SR fall Chinook salmon to 8.6% (3.2–15.1%) for UCR steelhead. Colony-wide estimates of predation rates by cormorants nesting on the AMB in 2022 were similar to, or greater than, those of cormorants that formerly nesting on ESI, despite the much smaller size of the cormorant colony on the AMB (4,054 pairs in 2022) compared with the cormorant colony on ESI (12,787 pairs on average during 2003–2014).

For the first time since the colony was discovered in 2012, predation rates were estimated for cormorants nesting at a colony on transmission towers near the town of Troutdale, OR in 2022. The colony has rapidly increased in size, from 26 breeding pairs in 2013 to 353 breeding pairs in 2022. The Troutdale transmission tower (TRT) colony is located in the freshwater zone of the lower Columbia River, 44 river kilometers (Rkm) downstream of Bonneville Dam. Per capita predation rates on salmonid smolts by TRT cormorants were significantly higher compared to those of cormorants nesting on ESI (10–20 times higher, depending on the ESU/DPS) and compared to those of cormorants nesting on the AMB (4–8 times higher, depending on ESU/DPS); colony-wide predation rates for cormorants nesting at the TRT colony ranged from 0.7% (0.1–2.0%) for SR fall Chinook to 4.4% (1.2–9.5%) for SR sockeye in 2022. Collectively, our results indicate that colony location is closely related to the magnitude of cormorant predation on salmonid smolts and its effects on smolt survival in the CRE and lower Columbia River. Due to the movement of cormorants from the marine-zone to the mixing- and freshwater-zones of the CRE and lower Columbia River, the impact of cormorant predation on smolt survival has remained unchanged or has increased since cormorant management actions on ESI were first implemented in 2015.

An investigation of predation on salmonid smolts by piscivorous colonial waterbirds nesting at other unmanaged colonies has indicated that smolt consumption rates by gulls and smolt predation rates by cormorants nesting at certain colonies in the CPR were higher than those of nearby managed tern colonies in 2022. Unlike terns and cormorants, 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 5.7% (2.7–11.8%) for SR steelhead and 6.6% (3.3–11.4%) for UCR steelhead were documented for gulls nesting at Badger Island in McNary Reservoir and Miller Rocks in The Dalles Reservoir, respectively. Estimates of cormorant predation rates as high as 3.7% (2.0–7.0%) for SR spring/summer Chinook were documented at the Foundation Island colony in McNary Reservoir. Estimates of consumption/predation rates by gulls and cormorants nesting at colonies in the CPR during 2022 were generally lower than those observed in 2021, but similar to those in several other years dating back to 2008. Despite the relatively high predation/consumption rates on smolts observed at some gull and cormorant colonies in the CPR during 2022, impacts from piscivorous waterbirds nesting at several other colonies in the CPR were low to non-existent. For example, estimates of predation/consumption rates by cormorants nesting on Shoal Island in Lenore Lake, by gulls nesting on Goose Island in Potholes Reservoir, by American white pelicans nesting on Miller Sands Spit in the CRE, and by Brandt's cormorants (*Phalacrocorax penicillatus*) nesting on the AMB in the CRE were often < 0.2% per colony, per salmonid ESU/DPS in 2022. These were also all relatively large colonies (hundreds to thousands of breeding adults) and results indicate that not all piscivorous waterbird colonies in the CRB pose a threat to smolt survival.

An investigation of cumulative predation/consumption rates for SR and UCR salmonid smolts by piscivorous waterbirds from all colonies combined indicated that avian predation/consumption rates represented a substantial proportion of available steelhead smolts; an estimated 25.3% (21.9–30.2%) of SR steelhead and 29.4% (24.1–36.2%) of UCR steelhead were consumed by piscivorous colonial waterbirds in 2022. Cumulative predation/consumption rates were significantly lower for salmon ESUs, with an estimated 8.2% (6.2–10.6%) of SR sub-yearling Chinook, 11.1% (9.1–14.3%) of SR yearling Chinook, and 13.4% (10.5–18.8%) of SR sockeye consumed by birds in 2022. By river reach, predation/consumption rates were highest for smolt between Bonneville Dam and the Pacific Ocean, followed by Rock Island Dam (for UCR smolts) or Lower Monumental Dam (for SR smolts) to McNary Dam for most of the salmonid ESUs/DPSs evaluated in 2022. Reach-specific predation/consumption rates in 2022 were similar to those of previous years, although estimates in previous years often did not include predation/consumption rates for all active piscivorous waterbird colonies, resulting in minimum estimates of the cumulative level of predation/consumption in some years. Collectively, results indicate that the cumulative effects of avian predation/consumption remain a substantial source of smolt mortality, particularly steelhead mortality, during outmigration to the Pacific Ocean.

Previously published research has demonstrated that predation by terns was largely an additive source of mortality for steelhead smolts, with significantly more steelhead estimated to survive outmigration in the absence of tern predation. Results of the relationship between gull consumption and smolt survival in these same studies, however, were inclusive as of 2018. In 2022, we updated the time-series available for these analyses to include data from 2019-2022 and we investigated the relationship at a smaller spatial-scale to evaluate to what degree consumption by gulls was associated with lower steelhead survival during smolt outmigration from McNary Dam to Bonneville Dam. Our results indicated that higher levels of gull consumption of both UCR and SR steelhead smolts were, on average, associated with lower levels of steelhead survival. The relationship was statistically significant for both steelhead DPSs when data from all years were considered. The statistical power to accurately determine to what degree smolt consumption by gulls influenced smolt survival was, however, limited by uncertainty in estimates of gull consumption rates and a lack of weekly variation in estimates of gull consumption rates in some years. As a result, estimates of the relationship between gull consumption and steelhead survival, although statistically significant, were imprecise. With these caveats in mind, our results provide evidence that some proportion of the steelhead smolts consumed by gulls that forage between McNary and Bonneville dams would have survived outmigration to Bonneville Dam in the absence of gull consumption. Because gulls will readily consume stunned/disoriented, dead, and moribund fish, however, additional research to understand to what degree gull consumption of smolts in the tailrace of dams and elsewhere represents additive mortality, and therefore limits smolt survival, is warranted.

Lastly, as part of an ongoing analysis, we investigated the effects of biotic and abiotic factors or covariates on steelhead smolt predation by terns and smolt survival from Rock Island Dam to McNary Dam during 2008-2021. Analyses represent a novel approach to assess a covariate-parameterized capture-recapture-recovery model and to assess covariates in the additive-mortality framework. We used the additive-mortality framework to evaluate possible covariate associations with respect to not only measures of tern mortality, but also their potential association with "baseline" levels of morality (i.e. mortality in the absence of tern predation). Analyses also uniquely investigated data on a daily basis, necessitating an estimation of smolt passage times (days until exposure) to better characterize the environmental conditions experienced by smolts during outmigration. Results indicated that hatchery smolts were more likely to be consumed by terns and were less likely to survive outmigration to McNary Dam. Larger-sized smolts were also more likely to be consumed by terns than smaller-sized smolts. Other, non-tern sources of mortality were also associated with steelhead smolt survival to McNary Dam, including a fish's rear-type, size, and fish condition (injured or diseased). No statistically significant relationship, however, was identified amongst the abiotic factors investigated, including river discharge, % spill, elevation, and temperature. A lack of an identifiable relationship with abiotic factors could be due to uncertainty associated with estimated smolt exposure times or due to other, unidentified factors that influenced smolt survival, but that were not considered as part of the analysis. As part of future covariate studies we will build on the modelling effort developed herein to explore associations between smolt migration speed, survival, and predation and to better investigate the degree to which a fish's passage route at a dam and across multiple dams, influence downstream recapture, predation, and survival probabilities.

BACKGROUND

Avian predation on out-migrating juvenile salmonids (*Oncorhynchus* spp.) 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 in the Columbia River basin (CRB). Addressing the impact of avian predation on juvenile salmonid (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 ESI 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 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 ESI 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, terns and cormorants from managed colonies in the CRB are relocating to nest in large numbers at other colony sites within the basin where their per capita (per bird) impacts to smolt survival are as high or higher than at the managed colony sites. Second, management to reduce the size of the tern and cormorant colonies on ESI have led to complete colony failure/abandonment which is contributing to the movement of birds from managed to unmanaged sites within the CRB and may be affecting the conservation status of the managed populations/species. 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 increase 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 2022 were to (1) assess the distribution and size of piscivorous waterbird colonies, (2) estimate colony-specific predation/consumption rates, and (3) estimate that cumulative effects of predation/consumption (predation by all colonies combined) on juvenile salmonid survival. As part of these objectives in 2022 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 and colonies, plus survival rates for UCR steelhead, SR yearling Chinook, SR sub-yearly Chinook, and SR sockeye smolts; (d) estimated the additive effects of gull consumption on smolt survival; and (e) investigated the biotic and abiotic factors that influenced steelhead smolt survival. In addition to these primary objectives and tasks, we increased efforts to recover (detect) smolt PIT tags at piscivorous colonial waterbird nesting, loafing, and roosting sites in the CRE to provide data for use in regional smolt survival models (data used by our team, NOAA Fisheries, and other researchers).

METHODS & ANALYSES

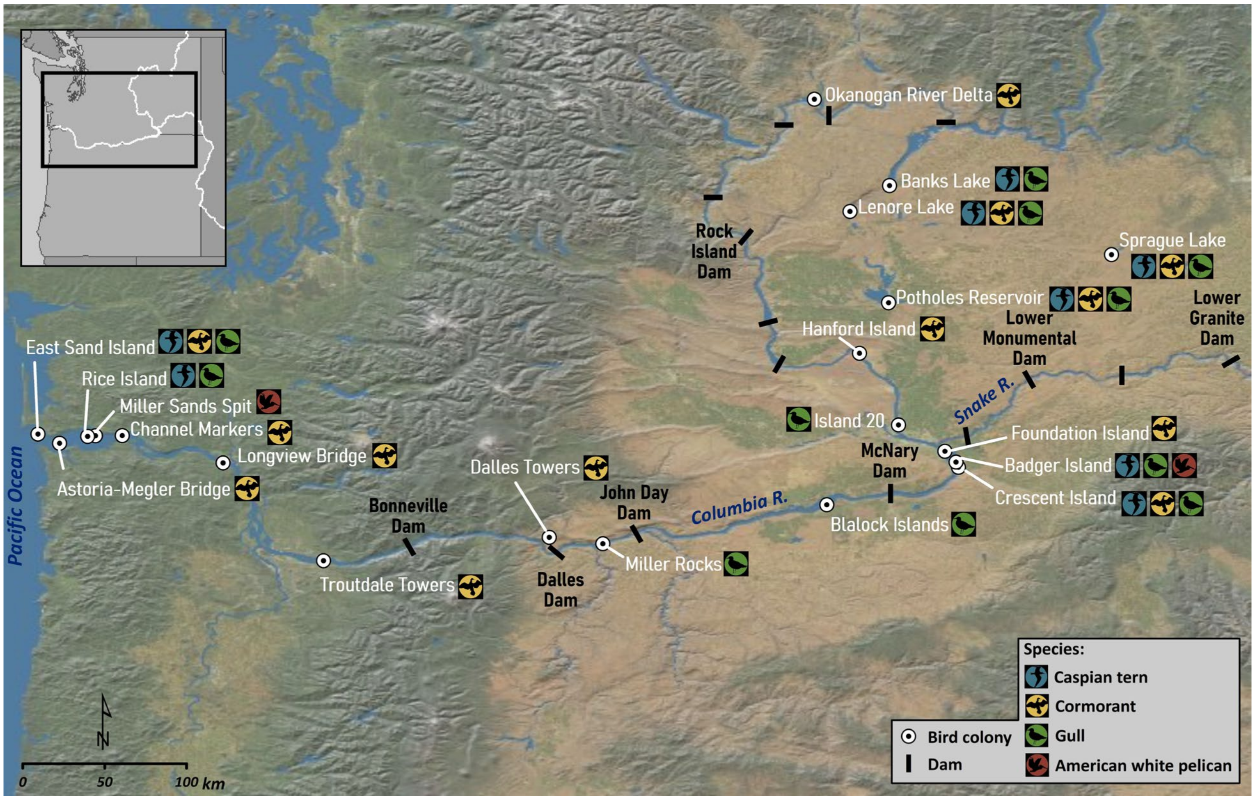
This work is part of a comprehensive program to implement, monitor, and evaluate avian predation management plans (including adaptive management) to reduce the impacts of avian predators on the survival of ESA-listed juvenile salmonids in the CRB. Action effectiveness monitoring was conducted in the CRB at both managed (i.e. East Sand Island, Crescent Island, Goose Island, and elsewhere in northern Potholes Reservoir) and at unmanaged sites where terns and cormorants might disperse to re-nest in response to management. The focus of this work 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 2022 (see *below*) 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 ACTIVITIES

In 2022, nest dissuasion activities on Goose Island and elsewhere in Potholes Reservoir were conducted by the BOR and its contractor, U.S. Department of Agriculture-Animal and Plant Health Inspection Service-Wildlife Services (USDA-APHIS-WS). These activities were summarized in an annual report completed by that agency (USDA-APHIS-WS 2022). The Fisheries Field Unit (FFU) at the USACE carried out nest dissuasion efforts outside the designated tern colony area on East Sand Island, and as an adaptive management component of the plan, on Rice Island in 2022, a description of those activities was summarized in two reports produced by that agency (USACE-FFU 2022a, 2022b). Nest dissuasion also occurred on sections of the AMB that were slated for repair/painting in 2022 (M. Alex, USDA-APHIS-WS, personal communication).

NESTING DISTRIBUTION & COLONY SIZE

Monitoring of piscivorous waterbird colonies in the CRB (*Map 1*) were conducted from the air (fixed-wing aircraft or drone), a boat, and/or from shore, with precautions taken to minimize disturbance to actively colonial waterbirds. 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 2022 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. 2021a; Evans et al. 2022b).



Map 1. Study area in the Columbia River basin in 2022.

Periodic (weekly to monthly) ground- and boat-based surveys were carried out throughout the breeding season to identify where piscivorous colonial waterbirds were nesting in the CRB. Three fixed-winged aerial surveys (8 May, 13-14 May, and 6-7 June) and numerous drone-based surveys were conducted to help identify all active nesting colonies of piscivorous waterbirds in the region, to assess nesting chronology, and to estimate peak 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 (i.e. incubating) 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 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. The remaining piscivorous waterbird colonies (*Map 1*) in the CPR were monitored one to three times per month throughout the breeding season. The cormorant colony on the Astoria-Megler Bridge in the Columbia River estuary was monitored by boat during three different occasions (9 May, 26 May, and 15 June) to determine nesting chronology and peak colony size. Finally, the size of the tern and cormorant colonies on East Sand Island and the tern colony on Rice Island were estimated by the USACE-FFU in 2022 (USACE-FFU 2022a, 2022b).

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 predation rates prior to and during management actions, with a focus on data collected in 2022. Comparisons between current (2022) and previous predation rates (2007-2021) were made in the context of management initiatives for terns in CPR 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 investigated the cumulative effects of avian predation (predation from multiple piscivorous colonial waterbird species and colonies combined) on PIT-tagged smolts and compared mortality due to avian predation to total smolt mortality (1 - survival). Results provided important information on the system-wide effects of avian predation and identified which predator species and colonies pose the greatest potential threat to smolt survival in the CRB. The analytical framework and groups of fish used to estimate the cumulative effects of avian predation differ in various ways from those used to estimate ESU/DPS-specific predation rates (see *below* for more details).

PIT-tagging of Upper Columbia River Steelhead

To ensure adequate numbers of ESA-listed 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 2022. Previously research has demonstrated that the UCR steelhead population is highly susceptible to predation by terns, cormorants, and gulls (Evans et al. 2012, Evans et al. 2019, Payton et al. 2020) and is therefore a suitable group to evaluate the efficacy of management actions aimed at reducing avian predation. Efforts to tag steelhead smolts at RIS as part of avian predation studies have also been ongoing since 2008, providing a long-term dataset in which to evaluate relative changes in predation rates associated with both managed and unmanaged piscivorous waterbird colonies (Evans et al. 2019, Evans et al. 2022b). The tagging of steelhead smolts at RIS also provide a means to investigate factors that influence smolt susceptibility to avian predation (e.g., fish length, condition; Hostetter et al. 2023) and to determine to what degree avian predation is an additive versus compensatory source of mortality (Payton et al. 2020; see also *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 July. 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). 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 naturally 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; 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 2022. 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 was estimated to generate predation rates estimates with a precision of approximately ± 2% for those colonies that forage on smolts in the middle Columbia River, like the managed tern colony on Goose Island in Potholes Reservoir.

For several 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 on the lower Snake River or upstream of McNary and Bonneville dams on the Columbia River. 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 (Roby et al. 2021a). In brief, to recover (electronic detection) fish 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. 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 $\varepsilon_w \sim normal(0,\ \sigma_\theta^2)\ \forall\ w$. 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:

$$\left. \sum_{i \in breeding\ season} (\theta_i * n_i) \middle/ \sum_{i \in breeding\ season} (n_i) \right.$$

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).

For some colonies included in the study, not all nesting areas or habitat used by birds during the breeding season were accessible to researchers (e.g., truss sections of the Astoria-Megler

Bridge cormorant colony) or not all accessible areas were scanned following the breeding season (e.g., Troutdale Transmission Tower cormorant colony and Goose Island gull colony). In these few cases, per capita predation rates ($\dot{\theta}_p$) were calculated in areas or plots (p) that were scanned for PIT tags and where a known number of birds nested as:

$$\dot{\theta}_p = \sum_p \sum\nolimits_i \left(\theta_{ip} * n_i\right) / \sum\nolimits_i \left(n_i\right) \Big/ C_p$$

where C_p is colony size within plot p. Colony-wide estimates of predation (predation by all breeding pairs at that colony) were then calculated by multiplying the sum of the per capita predation rates by the peak colony-wide measure of colony size as:

$$\theta = \frac{\sum_i \sum_p \left(\theta_{ip} * n_i\right) \Big/ \sum_i n_i * C_{all}}{\sum_p C_p}$$

Models were analyzed using the software STAN (2022), accessed through R version 3.6.3 (RDCT 2014), and using the rstan package (version 2.30; SDT 2022). 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. Data to address the efficacy of these management plans varied by plan, funding, and year and are detailed below:

Inland Avian Predation Management Plan (IAPMP) – 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, per year (USACE 2014). To help evaluate the efficacy of the *IAPMP* at reducing predation impacts to those levels, predation rates were compared between the pre-management period (2007–2013) and the management period (2014–2022) at both managed and unmanaged tern colonies in the CPR.

Estuary Caspian Tern and Double-crested Cormorant Management Plans – A stated goal of estuary tern and cormorant management plans were to reduce the size of the tern and cormorant colonies on East Sand Island by about 60% and thereby reduce tern and cormorant predation rates on ESA-listed salmonids in the CRE by about 60% (USFWS 2005, USACE 2015). The USACE and its contractors recovered smolt PIT tags from both the tern and cormorant colonies on East Sand Island in 2022 (USACE-FFU 2022a) but they had not estimated per capita and colony-specific predation rates at the time this report was written. Similarly, tags were also recovered by the USACE in 2021, but no predation rate analyses using these data have occurred.

While salmonid ESU/DPS-specific rates by terns and cormorants on East Sand Island in 2022 are currently unavailable, as part of this study, we incorporated smolt PIT tags recovered by the USACE on East Sand Island to evaluate the cumulative effects of avian predation on select groups of PIT-tagged smolts in 2022. Estimates of cumulative predation effects, however, rely on a different analytical framework (see *Cumulative Predation and Survival* section below) than those of colony- and ESU/DPS-specific predation rates (see *Predation Rate Estimates* section above), so the two estimates are not directly comparably to one another, although the two methods often generate similar estimates of predation at the colony-specific level (Payton et al. 2019, Evans et al. 2021).

Cumulative Predation and Survival: We evaluated the cumulative and reach-specific effects of avian predation on UCR steelhead tagged at or detected (i.e. previously-tagged recaptures) at RIS and on SR steelhead, yearling Chinook, sub-yearling Chinook, and sockeye that were tagged/detected at Lower Granite Dam (LGR) in 2022. Downstream river reaches were defined by locations where PIT-tagged smolts were detected alive following passage at RIS and LGR, which include Little Goose Dam and Lower Monumental Dam (LMN) for SR migrants and McNary Dam (MCN), John Day Dam (JDJ), Bonneville Dam (BON) and a net trawl detection system downstream of BON for both SR and UCR migrants. Predation rates were based on the proportion of available smolts consumed by birds within each river reach or from all reaches combined, and survival rates were based on the proportion that survive out-migration through each river reach or all reaches combined. In addition to avian predation, 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. 2016, Payton et al. 2019). As such, estimates of avian predation were compared to estimates of total mortality (1-survival) to determine what proportion of all mortality sources were due to bird predation. To help describe recent trends in predation and survival, results from 2022 were compared with those from years past, including the results of Evans et al. (2022a), which date back to 2008.

Tagged smolts included in these analyses were both hatchery and wild (natural origin) fish. Not all hatcheries are included in the ESA-defined ESUs/DPSs, and in the case of Chinook salmon, yearlings may be a mixture of two distinct ESA-listed populations (spring- and fall-run; NOAA 2021); thus, populations reflected the same species and age-classes but some unknown proportion of the fish were likely not part of the ESA-listed ESU or were a mixture of two ESA-listed ESUs. Only in-river out-migrating smolts within each river reach were included, with all transported smolts excluded following, but not prior to, their removal from the river in fish barges or trucks. Adults returning to the Columbia River following ocean residency were also detected at PIT tag arrays located in fishways at Bonneville Dam, the first dam encountered by

UCR and SR adults following ocean residency (see also Evans et al. 2022a for additional details).

The joint mortality and survival (JMS) estimation methods of Payton et al. (2019) were used to independently estimate reach-specific and cumulative predation and survival probabilities for each salmonid species and population (hereafter simply "population") evaluated. This hierarchal state-space Bayesian model incorporated both live and dead detections of PIT-tagged fish in space and time to simultaneously estimate consumption and survival through up to nine sequential river reaches (or segments), defined by passive recapture opportunities in which smolts were assumed to only travel downstream. In brief, the model used two vectors, $\boldsymbol{y}$ and $\mathbf{r}$, to describe each fish's recapture and recovery history throughout each downstream river reach and each of the bird colony recovery sites under consideration. Each vector $\mathbf{y}$ was a J-length vector –$J = 9$ for SR fish and $J = 5$ for UCR fish– where y_j was an indicator variable of a fish's recapture at recapture opportunity j for $j \in \{1,2,\ldots,J-1\}$ and $y_J = 0$ as there was no live recapture site downstream of the net detector in the Columbia River estuary. Recoveries were indicated by $\mathbf{r}$, a D-length vector, where D represents the number of recovery areas each year, with a single element equal to one and the rest of the elements are zero, where $r_d = 1$ indicated recovery on colony *d* for $d \in \{1,2,\ldots,D-1 = 14\}$, and $r_D = 1$ indicated a fish was unrecovered. Parameters used in the model included:

> $\boldsymbol{\Theta}$**,** a DxJ matrix where $\Theta_{d,j}$ represented the probability (from release) that a fish survived to recapture opportunity *j-1* –where $j = 0$ represents release from RIS/LGR– and then subsequently succumbed to depredation by colony *d* for $d \in \{1,2,\ldots,D-1\}$ or some other cause of mortality for $d = D$, prior to arrival at recapture opportunity $j + 1$. Implicit from this parameterization is that survival from release through segment k is equal to $1 - \sum_{j \leq k} \sum_d \Theta_{j,d}$.
>
> **p,** a J-length vector where p_j represented the probability that a fish alive at recapture opportunity *j* was successfully recaptured. We define $p_J = 0$, as there is no recapture opportunity downstream of the Net Detector.
>
> $\boldsymbol{\gamma}$, a D-length vector where γ_d represented the probability of recovering a fish which died due to depredation by colony *d* for $d \in \{1,2,\ldots,D-1\}$, and γ_{15} = 0 represented the lack of recovery opportunity for fish which died from all other unspecified causes.

The model employed can be expressed by incorporating these parameters into recursive functions, $\chi_{j,d}$, defined to represent the probability a fish entering segment j is not subsequently recaptured and is recovered on colony d (i.e., r_d=1), such that

$$\chi_{j,d} = \theta_{j,d} * \gamma_d + \left(1 - p_{j+1}\right) * \chi_{j+1,d} \; for \; d \in 1,\ldots,D-1,$$

or not recovered at all (i.e., r_{15}=1), such that

$$\chi_{j,D} = \sum_{d} \theta_{j,d} * (1 - \gamma_d) + \left(1 - p_{j+1}\right) * \chi_{j+1,D}.$$

Then, if we define m to be the final recapture opportunity at which the fish was seen, with $m = 0$ representing a fish never reseen following release, the portion of the aggregate likelihood associated with each fish's recapture/recovery history can be expressed as

$$L = \prod_{j \le m} \left(p_j^{y_j} * \left(1 - p_j\right)^{(1-y_j)}\right) * \prod_{d} {\chi_{m+1,d}}^{r_d},$$

where the former product describes a fish's recapture history prior to its final recapture and the latter product describes the fish's subsequent recovery or lack thereof following its final recapture.

Each year, a subset of tagged smolts were collected and removed from the river in fish barges or trucks at one of the first three capture/recapture sites on the lower Snake River: Lower Granite Dam, Little Goose Dam, or Lower Monumental Dam. Once collected for transportation, these fish were no longer available in-river and, as such, the capture-recapture-recovery history for these fish was truncated following their removal at each dam. The likelihood associated with the truncated capture-recapture history of each of these fish can be expressed as:

$$L = \prod_{j<m} \left(p_j^{y_j} * \left(1 - p_j\right)^{(1-y_j)}\right) * \left(1 - \sum_{d} \sum_{j<m} \theta_{j,d}\right)$$

Two further modelling considerations beyond those of Payton et al. (2019) were included to better inform our spatially explicit estimates of predation effects. First, the informed partitioning methods of Evans et. al (2022a) were also used to allow for a sharing of information among years to increase the precision of segment-specific estimates. In brief, a vector of aggregate life-path possibilities is constructed including the probability of survival to return as an adult, the cumulative probability (across all segments) of depredation by each colony, and segment specific probabilities of death from unspecified sources to be the basis for modelling variations across days. The cumulative probability of depredation by each colony is subsequently partitioned across river segments with proportionate impacts among reaches assumed to be similar among years. Second, pelicans and gulls established multiple nesting areas on Badger Island (BGI) during 2015–2022, with portions of each genus' colonies overlapping spatially creating a "mixed" or co-nesting area on BGI. We employed the methods of Payton et al. (2023) to incorporate supplemental data (i.e. aerial nest count surveys) to inform what proportion of each genus was nesting in the "only" areas versus the "mixed" areas. Then, by assuming the odds of a tag consumed by a given genus was deposited in the single genus portions of each colony versus in the "mixed" area was similar to the odds of a bird of that genus nesting in the single genus portion, we were able to estimate the portion of tags recovered from the "mixed" portion of the colony were attributable to each predator genera.

To measure inter-annual temporal variation in probabilities, fish were partitioned into weekly release groups with the assumption that fish released within the same week experienced similar rates of mortality/survival, recapture, and recovery (see Payton et al. 2019). While all rates were assumed to be independent among years, weekly cohorts closer in time were assumed to be more alike than those further apart. The serial correlation in probabilities were assumed and accounted for as described by Payton et al. (2019). The prior distribution for the initial week's detection probability in each year was defined to be $\mathrm{uniform}(0,1)$. Analogously, the prior distribution assigned for the life paths simplexes in the initial week of each year was assumed to be $\mathrm{D}(\mathbf{1})$, where $\mathbf{1}$ was an appropriately sized vector of ones. Weakly-informative priors of $\mathrm{half}-\mathrm{normal}(0, 1.5)$ were implemented for the variance parameters describing inter-weekly variation.

The recovery parameters, γ_d, represent the combined probability that a consumed tag was deposited on-colony, *d*, and the probability that the tag is subsequently detected (recovered) by researchers following the breeding season given tag deposition on a colony. The simulated posterior distributions of deposition probabilities and colony-specific detection probabilities which were derived, summarized, and presented in previous studies were employed here as informative prior distributions in the derivation of predation probability estimates. Informative prior distributions used in this study are provided as Supplemental Materials (see also Evans et al. 2022a for recovery probabilities from each colony in each year).

Models were analyzed using the software STAN (SDT 2022), accessed through R version 3.6.2, and using the rstan package (version 2.19.3). To simulate random draws from the joint posterior distribution, we ran four Hamiltonian Monte Carlo (HMC) Markov Chain processes. Each chain contained 4,000 warm-up iterations followed by 4,000 posterior iterations thinned by a factor of 4. Chain convergence was visually evaluated and verified using the Gelman-Rubin statistic (Gelman et al. 2013); only chains with zero reported divergent transitions were accepted. Posterior predictive checks compared simulated and observed annual aggregate raw recapture and recovery numbers to ensure model estimates reflected the observed data. Reported estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% Credible Interval [CRI]) calculated using the HDInterval package (version 0.2.0).

Additive Effects of Predation: The degree to which avian predation limits fish survival is paramount to evaluating the efficacy of management actions to increase fish survival. In particularly, understating if reductions in predation rates are associated with higher rates of fish survival (i.e. avian predation adds to total mortality) or if most fish consumed by birds are 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).

Previously published research indicated that predation by terns was an additive source of smolt mortality during the smolt life-stage and a partially additive source of mortality to adulthood, with significantly more smolts estimated to survive outmigration to the Pacific Ocean and to return to Bonneville Dam as an adult in the absence of tern predation (Payton et al. 2020, Payton et al. 2021, Evans et al. 2022a). Results of the relationship between gull predation and fish survival, however, were inclusive using data collected during 2013-2018 (Payton et al. 2021). In 2022, to further investigate the relationship between gull predation and fish survival we (1) updated the time-series to included data from 2019-2022, (2) we investigated the relationship between gull predation and steelhead survival at a smaller spatial-scale (McNary Dam to Bonneville Dam), (3) we analyze predation by gull colonies (Blalock islands and Miller Rocks; see *Map 1*) that presumably kleptoparasitize very few smolts from terns, and (4) we modified the model assumptions of Payton et al. (2020) to better reflect the system under consideration (see *below*).

Weekly and annual estimates of predation and survival probabilities on PIT-tagged UCR and SR steelhead smolts that were initially tagged/recapture/released at RIS and LGR (see *above*) and survived passage to McNary Dam (MCN) were used to investigate to what degree gull predation was an additive source of mortality during outmigration to Bonneville Dam (BON). We use the mark-recapture-recovery model of Payton et al. (2020) to assess the strength, magnitude, and direction of the relationship between gull predation on steelhead smolt survival between MCN and BON and steelhead smolt survival to BON during 2013-2022. The time series commenced in 2013 because this was the first year the gull colonies from both the Blalock Islands and Miller Rocks were scanned for smolt PIT tags. 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 McNary Dam (MCN) to Bonneville Dam (BON). We aggregated 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 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 simplex weekly rates of survival and aggregated predation as,

$$\begin{bmatrix} s_w^* & \theta_w^{\{GENUS\}} & \theta_w^{\{OTHER\}} \end{bmatrix}^T.$$

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 observed in survival is assumed to be proportional to the level of genus-specific predation associated with that reach. The magnitude of this proportional relationship is measured with the parameter $a.$ However, in cases in which $\theta_w^{\{GENUS\}}$ accounts for all unmeasured mortality, the relationship between survival and predation necessarily becomes directly inverse. In cases in which survival is zero, there is necessarily no relationship between predation and survival. Therefore, "observed" weekly survival, s_w^*, can be expressed as

$$s_w^* = \begin{cases} 0;\ if\ \Theta_w^{\{\mathrm{GENUS}\}} < \dfrac{s^0 + \epsilon_w}{a} \\ 1 - \Theta_w^{\{\mathrm{GENUS}\}} - \Theta_w^{\{\mathrm{OTHER}\}};\ if\ \Theta_w^{\{\mathrm{GENUS}\}} > \dfrac{1 - (s^0 + \epsilon_w) - \Theta_w^{\{\mathrm{OTHER}\}}}{1 - a} \\ s^0 + \epsilon_w - a\Theta_w^{\{\mathrm{GENUS}\}};\ \ o.w. \end{cases}$$

An inherent consequence of the bounded nature of the parameter space is that any portion of joint posterior distribution of s_w^* and $\Theta_w^{\{\mathrm{GENUS}\}}$, which abut the parameter space boundary, is associated with little to no added information respecting the values of the additivity parameters, s^0 and a. Such considerations were of negligible consequence in our previous research (where large spatial scales were explored). However, with the smaller spatial scale considered in the analysis and the reduced level of unmeasured mortality, much greater portions of the joint posterior distribution will abut the parameter space boundary. Results herein are still presented with respect to a for the sake of comparability with previously published studies.

In the previous research cited above, we made the very weak assumption that baseline survival, s^0, was completely independent from year to year while letting a be "similar" among years (i.e. $a_y \sim normal(\mu_a, \sigma_a)$ for each year y). However, smolt survival among most reaches in the CRB is relatively similar among years rather than uniformly distributed between zero and one each year. We therefore model this similarity $as\ s_y^0 \sim normal(\mu_{s^0}, \sigma_{s^0})$ for each year y. This adjusted assumption provokes a reassessment of our assumptions with respect to additivity, as additivity is inherently a function of baseline survival, wherein the portion of fish "destined to die" is related to the baseline survival rate. Payton et al. (2020), reparametrized the Sandercock et al. (2011) model, in order to directly measure the "portion of depredated fish that would not have died if not for avian predation", defined as the parameter $a.$ As a result, the parameter a encompasses both the portion fish "destined to die" and the extraneous super-additive/compensatory factors which may be impacting the system (e.g., latent mortality, kleptoparasitism; see Payton et al. 2020). For this analysis we disentangle these processes by defining $a = s^0 * b$, where b measures these latter effects; effectively reverting the parameterization to the original Sandercock et al. (2011) model. We thus assume b is also similar among years; $b_y \sim normal(\mu_b, \sigma_b)$. Results herein are still presented with respect to a for the sake of comparability with previously published studies. To evaluate the consequences of these strengthened model assumptions, we reanalyzed the additive effects of tern predation on steelhead smolts from RIS (for UCR smolts) and LMJ (for SR smolts) to BON (the full spatial scale over which tern predation occurred) with both these assumptions and those of Payton et al. (2020).

Biotic and Abiotic Factors: It is well documented that biotic and abiotic factors or conditions experienced by smolts during outmigration 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. 2023). Furthermore, even within the same salmonid population, differences in predation based on a fish's rear-type (hatchery, wild), size (fork length), condition (presence or absence of injuries and other types of damage), abundance, 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. 2023). Environmental factors experienced by smolts also vary within and across years, including hydroelectric dam operations (e.g., spill and discharge) and measures of river flow and speed, factors that may be directly or indirectly related to smolt survival (Petrosky and Schaller 2010, Hostetter et al. 2023). 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 avian predation or other factors that limit smolt survival.

As part of this ongoing analysis, using data collected from 2008-2021, we investigated which biotic and abiotic factor(s) best explained variation in UCR steelhead survival during smolt outmigration from RIS to MCN. Because previous research indicated that mortality associated with tern predation was substantial in this river reach, and was largely an additive source of steelhead mortality (Payton et al. 2020), covariates were investigated with and without the effects of tern predation. The association of covariates with predation presumably differs by predator species (tern, cormorant, gull, pelican), so we limited this analysis to predation by terns due to their disproportionate impacts on UCR steelhead smolts (Evans et al. 2012). Tern colonies evaluated included those on Goose Island and other islands in Potholes Reservoir, islands in Banks Lake and Lenore Lake, and Badger Island and Crescent Island in McNary Reservoir (see *Map 1*). Biotic covariates investigated included fish size (mm, fork length), rear-type (hatchery, wild), and external condition (classified as compromised {fish with injuries, descaling, fin damage, or disease} or not compromised {fish that lack these anomalies}; see also Evans et al. 2014). Abiotic covariates included river discharge, spill percentage, elevation, and temperature; all quantitative metrics that were regularly measured at Wanapum Dam, Priest Rapids Dam, and an in-river monitoring station near Pasco, WA. Data were obtained from the Data Access in Real Time website (DART 2022).

Our ability to address the effects of abiotic factors on smolt survival and/or tern predation may be limited by the ability to accurately characterize the environmental conditions experienced by smolts during passage from RIS to MCN; a large spatial-scale, with multiple dams and reservoirs. To best characterize where and when environmental factors were experienced by smolts, it was necessary to account for the time elapsed between a fish's release from RIS and the time of exposure to each covariate factor assessed. Similarly, it was necessary to account for the time elapsed from release to the time of depredation by an avian predator. In previous investigations of predation on UCR steelhead, smolt releases have been partitioned into weekly cohorts with an assumption that any depredated smolts was consumed – and the tag deposited on-colony – within the same week of release. For this more granular modelling effort we partitioned smolts into daily cohorts, necessitating an estimation of time until exposure to predators from each colony. To address both concerns, we constructed probabilistic estimation of the day of exposure for all smolts within each reservoir and for each avian colony using date-of-detection data from acoustic tagged (AT) steelhead during studies in 2008-2010, 2014-2016 (Evans et al. 2022a).

For river metrics, we collected the observed time-to-exposure distributions, comprising the elapsed travel times (measured in days) from release until detection at the AT array in closest proximity to each given dam or monitoring station. For tags recovered on tern colonies, we collected the observed distribution of elapsed travel times from release until last successful detection at an AT array with the assumption that the smolt was then consumed within the same day as their last detection. Due to the limited number of AT tags recovered on some of the colonies evaluated, we combined the distributions of multiple colonies based on geographic similarity. For example, the time-to-exposure data for tags recovered from islands in Banks Lake and Lenore Lake were combined, data for tags recovered from Goose Island and an unnamed island in Northeast Potholes Reservoir were combined, and data for tags recovered from Badger Island and Crescent Island were combined. For each time-to-exposure dataset, we fit a Poisson distribution using the fitdistrplus R package (Muller and Dutang 2015). With these estimates, we were able to assign probabilities for possible days until exposure for each river covariate and avian colony following a smolts release from RIS. Specifically, we constructed a geometric average of each river metric, with weights defined by the probability a released smolt was exposed to that covariate on each day following release. Further, we constructed a geometric average of the probability of detecting a deposited tag on each colony, with weights defined by the probability a released smolt was consumed by a tern from that colony each day following release.

To begin the investigation of which covariates were most closely associated with variation in smolt survival and tern predation probabilities, we employed simplified versions of the JMS models of Payton et al. (2019) and Payton et al. (2020; see also *above*), evaluating a single river segment, such that smolts released from RIS could potentially be recaptured at MCN, potentially recovered from any of six tern colonies known to forage upstream of MCN, and potentially recaptured or recovered downstream of MCN. To add covariates to the JMS parameterization, we assumed, θ_{c_i} represents the probability that smolt i is depredated by a tern from colony c prior to arrival MCN and θ_{M_i} represent the probability that smolt i succumbs to some other mortality source. We model the inter-weekly variation in these parameters using AR(1) logistic regression model, evaluating the relative log-odds of these hazards vs survival to MCN. That is, given the passage of smolt i on day d_i, we model the probabilities of smolt mortality prior to MCN as

$$\log\left(\theta_{c_i}\right) - \log\left(S_{1_i}\right) = \mu_{\theta_c} + \boldsymbol{x}_{\theta,i}^{T}\boldsymbol{\beta}_{\theta} + \sum_{j<d_i} \epsilon_{\theta_c,j} \qquad for\ c \in \{0,\dots,C\}$$

and

$$\log\left(\theta_{M_i}\right) - \log\left(S_{1_i}\right) = \mu_{\theta_M} + \boldsymbol{x}_{\theta_M,i}^{T}\boldsymbol{\beta}_{\theta_M} + \sum_{j<d_i} \epsilon_{\theta_M,j}$$

where $\mathbf{x}_{\theta}$ and $\mathbf{x}_{\theta_M}$ consist of the rows of $\mathbf{X}_{cov}$ associated with the covariates applicable to process, $\mathbf{x}_{\theta}$ and $\boldsymbol{\beta}_{\theta}$ are equivalent for all tern colonies, and $\boldsymbol{\epsilon}_{\theta_c} \sim \text{normal}\left(0, \sigma_{\theta_c}\mathbf{I}\right)\ \forall c$ and $\boldsymbol{\epsilon}_{\theta_M} \sim \text{normal}\left(0, \sigma_{\theta_M}\mathbf{I}\right)$.

Similarly, we model the probabilities of smolts which survived to MCN being subsequently recaptured (i.e. interrogated) at MCN, $\boldsymbol{p}$, as

$$\mathrm{p}_i = \text{logit}^{-1}\left(\mu_p + \boldsymbol{x}_{p,i}^{T}\boldsymbol{\beta}_p + \sum_{j<d_i} \epsilon_{p,j}\right)$$

the probabilities of smolts which survived to MCN being subsequently recovered or recaptured after MCN, $\boldsymbol{\nu}$, as

$$\nu_i = \text{logit}^{-1}\left(\mu_\nu + \boldsymbol{x}_{\nu,i}^{T}\boldsymbol{\beta}_\nu + \sum_{j<d_i} \epsilon_{\nu,j}\right)$$

where $\boldsymbol{\epsilon}_{p_1} \sim \text{normal}\left(0, \sigma_{\mathrm{p}_1}\mathbf{I}\right)$ and $\boldsymbol{\epsilon}_{\nu} \sim \text{normal}(0, \sigma_{\nu}\mathbf{I})$.

We first assessed which biotic covariates demonstrated significant associations with each parameter. We then evaluated possible covariate associations with the probabilities of recapture at MCN, then possible covariate associations with the probabilities of predation by terns, and finally we evaluated possible covariate associations with other, non-tern sources of mortality. Upon the construction of a full biotic covariate model, we then evaluated the statistical evidence of associations between the tern predation and mortality parameters and the abiotic covariates.

Estimation of the probability of recapture at MCN relied on data provided by downstream detections. The probability of detection downstream of MCN may have associations with the covariates evaluated. To avoid any possible bias which may be introduced by neglecting to include the relevant covariates, we simply assumed a saturated model for the probability of downstream recapture/recovery. For every other parameter evaluated, we used a stepwise approach to building a covariate model. All covariate regression parameters significantly different from zero were included (with Bayesian p-values defined as $\min(p(\beta) \geq 0, p(\beta) \leq 0)$ and 0.05 considered significant). Regression parameter estimates may vary considerably among years such that their average impacts are no different from zero. We therefore started with analyzing parameters by year and, given they all trended in the same direction (i.e. demonstrate a direct versus inverse relationship) and without evidence of statistically significant variation among them, we aggregated to a single, across year average.

For the evaluation of possible covariate associations with other, non-tern mortality, we employed two different frameworks: (1) a standard partitioned JMS model (as expressed above) and (2) an additive-mortality JMS model. The additive-mortality JMS model allows us to differentiate between direct impacts of specific mortality sources (e.g., tern predation) on survival versus more indirect or proximate causes of mortality (e.g., fish length) as represented by covariates and classified as “other mortality”. That is, these models simultaneously measure the additive effects of tern predation, a necessarily linear relationship, and measure the extent to which variation in “baseline” survival (i.e. survival in absence of tern predation) 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 for smolts released on 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 for release 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

$$s_d = \text{logit}^{-1}\left(\mathbf{X}_{\boldsymbol{d}}^{\mathrm{T}}\boldsymbol{\beta} + \mathbf{Z}_{\mathrm{d}}^{\mathrm{T}}\boldsymbol{\epsilon}\right) - a\Theta_d^{\{\text{predation}\}}$$

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

Models were implemented using the software STAN accessed through R version 4.1.0 (RCDT 2021) using the rstan package (version 2.30 SDT 2022). 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. 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

As was the case in 2020-2021, the U.S. Bureau of Reclamation and its contractor (USDA-APHIS-WS) carried out all nest dissuasion activities of Caspian terns on Goose Island and elsewhere in Potholes Reservoir in 2022. A passive nest dissuasion array (stakes, cable, and flagging) was installed on all upland habitat that might be suitable for tern nesting on Goose Island prior to the onset of the 2022 breeding season. To create a passive nest dissuasion array that would require less in-season and between-season maintenance, the braided rope or wire that was previously used to connect the stakes was replaced by stainless steel cable, and the plastic caution tape (i.e. flagging) was replaced with UV-protected polyester flags with grommets that were attached to the cable using zip ties (USDA-APHIS-WS 2021). The passive nest dissuasion array installed in 2022 was an improvement over the array that was installed the previous year, when the flagging was observed to move down the wire in windstorms, leaving large swaths of suitable tern nesting habitat without flagging. 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-APHIS-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 the nest dissuasion efforts performed by USDA-APHIS-WS on Goose Island and elsewhere in Potholes Reservoir can be found in annual reports prepared by the contractor (USDA-APHIS-WS 2021, 2022).

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. 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. The growth of the gull colony on Crescent Island from 400 individuals in 2020 to over 2,000 individuals in 2021 led to further die back of willows and other native vegetation on Crescent Island, leaving large areas (primarily in the location of the historic tern colony) unvegetated, and hence, suitable for tern nesting (see *below* for further discussion). No active hazing of prospecting terns was conducted on Crescent Island in 2022, or in any other year since 2015.

As part of the *IAPMP*, provisions were made for adaptive management at tern colonies in the CPR that might grow as a direct result of management or for other reasons (USACE 2014). Tern colonies located at the Blalock Islands in John Day Reservoir and Badger Island in McNary Reservoir met the criteria for adaptive management outlined in the *IAPMP* beginning in 2021 at the Blalock Islands and in 2022 at Badger Island. In 2021-2022, the John Day pool level was raised during the tern breeding season to eliminate all upland habitat previously used by nesting terns at the Blalock Islands (USACE 2021,2022). At Badger Island, woody debris was spread prior to the 2022 breeding season on one of two nesting areas used by terns in 2021. The other nesting area used by terns on Badger Island in 2021 was left as is to allow unimpeded pathways for pelican chicks to access the water.

Most sections of the AMB were available for nesting cormorants in 2022. On two spans of the AMB, where active dissuasion efforts were performed during the breeding season, only 15 double-crested cormorants attempted to nest, all nesting attempts which later failed. Compared to 2021, there were over 450 breeding pairs of cormorants on these two sections of the AMB. It is presumed that cormorants which either attempted to nest or would have nested on these two sections moved to other areas of the bridge where dissuasion efforts were not occurring. Areas where dissuasion efforts occurred in 2022 are expected to be available again for cormorants to nest during the 2023 breeding season. Other dissuasion efforts on the AMB in 2023 for purposes of painting the structure are planned in 2023 (M. Alex, USDA-APHIS-WS, personal communication).

As was the case in 2019-2021, the USACE-FFU conducted tern monitoring and tern nest dissuasion efforts on East Sand Island outside the 1-acre designated colony area and on Rice Island in 2022 (USACE-FFU 2022a, 2022b). 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, active human hazing using colony walkthroughs and an autonomic laser, and the collection of any tern eggs laid (under permit). No active nest dissuasion activities were performed at the former cormorant colony on East Sand Island in 2022 (USACE-FFU 2022a). For further details on avian predation monitoring and management efforts on East Sand Island and Rice Island in 2022, see USACE-FFU (2022a, 2022b).

NESTING DISTRIBUTION & COLONY SIZE

A total of 34 active breeding colonies of piscivorous waterbirds were detected in the CRB during the 2022 nesting season. Of those, cormorant and gull colonies were the most prevalent (13 and 11 colonies, respectively), followed by terns (8 colonies), and pelicans (2 colonies; *Table 1* and *Table 2*). Most of these breeding colonies (24) were in the CPR, with 8 and 2 colonies located in the CRE and on the lower Columbia River, respectively. Certain tern and cormorant colonies continue to be managed as part of ongoing management plans and management has resulted in a shift in the nesting distribution of these birds, prompting adaptive management at several colony sites (i.e. Rice Island, the Blalock Islands, and Badger Island; see *below*). As has been the case in the past, gulls are the most numerous (ca. 48,000 individuals) of all the piscivorous colonial waterbird taxa in the CRB, followed by cormorants (ca. 6,200 breeding pairs), pelicans (ca. 4,500 individuals), and terns (ca. 2,500 breeding pairs).

Table 1. Annual colony size (peak number of breeding pairs) for Caspian terns in the Columbia River basin during 2005-2022. "Active" denotes a colony that was active but not counted to determine peak number of breeding pairs. Estimates of colony size for the East Sand Island and Rice Island tern colonies in 2020-2022 were provided by the U.S. Army Corps of Engineers – Fisheries Field Unit (see USACE-FFU 2020, 2021, 2022a, 2022b for more details).

Colony	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Columbia River Estuary																
East Sand Is.	9,623	10,668[1]	9,854[1]	8,283[1]	6,969[1]	6,416[1]	7,387[1]	6,269[1]	6,240[1]	5,915[1]	3,500[1]	4,959[1]	3,861[1]	2,387[1]	2,050[1]	1,725[1]
Rice Is.	0[1]	0[1]	0[1]	0[1]	3[1]	0[1]	0[1]	0[1]	0[1]	0[1]	0[1]	0[1]	0[1]	0[1]	Active[1]	Active[1]
Columbia Plateau Region																
Goose Is. (Potholes Reservoir)	282	293	487	416	422	463	340	159[1]	2[1]	0[1]	0[1]	0[1]	0[1]	6[1]	22[1]	16[1]
Crescent Is. (Columbia River)	355	388	349	375	419	422	393	474	0[1]	0[1]	0[1]	0[1]	0[1]	0[1]	1	149
Blalock Islands (Columbia River)	43	104	79	136	20	6	26	45	677	483	449	313	379	150	0[1]	0[1]
Badger Is. (Columbia River)	0	0	0	0	33	60	0	0	0	0	41	8	0	0	231	267[1]
Twinning & Goose Is. (Banks Lake)	31	27	61	34	19	22	13	67	64	6	0	0	0	0	0	1
Harper Is. (Sprague Lake)	0	11	4	4	4	30	1	8	10	3	92	79	18	0	85	2
North Rocks & Shoal Is. (Lenore Lake)	0	0	0	0	0	0	0	0	0	0	123	91	48	53	61	76
Total (CPR)	**711**	**823**	**980**	**965**	**917**	**1003**	**773**	**755**	**769**	**675**	**705**	**491**	**445**	**209**	**400**	**511**

[1] Nest dissuasion actions implemented to prevent or reduce nesting attempts by terns.

Table 2. Size of California/ring-billed gull (LAXX), double-crested cormorant (DCCO), and American white pelican (AWPE) nesting colonies in the Columbia River basin in 2022. "Active" denotes a colony that was active but not counted in 2022.

Colony	LAXX[1]	DCCO[2]	AWPE[1]
Columbia River Estuary			
East Sand Is.	active[3]	58[4]	
Astoria-Megler Bridge		4,054[5]	
Rice Island	active[3]		
Channel Markers		285[6]	
Miller Sands Spit			992
Lower Columbia River			
Longview Bridge		221	
Troutdale Transmission Towers		352	
Columbia Plateau Region			
The Dalles Transmission Towers (Columbia River)		67[7]	
Miller Rocks (Columbia River)	5,832		
Blalock Is. (Columbia River)	1,792		
Crescent Is. (Columbia River)	4,516	47	
Badger Is. (Columbia River)	3,502		3,486
Foundation Is. (Columbia River)		266	
Island 20 (Columbia River)	7,284		
Hanford Is. (Columbia River)		87	
Okanogan River Delta (Columbia River)		58	
Goose Is. (Potholes Reservoir)	15,583	5	
North Rocks, Shoal Is., and unnamed island (Lenore Lake)	1,717	159	
Harper Is. (Sprague Lake)	3,187	589	
Twinning Is. and Goose Is. (Banks Lake)	3,888		

[1] *Number of individuals.*
[2] *Number of breeding pairs.*
[3] *California gulls are replaced by glaucous-winged/western gulls as the dominant gull species nesting in the Columbia River estuary.*
[4] *Data provided by the U.S. Army Corps of Engineers – Fisheries Field Unit (see USACE-FFU 2022a, 2022b for further details).*
[5] *Minimum estimate, some nesting cormorants within the structure of the bridge were not visible.*
[6] *Minimum estimate, some channel markers located in the upper Columbia River estuary were not counted.*
[7] *Data provided by J. Day (U.S. Army Corps of Engineers).*

Columbia Plateau Region

Caspian Tern Colonies: In 2022, terns nested at six locations in the CPR, including Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir (both “managed colonies” as part of the *IAPMP*; see USACE 2014), Badger Island (where adaptive management was implemented in 2022 as part of the *IAPMP*), and three unmanaged colony sites (Harper Island in Sprague Lake, Shoal Island in Lenore Lake, and Goose Island in Banks Lake; see *Table 1* above). As in 2021, adaptive management to prevent terns from nesting at the Blalock Islands in John Day Reservoir by raising reservoir elevation to inundate all previously used nesting habitat for terns was successful in preventing tern colony formation at that site in 2022 (USACE 2021, 2022).

Compared to previous years, fewer terns were observed roosting and nesting along the shoreline of Goose Island in Potholes Reservoir during 2022 (*Figure 1*). This was likely due to an increase in passive and active nest dissuasion efforts along the shoreline (USDA-APHIS-WS 2022) and higher reservoir water levels during the nesting season, which flooded most, if not all, of the shoreline habitat where terns have nested in previous years. Typically, reservoir water levels in Potholes Reservoir begin to recede in May and expose more shoreline nesting habitat for terns through June; this was not the case in 2022. From 10 May to 26 June, reservoir water levels remained above 318 m (1,042 ft), higher than was observed in previous years during this same period (USDA-APHIS-WS 2022). While shoreline nesting habitat for terns was mostly unavailable during the 2022 breeding season, terns continued to attempt nesting in open upland habitat elsewhere on the island. These habitats were mostly unvegetated areas that lacked passive nest dissuasion (stakes, cable, and flagging) on the higher elevation part of Goose Island, plus surrounding islets, and consisted of small, isolated groups of terns that were actively nesting over multiple weeks.

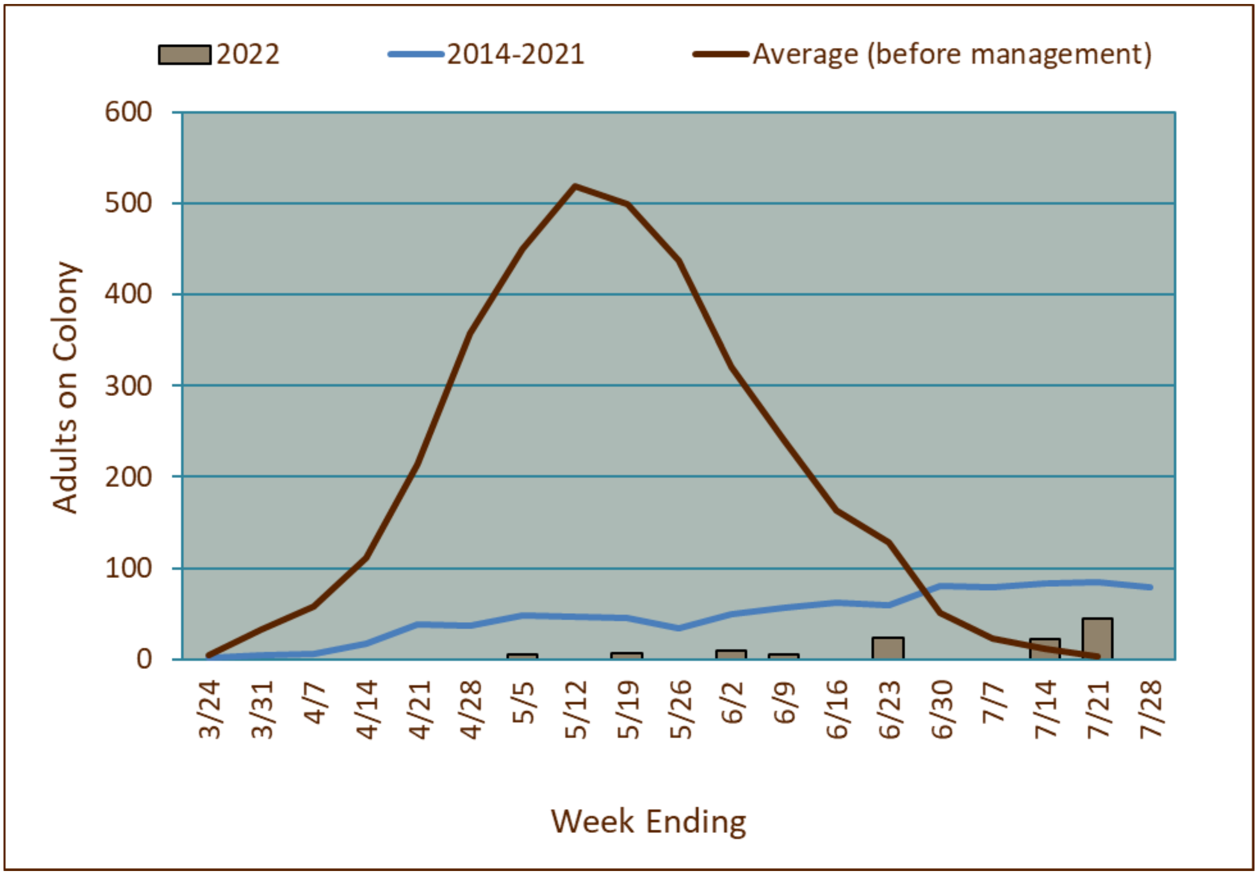


Figure 1. Weekly estimates of colony attendance by adult Caspian terns on Goose Island and surrounding islets in Potholes Reservoir. Counts are from boat-based surveys and show the average number of adults on colony throughout the breeding season prior to tern management (2010-2013), during tern management (2014-2021), and this past breeding season (2022). During the management period, counts of adults on colony do not include terns present on the island but obscured from view from the boat (i.e. at the highest elevations of the island near the historical colony). Counts of terns on this portion of Goose Island were conducted using drones and fixed-wing surveys and are reported elsewhere (see below).

In 2022, passive and active nest dissuasion on Goose Island, plus the collection of tern eggs (n = 12) under permit (USDA-APHIS-WS 2022), were effective in preventing terns from reaching the threshold colony size stipulated the *IAPMP* as the maximum allowable colony size for that and other tern colonies in the CPR (40 breeding pairs). We estimate the peak colony size at Goose Island in 2022 was 16 breeding pairs (*Figure 2*). This count was of incubating terns in drone photography taken in early June, with most of these nesting attempts occurring on or near the historical colony site on top of the island. Three tern fledglings were observed on Goose Island in July, but because these birds were capable of flight, we are unsure whether these birds were raised on Goose Island or at another colony in the region.

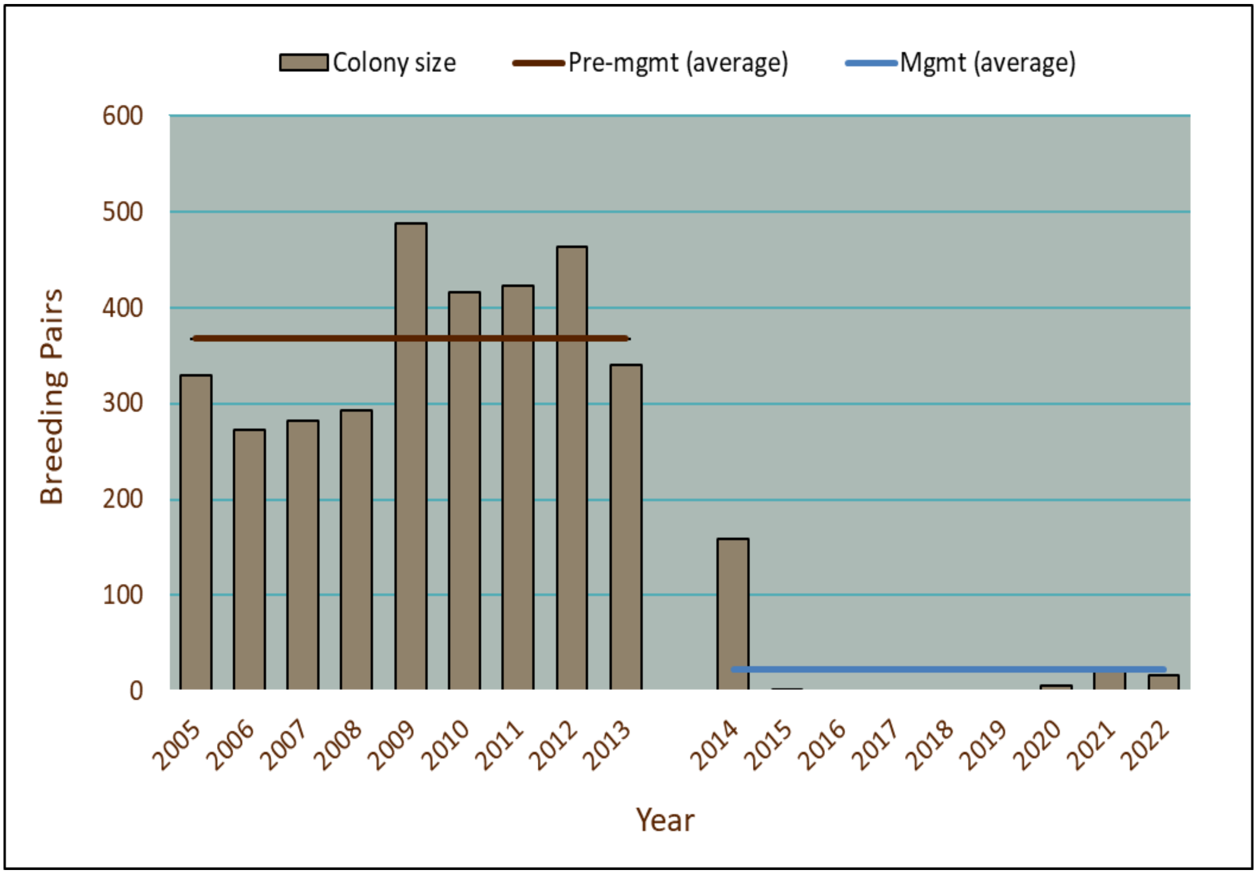


Figure 2. Annual colony size (number of breeding pairs) of Caspian terns at Goose Island and surrounding islets in Potholes Reservoir before tern management (Mgmt; 2005-2013) and during tern management (2014-2022). No Caspian tern nesting occurred at Goose Island or surrounding islets during 2016-2019.

Prospecting terns were documented at 7 different sites in northern Potholes Reservoir during the 2022 nesting season (USDA-APHIS-WS 2022), down from 12 sites used by prospecting terns in 2021 (USDA-APHIS-WS 2021). In 2022, the total number of terns observed in northern Potholes Reservoir during independent surveys conducted by our team and USDA-APHIS-WS was often < 20 individuals, although higher numbers of terns (~50–80 terns) were observed post-breeding season during August (USDA-APHIS-WS 2022). Egg-laying by terns was not detected at any of the sites surveyed in northern Potholes Reservoir during 2022, and active hazing was successful in preventing tern colony formation at any of these sites. While these results are encouraging, terns continue to show strong fidelity to Goose Island and other islands in Potholes Reservoir as evidenced by resightings of terns in 2022 that were originally banded on Goose Island in 2007-2013. Terns could reestablish colonies at any number of potential colony sites in Potholes Reservoir if adaptive management actions were discontinued.

An unexpectedly large breeding colony of terns reformed on Crescent Island in McNary Reservoir in 2022. Not since 2014, prior to implementation of management to eliminate the Crescent Island tern colony, has a tern colony been present on Crescent Island. The re-establishment of a tern colony on Crescent Island was facilitated by the recent reduction in vegetative cover on the island, apparently due to a combination of factors including resumption of gulls nesting on the island, beaver herbivory, and weather-related events (e.g., windstorms, drought). These factors have created patches of open, unvegetated habitat that is suitable for tern nesting. Gulls, both California and ring-billed, have successfully re-establishing a large breeding colony on Crescent Island (approximately 4,500 breeding individuals in 2022). In 2021 a single breeding pair of terns nested at the historical tern colony site on Crescent Island, the first successful tern nesting attempt on Crescent Island since 2014. In 2022, a tern colony consisting of 149 breeding pairs formed on the former tern colony site (*Figure 3*). In the absence of adaptive management, we expect the tern colony on Crescent Island to continue to grow as the available nesting habitat for terns reverts back to the pre-management condition; gulls not only provide strong social attraction to nesting terns but also cause a die-back in willows and other native vegetation that dissuades terns from nesting there.

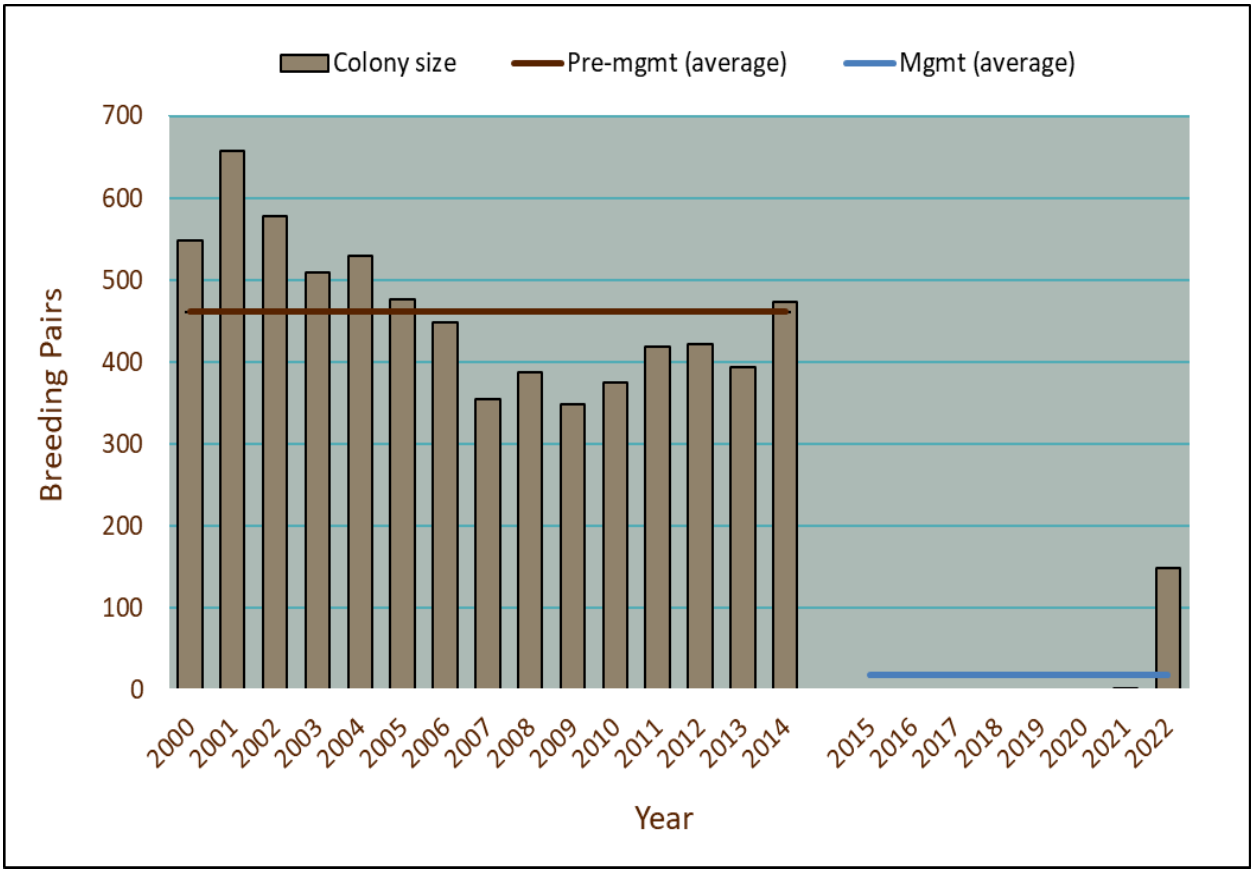


Figure 3. Annual colony size (number of breeding pairs) of Caspian terns at Crescent Island in the mid-Columbia River before tern management (Mgmt; 2005-2014) and during tern management (2015-2022). No terns nested at Crescent Island during 2015-2020; one tern breeding pair successfully nested on Crescent Island in 2021.

Management implemented at the two largest tern colonies in the CPR (Goose and Crescent islands) has resulted in a shift in the distribution of nesting terns in the region. In the first year of tern management on Crescent Island (2015), most of the terns prevented from nesting at that site relocated to nearby islands in the Blalock Islands (see *Table 1* above). During the 2021 and 2022 nesting seasons, adaptive management (i.e. raising the John Day Reservoir elevation to inundate all nesting habitat formerly used by terns) was successful in preventing tern colony formation in the Blalock Islands (USACE 2021,2022). These actions resulted in a shift of terns away from the Blalock Islands and back to Crescent Island (where management actions had ceased; see *above*) and to Badger Island, located on the Columbia River immediately upstream of Crescent Island. Although adaptive management implemented in 2022 to prevent/reduce tern nesting on Badger Island (i.e. placement of large woody debris on one of two nesting areas used by terns in 2021) had some success, the largest tern colony in the CPR over the past two years has been on Badger Island (see *Table 1* above). In 2022, the interior colony area on Badger Island that was covered with woody debris was not used by nesting terns; however, the other colony area used by terns the previous year (on northeastern shoreline of the island) expanded to ultimately support a larger number of nesting terns on Badger Island in 2022 (267 breeding pairs) compared to 2021 (231 breeding pairs; see *Table 1* above). Together with the newly re-established tern colony on Crescent Island, a total of 416 breeding pairs of terns nested in this reach of the Columbia River in 2022, greater than the average size of the Crescent Island tern colony in the 8 years prior to implementation of the IAPMP (397 breeding pairs).

A total of 511 breeding pairs of terns nested in the CPR in 2022 (see *Table 1* above, *Figure 4*), the highest regional breeding population size observed since 2017, and more than double the maximum regional population size stipulated in the *IAPMP* (200 breeding pairs; USACE 2014). The regional population size in 2022 represented a 42% reduction in the size of the regional tern breeding population compared to the pre-management average (875 breeding pairs; *Figure 5*) but has been increasing in the last two years. Of the USACE-constructed islands as alternative tern nesting sites at Malheur Lake, Crump Lake, and Sheepy Lake, only Sheepy Lake supported a tern colony in 2022 with approximately 150 breeding pairs. 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. 2022). Our team observed two banded terns with symptoms of the Highly Pathogenic Avian Influenza (HPAI) virus on Foundation Island in the McNary Reservoir in 2022. Several days later, both terns were found dead in the same location, recovered, and samples were sent to the Washington Animal Disease Diagnostic Lab to test for Avian Influenza virus (AIV) and HPAI. Tests came back positive for AIV and HPAI, which were the first known positive cases for HPAI in Caspian terns in the Pacific Flyway and indicates a new and emerging threat to the Flyway population. Assessing new and emerging threats, along with continued effectiveness monitoring and further adaptive management, will be needed at both managed and unmanaged tern colonies in the CPR in the future if the goals and objectives of the *IAPMP* are to be met (see *Management Recommendations* section below).

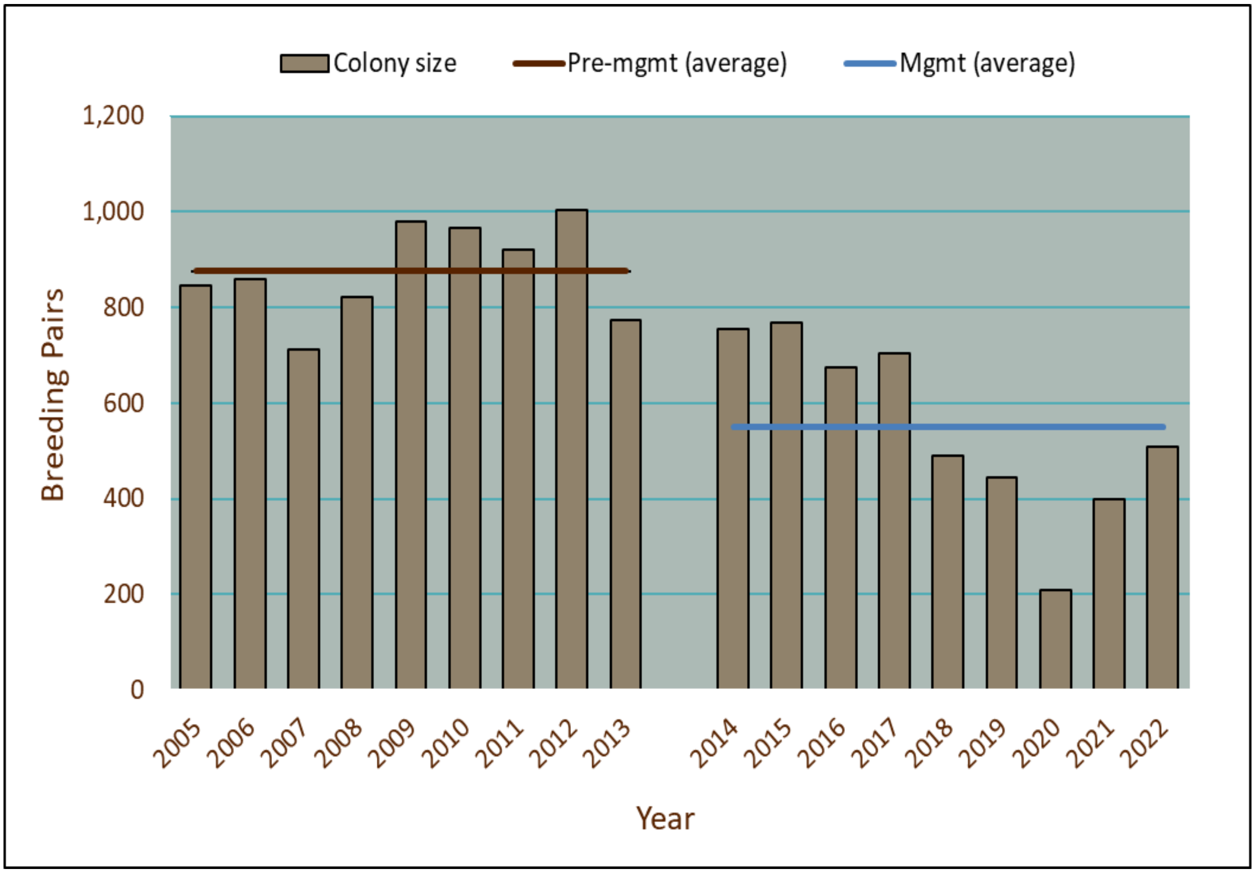


Figure 5. Annual and average number of Caspian tern breeding pairs nesting at all known colonies before tern management (Mgmt; 2005–2013) and during tern management (2014-2022) in the Columbia Plateau region.

Other Piscivorous Waterbird Colonies: A total of nine active California and ring-billed gull colonies were detected in the CPR region in 2022, ranging in size from ca. 1,700 individuals (Lenore Lake) to over 15,000 individuals (Goose Island in Potholes Reservoir; see *Table 2* above). Double-crested cormorants were confirmed to be nesting at eight colonies in the CPR in 2022, with colony size ranging from 5 breeding pairs (Goose Island in Potholes Reservoir) to 589 breeding pairs (Harper Island in Sprague Lake; see *Table 2* above). Finally, the size of the American white pelican colony on Badger Island, the only known nesting colony for the species in the CPR, was estimated to be 3,486 individuals in 2022 (see *Table 2* above).

Lower Columbia River

Two active double-crested cormorant nesting colonies were identified below Bonneville Dam on the Lower Columbia River, one on the Longview Bridge (also referred to as the Lewis and Clark Bridge; 221 breeding pairs) and the other on transmission towers near Troutdale, Oregon (352 breeding pairs; see *Table 2* above). Although these colonies have been active for many years now, 2022 marks the first year that PIT tags were recovered below the Troutdale towers to assess the impacts of that colony on smolt survival (see *below*). Cormorant nests on the Lewis and Clark Bridge are located directly over the river, so no attempt was made to recover PIT tags from that colony site. Except for several channel marker sites between river kilometer (RKM) 51–234 which were not surveyed in 2022 but have been cormorant nesting sites in previous years (Lawonn 2023), no other piscivorous waterbird colonies were identified on the Lower Columbia River above the tidally influenced estuary in 2022.

Columbia River Estuary

Caspian Tern Colonies: The USACE-FFU estimated that 1,725 breeding pairs of terns nested on the prepared 1-acre colony site on East Sand Island in 2022 (USACE-FFU 2022a). This is the smallest colony size observed on East Sand Island since 1999, the first year of efforts to move the Rice Island tern colony to East Sand Island (Roby et al. 2021b; see *Table 1* above for colony size estimates during 2007-2022). Efforts to prevent terns from nesting outside the designated 1-acre colony area on East Sand Island were successful in 2022 using passive and active nest dissuasion and egg collection (under permit). However, the tern colony on the designated site on East Sand Island once again failed to produce any young; complete nesting failure has been observed at that colony in 5 out of the last 7 years. The factor(s) causing colony failure in recent years are not clearly understood because close monitoring of the colony ceased after the 2019 nesting season. It is likely, however, that disturbance of the tern colony by bald eagles and concurrent predation on tern eggs and chicks by glaucous-winged/western gulls was the proximate cause of colony failure. During late May the designated tern colony area on East Sand Island was mostly abandoned, and thousands of terns were counted on Rice Island in the freshwater zone of the estuary. When terns attempted to nest on Rice Island, active and passive dissuasion of the incipient tern colony was used to prevent a nesting colony from forming on Rice Island. This adaptive management was eventually successful in dispersing the terns from the incipient colony site, but only after 2.5 weeks of island residency, which corresponded with the peak of the steelhead run in the CRE (see USACE-FFU 2022a, 2022b for further details). Terns were also observed in the hundreds at the Port of Astoria (Pier 3) and a condemned pier at the U.S. Coast Guard facility at Tongue Point, although no nesting was confirmed at either location.

Double-crested Cormorant Colonies: Ongoing management at the East Sand Island cormorant colony has resulted in the dispersal of cormorants away from East Sand Island, with most nesting cormorants now located upriver in the mixing zone of the CRE on the AMB (see *Table 2* above). 2022 was the fourth year in a row when there was little to no nesting attempts by cormorants on East Sand Island; in all four years all nesting attempts by cormorants on East Sand Island failed to produce any young. Meanwhile, cormorant colony size on the AMB has increased dramatically, coinciding with implementation of the *Estuary Cormorant Management Plan*; by 2020 over 5,000 breeding pairs of cormorants nested on the bridge. In 2021 and 2022 the cormorant colony on the AMB was less than 5,000 breeding pairs, apparently because of disturbance due to maintenance work on the bridge.

Other Piscivorous Colonial Waterbird Colonies: There were two active gull nesting colonies in the CRE during 2022, a mixed glaucous-winged/western gull and ring-billed gull colony on East Sand Island and a glaucous-winged/western gull colony on Rice Island. Also, there was one active nesting colony of American white pelicans on Miller Sands Spit in 2022 (see *Table 2* above). The Miller Sands Spit pelican colony is one of only two white pelican colonies in the CRB; the other colony is on Badger Island in McNary Reservoir in the CPR (see *Table 2* above). The Miller Sands Spit pelican colony first became established in 2010, when 42 individuals attempted to nest there, and has relocated to nearby Rice Island in some years (Cramer et al. 2021a). In 2022 the colony on Miller Sands Spit had grown to about 992 individuals (see *Table 2* above). In addition to double-crested cormorants nesting on the AMB, Brandt's cormorants (BRAC) and pelagic cormorants (*Phalacrocorax pelagicus*; PECO) also nested on the AMB in 2022; all cormorant species were separately enumerated during bridge-wide counts, but predation rates were only generated for double-crested cormorants and BRAC. Peak counts for BRAC were 935 breeding pairs and for PECO were 57 breeding pairs in 2022.

AVIAN PREDATION RATES

PIT-tagging of Upper Columbia River Steelhead

A total of 6,217 UCR steelhead were sampled and available for predation and survival analyses in 2022, which including 5,676 tagged smolts and 541 recaptured (i.e., previously tagged) smolts. Of these, 4,579 and 1,638 were classified as hatchery and wild, respectively (see also *Appendix B*). The number of steelhead smolts collected at the RIS fish trap in 2022 were well below average, so the target goal of 7,000 steelhead was not met in 2022, but samples were still sufficient to estimate predation and survival probabilities (see *below*). Steelhead were tagged and released from 15 April to 3 July 2022, a period which accounted for > 99% of all steelhead encountered in the trap. Run-timing was delayed (right-shifted) relative to years past (2008-2021) with large numbers of smolts still out-migrating in late-June. Mean fork length was 193 mm (standard deviation [SD] = 26 mm; range = 83–303 mm). An evaluation of external fish condition indicated that most steelhead were in good over-all external condition in 2022, with 5.2% of steelhead observed with disease (bacterial, fungal, or viral infections), severe body injuries (subcutaneous wounds/scars), severe descaling (>20% of scales missing), and/or major fin damage (>50% of fin tissue missing). For comparison, on average, 9.7% of steelhead tagged at RIS in previous years were in compromised conditions (Evans et al. 2014, Evans et al. 2022b). The most common type of anomaly in 2022 was disease, followed by severe descaling, and body injuries. Body injuries and descaling were often the most common type of damage observed in years past. In 2022, increases in disease were due to a presumed bacterial outbreak of cold-water disease (T. Jackson, Chelan County PUD, personal communication).

PIT Tag Recovery

A total of 14 genera-specific avian colonies in the CPR were scanned for smolt PIT tags following the 2022 breeding season, including 4 tern colonies, 6 gull colonies, 3 cormorant colonies, and 1 pelican colony (*Table 3*). Scanning was also conducted at 6 avian loafing/roosting sites, areas where large numbers of piscivorous colonial waterbirds were observed during the smolt out-migration period. A total of 22,816 PIT tags from 2022 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead combined) were recovered from both breeding and loafing sites in the CPR (*Table 3*). Over 95% of smolt tags were recovered from breeding sites, with the largest number recovered on the gull, tern, and pelican colonies at Badger Island (n = 11,942), followed by the cormorant colony at Foundation Island (n = 2,754), the gull colony at Miller Rocks (n = 2,392), and tern colony on Crescent Island (n = 1,124; *Table 3*). A record high number of smolt PIT tags were also recovered at the tern colony on Shoal Island in Lenore Lake in 2022 (n = 531; *Table 3*). Only 29 smolt tags were recovered from the tern colony on Goose Island in Potholes Reservoir in 2022 (*Table 3*).

In the CRE, a total of 8 avian colonies (2 tern colonies, 5 cormorant colonies, and 1 pelican colony) and 2 avian loafing/roosting sites were also scanned for smolt PIT tags in 2022, yielding a total of 8,629 PIT tags from 2022 migration year smolts (*Table 3).* The largest number of tags were found on the Rice Island tern colony (n = 2,385), followed by the Troutdale Transmission Towers (TRT) cormorant colony (n = 1,588), the AMB cormorant colony (n = 1,315), cormorant loafing sites on the AMB (n = 891), and a tern loafing site on Pier 3 located in Astoria, OR (n = 390; *Table 3).* Only small numbers of smolt PIT tags were recovered from the East Sand Island cormorant colony (n = 99), the Miller Sands Spit pelican colony (n = 58), and the AMB Brandt's cormorant colony (n = 53; *Table 3).* Scanning at AMB and TRT was limited to areas accessible to researchers following the breeding season, resulting in a subsample of all available nests (*Table 3*; see *below* for additional details).

In addition to PIT tags from 2022 migration year smolts, a total of 28 tags from other salmonid species and age-classes were also recovered on avian colonies in 2022. This included PIT tags from 12 adult sockeye salmon, 6 adult jack Chinook salmon, 3 adult steelhead, and 3 bull trout (*Salvelinus confluentus)* from colonies in the CPR (*Table 3*). A total of 4 tags implanted in sea-run cutthroat trout (*O. clarkii clarkii*) were recovered on bird colonies in the CRE (*Table 3*).

Table 3. Numbers of 2022 migration year smolt PIT tags (Chinook salmon, sockeye salmon, coho salmon, and steelhead combined) and other fish species and life stages recovered at avian breeding 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-species sites represent an unknown combination of the above and possibly other avian species. The number of PIT tags recovered was not adjusted to account for tag loss due to on-colony PIT detection and deposition probabilities (see Table 4) and thus represents the minimum number of tagged fish consumed by birds.

Location	Rkm	Bird Species	Area Use	Smolt PIT Tags	Other PIT Tags[1]
Goose Island (Potholes R.)	Off-river	CATE	Breeding	19	
		LAXX [2]	Breeding	3	
		Mixed	Loafing	29	
Lenore Lake	Off-river	CATE	Breeding	531	
		DCCO	Breeding	15	
Hanford Island	592	DCCO	Breeding	358	
		Mix	Loafing	123	
Island 20	549	LAXX	Breeding	1,051	
		Mix	Loafing	48	1
Foundation Island	518	DCCO	Breeding	2,754	
Badger Island	512	LAXX	Breeding	734	
		AWPE	Breeding	915	2
		CATE	Breeding	2,268	
		Mixed	Breeding	8,025	21
		Mixed	Loafing	515	
Burbank Slough	Off-river	CATE	Loafing	119	
Crescent Island	510	CATE	Breeding	1,124	
		LAXX	Breeding	669	
Blalock Islands	441-439	LAXX	Breeding	999	
		Mixed	Loafing	125	
Miller Rocks	331	LAXX	Breeding	2,392	
Troutdale Towers	189	DCCO [2]	Breeding	1,588	
Miller Sands Spit	38	AWPE	Breeding	58	
Rice Island	34	CATE	Breeding	2,385	1
Astoria-Megler Bridge	23	DCCO [2]	Breeding	1,315	
		BRAC [2]	Breeding	53	
		Mixed [2]	Breeding	126	1
		Mixed [2]	Loafing	891	
Pier 3	21	CATE	Loafing	390	
East Sand Island [3]	8	CATE	Breeding	1,724	2
		DCCO	Breeding	99	
Total				31,445	28

[1] *Includes adult sockeye (n=12; Badger Island), adult steelhead (n=3; Badger Island), adult Chinook (n=6; Badger Island and Island 20), bull trout (n=3; Badger Island), and sea-run cutthroat (n=4; East Sand Island, Astoria-Megler Bridge, and Rice Island).*

[2] *Tags recovered from a sub-sample of nests.*

[3] *Tags recovered by the USACE (USACE-FFU 2022a).*

Smolt Survival to Bonneville Dam

In addition to providing information on predation effects, recoveries of smolt PIT tags on avian colonies can also be used to increase the precision and accuracy of smolt survival estimates by increasing the sample sizes of tagged fish used in mark-recapture-recovery survival models (Hostetter et al. 2018, Payton et al. 2019, Payton et al. 2023). To provide information for use in smolt survival models in 2022, we recovered smolt PIT tags from several additional avian breeding and loafing sites, 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 resulted in the detection of 6,840 PIT tags from 2022 migration year smolts that survived outmigration to below Bonneville Dam prior to being consumed by a bird in the CRE. This information, coupled with 2022 migration year smolt tags recovered by the USACE on the East Sand Island tern and cormorant colonies (n = 1,823), tags of live fish detected at antennas located on pile dikes in the CRE (n = 3,716), and tags of live fish detected at the National Marine Fisheries Service net trawl detection system (n = 9,661) in the CRE, were used as part of this study and other federally-funded projects to estimate smolt survival to Bonneville Dam in 2022.

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 (Hostetter et al. 2015, Evans et al. 2022c). Detection probabilities were directly measured in 2022 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 for details). Like 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*).

Table 4. Detection efficiency (range during breeding season) and deposition (95% credible interval) estimates (depicted as a proportion) for smolt PIT tags on bird colonies during 2022. 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), Brandt's cormorants (BRAC), and American white pelicans (AWPE).

Location	Rkm	Bird Species	Detection (Range)	Deposition (95% CRI) [1]
Lenore Lake	Off-river	CATE [2]	0.68–0.68	0.71 (0.51–0.89)
		DCCO	0.99	0.51 (0.34–0.70)
Goose Island	Off-river	CATE	0.60–0.99	0.71 (0.51–0.89)
		LAXX	0.60–0.96	0.15 (0.11–0.21)
Hanford Island	592	DCCO	0.56–0.72	0.51 (0.34–0.70)
Island 20	549	LAXX	0.82–0.92	0.15 (0.11–0.21)
Foundation Island	549	DCCO	0.26–0.30	0.51 (0.34–0.70)
Badger Island	512	AWPE	0.82–0.96	0.47 (0.24–0.73)
		LAXX	0.64–0.96	0.15 (0.11–0.21)
		CATE [2]	0.64–0.94	0.71 (0.51–0.89)
Crescent Island	510	LAXX	0.76–0.99	0.15 (0.11–0.21)
		CATE	0.68–0.96	0.71 (0.51–0.89)
Blalock Islands	441-439	LAXX	0.86–0.90	0.15 (0.11–0.21)
Miller Rocks	331	LAXX	0.52–0.86	0.15 (0.11–0.21)
Troutdale Towers	189	DCCO	0.34–0.72	0.51 (0.34–0.70)
Miller Sands Spit	38	AWPE [2]	0.80–0.98	0.47 (0.24–0.73)
Rice Island	34	CATE	0.68–0.99	0.71 (0.51–0.89)
Astoria-Megler Br.	23	DCCO	0.61–0.87	0.51 (0.34–0.70)
		BRAC [3]	0.30–0.99	0.51 (0.34–0.70)
East Sand Island [5]	8	CATE	0.51–0.43	0.71 (0.51–0.89)
		DCCO [2]	0.67–0.92	0.51 (0.34–0.70)

[1] *Deposition estimates for CATE, LAXX, and DCCO are those of Hostetter et al. (2015); estimates for AWPE are those of Evans et al. (2022c).*

[2] *Variation in detection was partially inferred from other years (see Payton et al. 2019)*

[3] *BRAC deposition was assumed to be the same as DCCO (see Cramer et al. 2021a).*

Efficacy of Avian Predation Management Plans

Inland Avian Predation Management Plan (IAPMP): In 2022, estimates of tern predation rates were above the 2% threshold identified in *IAPMP* for UCR and SR steelhead by terns breeding on Shoal Island in Lenore Lake and on Badger Island in McNary Reservoir, respectively (*Table 5*). Estimates of predation by Crescent Island terns in 2022 were also substantially higher than those in 2021, but estimates were below the 2% threshold for all ESUs/DPSs evaluated, with the highest estimate of 1.3% (0.9–2.1%) observed in SR steelhead (*Table 5*).

Table 5. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations, 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 2022. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable predation estimates (see Methods & Analysis section). See Appendix A 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%	0.2% (0.1–0.4)	0.1% (<0.1–0.2)
UCR Sp Chinook	NA	NA	NA	NA
SR Fall Chinook	< 0.1%	< 0.1%	0.6% (0.3–1.0)	0.2% (0.1–0.4)
SR Steelhead	< 0.1%	< 0.1%	2.8% (2.0–4.4)	1.3% (0.9–2.1)
UCR Steelhead	0.1% (<0.1-0.2)	2.1% (1.4–3.4)	0.7% (0.4–1.2)	0.6% (0.2–1.0)

Increases in tern predation rates on salmonid ESUs/DPSs in 2022 compared with 2021 were associated with terns on Crescent Island in McNary Reservoir and increases in predation by terns on Shoal Island in Lenore Lake. Terns also nested on Badger Island in McNary Reservoir, where the highest tern colony-specific predation rate of 2.8% on SR steelhead was observed (see *Table 5* above), however, estimates in 2022 were similar to those in 2021 (Evans et al. 2022b). Predation rates by terns on Goose Island in Potholes Reservoir in 2022 (ca. 0.1%; see *Table 5* above) were significantly lower than those observed in 2021 (ca. 3.9%; *Appendix A*). Reductions in Goose Island tern predation rates were presumably due to a (1) smaller sustained breeding colony in 2022 (ca. 16 pairs) compared with 2021 (ca. 29 pairs) and a (2) large reduction in the number of terns prospecting for a nesting site during the peak smolt out-migration period in 2022 (~ 10–20 terns per day during late-April to early-June; USDA-APHIS-WS 2022) compared with 2021 (~ 20–200 terns per day during late-April to early-June; USDA-APHIS-WS 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 CPR, averaging 15.7% (14.1–18.9%) during 2007-2013 (*Table 6*). Estimated average annual predation rates on UCR Spring Chinook were 2.5% (1.7–3.6%) during 2007-2013 (*Table 6*). In 2016, a colony of 144 breeding pairs formed on an unnamed island in northern Potholes Reservoir where recoveries of smolt PIT tags indicated that terns consumed an estimated 4.1% (2.9–6.3%) of UCR steelhead in 2016 (*Appendix A*), impacts that prompted adaptive management actions at this and other surrounding islands in northern Potholes Reservoir during 2017-2022. Since then, active and passive dissuasion techniques have been successful at preventing terns form nesting on islands in northern Potholes Reservoir, thereby reducing impacts from tern predation. Due to the large number of terns that continue to prospect for nest sites on Goose Island and the re-establishment of the Goose Island tern colony in recent years, continued efforts to prevent terns from nesting using both passive and active dissuasion techniques will likely be necessary if the goals and objectives of the *IAPMP* are to be reached and maintained year after year.

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 (2014-2022 for Goose Island and North Potholes; 2015–2022 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 were evaluated. Time periods are denoted as the average of all years with data or data from the last three years of the study period (2020-2022). See Appendix A for annual estimates. NC denotes that no colony existed during that period.

	Goose Is.			North Potholes Is.			Crescent Is.		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2022 [1]	Last 3-years 2020–2022 [1]	Pre-Management 2007–2013	Management 2016 [2]	Last 3-years 2020–2022	Pre-Management 2007–2014	Management 2015–2022	Last 3-years 2020–2022 [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)	0.2% (0.1–0.2)	0.4% (0.3–0.7)
UCR Steelhead	15.7% (14.1–18.9)	1.1% (0.7–1.8)	1.3% (0.8–2.2)	NC	4.1% (2.9–6.3)	NC	2.5% (2.2–2.9)	0.1% (0.1–0.2)	0.2% (0.1–0.3)

[1] Small numbers of terns were observed during the peak smolt outmigration period in 2020 (USDA-APHIS-WS 2021) but tags were not recovered.

[2] Nesting colony formed in 2016 and was successfully managed in all subsequent years.

[3] One pair of terns was observed in 2021, the first year nesting occurred since management was implemented on Crescent Island in 2015.

For the second consecutive year, terns nested on Crescent Island in McNary Reservoir with the colony increasing from just 1 pair in 2021 to 149 pairs in 2022 (see *Table 1* above). Not surprisingly, predation rates were significant higher in 2022 compared with 2021, with the highest estimate observed on SR steelhead at 1.3% (0.9–2.1%, 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, 2021b). 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 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 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 at this site, with the greatest benefits to SR salmonids due to the greater susceptibility of SR 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 rapid growth of both the tern and gull colonies on Crescent Island in 2020-2022, however, future management to prevent terns from nesting on Crescent Island will likely be necessary to achieve the goals and objectives of the *IAPMP*.

Estimates of predation rates on smolts at the unmanaged tern colony on Shoal Island in Lenore Lake were above the 2% threshold for the first time since the colony formed in 2015, with an estimated 2.1% (1.4–3.4%) of UCR steelhead consumed in 2022 (see *Table 5* above). Although tags were recovered from UCR spring Chinook in 2022, due to inadequate sample sizes of available tagged fish at RIS, estimates of predation were unavailable for this critical endangered ESU; however, predation rates on UCR spring Chinook have historically been low by Lenore Lake terns (< 0.5%; see *Table 6* above). Increases in UCR steelhead predation rates in 2022 were associated with increases in colony size, with the colony growing from 16 pairs in 2015 to 76 pairs in 2022 (see *Table 2* above). An estimated 123 pairs, however, nested in Lenore Lake in 2017, yet predation rates on UCR steelhead were estimated to be 1.0% (0.6–2.0; *Appendix A*), suggesting that factors other than colony size alone may be related to predation effects by terns breeding in Lenore Lake. Results from 2022 suggest that future monitoring and possible management of the tern colony in Lenore Lake may be necessary to active the goals of *IAPMP*.

Based on aerial imagery taken during the peak of the breeding season in May, there was one pair of terns confirmed nesting at an unmanaged colony site in Banks Lake, WA in 2022. The last time a tern colony was observed on Banks Lake was in 2016 (Collis et al. 2021b). In 2014 and 2015, following implementation of management actions at the tern colony on Goose Island in Potholes Reservoir, 66 pairs and 64 pairs nested on Twinning Island in Banks Lake, respectively, and predation rates on UCR steelhead were 1.2% (0.3–6.4%) and 2.6% (1.8–3.9%), respectively (*Appendix A*). These results demonstrated 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. As such, continued monitoring of Twinning Island and other islands in Banks Lake to detect tern nesting and to estimate the impacts of nesting terns on smolt survival is warranted.

There was a very small tern colony on Harper Island in Sprague Lake, WA in 2022, with just 2 breeding pairs documented. Harper Island has had a relatively small tern colony (< 40 breeding pairs) in most years (Collis et al. 2021b), except for 2021 when estimated 85 breeding pairs nested there (Evans et al. 2022b). Because the island is privately owned, we were not able to scan the island for PIT tags in 2022. 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 UCR and SR ESUs/DPSs, with the highest rates observed on SR steelhead at 0.2% (Roby et al. 2021a). 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. Future monitoring of predation rates may be warranted if a large 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 2022, ranged from 0.2% (0.1–0.4%) on SR spring/summer Chinook to 2.8% (2.0–4.4%) on SR steelhead (see *Table 5* above). Estimates were 0.7% (0.4–1.2%) on UCR steelhead (see *Table 5* above). Estimates of predation rates were consistent with, but slightly lower than, what might be expected based on a colony size of 267 breeding pairs in McNary Reservoir. For example, prior to management actions, predation rates by terns nesting on nearby 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 A*).

For the second consecutive year (2021–2022), there was no 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 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 A*). 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 (Collis et al. 2021a, 2021b). During the management period, estimated average annual predation rates by terns nesting on the Blalock Islands were comparable to or higher than that of terns nesting on Crescent Island during the pre-management period for most of the ESUs/DPSs evaluated, particularly for ESUs/DPSs originating from the SR (*Table 7*). 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 actions at Crescent Island to an average of 4.0% (3.3–4.9%) following management at Crescent Island during 2015-2020 (Collis et al. 2021b). 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 most of the management period, the impacts of predation by terns from this colony were greatly reduced (if not eliminated) in 2021 and 2022 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 at this site in recent years.

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 (2014-2022). 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 A for annual estimates. Time periods represent the average of all years or data from the last three years of the management (2020-2022). NA denotes that predation estimates were not available. NC denotes that no colony existed during that period.

	Twinning Island			Badger Island			Blalock Islands		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2016	Last 3-years 2020–2022	Pre-Management 2007–2013 [1]	Management 2014-2022[2]	Last 3-years 2020-2022[2]	Pre-Management 2007–2013	Management 2014–2022[3]	Last 3-years 2020–2022[3]
SR Sockeye	< 0.1%	0.1% (0.0–0.5)	NC/NA	NC/NA	NA	NA	0.2% (0.1–0.4)	1.8% (1.3–2.8)	0.2% (0–01.2)
SR Sp/Su Chinook	< 0.1%	< 0.1%	NC/NA	NC/NA	0.2% (0.1–0.4)	0.3% (0.1–0.8)	0.1% (0.1–0.2)	0.3% (0.2–0.4)	< 0.1%
UCR Sp Chinook	< 0.1%	0.2% (0.0–0.7)	NC/NA	NC/NA	< 0.1%	< 0.1%	< 0.1%	0.4% (0.2–0.5)	< 0.1%
SR Fall Chinook	< 0.1%	< 0.1%	NC/NA	NC/NA	0.2% (0.1–0.3)	0.3% (0.2–0.5)	< 0.1%	0.5% (0.3–0.7)	0.1% (0–0.3)
SR Steelhead	< 0.1%	< 0.1%	NC/NA	NC/NA	0.8% (0.6–1.1)	1.4% (1.1–2.1)	0.5% (0.4–0.9)	1.8% (1.4–2.3)	0.7% (0.3–1.8)
UCR Steelhead	0.1%	1.1% (0.8–1.6)	NC/NA	NC/NA	0.4% (0.3-0.6)	0.7% (0.5–1.0)	0.5% (0.3–0.7)	2.1% (1.7–2.8)	0.5% (0.1–1.5)

	Lenore Lake Islands			Harper Island		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2022	Last 3-years 2020–2022	Pre-Management 2007–2013	Management 2014–2022	Last 3-years 2020–2022
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% (0.1–0.4)	0.2% (0.0–0.9)	< 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	0.8% (0.6–1.0)	1.3% (1.0–1.8)	< 0.1%	NA	NA

[1] *Colonies existed in 2011 and 2012 but no estimates of predation were available.*

[2] *Colonies and predation estimates were available in 2017 and 2021.*

[3] *No established tern nesting colony was present in the Blalock islands during 2021-2022.*

In summary, predation rates by terns nesting at colonies in the CPR in 2022 were generally higher than those observed in 2021, commensurate with increases in the size of several tern colonies in the CPR in 2022. 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 successfully preventing tern nesting on the Blalock Islands in both 2021 and 2022. Some proportion of terns that formerly nested at the Blalock Islands, however, presumably renested at Badger Island and Crescent Island. Predation rates by terns nesting at Badger Island and Shoal Island were above the 2% threshold for SR and UCR steelhead, respectively, in 2022. Thus, continued adaptive management at tern nesting sites will likely be needed in 2023 and beyond to achieve the goals and objectives of the *IAPMP*.

Estuary 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 (ESI) in 2022 were not available, although smolt PIT tags were recovered on ESI by the USACE-FFU following the 2022 breeding season (see *Table 3* above). It should be noted that the tern colony on ESI in 2022 was the smallest recorded (1,725 breeding pairs; USACE-FFU 2022a) since the colony was fully established in 2000 and down from its peak colony size in 2008 (10,668 breeding pairs; Roby et al. 2021b). For the fourth consecutive year, the ESI tern colony failed to fledge chicks in 2022 (i.e. colony failure). Large numbers of terns (hundreds) continued to attempt to nest outside of the designated 1-acre nesting area on ESI, as well as nesting attempts by thousands of terns on Rice Island during the peak smolt outmigration period in 2022 (USACE-FFU 2022b). On average, approximately 700 terns were observed on Rice Island each day during the peak smolt outmigration period, with upwards of 3,500 terns observed in late-May (USACE-FFU 2022b). Approximately 450 terns were also observed at two separate loafing sites, on Pier 3 (Rkm 20) and at a condemned pier at the U.S. Coast Guard Tongue Point Facility (Rkm 26), but nesting was not confirmed at either site in 2022. At Pier 3, 390 current year (2022) smolt PIT tags were recovered (see *Table 3* above) but tags were not recovered from the condemned pier in Tongue Point.

Although ESU/DPS-specific predation rates by terns on nesting on ESI in 2022 were not available (estimate generated with funding from the USACE in years past), predation rates were generated by terns that attempted to nest on Rice Island in 2022 as part of this study. Results of predation analyses indicated that predation rates ranged from 0.1% (<0.1–0.4%) on SR Fall Chinook to 2.9% (1.6–5.1) on Middle Columbia River steelhead (*Table 8*). Rates were also elevated on SR and UCR steelhead at 2.3% (1.6–3.6%) and 2.7% (1.7–4.2), respectively (*Table 8*). Results indicate that despite dissuasion efforts on Rice Island, predation on steelhead smolts in 2022 were appreciable. Collectively, results suggest that continued implementation of nest dissuasion activities and continued monitoring of tern nesting sites throughout the CRE will be necessary to ensure that smolt survival gains achieved in implementing the estuary tern management plan are not offset by increased tern predation rates at other sites in the CRE.

Table 8. Estimated predation rates (95% credible interval) on Snake River (SR), Upper Columbia River (UCR), and Middle Columbia River (MCR) salmonid populations (ESUs/DPSs), with runs of spring (Sp), summer (Su), and Fall fish, by Caspian terns (CATE) on Rice Island and double-crested cormorants (DCCO) nesting at the Astoria-Megler Bridge (AMB) during 2022. See Appendix A for estimates of predation by these colonies in previous years. Smolt PIT tags were collected by the U.S. Army Corps of Engineers from both the East Sand Island (ESI) CATE and DCCO colonies in 2022 but ESU/DPS-specific predation rate estimates were not available (NA) as part of that study.

ESU/DPS	Rice Island CATE	AMB DCCO [1]	ESI CATE	ESI DCCO
SR Sockeye	0.1% (<0.1–0.5)	6.6% (1.7-14.7)	NA	NA
SR Sp/Su Chinook	0.3% (0.2–0.6)	4.9% (2.6-8.1)	NA	NA
UCR Sp Chinook	0.2% (<0.1–0.5)	5.2%(2.0-10.3)	NA	NA
SR Fall Chinook	0.1% (<0.1–0.4)	3.1% (2.1-7.9)	NA	NA
SR Steelhead	2.3% (1.6–3.6)	7.2% (3.5-12.0)	NA	NA
UCR Steelhead	2.7% (1.7–4.2)	8.6% (3.2-15.1)	NA	NA
MCR Steelhead	2.9% (1.6–5.1)	7.4% (2.1-15.5)	NA	NA

[1] *Based on subsample of all available nests (see Methods & Analysis section).*

For the third consecutive year, there was no evidence of a sustained breeding colony of double-crested cormorants on ESI in 2022. Several thousand cormorants (n = 2,317) were observed in late-June on areas where birds nested on ESI in years past, but attendance was sporadic and brief through the traditional breeding season (USACE-FFU 2022a). Similar to the tern colony on ESI, smolt PIT tags were recovered by USACE-FFU following the breeding season but estimates of ESU/DPS-specific predation rates were not available, work that has been funded by USACE in the past. Predation rates by cormorant on ESI in 2022, however, were presumably low based on the small number of smolt PIT tags (n = 99) recovered on-colony following the breeding season by the USACE-FFU (see *Table 3* above).

An unintended consequence of management actions at the cormorant colony on ESI during 2015-2019 was the complete abandonment of the colony site and the subsequent rapid expansion of the cormorant colony on the AMB (Lawes et al. 2021b). The AMB is located approximately 10 Rkm upstream of ESI in the freshwater mixing zone of the CRE, an aquatic environment where fewer marine forage and a greater proportion of juvenile salmonids are available relative to the waters surrounding ESI, which is in the marine zone of the CRE (Cramer et al. 2021b, Evans et al. 2022b). To estimate predation rates on salmonid smolts by cormorants breeding on the AMB in 2021 and 2022, we recovered smolt PIT tags deposited by cormorants nesting on a concrete footing of the bridge (referred to as the North Crib), an area where smolt PIT tags could be recovered by researchers following the breeding season (see also Evans et al. 2022b). Estimates of per capita predation rates derived from cormorants nesting on the North Crib were then extrapolated to account for all cormorants nesting on the bridge to generate colony-wide estimates of predation. A total of 506 breeding pairs were counted on the North Crib and a total of 4,054 pairs were counted throughout the entire AMB colony in 2022. Results of predation analyses indicated that per capita predation rates of cormorants nesting on the

AMB ranged from 0.0008% (0.0001–0.0038%) on SR Fall Chinook to 0.0021% (0.0008–0.0030%) on UCR steelhead. Colony-wide estimates of predation ranged from 3.1% (2.1–7.9%) on SR Fall Chinook to 8.6% (3.2–15.1%) on UCR steelhead (see *Table 8* above). Estimates of predation were similar amongst and between salmon and steelhead ESUs/DPSs, with no statistically significant difference detected in most relative comparisons of predation rates (see *Table 8* above). It should be noted, however, that estimates of predation were often imprecise, in part due to the process of extrapolating estimates from a subsample of breeding pairs on North Crib to all breeding pairs on AMB and by limiting the analysis to known ESA-listed PIT-tagged smolts last detected passing Bonneville Dam (see *Methods & Analysis* section for details).

Estimates of per capita predation rates on salmonid smolts by cormorants nesting on the AMB in 2022 were very similar to those observed in 2021 (Evans et al. 2022a). For instance, per capita predation on all steelhead smolts last detected passing Bonneville Dam were 0.0014% (0.0007–0.0025%) and 0.0018% (0.0011–0.0027%) in 2021 and 2022, respectively. Estimates of per capita predation by cormorants on AMB in 2021 and 2022 were approximately 2 to 4 times greater (depending on salmonid species and ESU/DPS) than average annual per capita predation rates by cormorants that nested on ESI prior to implementation of management actions (2003-2014; see also *below*). For example, per capita predation rates on steelhead last detected passing Bonneville Dam by cormorants on ESI were, on average, 0.0006% (0.0005–0.0007; Lawes et al. 2021b), significantly lower than those of cormorants breeding on the AMB in both 2021 and 2022. As such, despite the much smaller size of the cormorant colony on the AMB (4,054 and 4,151 breeding pairs during 2021 and 2022, respectively) compared with the cormorant colony on ESI (12,787 breeding pairs on average during 2003-2014), colony-wide predation rates by cormorants on the AMB were similar to or greater than those of cormorants on ESI. For instance, average annual predation rates on SR spring/summer Chinook and SR steelhead by cormorants breeding on ESI during 2003-2014 (prior to management on ESI) were 4.6% (4.1–5.3%) and 7.2% (6.3–8.5%), respectively (Lawes et al. 2021b), compared with 4.9% (2.6–8.1%) and 7.2% (3.5–12.0%), respectively, by cormorants on the AMB in 2022 (*Table 8*). Collectively, results suggest that colony location within the CRE is a key factor associated with cormorant predation rates on salmonid smolts, and that currently, predation rates by cormorants nesting in the CRE are similar, if not higher, than the rates observed prior to management (see also Cramer et al. 2021b and Evans et al. 2022a for a more detailed description of cormorant predation effects by colony location). Due to small sample sizes of PIT-tagged smolts from Lower Columbia River (LCR) ESUs/DPSs in 2022, estimates of predation effects by cormorants breeding on the AMB were lacking, but estimates may be substantial given predation effects on LCR salmonids by cormorants on ESI in previous years. For instance, predation effects by cormorants nesting on ESI were, on average, 15.0% (12.2–18.2%) and 27.5% (24.3–30.7) on ESA-listed LCR coho and Chinook, respectively (Lawes et al. 2021b). Efforts to PIT tag larger numbers of LCR smolts would be necessary to evaluate predation effects on LCR salmonid populations by cormorants in the CRE in the future.

Annual colony-specific and ESU/DPS-specific predation rates by terns and cormorants breeding on ESI dating back to 2000 for terns and 2003 for cormorants are provided in the *Appendix A*.

Predation Rates by Other Piscivorous Colonial Waterbirds

Predation/consumption rates of juvenile salmonids by gulls and cormorants nesting at some colonies in the CPR were equal to or greater than those of terns nesting at colonies in the CPR, with estimates in excess of 5.0% of available smolts observed in some of the ESUs/DPSs evaluated in 2022 (see *below*). Estimates were particularly high at some of the gull colonies in the CPR. It should be noted, however, that gulls 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 are more indicative of consumption rates, rather than predation rates (Cramer et al. 2021a; Evans et al. 2022a, 2022b). Unlike gulls, other species (terns, cormorants, and pelicans) are strictly piscivorous and 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 2022, along with comparisons to estimates of predation/consumption rates from previous years.

Gulls - Goose Island: PIT tags were recovered from the mixed California and ring-billed gull colony on Goose Island in Potholes Reservoir in 2022, where just 3 smolt tags from migration year 2022 fish were recovered from a subsample of nests (ca. 10% of all available gull nesting habitat was scanned for PIT tags; see *Table 3* above). Due the paucity of tags recovered, even after adjusting estimates for all gull nesting habitat on the island (see *Methods & Analysis* section for details), consumption r ate estimates were less than 0.1% for all available salmonid ESUs/DPSs evaluated (*Table 9*). Results from 2022, plus those of 2020 (where gull consumption estimates were also < 0.1% per ESU/DPS; *Appendix A*), suggest that despite the large size of the gull colony on Goose Island (15,583 adults counted in 2022; see *Table 2* above), gulls consumed very few anadromous salmonids. In 2012, the first year the Goose Island gull colony was scanned for smolt PIT tags, estimated consumption rates by gulls were as high as 2.8% of available UCR steelhead smolts (*Appendix A,* see also Roby et al. 2015). A large colony of terns (463 breeding pairs), however, also nested on Goose Island in 2012 and we hypothesized that most of the tags deposited on the gull colony in 2012 were from smolts that were stolen by gulls (kleptoparasitized) from terns. The low consumption estimates of gulls in both 2020 and 2022 – years when small numbers of terns (< 20 pairs) nested on Goose Island – support this hypothesis and indicate that gulls breeding on Goose Island pose little to no threat to the survival of anadromous juvenile salmonids in the middle Columbia River. If a large tern colony were to reform on Goose Island in the future, however, gulls on Goose Island may still pose a threat to steelhead survival because the energy demands of terns are presumably higher in the presence of gulls that kleptoparasitize smolts from terns (see Cramer et al. 2021a for a more detailed discussion).

Table 9. Estimated 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 Goose Island in Potholes Reservoir; Island 20, Badger Island, and Crescent Island in McNary Reservoir; Blalock Islands in John Day Reservoir; and Miller Rocks in The Dalles Reservoir during 2022. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods). See Appendix A for estimates of predation/consumption by these colonies in previous years.

ESU/DPS	Goose Is.[1]	Island 20	Badger Is.	Crescent Is.	Blalock Is.	Miller Rocks Is.
SR Sockeye	NA	NA	NA	NA	NA	NA
SR Sp/Su Chinook	< 0.1%	0.5% (0.2–1.3)	1.3% (0.3–3.0)	0.1% (0.1–0.5)	0.4% (0.1–1.4)	0.8% (0.2–2.3)
UCR Sp Chinook	NA	NA	NA	NA	1.0% (0.1–4.9)	1.2% (0.1-6.4)
SR Fall Chinook	< 0.1%	0.1% (<0.1–0.6)	0.2% (0.1–0.8)	0.1% (<0.1–0.7)	0.3% (0.1–1.2)	1.9% (0.8–4.1)
SR Steelhead	< 0.1%	1.0% (0.4–2.1)	3.4% (1.6–7.5)	2.4% (1.2–4.3)	5.5% (2.8–10.2)	5.7% (2.7–11.8)
UCR Steelhead	< 0.1%	3.4% (2.1–5.6)	6.6% (3.3–11.4)	1.3% (0.6–2.5)	5.2% (1.5–13.0)	5.4% (1.9–19.3)

[1] *Based on a subsample of all available nests (see Methods and Analysis section).*

Gulls - Island 20: Similar to years past, consumption rates of smolts by gulls nesting on Island 20 varied by salmonid ESU/DPS, with rates ranging from 0.1% (<0.1–0.6%) for SR Fall Chinook salmon to 3.4% (2.1–5.6%) for UCR steelhead (see *Table 9* above). No terns nested on Island 20 in 2022 or any other year, so smolts were presumable directly captured by gulls nesting at that site. 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 relative to salmon ESUs, factors that increase the susceptibility of steelhead to plunge-diving (surface oriented) predators like gulls and terns (Evans et al. 2016, Evans et al. 2019, Cramer et al. 2021a, Hostetter et al. 2023). Estimates of smolt consumption by gulls nesting on Island 20 in 2022 were lower than those observed in 2021, but similar to those in several other years, with estimates generally less than 1.0% per salmon ESU and approximately 3–4% per steelhead DPS in previous years (*Appendix A*). Years with the highest estimates were those of 2015 and 2021, where estimates in excess of 5% and 7% of available SR and UCR steelhead, respectively, were observed (*Appendix A*). These increases 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 outmigration (Hostetter et al. 2012, Roby et al. 2016, Payton et al. 2016, Hostetter et al. 2023).

Gulls - Badger Island: Consumption rates on salmonid smolts for gulls nesting at Badger Island ranged from 0.2% (0.1–0.8%) for SR Fall Chinook salmon to 6.6% (3.3–11.4%) for UCR steelhead. Consumption rates by gulls nesting at Badger Island were also high for SR steelhead with an estimated 3.4% (1.6–7.5%) of available fish consumed in 2022 (see *Table 9* above). Similar to consumption estimates for gulls from the colony on Island 20, predation rates by gulls breeding on Badger Island were consistently higher on steelhead DPSs relative to salmon ESUs (see *Table 9* above and *Appendix A*). Unlike the gull colony at Island 20, however, terns were co-nesting with gulls on Badger Island in 2022, so an unknown number of smolt PIT tags recovered on the Badger Island gull colony were from smolts initially captured by terns and subsequently kleptoparasitized by gulls.

Gulls - Crescent Island: Consumption rates on salmonid smolts for gulls nesting on Crescent Island, a colony that reformed in 2020, ranged from 0.1% (<0.1–0.7%) for SR Fall Chinook salmon to 2.4% (1.2–4.3%) for SR steelhead in 2022 (see *Table 9* above). Estimates were 1.3% (0.6–2.5%) on UCR steelhead and were less than 0.2% for all salmon ESUs evaluated (see *Table 9* above). Consumption estimates by gulls on Crescent Island in 2022 were similar to those observed in 2021 but were substantially lower than those observed during 2007-2015 (*Appendix A*). Lower estimates of smolt consumption since colony reformation in 2020 were likely related to the smaller size of the gull colony on Crescent Island (average of 3,509 adults counted during 2021-2022, compared with 7,379 adults during 2007-2015) and/or the smaller size of the tern colony on Crescent Island, whereby kleptoparasitism rates were also potentially lower during 2021-2022 due to the smaller size of the tern colony on Crescent in these years.

Gulls - Blalock Islands: In most years dating back to 2012, there have been gull colonies on two separate islands (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 (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). Data from Hostetter et al. (2015) and Cramer et al. (2021a) also indicated that per capita 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 (Cramer et al. 2021a).

In 2022, a gull colony never formed on Straight Six Island but gulls did nest on Anvil Island. Consumption rate estimates from Anvil Island ranged from 0.3% (0.1–1.2%) in SR Fall Chinook to 5.5% (2.8–10.2%) in SR steelhead (see *Table 9* above). Similar to other nearby gull colonies, consumption estimates were highest on steelhead DPSs compared with salmon ESUs. Estimates of consumption rates for gulls nesting at the Blalock Islands in 2022 were similar to those observed in 2021, which was higher than those observed in most, but not all, previous years dating back to 2013, the first year gull colonies in Blalock Islands were scanned for smolt PIT tags (*Appendix A*).

Gulls - Miller Rocks: Estimates of smolt consumption rates by gulls breeding on Miller Rocks in The Dalles Reservoir ranged from 0.8% (0.2–2.3%) for SR spring/summer Chinook to 5.7% (2.7–11.8%) for SR steelhead (see *Table 9* above). An estimated 1.9% (0.8–4.1%) of SR Fall Chinook were also consumed by Miller Rocks gulls, the highest estimate by a gull colony on a salmon ESU in 2022 (see *Table 9* above). Although the sample size of PIT-tagged SR sockeye were too small to generate reliable estimates of consumption, results from previous years indicate that, among available salmon ESUs, SR sockeye smolts were 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%) of SR sockeye observed in some years (*Appendix A*).

Estimates of smolt consumption rates by gulls nesting at Miller Rocks in 2022 were consistently lower than those observed in 2021 but similar to those in several other years dating back to 2007 (*Appendix A*). In 2022, as part of a pilot study, the Confederated Tribes and Bands of the Yakama Nation used human hazing and falconry to dissuade gulls from breeding on Miller Rocks during 1 April to 25 April (B. Sharp, Yakama Nation, personal communication). Although efforts were successful at preventing colony formation while dissuasion occurred, a colony quickly formed once dissuasion activities ceased in late-April. By late-May a peak count of 5,832 adult gulls was observed in 2022 (see *Table 2* above). The average annual peak count was 4,466 adult gulls on Miller Rocks during 2007-2021 (Cramer et al. 2021a, Evans et al. 2022b).

Miller Rocks is located in The Dalles Reservoir, 23 Rkm upstream from The Dalles Dam and 18 Rkm downstream from John Day Dam. Evans et al. (2016) observed that gull 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 also no terns nesting on Miller Rocks, so salmonid smolts were presumably captured and consumed by gulls and not kleptoparasitized from terns. Nevertheless, the proportion of smolts consumed by gulls in tailraces of The Dalles and John Day dams that were dead, moribund, or otherwise compromised when consumed, rather than depredated when alive and in a healthy state, is currently unknown. Addressing this uncertainty is paramount to understanding additive mortality of salmonid smolts due to gull depredation (see *Additive Effects of Predation* section below for additional details).

Cormorants - Lenore Lake: Despite an estimated colony size of 159 breeding pairs, only 15 smolt PIT tags were recovered from the Lenore Lake cormorant colony following the breeding season in 2022 (see *Table 5 above*), resulting in predation rate estimates of < 0.1% per salmonid ESU/DPS (*Table 10*). The colony was scanned for smolt PIT tags during 2018-2020, when just two PIT tags were recovered, one from a 2018 migration year smolt and one from a 2020 migration year smolt. Results provide strong evidence that Lenore Lake cormorants pose little or no threat to out-migrating juvenile salmonids. Unlike terns breeding on islands in Lenore Lake (see *above*), cormorants apparently do not regularly commute to the middle Columbia River to forage on anadromous juvenile salmonids during the breeding season (Cramer et al. 2021a).

Table 10. Estimated predation rates (95% credible interval) on Snake River (SR), Upper Columbia River (UCR), and Middle Columbia River (MCR) salmonid populations (ESUs/DPSs), with runs of spring (Sp), summer (Su), and fall Fish, by double-crested cormorants nesting at North Rock in Lenore Lake, Hanford Island in the middle Columbia River, Foundation Island in McNary Reservoir, and the Troutdale Transmission Towers in the lower Columbia River during 2022. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section). 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 from that ESU/DPS.

ESU/DPS	Lenore Lake	Hanford Island	Foundation Island	Troutdale Towers [1]
SR Sockeye	NA	NA	NA	4.4% (1.2–9.5)
SR Sp/Su Chinook	< 0.1%	< 0.1%	3.7% (2.0–7.0)	2.4% (1.3–4.3)
UCR Sp Chinook	NA	NA	NA	1.7% (0.4–3.6)
SR Fall Chinook	< 0.1%	< 0.1%	1.1% (0.4–2.7)	0.7% (0.1–2.0)
SR Steelhead	< 0.1%	< 0.1%	3.5% (1.9–6.5)	2.8% (1.2–5.0)
UCR Steelhead	0.2% (<0.1-0.4)	0.1% (<0.1-0.2)	0.5% (0.1–1.2)	3.2% (1.2–6.1)
MCR Steelhead	-	-	-	3.2% (0.8–7.0)

[1] *Based on a subsample of all available nests (see Methods & Analysis section).*

Cormorants - Hanford Island: The cormorant colony on Hanford 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 previous years, estimates of predation rates in 2022 were low, with less than 0.2% of available fish consumed per ESA-listed ESU/DPS. The smaller size of this cormorant colony (87 pairs; see *Table 2* above) is presumably responsible, in part, for the low predation rates on ESA-listed juvenile salmonids. However, most of the smolt PIT tags recovered from this colony were from non-listed sub-yearling Chinook from the Upriver Bright population, which accounted for 73% (261 out of 358 recovered tags) of the 2022 migration year smolt tags recovered from this colony following the breeding season. Recently published research indicated that Upriver Bright Fall Chinook were especially susceptible to predation by cormorants breeding on Foundation Island and American white pelicans breeding on Badger Island in McNary Reservoir due to the close proximity of these colonies to areas where sub-yearling Chinook are released from hatcheries (e.g., Priest Rapids Hatchery) and/or where wild fish rear and congregate in the Hanford Reach (Payton et al. 2023). The effects of Hanford Island cormorants on non-listed sub-yearling Chinook, however, are currently unknown but maybe appreciable given the relatively large number of tags recovered in 2022.

Cormorants - Foundation Island: Estimates of predation rates for cormorants nesting at the colony on Foundation Island, the largest cormorant colony located on the Columbia River upstream of Bonneville Dam, ranged from 0.5% (0.1-1.2%) on UCR steelhead to 3.7% (2.0–7.0%) on SR spring/summer Chinook salmon in 2022 (see *Table 10* above). Predation rates on SR steelhead were also substantial at 3.5% (1.9–6.5%) of available tagged smolts. Similar to previous years, predation rates by Foundation Island cormorants on UCR smolts were substantially lower than those of SR smolts (see *Table 10* above). 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, Evans et al. 2016).

Predation rates for cormorants nesting at the Foundation Island colony in 2022 were similar to those of previous years (see *Appendix A*), as were estimates of colony size (266 breeding pairs in 2022; see *Table 2* above). 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 have averaged 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 A*). 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).

Cormorants - Troutdale Transmission Towers: For the first time since the colony site was discovered in 2012, smolt PIT tags were recovered from a cormorant colony located on five transmission towers near the town of Troutdale, OR approximately 45 Rkm downstream of Bonneville Dam. Counts of the number of breeding pairs on TRT indicate that the colony has grown from an estimated 26 breeding pairs in 2013 to 352 pairs in 2022 (Lawes et al. 2021b, Lawonn 2023, see *Table 2* above). Smolt PIT tags were recovered in the area underneath three of five transmission towers with nesting cormorants on them, with per capita predation rates used to generate colony-wide predation rates (see *Methods & Analysis* section for details). Per capita predation rates ranged from 0.0019% (0.0001–0.0058%) on SR Fall Chinook to 0.0126% (0.0036–0.0271%) in SR sockeye (*Table 11*). Colony-wide estimates of predation rates ranged from 0.7% (0.1–2.0%) on SR Fall Chinook to 4.4% (1.2–9.5%) on SR sockeye (see *Table 10* above). Analogous to predation by cormorants nesting on the AMB colony site, predation rates by cormorants breeding at the TRT were similar across steelhead and salmon ESUs/DPSs (see *Table 10* above).

Table 11. Estimated per capita (per breeding pair) 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 Foundation Island in McNary Reservoir, Troutdale Transmission Towers in the lower Columbia River, the Astoria-Megler Bridge in the Columbia River estuary, and East Sand Island in the lower Columbia River estuary. Estimates from East Sand Island are those of Lawes et al. (2021b) and represent average annual estimates prior to management on East Sand Island during 2003-2014. Estimates from other sites are from 2022 only. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section).

ESU/DPS	Foundation Island, 2022	Troutdale Towers, 2022	Astoria-Megler Bridge, 2022	East Sand Island, 2003-2014
SR Sockeye	NA	0.0126% (0.0036–0.0271)	0.0016% (0.0004–0.0036)	0.0003% (0.0002–0.0004)
SR Sp/Su Chinook	0.0140% (0.0076–0.0263)	0.0069% (0.0036-0.0124)	0.0012% (0.0007–0.0020)	0.0004% (0.0004–0.0005)
UCR Sp Chinook	NA	0.0047% (0.0011–0.0104)	0.0013% (0.0005–0.0025)	0.0003% (0.0002–0.0004)
SR Fall Chinook	0.0041% (0.0015–0.0100)	0.0019% (0.0001–0.0058)	0.0008% (0.0001–0.0019)	0.0003% (0.0002–0.0003)
SR Steelhead	0.0132% (0.0071–0.0245)	0.0081% (0.0035–0.0144)	0.0018% (0.0008-0.0030)	0.0006% (0.0005–0.0007)
UCR Steelhead	0.0017% (0.0005–0.0046)	0.0091% (0.0034–0.0176)	0.0021% (0.0008–0.0037)	0.0005% (0.0005–0.0006)

Results from this and several other studies indicate that per capita predation rates are closely associated with colony location and the relative availability of juvenile salmonids compared with other prey/forage fish (Collis et al. 2001, Roby et al. 2002, Cramer et al. 2021a, Evans et al. 2022a, Hostetter et al. 2023). In the present study, per capita cormorant predation rates on ESA-listed salmonid smolts significantly increased the further upstream the colony was located from marine zone of the CRE (see *Table 11* above). For example, per capita predation rates by cormorants breeding on the AMB in the marine freshwater mixing zone of the CRE were approximately 4–5 times greater (depending on species and ESU/DPS) than those of cormorants that nested on ESI in the marine zone of the CRE (see *Table 11* above; see also *Estuary Management Plans* section above). Per capita predation rates by cormorants nesting on the TRT in the freshwater zone of the lower Columbia River were approximately 2–5 and 9–10 times greater (depending on the species and ESU/DPS) than those of cormorants on the AMB and ESI, respectively (see *Table 11* above). Of all the cormorant colonies evaluated in 2022, per capita predation rates were consistently the highest by cormorants breeding on Foundation Island in McNary Reservoir, located over 500 Rkm from the marine-zone of the CRE, where per capita predation rates were nearly 20 times greater than those of cormorants on ESI (see *Table 11* above).

Brandt's Cormorants - Astoria-Megler Bridge: For the first time since the Brandt's cormorant (BRAC) colony was documented on the AMB in 2018, per capita and colony-wide predation rate estimates were generated for BRACs on the AMB in 2022. The same methods used for double-cormorants on the AMB were used for BRAC, whereby smolt PIT tags were recovered from a concrete footing where a known number of BRAC cormorants nested (188 breeding pairs). Per capita predation rates were then used to generate colony-wide predation rates based on the total number of BRAC nests on the AMB (935 pairs; see *Methods & Analysis* section for details). The AMB nesting habitat was complex (e.g., cormorants of different species nested on different vertical levels of the bridge). Although we limited PIT-tag recovery efforts to concrete footings where only BRACs nested, a small number of recovered tags may have been deposited by double-crested cormorants, tags that could have biased BRAC predation estimates high to an unknown degree. Due to the small numbers of smolt PIT tags recovered from BRAC nests in 2022 (n = 53; see *Table 3* above) predation rates were based on the number of PIT-tagged smolts by species (steelhead, Chinook, sockeye) and age/class (yearling, sub-yearling; for Chinook) for tagged smolts last detected alive passing Bonneville Dam and not by ESA-listed ESUs/DPSs (as was the case for some other colonies in the CRE). This approach provided a substantially larger sample size of available tagged fish from which to generate predation rates by BRAC.

Results indicated that even with some double-crested cormorants likely contributing tags to the scanned area on the BRAC colony, per capita and colony-wide predation rates by BRAC on smolts were amongst the lowest of all colonies evaluated in the CRE in 2022, with per capita predation rates ranging from 0.0002% (<0.0001–0.0005) on yearling Chinook to 0.0007% (0.0001–0.0019) on sub-yearling Chinook. Colony-wide predation rates ranged from 0.2% (< 0.1%–0.5%) for yearling Chinook to 0.7% (0.1–1.7%) for sub-yearling Chinook (*Table 12*). Comparisons of colony-wide predation rates by BRAC on the AMB compared with BRAC that formerly nested on ESI (prior to colony abandonment), indicate that predation effects by BRAC on the AMB in 2022 were similar to, but slightly higher than, those of BRAC on ESI during 2012–2015 (Cramer et al. 2021a). Although colony-wide predation rates for BRAC were similar between the AMB and ESI, the BRAC colony on the AMB in 2022 was substantially smaller (935 breeding pairs) compared to the BRAC colony during 2012–2015 on ESI (average of 1,727 breeding pairs). As such, per capita predation rates on yearling Chinook, steelhead, and sub-yearling Chinook for BRAC nesting at the AMB were 1.7, 1.8, and 4.1 times higher, respectively, compared to per capita predation rates on the same species and age classes for BRAC nesting on ESI. Despite higher per capita predation effects by BRAC on AMB versus ESI, however, colony-wide predation rate results indicated that Brandt's cormorants nesting on the AMB posed little risk to the survival of juvenile salmonids in 2022 (*Table 12*).

Table 12. Estimated predation rates (95% credible interval) on yearling Chinook, sub-yearling Chinook, steelhead, and sockeye by Brandt's cormorants (BRAC) nesting on the Astoria-Megler Bridge in 2022. Estimates are based on all PIT-tagged smolts by species and age-class last detected passing Bonneville Dam in 2022 (see Methods).

Species/Age-class	Predation Rate
Yearling Chinook	0.2% (<0.1–0.5)
Sub-yearling Chinook	0.7% (0.1–1.7)
Sockeye	NA
Steelhead	0.2% (<0.1–0.6)

Pelicans - Badger Island: Estimates of predation rates by American white pelican breeding on Badger Island, the largest pelican colony in the CRB, were consistently lower than those for terns, gulls, and cormorants nesting at nearby colonies in McNary Reservoir, with estimates ranging from 0.3% (0.1–0.9%) for SR Fall Chinook salmon to 0.9% (0.1–2.8%) for SR steelhead (*Table 13*). Results from 2022 where very similar to those in 2020 and 2021 (*Appendix A*) and suggest that pelicans breeding on Badger Island posed little threat to actively migrating 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 are migrating (Stinson 2016; Payton et al. 2023).

Table 13. 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 2022. 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	0.4% (0.1–1.3)	< 0.1%
UCR Sp Chinook	NA	< 0.1%
SR Fall Chinook	0.3% (0.1–0.9)	< 0.1%
SR Steelhead	0.9% (0.1–2.8)	< 0.1%
UCR Steelhead	0.7% (0.2–2.6)	< 0.1%
MCR Steelhead	-	< 0.1%

In a recently published study of predation on non-listed sub-yearling Chinook from the Upriver Bright stock in the Hanford Reach of the middle Columbia River, Payton et al. (2023) estimated that predation rates by pelicans nesting on Badger Island were substantial, with upwards of 25% of non-listed sub-yearling smolts consumed by pelicans in some years. Estimates of pelican predation rates were also significantly higher on wild sub-yearling Chinook compared with their hatchery counterparts, fish that rear and reside in the Hanford Reach of the middle Columbia River prior to outmigration (Payton et al. 2023). Large numbers of tags from smolts released into the Yakima River, which enters the middle Columbia River 28 Rkm upstream of Badger Island, have also been recovered on the Badger Island pelican colony in recent years (authors, unpublished data). Given evidence that particular salmon stocks and age-classes 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 (see also Payton et al. 2023).

Unlike gulls and terns, pelicans are capable of consuming adult-sized salmonids, with PIT tags implanted in adult sockeye, adult steelhead, 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 (adult steelhead) have been consumed by Badger Island pelicans (Roby et al. 2017). In 2021 and 2022, 47 and 12 tags, respectively, from adult sockeye that were tagged at the Bonneville Dam adult fishway were recovered on the Badger Island pelican colony (Evans et al. 2022b, see *Table 3* above). Given that relatively small numbers of adult sockeye that are annually tagged at Bonneville Dam (e.g., 1,534 adults in 2021; PSFMC 2022), predation rates by pelicans on adult sockeye may be greater than that on juvenile sockeye. Kock et al. (2021) observed that pelicans consumed upwards of 40% of adult sockeye 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 adult sockeye are migrating upstream. Thus, future research to estimate predation rates by pelicans on adult sockeye is warranted given the number of adult sockeye tags recovered on the Badger Island pelican colony and the increasing size of the Badger Island pelican colony, with the counts from 2021 (n = 3,624 individuals) and 2022 (n = 3,486) 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, pikeminnow (see *Table 3* above; see also Cramer et al. 2021a), and even adult shortnose suckers (*Chasmistes brevirostris*) from the Klamath Basin (U.S. Geological Survey Western Fisheries Research Center unpublished data). 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 pelicans breeding on Miller Sands Spit in the CRE were the lowest of those colonies evaluated in the mainstem Columbia River in 2022. Despite an estimated colony size of 992 individuals, only 58 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 13* above). Estimates of predation rates by pelicans on Miller Sands Spit in 2022 were very similar to those in 2021, with predation rates of < 0.1% per salmonid ESU/DPS also observed in 2021 (Evans et al. 2022a). Similar to the American white pelicans breeding on Badger Island, results suggest that pelicans breeding on Miller Sands Spit pose little threat to the survival of ESA-listed juvenile salmonids from UCR, MCR, and SR ESUs/DPSs, particularly predation on actively migrating, yearling smolts. However, of the 58 smolt PIT tags recovered on the Miller Sands Spit pelican colony, 37 or 64% were from sub-yearling Chinook, fish that were potentially still rearing and/or that congregated in shallow water habitats in the lower CRE (Sebring et al. 2013). Sample sizes of PIT-tagged sub-yearling Chinook from the Lower Columbia River ESU, however, were too small to generate reliable estimates of pelican predation in 2022. As such, future research may be warranted to determine to what extent pelicans breeding at colonies in the CRE limit the survival of sub-yearling Chinook or other smolts that originate from LCR tributaries.

Cumulative Predation & Survival

Avian predation effects were first observed in the river reach located downstream of RIS and downstream of LMN. For SR smolts, results indicating there was no measurable consumption of tagged fish between LGR and LMN associated with the bird colonies evaluated in this study. For UCR smolts, however, predation occurred following release/detection at RIS. Estimates of avian predation/consumption and total mortality (1 - survival) of smolts downstream of RIS and LMN – the first location where predation/consumption was documented – varied considerably by bird species, colony location, river reach, and year and are provided below by salmonid species and ag.

Upper Columbia River Steelhead: The cumulative effects of avian predation/consumption (predation by all avian predator species and colonies combined) on UCR steelhead smolts during passage from RIS to the Pacific Ocean in 2022 were substantial at an estimated 29.4% (24.1–36.2%; *Figure 6*). Of the avian predator species evaluated, estimated consumption by gull colonies was the greatest at 17.9% (13.0–23.9%), followed by cormorant predation at 5.9% (3.6–9.6%), tern predation at 4.8% (3.8–6.3%), and pelican predation at 0.5% (0.1–2.0%). It is important to note that cumulative estimates are based on smolt availability at RIS or LMN as opposed to the proportion that survive outmigration to within the foraging range of each downstream bird colony (see *Reach-specific Predation* section below for estimates that account for survival to within the foraging range of birds from each colony). The cumulative effects of avian predation/consumptions on UCR steelhead smolts in 2022 were lower than those observed in 2021 and lower than those observed in several, but not all, years dating back to 2008 (*Figure 6*). It should also be noted that estimates of cumulative predation by gull colonies on Island 20 and the Blalock Islands (Anvil and Straight Six) during 2008-2012 and by cormorants on FDI during 2013, 2015-2019, the AMB during 2016-2022, and the TRT during 2012–2021 were not available because these sites were not scanned for smolt PIT tags in these years. As such, cumulative and reach-specific predation estimates (see *below*) were minimum estimates of predation on UCR and SR ESUs/DPSs in several, but not all, years (as noted in figure descriptions, e.g., *Figure 6*).

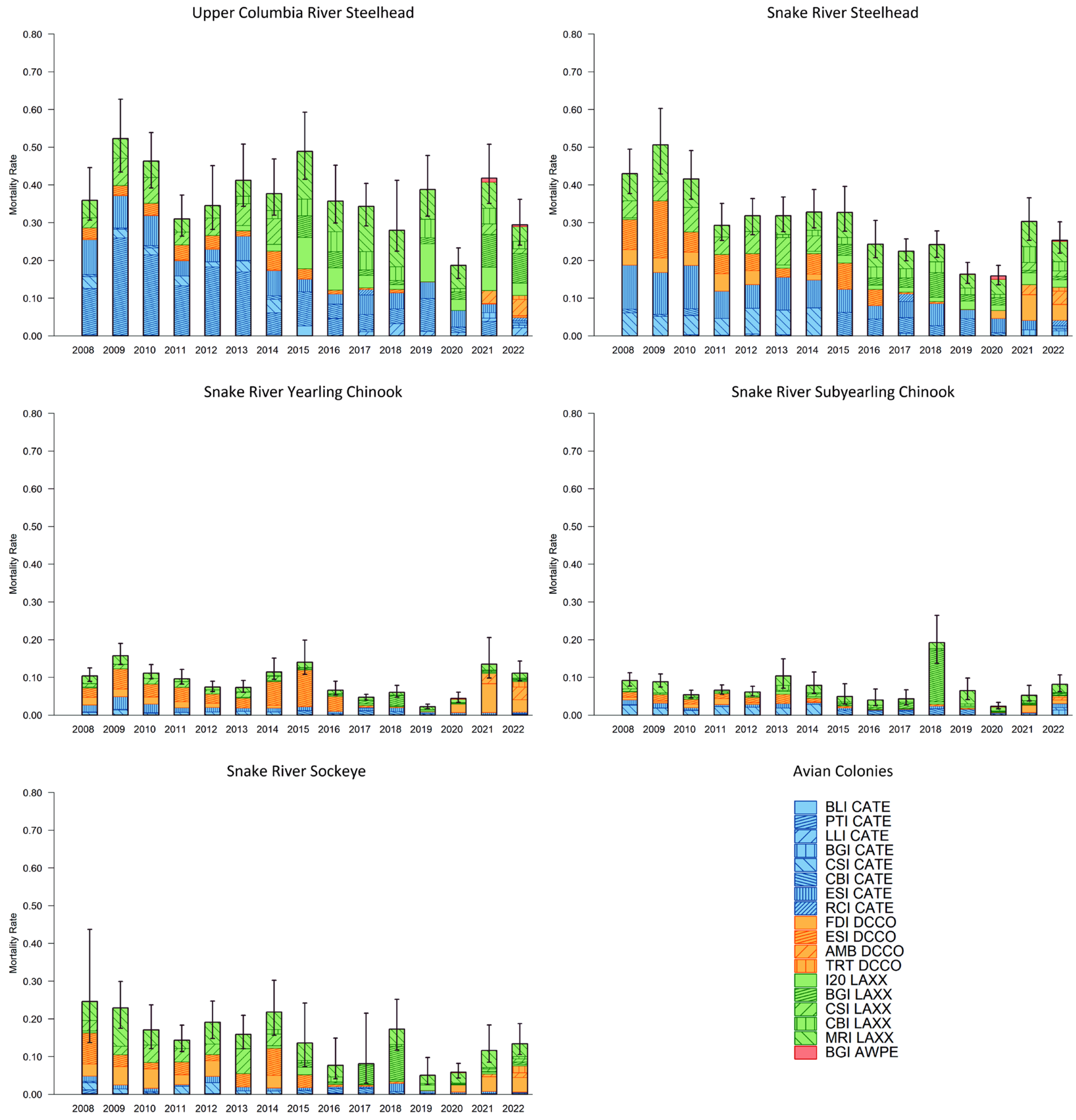


Figure 6. Estimated cumulative predation/consumption for Upper Columbia River steelhead smolts during passage from Rock Island Dam to the Pacific Ocean and for Snake River steelhead, yearling Chinook, subyearling Chinook, and sockeye smolts during passage from Lower Monumental Dam to the Pacific Ocean. Avian predator species include Caspian terns (CATE), double-crested cormorants (DCCO), California and ring-billed gulls (LAXX), and American white pelicans (AWPE; see Map 1 for colony locations). Estimates are proportions with error bars representing 95% credible intervals. No estimates of predation were available for IS20 and CBI LAXX during 2008-2012, for FDI DCCO during 2013, 2015-2019, AMB DCCO during 2016-2020, and TRT DCCO during 2012-2021, resulting in minimum estimates of cumulative predation in those years. Data from 2008-2018 are those of Evans et al. (2019, 2022b).

Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 65.2% (53.5–80.2%) of all UCR steelhead mortality sources during smolt passage from RIS to BON in 2022. Avian predation/consumption has accounting for more > 50% of all UCR steelhead smolt mortality sources in 12 of the last 15 years (*Figure 7*; see also Evans et al. 2022b). Results of this and several other studies (Evans et al. 2016, Payton et al. 2019, Evans et al. 2019, Evans et al. 2022b) provides strong evidence that avian predation/consumption was the dominate source of fish mortality during smolt outmigration through the middle and lower Columbia Rivers over the past 15 years.

The representative tagging (random and in proportion to the run at-large) of hatchery and wild UCR steelhead (see *PIT-tagging of Upper Columbia River Steelhead* section above) at RIS generated a unique dataset to make relative comparisons of predation and survival based on a fish's rear-type (see *Appendix B* for details). The process of capturing, handling, and tagging steelhead at RIS may also influence predation and survival probability estimates if smolts that are handled at RIS are less likely to survival outmigration and are more likely to be consumed by birds compared with unhandled smolts. To address this possibility, we investigated predation and survival probabilities of PIT-tagged steelhead smolts that were captured and tagged at RIS versus those that were passively detected (i.e., previously tagged and not handled) at the Rocky Reach Dam juvenile bypass facility, located 33 Rkm upstream of RIS (see *Appendix C* for details).

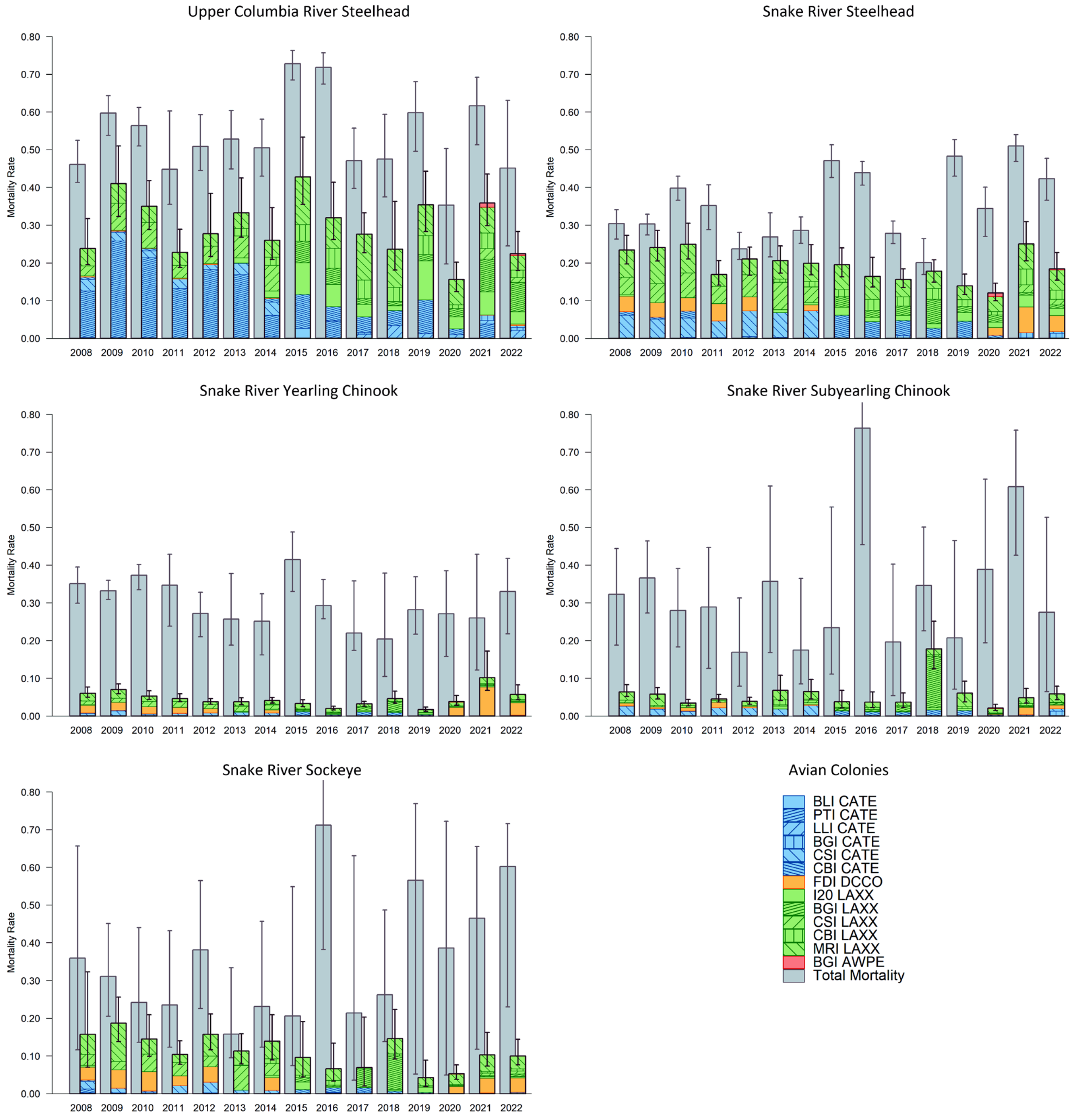


Figure 7. Estimated total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Upper Columbia River steelhead during passage from Rock Island Dam to Bonneville Dam or Snake River steelhead, yearling Chinook, sub-yearling Chinook, and sockeye during passage from Lower Monumental Dam to Bonneville Dam. Avian predator species include Caspian terns (CATE), double-crested cormorants (DCCO), California and ring-billed gulls (LAXX), and American white pelicans (AWPE; see Map 1 for colony locations). Estimates are proportions with error bars representing 95% credible intervals. No estimates of predation were available for IS20 and CBI LAXX during 2008-2012 and FDI DCCO in 2013, 2015-2019, resulting in minimum estimates of predation in those years. Data from 2008-2018 are those of Evans et al. (2019, 2022b).

Snake River Steelhead: The cumulative effects of avian predation/consumption on SR steelhead smolts during passage from LMN to the Pacific Ocean in 2022 were similar to those of UCR steelhead, with an estimated 25.3% (21.9–30.2%) of SR steelhead smolts consumed (see *Figure 6* above). The large numbers of PIT-tagged steelhead detected at LGR (n = 106,573) that were used in this analysis resulted in more precise estimates of predation and survival compared with steelhead smolts released/detected at RIS (n = 6,217). Of the avian predator species evaluated, estimated consumption by gull colonies was the greatest at 12.0% (9.9–15.2%), followed by cormorant predation at 8.7% (6.7–12.2%), tern predation at 4.1% (3.4–5.1%), and pelican predation at 0.3% (0.1–0.6%; see *Figure 6* above). Predator-specific aggregate predation effects in SR steelhead were similar to those of UCR steelhead, with the exception of predation by cormorant colonies, where predation by cormorants on SR steelhead was higher than that of cormorants on UCR steelhead due to predation by cormorants on Foundation Island in McNary Reservoir, a colony that disproportionately consumed SR migrates compared with those originating from the UCR (see also *Foundation Island Cormorants* section above). The cumulative effects of avian predation/consumptions on SR steelhead smolts in 2022 were lower than those observed in 2021 and lower than those in several, but not all, previous years (see *Figure 6* above).

Estimated total mortality (1 - survival) of SR steelhead during passage from LGR to LMN was relatively low at 8.3% (4.5–13.1) in 2022. Estimated total mortality of SR steelhead smolts, however, increased significantly downstream of LMN at an estimated 42.3% (36.6–47.7%) during passage from LMN and BON (see *Figure 7* above). Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 43.5% (32.4–62.0%) of all SR smolt mortality sources during passage from LMN to BON in 2022 (see *Figure 7* above). Analogous to results on UCR steelhead, avian predation/consumption accounting for more > 50% of all SR steelhead smolt mortality sources in 10 of the last 15 years (see *Figure 7* above; Evans et al. 2022a).

Snake River Yearling Chinook: The cumulative effects of avian predation/consumption on SR yearling Chinook during passage from LMN to the Pacific Ocean in 2022 were significantly lower than those of SR and UCR steelhead, with estimated 11.1% (9.1–14.3%) of SR yearling Chinook smolts consumed (see *Figure 6* above). Analogous to SR steelhead, large numbers of PIT-tagged yearling Chinook detected at LGR (n = 104,530) resulted in precise estimates of predation and survival downstream of LGR. Of the avian predator species evaluated, estimated predation/consumption was the highest by cormorant colonies at 8.1% (6.2–11.2%), followed by gulls at 2.2% (1.7–2.9%), terns at 0.7% (0.5–1.1%), and pelicans at 0.1% (< 0.1–0.4%; see *Figure 6* above). The cumulative effects of avian predation/consumptions on SR yearling Chinook smolts in 2022 were similar to estimates in previous years (see *Figure 6* above; Evans et al. 2022a). Again, however, a lack of predation estimates from cormorants on Foundation Island during 2013, 2015-2019, AMB during 2016-2020, and TRT during 2012-2021, resulted in minimum estimates of cumulative predation in these years.

Comparisons of avian predation/consumption to total mortality indicated that avian predation/consumption accounted 29.6% (19.1–47.8%) of all SR yearling Chinook smolt mortality sources during passage from LMN to BON in 2022 (see *Figure 7* above). Results are consistent with those in the past, with avian predation/consumption annually accounting for 7% (in 2016) to 40% (in 2021) of all SR yearling Chinook smolt mortality during passing from LMN to BON, depending on the year (see *Figure 7* above; see also Evans et al. 2022b). Collectively, results suggest that upstream of Bonneville Dam, predation/consumption on SR yearling Chinook by colonies included in this study posed a much lower risk to smolt survival than that observed on UCR and SR steelhead, albeit avian predation was a substantial source of smolt mortality in some river reaches and years (see also Evans et al. 2022b for a more detailed discussion).

Snake River Sub-yearling Chinook: The cumulative effects of avian predation/consumption on SR sub-yearling Chinook during passage from LMN to the Pacific Ocean in 2022 were similar to those of SR yearling Chinook, with estimated 8.2% (6.2–10.6%) of SR sub-yearling Chinook smolts consumed (see *Figure 6* above). Of the avian predator species evaluated, estimated predation/consumption were the greatest by gulls at 3.0% (1.9–4.5), following by terns at 2.9% (2.2–3.8%), cormorants at 2.1% (1.2–3.6%), and pelicans at 0.1% (<0.1–0.4; see *Figure 6* above). The cumulative effects of avian predation/consumptions on SR yearling Chinook smolts in 2022 were similar to those observed in 2021 (ca. 5.8%) and previous years (see *Figure 6* above; Evans et al. 2022a).

Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 21.4% (8.1–100%) of all SR sub-yearling Chinook smolt mortality sources during passage from LMN to BON in 2022 (see *Figure 7* above). Due to smaller sample sizes of sub-yearling Chinook detected at LGR (n = 17,066), however, estimate of survival and predation were much less precise than those SR steelhead and yearling Chinook, particularly estimates of survival to BON (see *Figure 7* above). It is also worth noting that unlike SR steelhead, SR yearling Chinook, and SR sockeye, a large proportion of SR sub-yearling Chinook originate downstream of LGR, and results presented herein apply only to those fish that originated upstream of LGR. With these caveats in mind, results from 2022 were consistent with those of years past, with avian/predation consumption annually accounting for 12% to 39% of all SR sub-yearling Chinook smolt mortality during passing from LMN to BON, depending on the year (see *Figure 7* above; Evans et al. 2022a).

Snake River Sockeye: The cumulative effects of avian predation/consumption on SR sockeye during passage from LMN to the Pacific Ocean were estimated at 13.4% (10.5–18.8%) of smolts in 2022 (see *Figure 6* above). Of the avian predator species evaluated, estimated consumption were the greatest by cormorants at 7.1% (4.8–10.8%), followed by gull colonies at 5.7% (4.2–8.4). Predation estimates were significantly lower by tern and pelican colonies at just 0.6% (0.5–0.9%) and 0.1% (<0.1–0.3%), respectively (see *Figure 6* above). The cumulative effects of avian predation/consumptions on SR sockeye in 2022 were similar to those in 2021 (ca. 11.6%) and similar to those in several other years dating back to 2008 (range = 5% to 24% annually; see *Figure 6* above).

Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 16.5% (10.6–62.8%) of all SR sockeye smolt mortality sources during passage from LMN to BON in 2022 (see *Figure 7* above). Similar to results from SR sub-yearling Chinook, relatively small samples size of PIT-tagged SR sockeye resulted in imprecise estimate of predation and, especially, survival (see *Figure 7* above). Consequently, comparisons of total mortality to mortality due to avian consumption were also relatively imprecise and results should be interpreted cautiously (see also Evans et al. 2022a for a more detailed discussion). With these caveats in mind, the percentage of total SR sockeye morality explained by avian predation during smolt passage from LMN to BON in 2022 (ca. 17%) was similar to, but generally lower than, those in past years (range = 10% to 71% annually, depending on the year; see *Figure 7* above).

In summary, the cumulative effects of avian predation/consumption on salmonid smolts from the UCR and SR and the proportion of all sources of mortality that were associated with avian predation was highly variable across salmonid species, avian predator species, and years. Our approach to jointly investigate multiple bird and prey species that share a common migration corridor revealed several important generalities, including (1) avian consumption was associated with the majority of mortality for steelhead smolts during outmigration, but a relatively small proportion of total mortality for yearling and sub-yearling Chinook smolts; and (2) the species and colony location of piscivorous waterbirds nesting in the Columbia River basin dramatically influenced the magnitude of consumption, with some colonies posing little threat to smolt survival, while others were associated with mortality of a large proportion of the available fish (see also Evans et al. 2019 and 2022b for a more detailed discussion of the cumulative effects of avian predation on CRB salmonids).

Results from this and several other published studies (e.g., Evans et al. 2016, Evans et al. 2019, Payton et al. 2019, Payton et al. 2020, Evans et al. 2022b) indicate that mortality of steelhead smolts associated with predation/consumption by piscivorous colonial waterbirds was greater than that from all other mortality sources combined in most years (2008-2022). For example, SR steelhead smolt losses associated with predation/consumption by piscivorous colonial waterbirds upstream of BON were greater than the combined direct losses associated with passage through five hydroelectric dams (Ice Harbor, 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 combined. 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. 2022a).

Reach-specific Predation/Consumption: An investigation of reach-specific predation/consumption effects, those that account for the survival of smolts to each downstream dam with PIT tag detection capabilities (see *Methods & Analysis* section), indicates that predation effects for UCR steelhead were the greatest during smolt passage from RIS to McNary Dam (MCN) at 15.8% (11.5–21.2%), followed by predation during passage from BON to

Pacific Ocean at 12.5% (7.3–22.7%), from MCN to John Day Dam (JDJ) at 4.3% (2.2–8.4%), and from JDJ to BON at 5.0% (2.3–9.8%) in 2022 (*Figure 8*). See *Figure 8* for estimates of predation/consumption by colony within each river reach. It's important to note that reach-specific estimates of predation/consumption may differ from ESU/DPS- and colony-specific estimates of predation/consumption (see *Avian Predation Rates* section above) because reach-specific estimates partition predation for those colonies capable of foraging in multiple river reaches, while the colony- and ESU/DPS-specific estimates depict the effects of predation on all smolts available to birds at each colony regardless of the river reach. For example, gulls breeding on Miller Rocks can forage on smolts both upstream and downstream of John Day Dam and reach-specific estimates account for this by partitioning predation accordingly (see also Payton et al. 2019 for additional details).

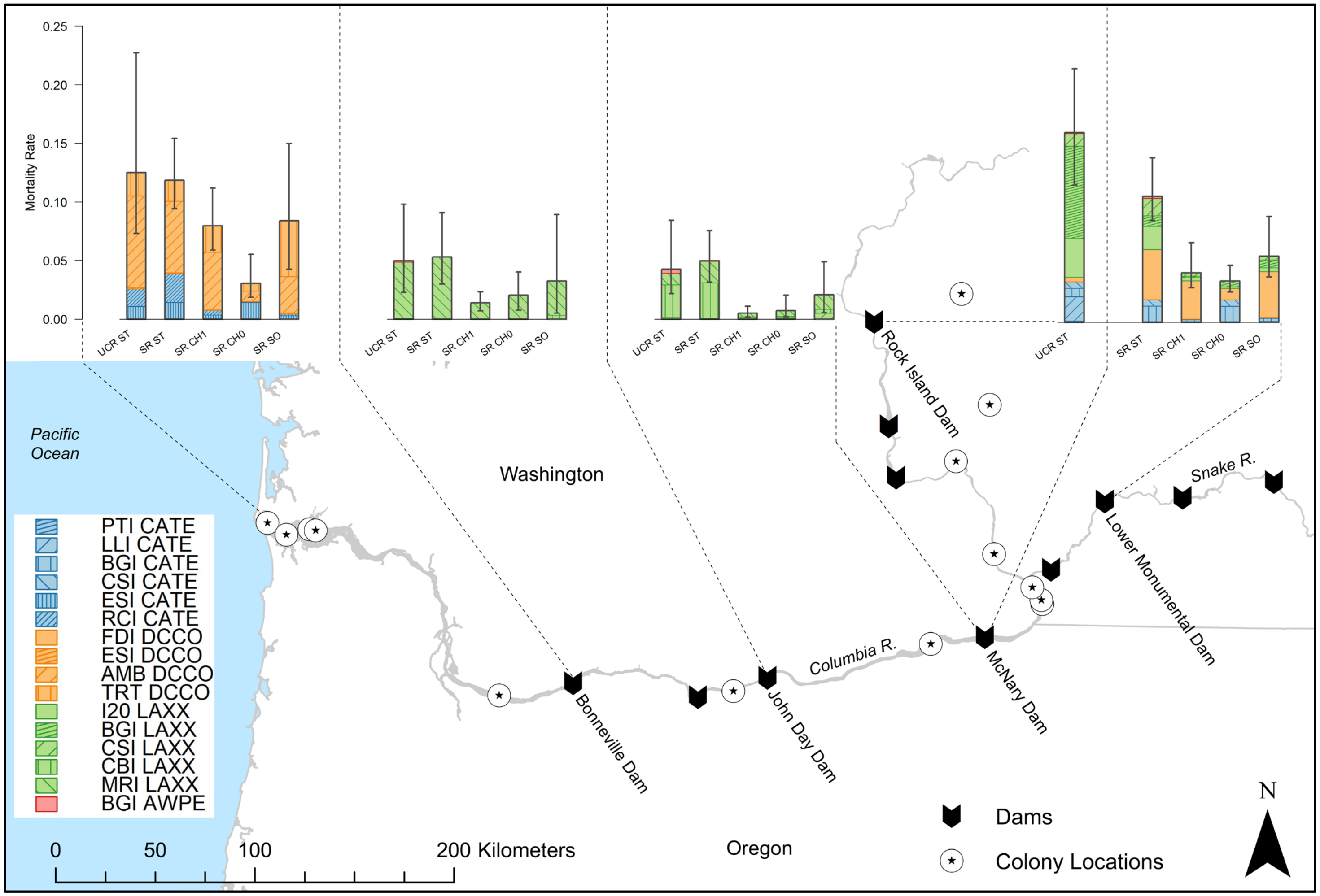


Figure 8. Reach-specific predation/consumption for Upper Columbia River (UCR) steelhead smolts during passage from Rock Island Dam to the Pacific Ocean and for Snake River (SR) steelhead, yearling Chinook (CH1), sub-yearling Chinook (CH0), and sockeye (SO) smolts during passage from Lower Monumental Dam to the Pacific Ocean. Avian predator species include Caspian terns (CATE), double-crested cormorants (DCCO), California and ring-billed gulls (LAXX), and American white pelicans (AWPE; see Map 1 for colony locations). Estimates are proportions with error bars representing 95% credible intervals.

Predation/consumption effects for Snake River steelhead were the greatest during smolt passage from BON to the Pacific Ocean at 11.9% (9.4–15.4%), followed by predation during passage from LMN to MCN at 10.5% (8.5–13.8%), from JDJ to BON at 5.3% (3.0–9.1%), and from MCN to JDJ at 5.0% (3.2–7.6%) in 2022 (see *Figure 8* above). Predation/consumption for Snake River yearling Chinook was also the greatest from BON to Pacific Ocean at 8.0% (5.9–11.1%), followed by LMN to MCN at 4.2% (2.9–6.7%), JDJ to BON at 1.4% (0.7–2.3%), and MCN to JDJ at 0.5% (0.1–1.1%) in 2022 (see *Figure 8* above). Predation/consumption for Snake River sub-yearling Chinook was the greatest from LMN to MCN at 3.5% (2.5–4.8%), followed by BON to the Pacific Ocean at 3.1% (1.5–5.5%), JDJ to BON at 2.1% (0.8–4.0%), and MCN to JDJ at 0.7% (0.2–2.1%) in 2022 (see *Figure 8* above). Finally, predation/consumption for Snake River sockeye was the greatest from BON to the Pacific Ocean at 8.4% (4.3–15.0%), followed by LMN to MCN at 5.6% (3.8–8.9%), JDJ to BON at 3.3% (0.5–8.9%), and MCN to JDJ at 2.1% (0.7–4.9%) in 2022 (see *Figure 8* above).

Reach-specific predation estimates indicate that colonial waterbirds are consuming juvenile salmonids in all river-reaches downstream of RIS and LMN but that predation/consumption varied by reach and by salmonid species. In 2022, predation effects were often the greatest on smolts downstream of BON in the CRE, depending on the species. Since smolts that have survived outmigration through the Columbia River Power System are, on average, more likely to survive to adulthood than those that are yet to complete outmigration, the benefits of managing avian predators downstream of BON may be greater on per fish basis than managing inland avian predators (Roby et al. 2002, Payton et al. 2020). None-the-less, results of this and other studies indicates that predation by some, but not all, inland avian colonies were substantially, especially within the McNary Reservoir and especially on UCR and SR steelhead.

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 MCN and BON. 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 and 2022, however, increases in predation rates by terns at these and other colonies and increases in consumption rates by gulls were both coincident with lower estimates of UCR steelhead survival in this river reach compared with 2020. Terns nesting at the colonies on Goose, Lenore Lake, Crescent, and Badger islands foraged on smolts in the Columbia River between RIS and MCN, and an increase in predation rates by terns nesting at these colonies coincided with lower UCR steelhead smolt survival (*Figure 9*). On average, over the course of the entire management period (2014–2022), however, tern predation rates on UCR steelhead smolts have significantly declined following management actions at Goose and Crescent islands (*Figure 9*). 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 as tern predation rates increase in this river reach, as they did in 2021 and 2022, 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 MCN and BON, 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 MCN and BON during these years (*Figure 9*). During 2020-2022, however, predation rates on UCR steelhead by terns breeding on the Blalock Islands were the lowest recorded since management commenced in 2014 (*Figure 9*), largely due to adaptive management that eliminated the historical tern nesting site on the Blalock islands in 2021 and 2022. Comparisons of tern predation rates on steelhead smolts and survival rates of steelhead smolts downstream of BON were not available due to the lack of smolt survival estimates from BON to the Pacific Ocean (see *Methods & Analysis* section for details).

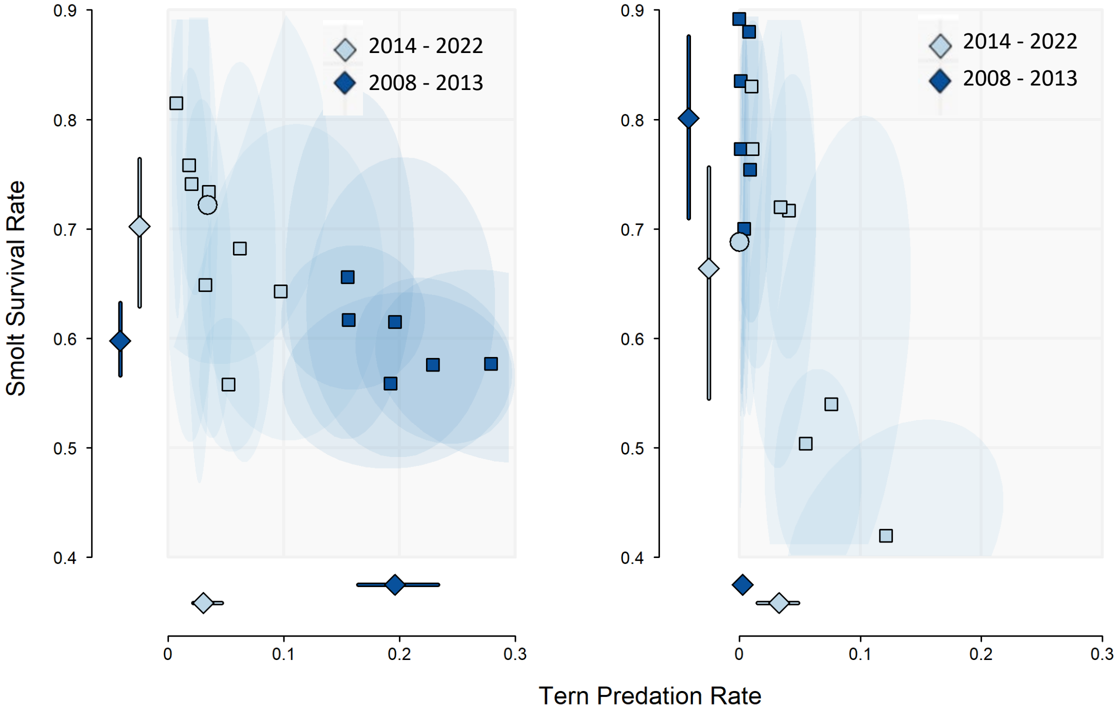


Figure 9. Caspian tern predation rates and survival rates of Upper Columbia River steelhead smolts during passage from Rock Island Dam (RIS) to McNary Dam (left panel) and from McNary Dam to Bonneville Dam (right panel) prior to (dark blue) and following (light blue) management actions that reduced the 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). Circles indicate estimates from 2022. Results are those of Evans et al. (2022b) updated with data from 2022.

In summary, the reach-specific effects of avian predation/consumption on survival of UCR and SR steelhead smolts in 2022 were substantial and continued to be one of the greatest sources of smolt mortality during outmigration. Avian predation/consumption rates by colonies residing upstream of BON in 2022 were generally lower than those observed in 2021 but 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 on UCR steelhead. Increases in tern predation rates on UCR steelhead in 2021 and 2022 were largely due to the reformation of a tern colonies on Crescent and Goose islands, along with predation by terns now nesting on Badger Island. Despite increases in avian predation/consumption and the subsequent decrease in survival rate of UCR steelhead smolts in 2021 and, to a lesser degree in 2022, 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 BON during the management period (2014-2022) as a whole (see *Figure 9* above).

Despite evidence that UCR salmonids have benefited from management actions associated with *IAPMP*, there is little evidence that SR salmonids have benefited from management actions, as predation and survival probabilities on SR smolts have remained largely unchanged since management actions were first implemented at Crescent Island in 2014 (see *Figures 7–8* above). This is due, in part, to the total number of terns foraging on SR smolts being similar in all years since 2008 (see *Table 2* above), with terns dissuaded from Goose and Crescent islands during 2014-2020 relocating to the Blalock islands and then terns dissuaded from the Blalock islands during 2021-2022 relocating to Crescent and Badger islands. All these breeding sites are located downstream of the confluence of Snake and Middle Columbia Rivers where SR smolts are susceptibility to predation.

In the CRE, there is evidence that predation by tern colonies has, on average, decreased following management actions that successfully reduced the size of the colony on ESI, but recent attempts by terns to nest at sites in the upper estuary are off-setting those benefits due to the higher per capita predation rates of terns breeding at colonies in the upper versus lower CRE. There is now also evidence that predation rates by cormorants in CRE are as great or greater than those observed prior to implementation of management actions on ESI in 2015 due to the rapid expansion of the cormorant colony on the AMB and increasing numbers of cormorants on TRT. There are also cormorants nesting on channel markers and on the Lewis and Clark Bridge in the CRE, colonies where predation estimates are not available, but colonies that are likely consuming appreciable numbers of juvenile salmonids based on results from the AMB and TRT colonies in 2022.

Consumption rates by gulls nesting at colonies in the CPR were the greatest of the four avian predator species evaluated in 2022. 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, the actual impact of gull consumption on smolt survival is largely unknown. Recent data suggests that some of the UCR and SR steelhead consumed by gulls between MCN and BON would survive outmigration to Bonneville Dam in the absence of gull consumption (see *Additive Effects of Predation* section below).

Additive Effects of Predation

An updated investigation of the additive effects of gull consumption on UCR and SR steelhead smolts indicated that increases in gull consumption on smolts between MCN and BON were associated with statistically significant decreases in smolt survival to BON (*Table 14* and *Figures 10–11*). Results suggest that significantly more steelhead smolts would, on average, have survived outmigration to BON in the absence of gull consumption, with an estimated difference between observed survival (i.e. survival with gull predation) and baseline survival (i.e. survival in the absence of gull predation) of 0.17 (0.01–0.31) and 0.11 (0.03–0.18) for UCR and SR steelhead, respectively (*Table 14*). For example, the average annual SR steelhead survival probability from MCN to BON during 2013-2022 was 0.73 (0.71–0.75) and results suggest that, if the additive relationship holds, survival would have been 0.84 (0.76–0.91) in the absence of all gull predation.

Table 14. Average annual predation/consumption probabilities for California and ring-billed gulls (LAXX) and survival probabilities for Upper Columbia River (UCR) and Snake River (SR) steelhead smolts during outmigration from McNary Dam to Bonneville Dam during 2013-2022. Estimates of the magnitude of the association between predation probabilities and survival probabilities ($\boldsymbol{\alpha}$, additivity) and the difference in survival probabilities from estimated baseline survival probabilities ($\boldsymbol{\Phi}^{\Delta}$) are also provided. Values are reported as medians with 95% credible intervals.

Predator	ESU/DPS	Survival	Predation	α	Φ^{Δ}
LAXX	UCR steelhead	0.65 (0.60–0.70)	0.15 (0.13–0.17)	0.74 (-0.56–1.90)	0.17 (0.01–0.31)
	SR steelhead	0.73 (0.71–0.75)	0.09 (0.08–0.10)	1.26 (0.23–2.51)	0.11 (0.03–0.18)

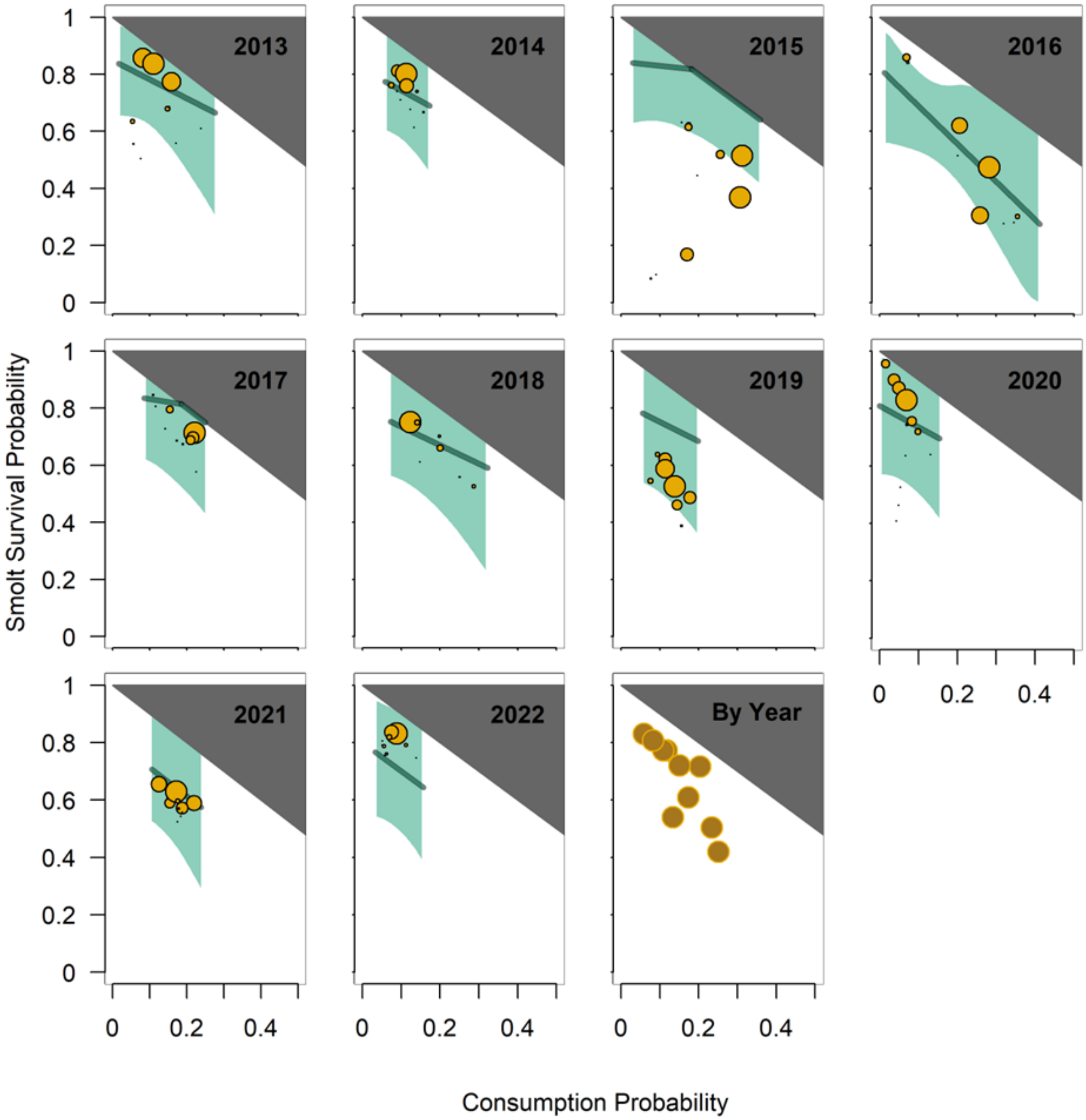


Figure 10. Weekly and annual probability estimates of Upper Columbia River steelhead smolt survival and gull consumption during smolt outmigration from McNary Dam to Bonneville Dam during 2013-2022. The size of circles depicts relative numbers of steelhead smolts available below McNary Dam. Dashed lines represent the best fit estimate of the linear relationship between survival and consumption and shading denotes 95% credible intervals around the best fit. The grey triangle represents the boundary space where consumption cannot be greater than survival.

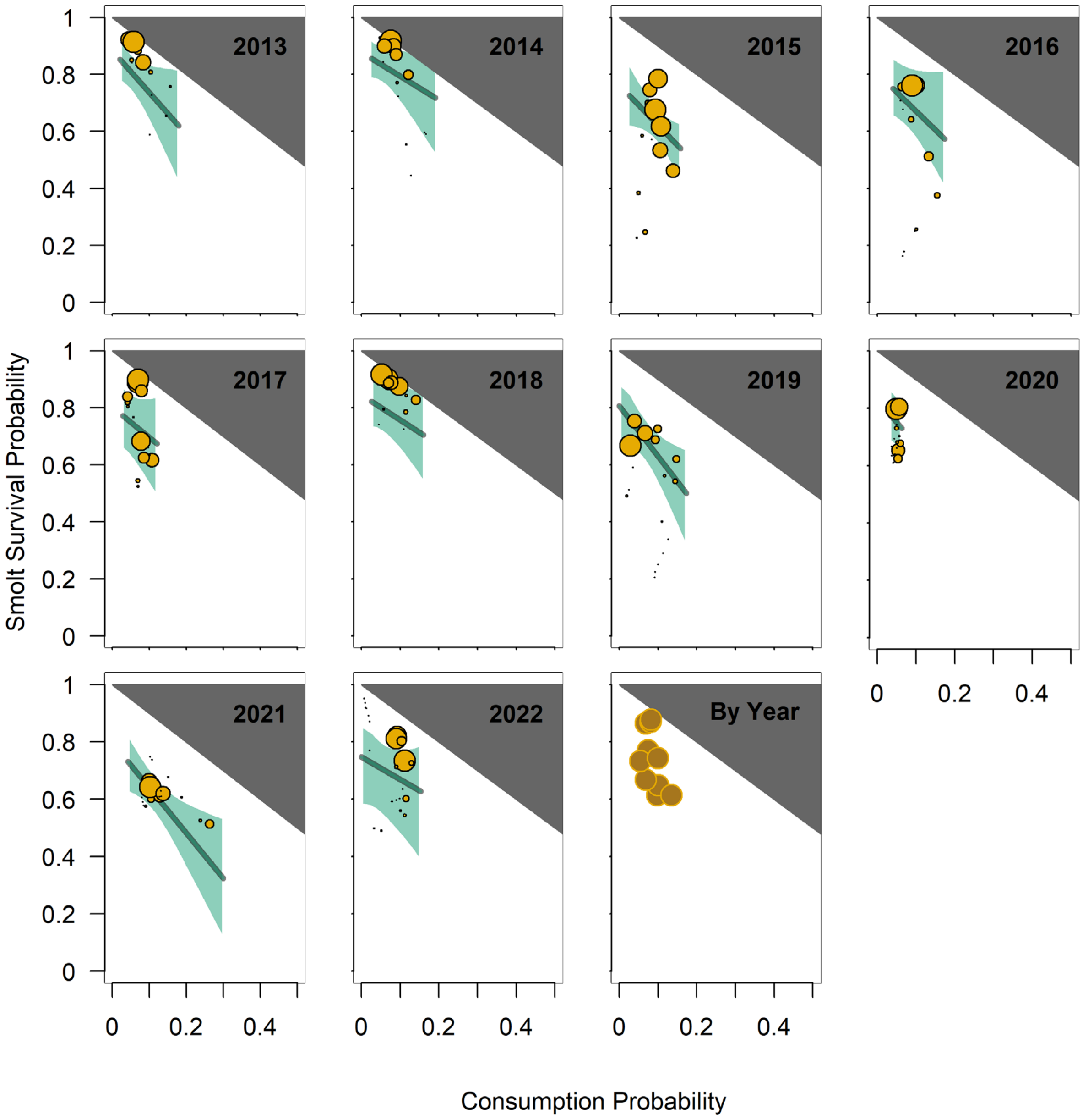


Figure 11. Weekly and annual probability estimates of Snake River steelhead smolt survival and gull consumption during smolt outmigration from McNary Dam to Bonneville Dam during 2013-2022. The size of circles depicts relative numbers of steelhead smolts available downstream of McNary Dam. Dashed lines represent the best fit estimate of the linear relationship between survival and consumption and shading denotes 95% credible intervals around the best fit. The grey triangle represents the boundary space where consumption cannot be greater than survival.

Previously published research demonstrated that predation by terns was an additive source of UCR and SR steelhead mortality during smolt outmigration to BON (Payton et al. 2020). Similar to predation rates by terns, gulls also consumed a substantial proportion of available UCR and SR steelhead smolts (see *Table 14* above). Despite high annual consumption probabilities by gulls, however, previous research was not able to quantify the relationship between gull consumption and fish survival. Weekly estimates of gull consumption were often less variable and less precise than estimates of tern predation and the time series available for gulls was truncated (2013-2018) relative to that of terns (2008-2018; Payton et al. 2021). Furthermore, previous analyses of gull consumption were conducted at larger spatial scales (e.g., RIS to BON) where other mortality factors (both avian [e.g., tern] and non-avian) influenced and possibly masked the effects of gull consumption on smolt survival. Previous analyses also included gulls that were co-nesting with terns (e.g., on Crescent and Goose islands) whereby some proportion of the fish consumed by gulls were those captured by terns, consumption that was necessarily compensatory.

Results suggest that gull consumption was a super-additive (a = 1.26) source of mortality in SR steelhead but a partially additive source in UCR steelhead (a = 0.74; see *Table 14* above). A super-additive source of mortality suggest that a particular probability of gull consumption was associated with an even greater probability of smolt mortality. Payton et al. (2020) observed the same relationship with tern predation on UCR and SR steelhead (a = 1.41). Several factors may contribute to the finding of super-additivity in both terns and gulls. First, super-additivity may be related to the foraging behavior of terns and gulls, where unsuccessful predation attempts result 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). Second, terns and gulls disproportionally consume larger-sized steelhead smolts, smolts that are more likely to survive out-migration (Evans et al. 2014). As such, estimates of avian predation/consumption probabilities reported herein may underestimate the full impact of avian predation on steelhead smolts during out-migration, and thus produce estimates of super-additive mortality during the smolt life-stage (Payton et al. 2020). In the case of gulls, estimates of additivity were imprecise (uncertain), with estimates ranging from < 0 (super-compensation) to > 1 (super-additivity) in UCR steelhead. As such, estimates of additivity should be interpreted cautiously. The difficulties in calculating unbiased estimates of additivity and the ambiguity associated with the interpretation of this parameter are the primary motivation for focusing on the differences in survival probabilities with and without predation ($i.e.$ $\boldsymbol{\Phi}^{\Delta}$). Unlike estimates of additivity, this metric provides management-relevant information with which to quantify the degree to which predation limits fish survival (see also Payton et al. 2020).

Previous research 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 temporarily stunned or moribund in the tailraces of dams, however, were unknown. Gull predation/consumption was not limited to the tailraces of dams, with gulls breeding on Miller Rocks and Blalock Islands foraging both upstream and downstream of John Day Dam (see *Figure 8* above). Evans et al. (2016) documented that the majority (numerically) of the smolts consumed by gulls breeding at Miller Rocks in 2012 and 2014 were consumed in the open reservoirs upstream and downstream of John Day Dam. As such, although gulls breeding at colonies that are in close proximity to dams may disproportionately consume smolts at those dams, they do not exclusively forage at dams. Thus, additional research is warranted to quantify what proportion of smolts consumed at dams by gulls were moribund and, ultimately, to what degree gull management at dams is likely to increase smolt survival.

The results of our updated analysis include data from four additional years (2019-2022), are based on gulls that presumably capture and consume the fish themselves and apply to a smaller spatial-scale where other sources of mortality were less prevalent. Updated results also relied on a reconsideration of some of the underlying assumptions of Payton et al. (2020) to better reflect fundamental properties of the system under consideration. As the majority of all mortality in the river reach investigated was attributable to gulls, many weeks of data provided little information with respect to estimates of annual baseline survival and the rate of additivity, due to the properties inherent to a restricted parameter space (see also Payton et al. 2020). By noting that survival in any given year is more likely to be similar to the average survival rate in past years than it is to being near zero or 1 (or 0% or 100%), we have updated our priors to share additional information among years which reduced the influence of the assigned priors and allowed for the development of estimates that better reflect the information provided by the data (see *Methods & Analysis* section for details). As expected, the effects of the new assumption framework are most readily apparent in results from years in which gull predation is largely responsible for all estimated mortality between MCN and BON. For example, in 2018, we see that most of the data points for SR steelhead are lying along the boundary of the parameter space (see *Figure 11* above), indicating little information about the nature of additivity within the year, with model fit largely reflecting the shared information across years (2012-2022). Under the former model assumption, the baseline survival estimate for 2018 would have been very close to 0.50 (or 50%) with an increasing slope (indicating a super-compensatory relationship). The updated assumptions, however, results in a much closer reflection of the true apparent relationship between predation/consumption and survival probabilities. Our reanalysis of the additive nature of tern predation on UCR and SR steelhead demonstrated that the alteration of these priors (assumptions) had very little influence on previously published results, with the relationship between tern predation on UCR and SR steelhead and steelhead survival being statistically significant, regardless of the model assumptions. For instance, in UCR steelhead, differences in observed and baseline survival were 0.15 (0.10–0.21) with the former prior assumptions and 0.12 (0.06–0.18) with the updated assumptions. For SR steelhead, differences in observed and baseline survival were 0.07 (0.06–0.09) with the former prior assumptions and 0.07 (0.05–0.08) with the updated assumptions.

The ability to accurately assess the relationship between predation/consumption 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/consumption and survival probabilities (see also Payton et al. 2021). Accounting for annual variation in predation/consumption 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/consumption 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. 2021). 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 gull consumption probabilities were estimated to be 0.04, reductions in gull consumption would do little to increase smolt or, especially, smolt-to-adult survival.

Results of the additive effects of avian predation/consumption by other avian predator species on other ESA-listed salmonid species (i.e. Chinook and sockeye salmon) and populations are provided in Payton et al. (2021a). It is important to note that the magnitude of avian predation/consumption rates differ dramatically amongst salmonid ESUs/DPSs, as does the potential benefits of managing avian predators to increase salmonid survival in the CRB. For instance, predation effects are generally, but not always, lower on salmon compared with steelhead, so even if the additive effects of predation are high, the potential benefits of managing predators to increase salmon survival maybe substantially less than that of steelhead due to lower levels of predation in salmon. The exception appears to be predation by cormorants on salmon in the CRE, where predation rates are often as great or greater (e.g., for Lower Columbia River ESUs/DPSs; Lawes et al. 2021b) on salmon compared with steelhead. 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).

Biotic & Abiotic Factors

Covariate analyses are ongoing and herein we provide results of our initial investigation into which biotic and abiotic factors or conditions experienced by steelhead smolts during outmigration from RIS to MCN best explain variation in survival or total mortality (1 - survival), mortality due to tern predation, and mortality due to other, non-tern related mortality sources. Information regarding the number, rear-type (hatchery, wild), size (fork length), and external condition (compromised, uncompromised) of RIS tagged steelhead used in the analysis are provided in *Table 15*.

Table 15. Numbers of PIT-tagged Upper Columbia River steelhead smolts released into the tailrace of Rock Island Dam by rear-type (hatchery, wild), condition (compromised, uncompromised), and average fork-length (interquartile range) during 2008-2021.

Year	Hatchery			Wild		
	No. (%) Hatchery	No. (%) Compromised	Length (Range)	No. (%) Wild	No. (%) Compromised	Length (Range)
2008	5,373 (73.9%)	702 (13.1%)	196 (184–209)	1,898 (26.1%)	71 (3.7%)	174 (160–194)
2009	5,150 (72.4%)	372 (7.2%)	201 (188–215)	1,964 (27.6%)	62 (3.2%)	173 (161–189)
2010	5,387 (73.1%)	549 (10.2%)	204 (190–218)	1,978 (26.9%)	65 (3.3%)	174 (160–191)
2011	5,961 (76.9%)	1370 (21.3%)	208 (193–224)	1,795 (23.1%)	118 (6.6%)	182 (166–206)
2012	5,107 (76.1%)	559 (10.9%)	201 (187–217)	1,605 (23.9%)	74 (4.6%)	173 (160–188)
2013	4,284 (72.7%)	333 (7.8%)	198 (182–214)	1,609 (27.3%)	88 (5.5%)	172 (160–187)
2014	5,686 (74.2%)	215 (3.8%)	196 (179–212)	1,977 (25.8%)	61 (3.0%)	172 (160–188)
2015	5,105 (72.2%)	866 (17.0%)	196 (181–212)	1,964 (27.8%)	97 (4.9%)	178 (165–194)
2016	4,965 (73.4%)	339 (6.9%)	202 (187–218)	1,799 (26.6%)	90 (4.8%)	172 (158–188)
2017	5,776 (77.7%)	442 (7.8%)	195.5 (179–212)	1,660 (22.3%)	76 (4.4%)	179 (163–202)
2018	5,261 (72.7%)	659 (12.5%)	203 (186–220)	1,980 (27.3%)	148 (7.0%)	182 (167–200)
2019	3,201 (72.8%)	173 (5.4%)	197 (181–212)	1,196 (27.2%)	27 (2.3%)	177 (165–191)
2020	4,895 (71.6%)	247 (5.0%)	194 (179–210)	1,946 (28.3%)	53 (2.7%)	175 (162–192)
2021	6,366 (78.7%)	223 (3.5%)	191 (178–205)	1,724 (21.3%)	27 (1.6%)	170 (157–184)

To best characterize where and when environmental factors were experienced by smolts, we had to account for the uncertainty in the time elapsed between a smolts release from RIS and their exposure to each covariate and exposure to predation by each tern colony (see *Methods and Analysis* section above). Time-to-exposure to depredation was estimated to be Poison distributed with means of 1.36 days following release for colonies at Lenore Lake and Banks Lake, 2.74 days following release for the two colonies in Potholes Reservoir, and 5.55 days following release for the tern colonies on Badger and Crescent islands. Time-to-exposure was estimated to be Poison distributed with means of 2.33, 3.86, and 4.71 days for river metrics measured at Wanapum Dam, Priest Rapids Dam, and the Pasco Monitoring Station, respectively.

Recapture Probabilities: There was evidence of a statistically significant association between a fish's rear-type (hatchery, wild) and the probability of recapture at the MCN juvenile bypass facility. This association varied considerably by year, with the odds of recapture for wild fish being, on average, 1.15 times (95% credible interval = 1.06–1.25) greater than that of a hatchery fish (*Table 16*). The association with rear-type was not entirely explained by the disparity in sizes between wild and hatchery fish (see *Table 15* above), as we also found strong evidence of an association between fish length and the probability of recapture at MCN after accounting for differences in rear-type, with smaller-sized fish more likely to be recaptured on average (*Table 16*). Results were consistent with those of Hostetter et al. (2015b) which demonstrated that smaller-sized Snake River steelhead and yearling Chinook smolts were more likely to be detected at downstream bypass facilities, presumably because smaller-sized fish were less likely to escape entertainment in juvenile bypass facilities compared with their large-sized counterparts. Unlike Hostetter et al. (2015b), however, we found little evidence that fish in compromised condition were more likely to be recaptured at bypass facilities than non-compromised fish (for fish presumed to be alive at McNary Dam). By accounting for possible differences in recapture probabilities among smolt cohorts, we remove some possible sources of bias in our estimates of survival and consequently our estimates of "other mortality". For example, if smaller-sized fish are more likely to be entrained in the JBS and seen downstream, we would over-estimate their recapture probability and thus underestimate their survival probability. As such, the precise modelling of recapture is vital to measure mortality more precisely.

Table 16. Best fitting models of the association between biotic covariates and recapture probabilities, tern predation probabilities, and other sources of mortality during steelhead smolt passing from Rock Island Dam to McNary Dam. Odd ratios, 95% credible intervals, and statistical significance (p-value) are provided (see Methods and Analysis section for details). Odd ratios > 1.0 denote higher odds for wild fish, larger-sized fish, and compromised fish. Conversely, odds ratios < 1.0 denote greater odds for hatchery fish, smaller-sized fish, and non-compromised fish.

Association	Covariate	Odds Ratio	95% CRI	P-value
Recapture	Rear-type	1.15 [1]	(1.06–1.25) [1]	0.009
	Length	0.91 [1]	(0.90–0.94) [1]	<0.001
Total Mortality				
Tern Predation	Rear-type	0.79	(0.71–0.88)	<0.001
	Length	1.05	(1.03–1.07)	<0.001
	Length^2	0.98	(0.97–0.99)	<0.001
Other (or Non-Tern) Mortality	Condition	1.32	(1.08–1.59)	<0.001
	Length	0.96	(0.93–0.98)	<0.001

[1] Represents an average relationship across all years for covariate effects which varied significantly among years.

Tern Predation Probabilities: Like previous studies (Hostetter et al. 2012, Payton et al. 2016), we found evidence for statistically significant associations between the odds of tern predation and a fish's rear-type and between the odds of tern predation and fish length as a quadratic (*Figure 12*). We further found evidence of an association between tern predation and rear-type after accounting for length (p-value = 0.028). That is, although hatchery fish were more susceptible to tern predation due to their greater average size, the discrepancy in the odds of predation between hatchery and wild smolts was not fully explained by length alone. We found no evidence that these associations varied by year. This finding suggests that behavioral differences between hatchery and wild steelhead smolts may influence their relative susceptibility to tern predation during outmigration to McNary Dam. For instance, hatchery-rearing systems may select for individuals that are more surface oriented and naive to predators relative to their wild counterparts (as reviewed by Hostetter et al. 2023). After accounting for fish size and rear-type, we found no evidence of an association between smolt condition and the odds of tern predation (p-value > 0.10). Previous studies have individually assessed associations between the odds of tern predation and rear-type, length, and condition individually (Hostetter et al. 2012; Payton et al. 2017). This is the first study, however, to model these associations simultaneously. Only a small proportion of all smolts were in compromised condition, however, especially wild smolts (< 6.0% annually; *Table 15*), so estimates of condition effects were often imprecise. Nonetheless, results suggest that fish size and rear-type had a greater effect on tern predation probabilities than that of fish condition.

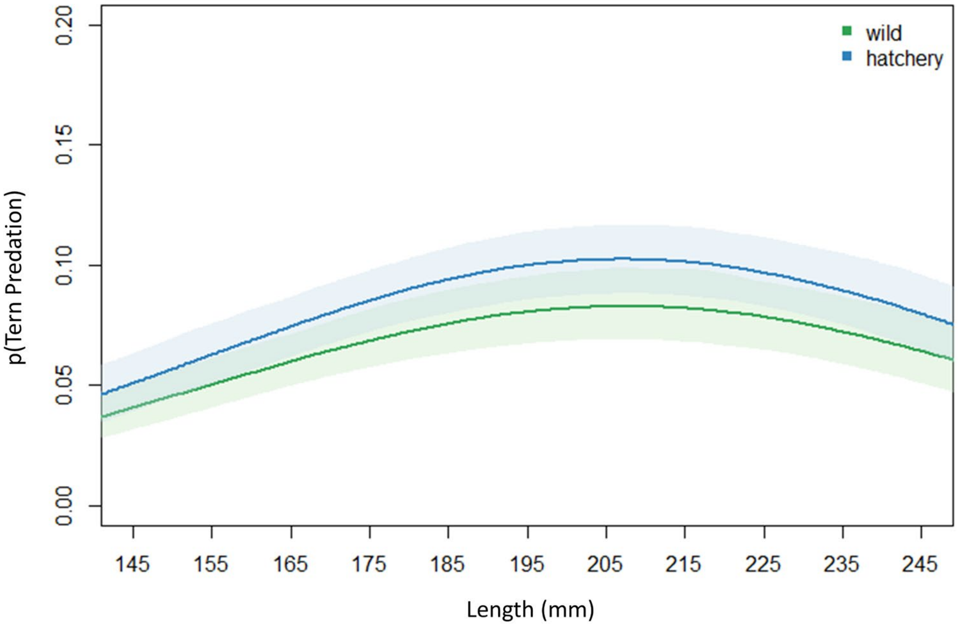


Figure 12. The best fit estimated relationships between fish length (fork length) and the average annual probability of predation by terns on Upper Columbia River steelhead smolts during passage between Rock Island Dam and McNary Dam.

Survival Probabilities: Our analysis of survival was an incremental process, using first a covariate capture-recapture-recovery model, then a covariate JMS model, and finally a covariate additive-mortality JMS model. The initial step of this analysis (the covariate capture-recapture-recovery model) investigated possible covariate associations with survival in aggregate, without attempting to partition the mortality due to tern predation versus other causes of mortality. In these initial analyses, we found significant associations between the probabilities of steelhead smolt survival and rear-type, fork-length, and condition individually. We found no evidence, however, of a quadratic relationship between length and the probabilities of smolt survival to MCN. We further found evidence that these associations remain statistically significant in a full model accounting for the other sources of mortality. We found no evidence that any of these associations varied significantly by year.

As the next step in our incremental investigation of covariate associations with survival, we partitioned total mortality into smolts consumed by terns and all mortality otherwise (a mortality-specific covariate model). The goal of this analysis was to distinguish the covariates associated with tern predation (evaluated above) and those associated with other, non-tern sources of mortality. In this analysis we found statistically significant associations between the odds of mortality due to other, non-tern mortality sources and fish length but we did not find significant associations with rear-type or condition, nor did we find the association with length to be quadratic.

The final step in our investigation of covariate associations with survival accounts for the additive effects of tern predation. As shown above, some portion of mortality not directly attributable to tern predation is nonetheless directly proportional to, and thus presumably related to, tern predation (e.g., due to latent mortality from tern injured smolts, gull kleptoparasitism, and other factors; Payton et al. 2020). In this investigation, we again found evidence of a significant association between the odds of mortality due to other causes and tern predation, suggesting a strong, super-additive effect of tern predation. Therefore, to accurately understand which factors impact mortality sources unrelated to tern predation (referred to as “baseline mortality” in the additivity-model), we must account for the additivity inherent to tern predation. After accounting for this additive effect, we still found a significant, but less pronounced, association between fish length and other (non-tern) related mortality. Additionally, after accounting for the additive effect of tern predation, the association between condition and other mortality became more pronounced (p-value = 0.007). We found week evidence of an association between rear-type and other non-tern mortality, after accounting for tern predation, length, and condition (p-value = 0.065).

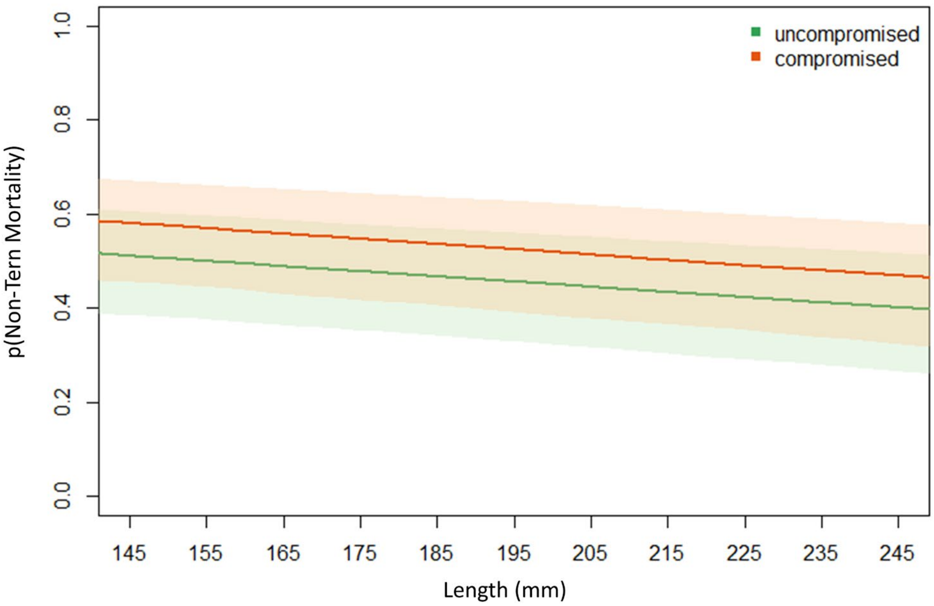


Figure 13. The best estimated relationship between fish length and condition (compromised, uncompromised), and the average annual probability of other, non-tern related mortality after accounting for the additive effects of tern predation for Upper Columbia River steelhead during smolt passage from Rock Island Dam to McNary Dam.

For our initial exploration of possible associations between abiotic covariates and smolt survival and tern predation, we investigated all applicable quantitative measures reported on DART (DART 2022). These investigations were largely uninformative. For instance, we found no evidence of an association between any of the evaluated abiotic covariates (discharge, percent spill, elevation, and temperature) and observed survival, tern predation, or baseline survival (i.e., survival in the absence of tern predation). As observed in this and other studies (Payton et al. 2016, Payton et al. 2020) there was a strong relationship between tern predation on steelhead smolts and steelhead survival to MCN (*Figure 14*; see also *Reach-specific Predation/Consumption* section above) but there was no apparent relationship between tern predation and abiotic factors (*Figure 14*) or between steelhead survival and abiotic factors (*Figure 15*). Previous research has documented a relationship between Snake River flows and tern predation (Hostetter et al. 2012) but that finding was not replicated with smolts traveling through the middle Columbia River as part of this study. Previous research has also observed a relationship between powerhouse operations (a measure turbine passage) and survival of acoustic-tagged steelhead smolts (Payton et al. 2016) but what proportion or number of PIT-tagged smolts used the powerhouse to pass Wanapum and Priest Rapids dams were unknown as part of this study.

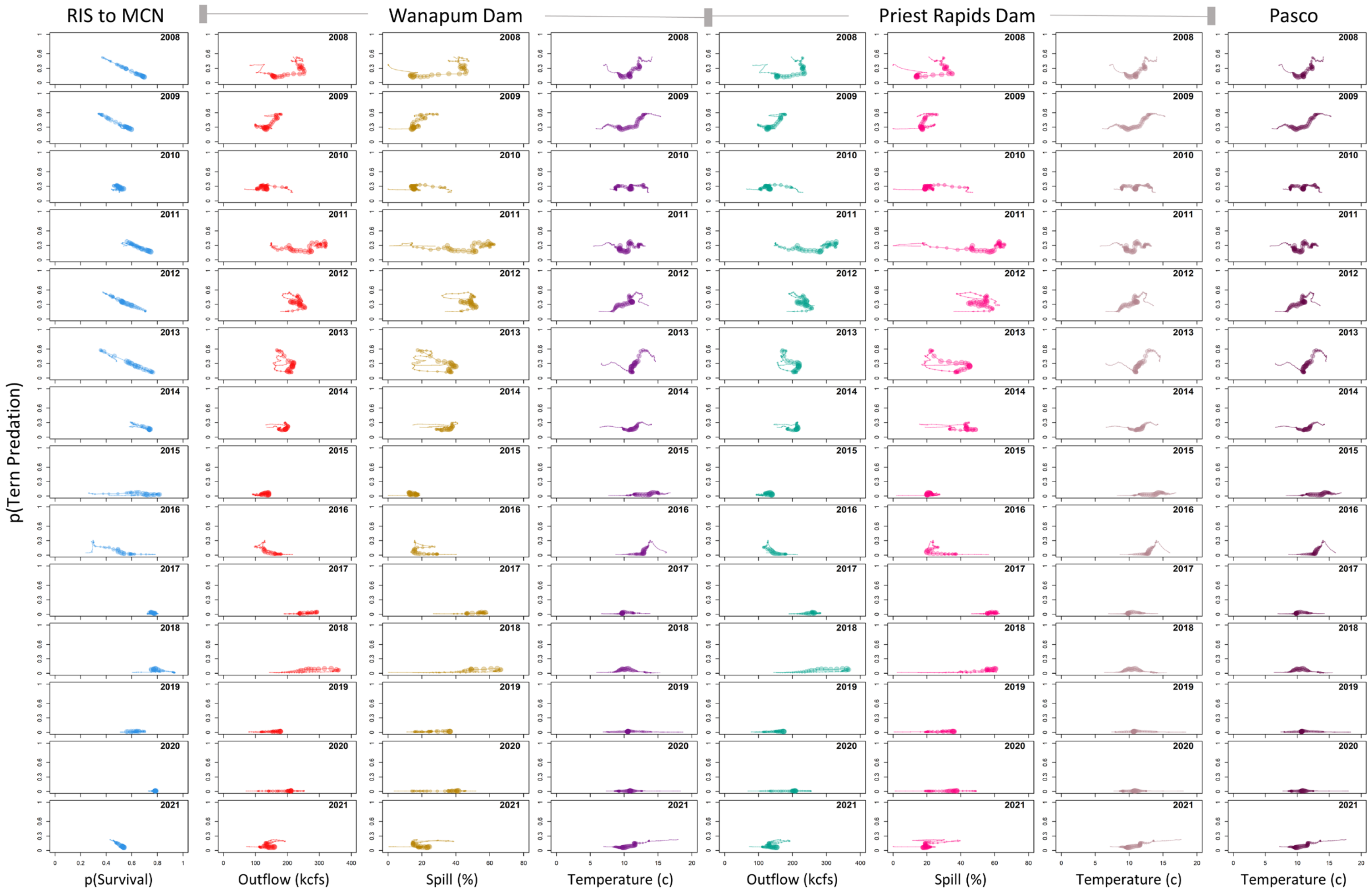


Figure 14. Daily estimates the probability of predation by terns on Upper Columbia River steelhead smolts during passage from Rock Island Dam (RIS) to McNary Dam (MCN) versus survival, along with tern predation versus abiotic covariates experienced by smolts as measured at Wanapum Dam, Priest Rapids Dam, and the Pasco Monitoring Station based the estimated day of passage through each river reach (see Methods and Analysis section). Point sizes are directly proportional to daily sample sizes.

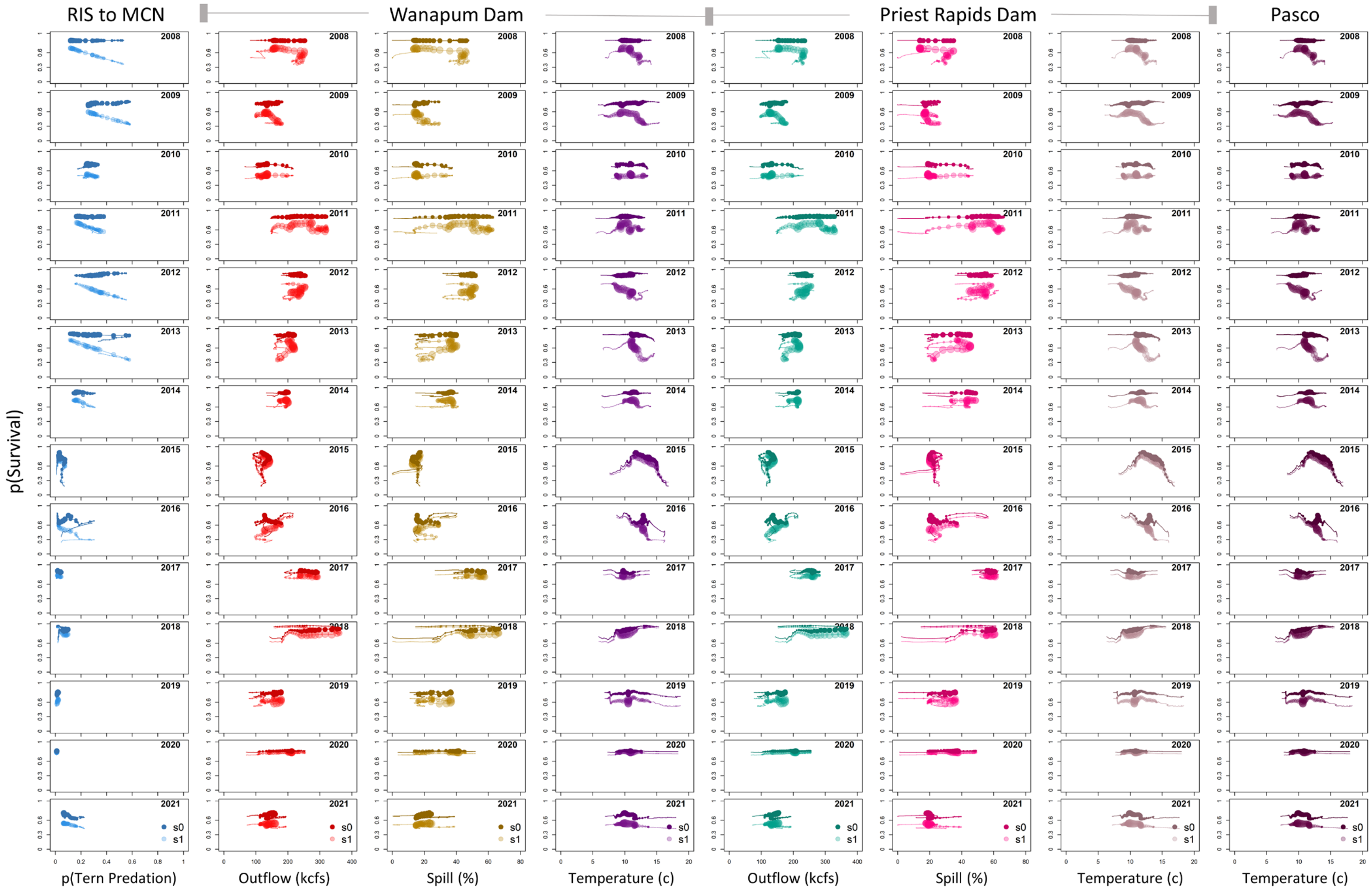


Figure 15. Daily estimates of survival (s1) and estimated baseline survival (s0; defined as survival in the absence of tern predation) of Upper Columbia River steelhead smolts versus abiotic covariates experienced by smolts during passage through the Wanapum, Priest Rapids, and Pasco Monitoring Station river reaches based the estimated day of passage through each river reach (see Methods and Analysis). Point sizes are directly proportional to daily sample size.

This modelling effort represents a novel attempt to assess covariate effects in an additive-mortality framework, wherein possible covariate associations were evaluated with respect to both the “baseline” morality sources and the additive mortality source (tern predation in this case). Results of this analysis demonstrate the benefits of such an approach. On an aggregate level, there is evidence of effects of rear-type, size, and condition on smolt survival, however, the advanced modelling approach used here allows for the identification of how the covariate impacts differ among mortality sources. Such tools may allow for a better estimation of the potential effects that may be expected with reductions or increases in mortality sources.

In summary, results of these analyses provide additional confirmation of the strong relationship between tern predation on steelhead smolts and smolt survival from RIS to MCN during 2008-2021. Additionally, results provided strong evidence that variation in tern predation was associated with fish size and rear-type, with large-sized smolts and hatchery smolts more likely to be consumed by terns. After accounting for both the direct and indirect impacts of tern predation, results provided strong evidence that fish size and condition were associated with levels of steelhead mortality due to other, non-tern causes of mortality. Results suggest that future survival studies should account for these, and possibly other, individual-level biotic factors. The abiotic factors evaluated demonstrated little association with any sources of mortality. However, the assessed abiotic factors and possible combinations thereof may simply represent proxies of the other, unidentified factors, which are influencing smolt survival but were not identified as part of this study. For instance, slower rates of travel during outmigration may be related to higher rates of different sources of mortality, but quantifying water transit time (a measure of river flow speed) is generally approximated using an arithmetic transformation of covariates evaluated in this study (reservoir elevation and dam outflow). The probability of smolts passing dams through the powerhouse versus other routes has also been approximated using an arithmetic transformation of covariates evaluated in this study (referred to as “powerhouse index”; dam outflow and spill percentage). Both these examples have been found to have significant associations with survival in previous studies (Payton et al. 2016) but could be addressed with more nuanced approaches over larger spatial scales with the tools developed here.

Ongoing Analyses: Additional analysis that build upon the methods developed herein will be included in the 2023 Annual Report. More specifically, we will more closely investigate the association between smolt travel times (using the newly developed models of smolt travel time throughout the CRB) and survival versus water transit times and survival. We will also investigate covariates that influence predation and survival probabilities of Snake River steelhead and yearling Chinook during passage from Lower Granite Dam to Bonneville Dam. Analyses will explore to what degree a fish’s passage route (spillway, juvenile bypass facility) influence recapture, predation, and survival probabilities of PIT-tagged smolts and smolt-to-adult returns.

MANAGEMENT RECOMMENDATIONS

Of the many phases of a natural resource management program, the adaptive management phase is perhaps the most important. This is when managers can apply what they have learned in implementing the management plan to improve outcomes and realize lasting benefits. Based on our results from 2022 and in previous years (Collis et al. 2021b, Roby et al. 2021b, Evans et al. 2022b), 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, while at the same time, minimizing the impacts of management on protected populations of migratory birds.

CASPIAN TERNS

The latest census of the Pacific Flyway breeding population of terns indicates that the population has declined by more than 50% since management began in 2008 (Lawes et al. 2022). This population decline is due primarily to the decline in size of the East Sand Island colony in recent years; in 2008 this colony numbered over 10,000 breeding pairs and represented two-thirds of the entire Pacific Flyway breeding population (Suryan et al. 2004), while in 2022 the colony consisted of just 1,725 breeding pairs. Exacerbating the Flyway-wide population decline is the increasing frequency of complete nesting failures at the tern colony on East Sand Island. A decline in the Pacific Flyway population of this magnitude was identified in the *Caspian Tern Management Plan for the Columbia River Estuary* as a trigger that would prompt adaptive management to reverse the decline and preclude putting the population at risk (USFWS 2005, Section 4.2.1.3). Unfortunately, this trigger point was reached in 2021, indicating the need for adaptive management to halt the decline. New emerging threats such as the HPAI virus have also begun to impact Pacific Flyway populations of terns as confirmed by two cases in McNary Reservoir in 2022. HPAI could become a new and major source of mortality for terns in the Pacific Flyway, and that added mortality could force an even steeper decline in the Flyway-wide population. In keeping with the goals and objectives of the *Estuary Tern Management Plan*, adaptive management to restore the tern colony at East Sand Island to the size range identified in the plan (3,125–4,375 breeding pairs) should be considered, as it will go a long way toward stabilizing the Pacific Flyway population. Adaptive management actions that would halt the decline of the East Sand Island tern colony include enhancing sandy substrate on the designated tern colony site, employing limited lethal control of gulls that are nesting on the tern colony site and depredating tern eggs and chicks, and deploying resident colony monitors at the tern colony site to dissuade bald eagles and detect other factors limiting colony size and nesting success. We also recommend providing alternative nesting sites for terns in Grays Harbor, the Strait of Juan de Fuca, and/or Puget Sound as a complement to the alternative sites already created/enhanced for terns at interior sites. These coastal regions have a history of supporting large tern colonies, have high connectivity with the tern colonies in the Columbia River basin, and pose little risk to salmonid stocks of conservation concern (Roby et al. 2021b).

Although there is evidence that reductions in tern predation rates on UCR steelhead have resulted in higher rates of steelhead survival, there continues to be high fidelity of terns to managed sites, and the CPR in general. This is evidenced by the re-establishment of a tern colony on Crescent Island in 2022, after management actions that had prevented tern nesting on the site since 2015 were discontinued in 2021. Adaptive management to eliminate suitable tern nesting habitat from Crescent Island must be implemented or the Crescent Island tern colony is likely to increase to its pre-management size. Also, the return each year of prospecting terns to the former colony site on Goose Island, despite ongoing efforts to prevent tern nesting, indicates strong fidelity to that nesting site. Without continued adaptive management of the Goose Island colony site, that former tern breeding colony is likely to become re-established as well. Finally, the shift in the distribution of nesting terns from managed colonies to unmanaged colony sites in the CPR (i.e. Blalock Islands, Badger Island, Shoal Island) supports the hypothesis of strong regional fidelity by nesting terns. Both waterbird ecologists and salmon managers were aware that breaking the attachment that nesting terns have to the region would be challenging, requiring a commitment to perennial adaptive management in order to assure lasting success, as suggested by the adaptive management provisions included in the *IAPMP* (USACE 2014, Section 3). Also, while Upper Columbia River salmonid populations have benefitted from tern management in the CPR, there is little evidence that Snake River salmonid populations have experienced significant benefits from the *IAPMP*. Maintaining and enhancing managed reductions in tern predation rates on ESA-listed juvenile salmonids from UCR and SR populations will require continued implementation of the *IAPMP* and adaptive management of new and expanding tern colonies within the CPR.

DOUBLE-CRESTED CORMORANTS

Cormorants have almost entirely abandoned East Sand Island as a nesting site and thousands are now nesting further upriver on the Astoria-Megler Bridge, an unintended consequence of implementation of the *Double-crested Cormorant Management Plan for the Columbia River Estuary*. The colony on the Astoria-Megler Bridge is currently the largest double-crested cormorant breeding colony anywhere in the Pacific Flyway of North America. Based on data collected during 2021 and 2022, per capita predation rates by cormorants nesting on the Astoria-Megler Bridge were 2–4 times higher (depending on salmonid species) than those of cormorants that formerly nested on East Sand Island. Our results suggest that predation rates on ESA-listed juvenile salmonids by cormorants nesting on the Astoria-Megler Bridge are as high or higher than those of cormorants nesting on East Sand Island prior to implementation of the *Management Plan*. These results indicate that management of cormorants in the CRE has not improved smolt survival, as intended, and that adaptive management to dissuade cormorants from nesting on the Astoria-Megler Bridge and to restore a cormorant colony on East Sand Island are needed to meet management objectives.

OTHER PISCIVOROUS COLONIAL WATERBIRDS

Our investigation of smolt consumption/predation by piscivorous waterbirds from other unmanaged colonies indicated that smolt consumption by gulls, primarily California gulls, was associated with substantial mortality of smolts during outmigration to Bonneville Dam. Although there is evidence that major reductions in the size of certain gull colonies (i.e. the Miller Rocks colony) could potentially increase smolt survival rates from McNary to Bonneville dam, the proportion of smolts consumed by gulls at dams that were dead or moribund is unknown and warrants additional investigation. Management actions to eliminate or reduce the size of the Miller Rocks gull colony are currently planned for 2023. 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 and islands in Lenore Lake) pose little threat to smolt survival and may be sites where gulls from managed colonies can relocate. Results of our study also indicate that predation by Brandt's cormorants nesting in the CRE and double-crested cormorants nesting on Shoal Island in Lenore Lake (an off-river location) pose little threat to the smolt survival; these colonies should not be of concern to fisheries managers. Similarly, predation rates by American white pelicans on ESA-listed salmonid smolts in the mainstem Columbia River were generally low in 2022, but additional research is warranted to better understand pelican predation on specific salmonid stocks from the Yakima River, Umatilla River, and other large tributaries of the Columbia River, as well as on certain age-classes of salmonids (e.g., juvenile sub-yearling Chinook and adult sockeye salmon).

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APPENDIX A: PREDATION RATE SUMMARY TABLES

Table A1. 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, islands in Lenore Lake, and Twinning Island in Banks 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 2007-2022 for spring (Sp), summer (Su), and Fall run fish when applicable. 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 2007-2021 are those previously reported by Evans et al. (2022a).

	Crescent Island, McNary Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	0.4% (0.3-0.6)	0.9% (0.4-1.7)	-	-	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
2022	0.1% (<0.1-0.2)	0.2% (0.1-0.4)	-	-	1.3% (0.9-2.1)	0.6% (0.2–1.0)
N	6,492	4,643			4,626	6,214

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
2022	0.2% (0.1–0.4)	0.6% (0.3–1.0)	-	-	2.8% (2.0–4.4)	0.7% (0.4–1.2)
N	6,492	4,643			4,626	6,214

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
2022	<0.1%	<0.1%	-	-	<0.1%	0.1% (<0.1-0.2)
N	6,492	4,643			4,626	6,214

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

Lenore Lake Islands, Lenore Lake						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2015	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	4,471	1,393	766	1,262	2,400	7,222
2016	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%
N	38,633	5,461	1,956	522	20,729	7,003
2017	<0.1%	<0.1%	0.3% (0.1-0.8)	-	<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
2022	<0.1%	<0.1%	-	-	<0.1%	2.1% (1.4–3.4)
N	6,492	4,643			4,626	6,214

Twinning Island, Banks Lake						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2008	<0.1%	<0.1%	-	<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 A2. 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 for spring (Sp) summer (Su) and Fall run fish when applicable. 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 2007-2021 are those previously reported by Evans et al. (2022a).

	Blalock Islands, John Day Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	<0.1%	0.1% (<0.1-0.2)	<0.1%	-	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 A3. 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 for spring (Sp) summer (Su) and Fall run fish when applicable. 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 and 2022, although smolt PIT tags were recovered following the breeding season (Table 3). Estimates are those previously reported by Evans et al. (2022a).

	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	NA	NA	NA	NA	NA	NA	NA	NA
N								
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	NA	NA	NA	NA	NA	NA	NA	NA
N								
2022	NA	NA	NA	NA	NA	NA	NA	NA
N								

Table A4. 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 2003-2018 for spring (Sp) summer (Su) and Fall run fish when applicable. 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. No estimates were available in 2020 -2022. Estimates from 2003-2018 are those previously reported by Evans et al. (2022a).

	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
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

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								
2022	NA	NA	NA	NA	NA	NA	NA	NA
N								

Table A5. 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 2007-2022 for spring (Sp) summer (Su) and Fall run fish when applicable. A dashed line indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates. Estimates from 2007-2021 are those previously reported by Evans et al. (2022a). Results from Goose Island gulls were available in select years only (2012, 2020, 2022) but gulls nested on Goose Island in all years during 2007-2022.

	Crescent Island, McNary Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	0.6% (0.3-1.1)	0.6% (0.1-2.3)	-	-	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
2022	0.1% (0.1–0.5)	0.1% (<0.1–0.7)	-	-	2.4% (1.2–4.3)	1.3% (0.6–2.5)
N	6,492	4,643			4,626	6,214

Badger Island, McNary Reservoir

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2015	0.1% (<0.1-0.5)	0.1% (0-0.9)	0.5% (<0.1-3.3)	1.1% (0.2-4.8)	2.9% (1.3-6.4)	5.2% (3.2-9.2)
N	7,706	3,449	766	1,262	3,601	7,222
2016	0.2% (0.1-0.4)	<0.1%	0.9% (0.1-3.3)	1.2% (0.1-7.7)	1.1% (0.6-1.9)	4.3% (2-13.8)
N	38,633	5,461	1,956	522	20,729	7,003
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
2022	1.3% (0.3–3.0)	0.2% (0.1–0.8)	-	-	3.4% (1.6–7.5)	6.6% (3.3–11.4)
N	6,492	4,643			4,626	6,214

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)
N	562	1,574	1,546		1,289	8,090
2022						
N	6,492	4,643			4,626	6,214
			Goose Island, Potholes Reservoir			
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2012	<0.1%	<0.1%	1.3% (0.3-4.0)	<0.1%	0.1% (<0.1-0.3)	2.8% (1.1-5.6)
N	41,258	24,772	1,812	2,884	25,841	6,845
2020	<0.1%	<0.1%	<0.1%	-	<0.1%	<0.1%
N	2,931	1,607	947		1,130	6,843
2022	<0.1%	<0.1%	-	-	<0.1%	<0.1%
N	6,492	4,643			4,626	6,214

Table A6. 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 2007-2022 for spring (Sp) summer (Su) and Fall run fish when applicable. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates. Estimates from 2007-2021 are those previously reported by Evans et al. (2022a).

	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

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
2022	0.8% (0.2–2.3)	1.9% (0.8–4.1)	1.2% (0.1-6.4)	-	5.7% (2.7–11.8)	5.4% (1.9–19.3)
N	3,278	3,645	617		1,634	554
			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
2022	0.4% (0.1–1.4)	0.3% (0.1–1.2)	1.0% (0.1–4.9)	-	5.5% (2.8–10.2)	5.2% (1.5–13.0)
N	3,278	3,645	617		1,634	554

Table A7. 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 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 2007-2022 for spring (Sp) summer (Su) and Fall run fish when applicable. 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 estimates in those years were unavailable. The Lenore Lake colony was also active in all years but was not scanned for smolt PIT tag from 2007-2018 and 2021, so estimate in those years were unavailable. Estimates from 2007-2021 are those previously reported by Evans et al. (2022a).

	Foundation Island, McNary Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2007	1.5% (1.0-2.4)	1.9% (1.0-3.6)	-	-	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
2022	3.7% (2.0–7.0)	1.1% (0.4–2.7)	-	-	3.5% (1.9–6.5)	0.5% (0.1–1.2)
N	6,492	4,643			4,626	6,214

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
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
2022	<0.1%	<0.1%	-	-	<0.1%	0.2% (<0.1-0.4)
N	6,492	4,643			4,626	6,214
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
2022	< 0.1%	<0.1%	-	-	<0.1%	0.1% (<0.1-0.2)
N	6,492	4,643			4,626	6,214

Table A8. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting at Astoria-Megler Bridge on yearling Chinook, sub-yearling Chinook, sockeye, and steelhead based on detections at Bonneville Dam during 2021-2022. Estimates were based on subsample of all cormorant nests in each year (see Methods). ESU/DPS-specific estimates were available in 2022 (see Table 8) but not 2021 due to small samples sizes of known ESA-listed smolts with experimental plots on the bridge in 2021 (see Evans et al. 2022b). Estimates from 2021 are those previously reported by Evans et al. (2022a).

	Astoria-Megler Bridge, Columbia River Estuary			
Year	Yearling Chinook	Sub-yearling Chinook	Sockeye	Steelhead
2021	4.8% (2.4-8.5)	17.6% (7.6-31.5)	13.1% (3.3-26.7)	7.1% (3.6-12.1)
N	50,411	10,380	4,848	46,961
2022	4.4% (2.6-7.0)	6.1% (2.9-10.7)	5.1% (1.2-11.3)	7.2% (4.4-11.2)
N	30,189	6,796	2,155	20,844

Table A9. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by American White Pelicans on Badger Island in McNary 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 2020-2022 for spring (Sp) summer (Su) and Fall run fish when applicable. A dashed line indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates. Estimates from 2020-2021 are those previously reported by Evans et al. (2022a).

	Badger Island, McNary Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2020	0.1% (<0.1-0.9)	0.4% (0.1-1.6)	0.2% (<0.1-1.3)	-	0.4% (0.1-4.2)	0.3% (0.1-1.0)
N	2,931	1,607	947		1,130	6,843
2021	0.2% (<0.1–1.4)	0.9% (<0.1–2.9)	< 0.1%	-	1.0% (0.1–7.7)	0.4% (0.1–2.4)
N	562	1,574	1,546		1,289	8,090
2022	0.4% (0.1–1.3)	0.3% (0.1–0.9)	-	-	0.9% (0.1–2.8)	0.7% (0.2–2.6)
N	6,492	4,643			4,626	6,214

Table A10. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by American White Pelicans on Miller Sands Spit in the Columbia River Estuary on ESA-listed salmonid populations originating from the Snake River (SR), Upper Columbia River (UCR), and Middle Columbia River (MCR) based on detections at Bonneville Dam during 2021-2022 for spring (Sp) summer (Su) and Fall run fish when applicable. A dashed line indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates. Estimates from 2021 are those previously reported by Evans et al. (2022a).

	Miller Sands Spit, Columbia River Estuary						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead	MCR Steelhead
2020	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
N	20,246	3,389	4,895	2,122	11,868	5,894	3,157
2021	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
N	14,345	2,069	3,390	1,677	7,935	3,294	1,791

APPENDIX B: PREDATION AND SURVIVAL OF UPPER COLUMBIA RIVER HATCHERY AND WILD STEELHEAD SMOLTS

One of the primary purposes of randomly selecting (regardless of size, condition, or rear-type) Upper Columbia River (UCR) steelhead smolts for tagging at Rock Island Dam (RIS) was to ensure that the sample of fish used to estimate predation and survival probabilities were representative of all steelhead smolts in the population at-large (tagged and untagged). Upper Columbia River steelhead smolts were also tagged in proportion to the run passing RIS, with more fish tagged when more fish were available in-river; an approach that eliminated the need to weight estimates of predation and survival based on the number and run-timing of steelhead smolts passing RIS each year (see *PIT-tagging of Upper Columbia River Steelhead* section for details). Data regarding the rear-type (hatchery, wild), external condition, and size (fork length) of steelhead smolts – coupled with survival and predation probabilities – provided a unique opportunity to investigate the relationship between individual fish characteristics and susceptibility to bird predation. Presented herein is an analysis of the odds of survival and predation of UCR steelhead smolts based on a fish's rear-type using all available data since tagging commenced at RIS in 2008.

Methods: Passive integrated transponder (PIT) tagged steelhead smolts tagged or recaptured (previously tagged) at RIS were classified as being either hatchery or wild. Hatchery fish were classified by the absence of an adipose fin or by characteristics associated with hatchery-rearing practices, including the removal or erosion of pectoral, pelvic, or dorsal fins or the presence of a tag [PIT {2008-2022} or coded wire tags {2013-2016}] that indicated the fish was of hatchery origin. Wild fish were classified by the presence of a fully intact fins, with no tags indicating hatchery origin. It is possibly, however, that some hatchery fish also met the criteria of being a wild fish, particularly in years when smolts were not scanned for coded wire tags, so we use the term "presumed wild" to denote this possibility.

Weekly and annual survival and predation probabilities (proportion of available fish) were generated using the joint mortality and survival (JMS) model of Payton et al. (2019) for the two cohorts (hatchery, presumed wild) of steelhead tagged/recaptured and released at RIS. Full details of the JMS model are provided in Payton et al. (2019; see also *Methods & Analysis* and *Cumulative Predation & Survival* sections above). In brief, for use in this analysis, we defined cohort specific parameters for survival, predation, and recapture, with,

> $\mathbf{\Theta}^c$**,** defined to be a DxJ matrix where $\Theta_{d,j}^c$ represented the probability (from release) that a fish from release cohort r succumbed to depredation by colony d for $d \in \{1,2,\ldots,D-1\}$ or some other cause of mortality for $d = D$, is the reach of river immediately preceding recapture opportunity j

and

$\boldsymbol{p^c}$, defined as a J-length vector where p_j^c represented the probability that a fish from release cohort r alive at recapture opportunity *j* was successfully recaptured (reseen)

and we assumed a common probability of tag recovery among cohorts with,

$\boldsymbol{\gamma}$, defined as a D-length vector where γ_d represented the probability of recovering a fish from any cohort which died due to depredation by colony *d* for $d \in \{1,2,\ldots,D-1\}$, and γ_{15} = 0 represented the lack of recovery opportunity for fish which died from all other unspecified causes.

Letting *m* represent the final recapture opportunity at which the fish was seen (with $m = 0$ representing a fish never reseen following release) the combined likelihood was then,

$$L = \prod_{r}\left(\prod_{j\leq m}\left({p_j^{c}}^{y_j} * \left(1-p_j^c\right)^{(1-y_j)}\right) * \prod_{d} {\chi_{m+1,d}^c}^{r_d}\right);$$

where,

$$\chi_{j,d}^c = \begin{cases} \theta_{j,d}^c * \gamma_d + \left(1-p_{j+1}^c\right) * \chi_{j+1,d}^c; & for\ d \in 1,\ldots,D-1 \\ \displaystyle\sum_{i=1}^{D} \theta_{j,i}^c * (1-\gamma_i) + \left(1-p_{j+1}^c\right) * \chi_{j+1,i}^c\,; & for\ d = D \end{cases}$$

To evaluate if survival and predation probabilities of tagged steelhead differed by rear-type, we compared the odds of predation among the cohorts across weeks and years and noted any differences that were statistically significant, defined as instances in which the log of the odds-ratio between the two cohorts had negligible overlap with zero (i.e. >95% of the posterior distribution of the difference lay above or below zero).

Comparisons of survival and predation probabilities by rear-type were investigated based on the avian predator species (Caspian terns [CATE}] California and ring-billed gulls [LAXX], and double-crested cormorants [DCCO], or all birds combined) and the river reach (RIS to McNary Dam [MCN]), MCN to Bonneville Dam [BON]) and BON to the Pacific Ocean) where predation occurred (see also *Methods & Analysis* and *Cumulative Predation and Survival* sections above).

Results & Discussion: Most UCR steelhead smolts captured, tagged/recaptured, and released at RIS were hatchery-reared, comprising 71.6–78.7% of all tagged fish sampled during 2008-2022 (*Table B.1*). Ratios of hatchery to presumed wild fish were also relatively consistent across the study period (*Table B.1*).

Table B.1. Rear-type (hatchery, presumed wild) of PIT-tagged Upper Columbia River steelhead smolts at Rock Island Dam during 2008-2022.

Year	Rear-type No. (%) Hatchery	 No. (%) Presumed Wild
2008	5,373 (73.9%)	1,898 (26.1%)
2009	5,150 (72.4%)	1,964 (27.6%)
2010	5,387 (73.1%)	1,978 (26.9%)
2011	5,961 (76.9%)	1,795 (23.1%)
2012	5,107 (76.1%)	1,605 (23.9%)
2013	4,284 (72.7%)	1,609 (27.3%)
2014	5,686 (74.2%)	1,977 (25.8%)
2015	5,105 (72.2%)	1,964 (27.8%)
2016	4,965 (73.4%)	1,799 (26.6%)
2017	5,776 (77.7%)	1,660 (22.3%)
2018	5,261 (72.7%)	1,980 (27.3%)
2019	3,201 (72.8%)	1,196 (27.2%)
2020	4,895 (71.6%)	1,946 (28.3%)
2021	6,366 (78.7%)	1,724 (21.3%)
2022	4,579 (73.7%)	1,638 (26.3%)

Predation/consumption probabilities by rear-type varied by avian predator species, river reach, and year (2008-2022; *Figure B.1-B.3*). Results indicated that hatchery fish were more likely to be consumed than their presumed wild counterparts (here after simply “wild”) by CATE and LAXX breeding at colonies that foraged on smolts between RIS and MCN (*Figure B1*). Differences were statistically significant when data from all weeks and years were considered. Of the two predator species evaluated, relative differences were often greater or more pronounces in LAXX compared with CATE. Taken together, predation/consumption probabilities by both CATE and LAXX (i.e. all birds combined) on hatchery steelhead averaged 19.7% (18.0–21.4%) compared with 12.1% (10.8–13.5%) on presumed wild steelhead. There was some evidence that wild fish were more likely, on average, to survival outmigration from RIS to MCN compared to hatchery fish (*Figure B.1*). Difference in survival between hatchery and wild steelhead were commensurate with the relative difference in predation/consumption probabilities on hatchery and wild steelhead in most, but not all, years.

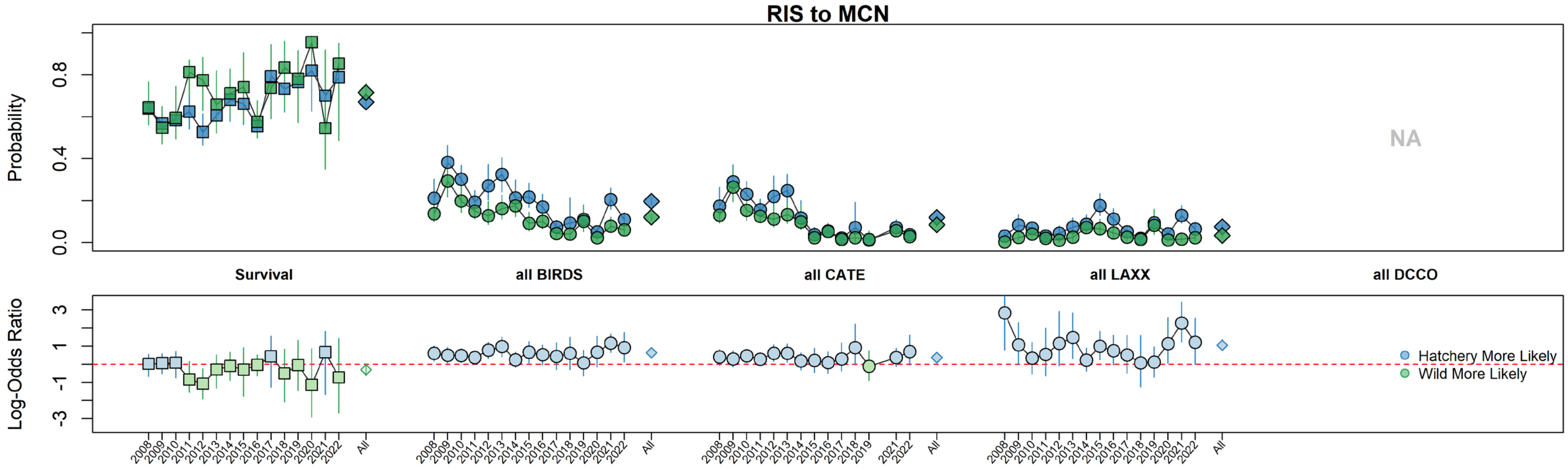

Figure B.1. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from Rock Island Dam (RIS) to McNary Dam (MCN) during 2008-2022 (upper panel). Statistical comparisons (lower panel) represent the log odds ratio of survival or predation, with values < 0 indicating greater odds for wild fish and values > 0 indicating greater odds for hatchery-reared fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not statistically significant. NA denotes that predation/consumption or survival probabilities were not available for relative comparisons. Wild fish were presumed to be wild based on fully intact fins and without a tag indicating the fish was of hatchery origin (see Methods).

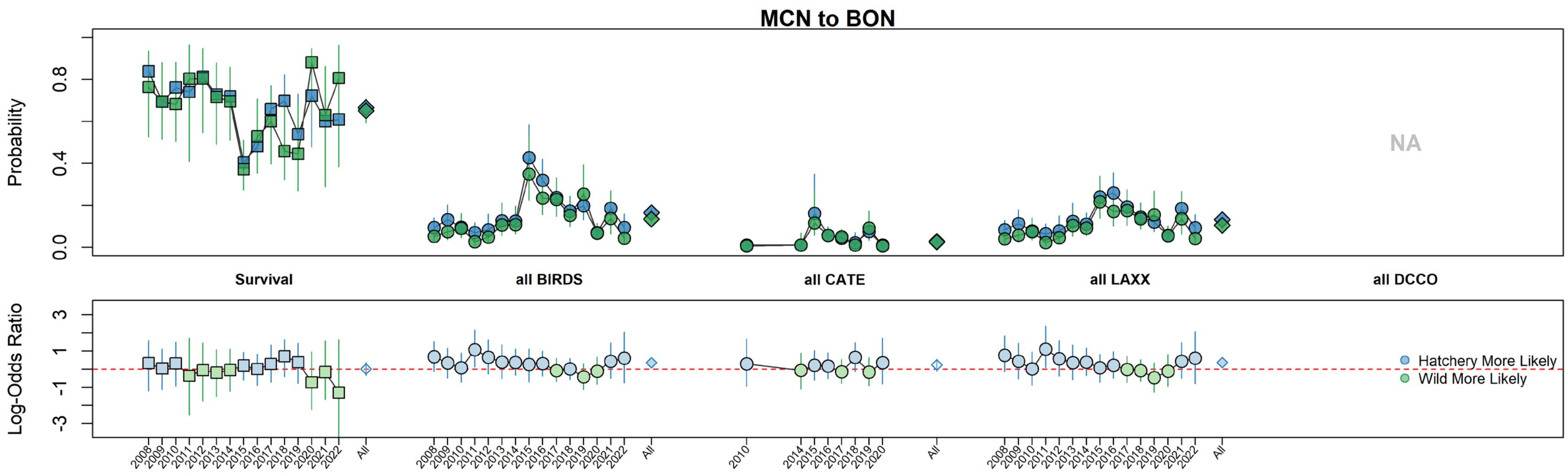

Figure B2. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from McNary Dam (MCN) to Bonneville Dam (BON) during 2008-2022 (upper panel). Statistical comparisons (lower panel) represent the log odds ratio of survival or predation/consumption, with values < 0 indicating greater odds for wild fish and values > 0 indicating greater odds for hatchery-reared fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not statistically significant. NA denotes that predation/consumption or survival probabilities were not available for relative comparisons. Wild fish were presumed to be wild based on fully intact fins and without a tag indicating the fish was of hatchery origin (see Methods).

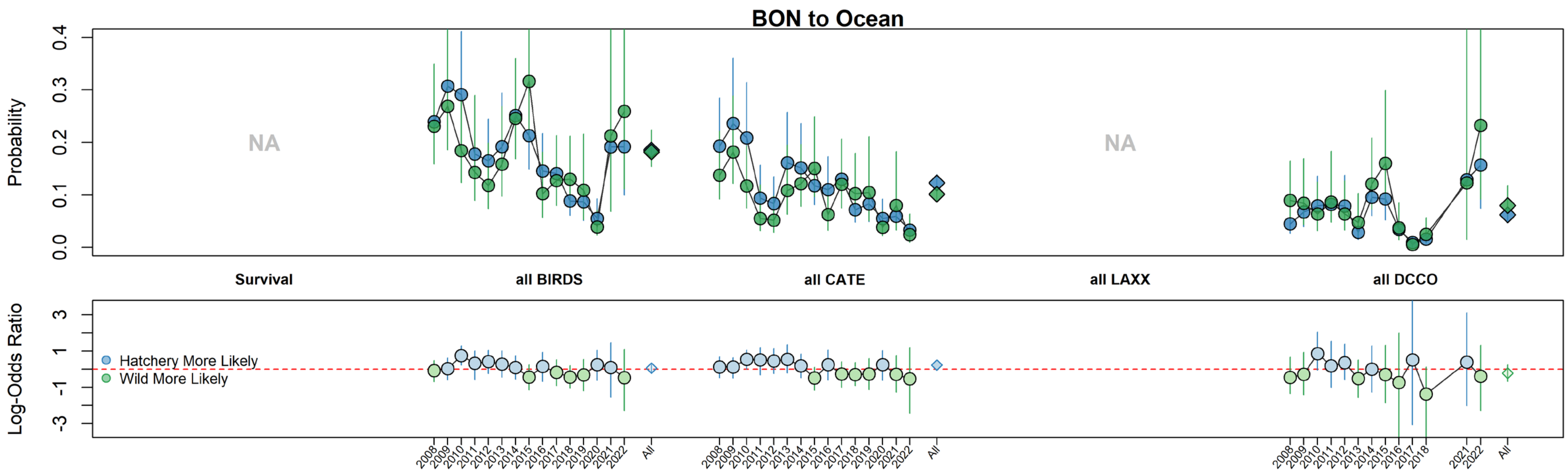

Figure B3. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from Bonneville Dam (BON) to the Pacific Ocean during 2008-2022 (upper panel). Statistical comparisons (lower panel) represent the log odds ratio of survival or predation/consumption, with values < 0 indicating greater odds for wild fish and values > 0 indicating greater odds for hatchery-reared fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not statistically significant. NA denotes that predation/consumption or survival probabilities were not available for relative comparisons. Wild fish were presumed to be wild based on fully intact fins and without a tag indicating the fish was of hatchery origin (see Methods).

Although there were active colonies of DCCO and American white pelican (AWPE) colonies that foraged on UCR steelhead between RIS and MCN (e.g., Hanford Island, Foundation Island, and Badger Island), predation probabilities were low (< 0.01), so adequate data to investigate relative differences in predation by rear-type due to DCCO and AWPE predation were not available, nor do birds from these colonies pose a substantial threat to UCR steelhead smolt survival upstream of Bonneville Dam.

There was some evidence that hatchery-reared steelhead smolts were more likely be predated/consumed than wild steelhead during smolt outmigration from MCN to BON (see *Figure B2* above). Relative differences, however, were less consistent than those observed upstream of MCN, with the occasional finding that wild fish were more likely to be predated/consumed than their hatchery counterparts, depending on the predator species and year. The magnitude of difference in predation/consumption probabilities based on a fish's rear-type were also consistently smaller (less pronounced) than that observed during smolt passage between RIS and MCN, with predation probabilities from all birds during smolt outmigration from MCN to BON averaging 16.5% (14.8–18.5%) and 13.5% (11.6–15.8%) for hatchery and wild smolts, respectively. There was no evidence that the odds of survival from MCN to BON were greater for wild smolts compared with hatchery-reared smolts when data from all weeks and years were considered.

There was no evidence that CATE and DCCO breeding at colonies downstream of BON disproportionately consumed hatchery and wild steelhead smolts, with both rear-types equally susceptible to bird predation in the Columbia River estuary (see *Figure B3* above). Taken together (all birds) the odds of predation by rear-type were nearly indistinguishable between hatchery and wild steelhead downstream of BON. Average annual predation/consumption probabilities by all birds were 18.6% (16.7–21.0%) and 18.2% (15.4–22.4%) for hatchery and wild smolts, respectively. Annual trends in CATE and DCCO predation also followed very similar patterns across the 15-year study period (see *Figure B3* above). Due to a lack of PIT tag detection sites downstream of East Sand Island in the lower Columbia River estuary, estimates of smolt survival to the Pacific Ocean were not available (see also *Methods & Analysis* and *Cumulative Predation and Survival* sections for details).

Our results are consistent with those of several other studies and indicate that steelhead susceptibility to colonial waterbird predation/consumption was associated a fish's rear-type for some, but not all, predator species and colonies in the Columbia River basin (Hostetter et al. 2012, Payton et al. 2016, Hostetter et al. 2023). When differences were observed, they often indicated higher levels of predation on hatchery-reared steelhead. Hostetter et al. (2012) also observed that hatchery-reared steelhead from the Snake River were more susceptible to CATE predation than wild steelhead, differences that were attributed to the larger average size of hatchery steelhead and to possible behavioral differences. For instance, hatchery-rearing systems may select for individuals that are more surface oriented, less able to endure sustained swimming, and naive to predators relative to their wild counterparts (as reviewed by Hostetter et al. 2023). In the present study, UCR hatchery-reared steelhead were, on average, larger than wild steelhead (hatchery-reared = 200 mm fork length, wild = 179 mm fork length).

There was some evidence that differences in the relative susceptibility of UCR hatchery and wild steelhead to CATE and LAXX predation/consumption decreased (lessened or diminished) during smolt passage from RIS to Pacific Ocean. For instance, relative differences in steelhead predation by rear-type were less pronounced and often not statistically different for CATE foraging on UCR steelhead downstream of MCN, even though difference were apparent upstream of MCN (i.e. following tagging and release at RIS). This may be due, in part, to changes in the size distribution of available hatchery and wild smolts to predators downstream, with larger-sized smolts disproportionally removed by plunge-diving predators upstream, functionally changing the length distribution of surviving steelhead to below BON. Additional research is warranted to better understand to what degree the disproportionate predation/consumption of larger-sized steelhead (both hatchery and wild) by some CATE and LAXX colonies upstream influenced the size distribution and subsequent survival of UCR steelhead during outmigration.

There was no evidence that DCCO disproportionately consumed UCR hatchery-reared steelhead compared with their wild counterparts. Similarly, Hostetter et al. (2012) found no evidence of a difference in the relative susceptibility of Snake River hatchery and wild steelhead to DCCO predation during smolt passage from Lower Monumental Dam to MCN. Unlike CATE and LAXX, which are surface feeders, DCCO are pursuit-diving predators. Also, unlike CATE and LAXX, there is no evidence that DCCO disproportionately consumed larger-size smolts (Hostetter et al. 2012, Roby et al. 2016), with smolts of all lengths equally susceptible to DCCO predation at both estuary and inland colony locations. This result emphasizes that predator-specific interactions can be dynamic and complex and should be considered when evaluating the over-all effects of predation/consumption on prey populations (Hostetter et al. 2023). A more detailed analysis of factors that potentially influence UCR steelhead smolt survival, including fish length and condition, are provided in the *Biotic and Abiotic Factors* section of this report.

APPENDIX C: RELATIVE COMPARISONS OF PREDATION AND SURVIVAL OF STEELHEAD SMOLTS PASSIVELY DETECTED AT ROCKY REACH DAM VERSUS THOSE TAGGED AT ROCK ISLAND DAM

As part of the Avian Predation Project, passive integrated transponder (PIT) tagging of juvenile steelhead commenced at the Rock Island Dam (RIS) fish trap in 2008 with the goal of estimating avian predation rates on smolts during outmigration through the Priest Rapids Project. Of specific concern was predation of steelhead by a colony of Caspian terns (hereafter "tern" or "terns") on Goose Island in Potholes Reservoir. The smolt trap at RIS was chosen as the most appropriate capture, tagging, and release site because the tailrace of RIS represents the start of the Priest Rapids Project and it represents the upper-most extent of the foraging range of terns on Goose Island (Roby et al. 2015). Smolts passing RIS can also be randomly selected for tagging (tagged regardless of size, condition, or rear-type) and tagged in-concert with, and in proportion to, the run at-large (tagged and untagged). This sampling approach eliminates the need to weight estimates of predation and survival based on the number and run-timing of steelhead smolts passing RIS each year (see also *PIT-tagging of Upper Columbia River Steelhead* section above for additional details). Rock Island Dam is also located downstream of all major steelhead spawning tributaries on the middle Columbia River (Okanogan, Entiat, Methow, and Wenatchee rivers), making it a location where all elements or characteristics of the Upper Columbia River (UCR) steelhead population can be sampled.

The process of capturing, handling, and tagging steelhead smolts at RIS, however, could negatively influence fish survival and fish susceptibility to avian predation due to stress associated with handling/tagging effects at RIS or other factors relating to the fitness of RIS sampled smolts. Conversely, PIT-tagged smolts that were passively detected (i.e., previously tagged and not handled) at the Rocky Reach Dam (RRJ) juvenile bypass facility, located 33 Rkm upstream of RIS, may be more likely to survive outmigration and less likely to be consumed by avian predators. These concerns were raised by the Fish Passage Center (FPC), whereby the FPC concluded there were "substantial and significant differences" in the survival and avian recovery rates of steelhead tagged at RIS versus those passively detected at RRJ (FPC 2021). The FPC investigated annual differences in smolt survival from RRJ and from RIS to Bonneville Dam and used avian recovery rates (based on the number of PIT tags recovered on the Goose Island tern colony) to infer annual differences in tern predation rates or probabilities. However, there are other confounding factors which may explain differences in annual rates of smolt survival and depredation which warrant further exploration.

Observed differences in both annual survival rates and tag recovery rates can be biased by run-timing differences between RRJ and RIS smolts, specifically, smolts seen at RRJ disproportionately migrate earlier in the year (see *Results* below). Past research suggests that Upper Columbia River Steelhead smolts migrating earlier in a season have relatively higher rates of survival (Evans et al. 2014). Furthermore, PIT tags deposited on tern colonies earlier in the year are much less likely to be recovered than those deposited by terns later in the year (closer to the date of recovery; see also Hostetter et al. 2015). Such bias can be readily addressed using corrections for PIT-tag deposition and detection probabilities on bird colonies (as described in the *Methods & Analysis* section above and in Hostetter et al. 2015). Furthermore, observed differences in both annual survival rates and annual tag recovery rates will be biased to some degree by the differences in spatial-scales or sampling locations – the rates being defined as "from RRJ" versus "from RIS"– and temporal differences in passage timing. As such, an accurate comparison between cohorts (RRJ, RIS) should compare smolts migrating through the same spatial reach, within the same time period. Toward this end, we assessed observed travel times of RRJ and RIS smolts to create a travel time model which enabled us to model survival and tern predation as a function of a smolt's probable date of passage at RIS.

Additionally, in considering whether differences in survival and tern predation rates may be biased in some degree by differences in the proportion of hatchery and wild fish from each cohort, we evaluated and accounted for any significant associations between a smolt's rear-type and any of the evaluated probabilities involved in the model (recapture, recovery, survival, and predation). Lastly, we also accounted for any significant associations between recapture probabilities at RIS (for RRJ smolts) or at Bonneville Dam (for both RRJ and RIS smolts) to account for biases. With these modelling considerations, we were able to assess potential differences between survival and tern predation probabilities of smolts passive detected (unhandled) at RRJ versus those tagged (handled) at RIS.

Methods: Weekly survival and tern predation probabilities from PIT-tagged steelhead smolts detected at RRJ and those tagged at RIS were generated using the Joint Mortality and Survival (JMS) model of Payton et al. (2019; see also *Methods & Analysis, Cumulative Predation & Survival* sections above). Estimates were generated in all years since the bypass facility at RRJ has been operational in 2010. All tern colonies capable of foraging on smolts during outmigration from RIS to McNary Dam (MCN) were included in the analysis (see *Map 1* above). To make ceteris paribus comparisons of survival and tern predation probabilities between cohorts (RRJ, RIS), we needed to account for differences in the run-timing, release timing (travel times between RRJ and RIS), and rear-type (hatchery, wild) of smolts from each cohort. Some unknown proportion of RRJ detected smolts also presumably die prior to reaching RIS, mortality that could influence relative comparisons of survival and tern predation between cohorts. To address these concerns, we developed a travel time model and covariate survival and predation model to account for inherent differences in the run-timing, rear-type, and survival of smolts from each cohort.

Travel Time Model – Independent of survival, we assessed travel times of steelhead smolts during outmigration from RRJ and RIS to BON. For each PIT-tagged steelhead smolt first seen (detected passing) at either RRJ or RIS we recorded at which recapture opportunity they were sighted (RRJ, RIS, MCN, JDA, and/or BON) and the day they were sighted. This modelling approach allowed us to build estimates of inter-recapture-sight travel times with all data, including recapture histories with missing observations (e.g., seen at RRJ on day 0, missed at RIS, seen at MCN on day 5, missed at JDA, seen at BON on day 7).

For each year, we defined a stochastic, state-transition matrix unique to each river reach j, $\mathbf{M}(j)$, where each smolt's transit time (in days) across the reach was assumed to be negative-binomial distributed with mean equal to a log-linear function of the day the fish entered the reach (up to a possible cubic term). Letting d_j represent the day a smolt enters reach j, the $(d_j, d_{j+1})^{\text{th}}$ entry of $\mathbf{M}(j)$ represents the probability of any smolt entering reach j on day d_j subsequently left reach j (i.e. entered reach $j+1$) on day, d_{j+1}, where

$$[M(j)]_{d_j, d_{j+1}} \sim \text{negative\_binomial}(d_{j+1} - d_j \mid \mu_j(d_j;\, \boldsymbol{\beta}_j), \sigma)$$

and

$$\mu_j(d_j;\, \boldsymbol{\beta}_j) = \beta_{j_0} + \beta_{j_1} * d_j + \beta_{j_2} * d_j^2 + \beta_{j_3} * d_j^3$$

where each $\beta_{j_k} \neq 0$ was included given sufficient statistical significance.

We assumed σ to be the same for all reaches, as the identifiability of reach specific σs was lacking in many years. We further assume the travel times among reaches to be mutually independent which implies that the probability a fish entering reach j on day d_j and then exiting reach $(j+n)$ on d_{j+n+1} can be expressed as a product of the appropriate $\mathbf{M}$ matrices,

$$\left[\prod_{k=0}^{n} \mathbf{M}(j+k)\right]_{d_j, d_{j+n+1}}.$$

For example, the collective likelihood for a fish seen at RRJ on day 0, missed at RIS, seen at MCN on day 5, missed at JDA, seen at BON on day 7 can be expressed by the product

$$[M(0) * M(1)]_{0,5} * [M(2) * M(3)]_{5,7}.$$

These matrices allowed us to create smolt specific vectors expressing the estimated probable time of passage at RIS, $\boldsymbol{\rho}$; such that the t^th^ entry of $\boldsymbol{\rho}$ represented the probability that the associated smolt passed RIS at time t. While the travel time model was based on days of recapture, in the covariate survival and predation modelling effort below, probabilities of survival, predation, recapture, and recovery are all calculated on a weekly basis (primarily to satisfy assumptions of tag recovery on tern colonies). To this end, the $\boldsymbol{\rho}$ vectors constructed with this travel-time analysis were calculated by summing the daily probabilities across weeks. So, for each smolt, i, in this comparative study we can aggregate their associated daily probabilities of passing RIS to construct a vector, $\boldsymbol{\rho}_i$, representing the probabilities of passing RIS in each week.

We let w_j represent the week a smolt enters reach j and $\{w_j\}$ represent the set of all days encompassed by w_j.

For smolts whom d_1 is known – as is the case for all smolts first seen at RIS and a small portion of those first seen at RRJ – $\boldsymbol{\rho}_i$ is a single-entry vector such that,

$$\rho_{i,w} = 1,$$

where $d_1 \in \{w\}$ and all other values 0.

For smolts seen at RRJ (on day d_0) but never recaptured, we have no further information with which to inform $\boldsymbol{\rho}_i$ and therefore,

$$\rho_{i,w} = \sum_{h\in\{w\}} [M(0)]_{d_0,h}.$$

However, for smolts with at least one downstream detection, we have added information as to the number of days it may have taken them to travel between RRJ and RIS; most explicitly an upper limit. Employing Bayes rule, we can make more informed estimates as to the probable day these smolts passed RIS. Using their first recapture (following passage at RRJ on d_0) at opportunity k on d_k we define $\boldsymbol{\rho}_i$ such that,

$$\rho_{i,w} = \sum\nolimits_{h\in\{w\}} \frac{[M(0)]_{d_0,h} * \left[\prod_{\mathrm{j=1}}^{\mathrm{k-1}} \mathbf{M}(j)\right]_{h,d_k}}{\left[\prod_{\mathrm{j=0}}^{\mathrm{k-1}} \mathbf{M}(j)\right]_{d_0,d_k}}.$$

Estimates of each $\boldsymbol{\beta}_j$ (and consequently each $\mathbf{M}$) were derived from all the inter-recapture travel times recorded each year for all steelhead smolts detected at RRJ or tagged and released from RIS. No consistent differences were observed in any year in the travel times between the first seen at RRJ and first seen at RIS cohorts. For each $\boldsymbol{\beta}_j$, coefficients lacking statistical evidence – $\mathrm{prob}(\beta_{j.} > 0) \in [0.05, 0.95]$ – were dropped. We assign priors of $\boldsymbol{\beta}_j \sim \mathrm{mvnorm}(0, \mathbf{I})$ for all j and $\sigma \sim \mathrm{half-normal}(0, 10)$. We defined estimates of each $\widehat{\boldsymbol{\beta}}_j$ as the median of the associated marginal posterior distribution.

Covariate Survival and Predation Model – We refer to probabilities of survival, predation, recapture, and recovery collectively as system parameters and parameters associated with the variation in these parameters as covariate parameters. All parameters were modeled independently for the cohorts of smolts first seen at RRJ and those first seen at RIS apart from recovery probabilities (i.e. the probabilities of tag deposition and detection on tern colonies). By allowing these latter probabilities to be equal, we can eliminate this uncertainty and directly compare predation probabilities (rather than recapture probabilities).

While recoveries are an important source of “downstream detections” useful for inference of survival probabilities, for the purposes of this analysis, we were only directly interested in the predation probabilities associated with tern colonies. Additional modelling would be necessary to account for the time until exposure to predation from other avian species and colonies (e.g., gulls on Miller Rocks in The Dalles Reservoir). We therefore aggregated the recoveries of tags from all these other avian colonies. We additionally aggregated all recoveries and recaptures downstream of BON – those from the trawl and adult returns at BON – as recoveries downstream of BON since the added utility of treating them as additional recapture opportunities was arguably negligible.

For clarity, we explain the model’s parameterization assuming a single example smolt, i, from a given release cohort (first seen at RRJ or RIS) from a given year. We further assume $\boldsymbol{x}_{cov,i}$ to be a vector of covariates values specific to this smolt with subsets of this vector constructed using the applicable covariate values to each system parameter. We limit our evaluation of covariates to those of specific applicability to the research question at hand. That is, the proportion of hatchery smolts is greater for smolts first seen at RRJ versus those first seen at RIS (see also *Results* section below) and previous research has indicated that probabilities of predation by terns can differ by rear-type, with hatchery fish often, but not always, more susceptible to tern predation (Hostetter et al. 2012; see also *Appendix B* above). Furthermore, previous research suggests that smolts initially detected at a juvenile bypass facility are more likely to be seen at other downstream bypass facilities and are less likely to survive outmigration (Hostetter et al. 2015b). Therefore, we test the statistical evidence for the association of rear-type and previous bypass recapture with the applicable parameters, first by year, then across years.

For the initial segment ($j = 0$), the parameterization is for a smolt, i, first seen at RRJ release cohort which was known to pass RRJ in a given week w_0.

We model the probabilities of smolts survival to RIS, $\mathbf{S}_0$, as

$$s_{0,i} = \mathrm{logit}^{-1}\left(\mu_{s_0} + \boldsymbol{x}_{\mathrm{s}_0,i}^{T}\boldsymbol{\beta}_{s0} + \sum_{t<w_{0,i}} \epsilon_{s_0,t}\right)$$

where $\boldsymbol{x}_{\mathrm{s}_0}$consist of the rows of $\boldsymbol{x}_{cov}$ associated with the covariates applicable to $\mathbf{S}_0$, and $\boldsymbol{\epsilon}_{s_0} \sim \mathrm{normal}(0, \sigma_{s0})$. Similarly, we model the probabilities of smolts which survived to RIS being subsequently recaptured at RIS, $\boldsymbol{p}_0$, as

$$\mathrm{p}_{0,i} = \mathrm{logit}^{-1}\left(\mu_{p_0} + \boldsymbol{x}_{\mathrm{p}_0,i}^{T}\boldsymbol{\beta}_{p_0} + \sum_{t<w_{0,i}} \epsilon_{p_0,t}\right)$$

where $\boldsymbol{\epsilon}_{p_0} \sim \mathrm{normal}\left(0, \sigma_{\mathrm{p}_0}\right)$.

The system parameters experienced by fish downstream of RIS are defined by a fish's week of arrival at RIS. For fish seen at RRJ but not at RIS, this requires the incorporation of our travel time estimates (explained *above*) applying our probabilistic estimation of when each smolt passed RIS. To this end, in the remaining model parameterization, we construct vectors of all weekly estimates associated with the covariate values of smolt i and then take a geometric average of these probabilities using the $\boldsymbol{\rho}_i$ vector constructed with the travel time model above.

Note that we model the inter-weekly variation in each of the two previous parameters using a random walk AR(1) model, as in the case with the JMS model, and as will be the case with all the subsequent system parameters. For these following derivations, it is useful to mathematically express this AR(1) model as a random effects design matrix $\mathbf{Z}$**,** whereby the w^{th} element of the product $\mathbf{Z}^{\mathrm{T}}\boldsymbol{\epsilon}$ equals a cumulative sum of the elements of $\boldsymbol{\epsilon}$. That is, we let

$$\mathbf{Z} = \begin{bmatrix} 0 & 1 & 1 & \dots & 1 \\ 0 & 0 & 1 & \dots & 1 \\ \vdots & \vdots & \vdots & \ddots & \vdots \\ 0 & 0 & 0 & \dots & 1 \end{bmatrix}$$

such that

$$[\mathbf{Z}^T\boldsymbol{\epsilon}]_w \ = \sum_{t<w} \epsilon_t.$$

For the river reach from RIS to BON ($j = 1$), we model survival and multiple cause-specific sources of mortality simultaneously, while considering the possible effects of covariates. We therefore let $\mathbf{L}_i$ be a $W \times C + 2$ matrix representing the odds from W weeks of survival and $C + 1$ of mortality sources between RIS to BON relative to their survival from RIS to BON. The c^{th} column of $\mathbf{L_i}$ represents the weekly odds of depredation by tern colony, c, relative to survival, which we can model as

$$[\mathbf{L_i}]_{\cdot c} = \exp\left(\mu_{\theta_c} + \ \boldsymbol{x}^T{}_{\theta,i}\boldsymbol{\beta}_\theta + \mathbf{Z}^T\boldsymbol{\epsilon}_{\theta_c}\right) \qquad \text{for } c \in \{1, \dots, C\}$$

where $\boldsymbol{\epsilon}_{\theta_c} \sim \mathrm{normal}\left(0, \sigma_{\theta_c}\right) \forall c$, the design matrix, $\boldsymbol{x}_\theta$, and fixed effects parameter vector, $\boldsymbol{\beta}_\theta$, are assumed equal for all colonies while μ_{θ_c}, $\boldsymbol{\epsilon}_{\theta_c}$, and σ_c are assumed to be unique to each colony. The $(C + 1)^{\mathrm{th}}$ column of $\mathbf{L_i}$ represents weekly odds of mortality due to all other causes relative to survival which we can model as

$$[\boldsymbol{L_i}]_{\cdot,C+1} = \exp\left(\mu_{\theta_M} + \ \boldsymbol{x}^T{}_{\theta_M,i}\boldsymbol{\beta}_{\theta_M} + \mathbf{Z}^T\boldsymbol{\epsilon}_{\theta_M}\right)$$

$\boldsymbol{\epsilon}_{\theta_M} \sim \mathrm{normal}\left(0, \sigma_{\theta_M}\right)$. Finally, as the last column of $\mathbf{L}_{\boldsymbol{i}}$ represent the odds of survival relative to survival (that is to say, relative to itself), we can simply write,

$$[\mathbf{L}_{\boldsymbol{i}}]_{\cdot,C+2} = \exp(\mathbf{0}) = \mathbf{1}.$$

To then derive for smolt i our best estimate of the probabilities of survival in reach 1, $\mathrm{s}_{1,i}$; depredation by colony $c, \theta_{C,i}$; and mortality by some other cause, $\theta_{M,i}$; using a geometric average (with weights defined by $\boldsymbol{\rho}_i$ as explained above) of the weekly estimates derived from the logistic regression equations:

$$[\theta_{1,i} \quad \dots \quad \theta_{C,i} \quad \theta_{M,i} \quad \mathrm{s}_{1,i}]^T = \boldsymbol{\rho}_\mathrm{i}^T \mathrm{diag}^{-1}(\mathbf{L}_{\boldsymbol{i}}^T \mathbf{1})\mathbf{L}_{\boldsymbol{i}}^T.$$

The probability of recovering a tag from each colony can be derived with the same detection and deposition parameters employed in the JMS model, however, we must additionally correct for the uncertainty in RIS passage week such that the probability of recovering a tag can be expressed $\theta_{1,c}\boldsymbol{\rho}_\mathrm{i}^T\boldsymbol{\gamma}_c$ where , $\boldsymbol{\gamma}_c$ is a vector a week-specific recovery probabilities for colony c.

For recapture at BON, the probabilities are derived according to a similar premise such that:

$$p_{1,i} = \boldsymbol{\rho}_\mathrm{i}^T \ \mathrm{logit}^{-1}\left(\mu_{p_1} + \ \mathbf{X}_{p_1}^T \boldsymbol{\beta}_{p_1} + \mathbf{Z}^T \boldsymbol{\epsilon}_{p_1}\right)$$

where $\boldsymbol{\epsilon}_{p_1} \sim \mathrm{normal}\left(0, \sigma_{p_1}\right)$.

For all other tag recoveries associated with reach $j = 1$ (i.e. from terns foraging between downstream of MCN and all gulls and cormorant colonies) and all recoveries and recaptures downstream of BON (i.e. in reach $j = 2$) we model a vector of weekly recovery probabilities, $\boldsymbol{\nu}_j$, as

$$\nu_{j,i} = \boldsymbol{\rho}_\mathrm{i}^T \ \mathrm{logit}^{-1}\left(\mu_{\nu_j} + \ \mathbf{X}_{\nu_\mathrm{j}}^T \boldsymbol{\beta}_{\nu_j} + \mathbf{Z}^T \boldsymbol{\epsilon}_{\nu_j}\right) \qquad \text{for } j \in \{1,2\}$$

where $\boldsymbol{\epsilon}_{\nu_j} \sim \mathrm{normal}(0, \sigma_\nu)$. Note, there were no recoveries between RRJ and RIS (segment $j = 0$) and therefore $\boldsymbol{\nu}_0 = \mathbf{0}$.

Likelihood Construction – The likelihood contribution of each smolt i can be expressed as:

$$Likelihood_i = p_i^{m_i}(1 - p_i)^{(1-m_i)} * prob(\boldsymbol{d}_i \mid f_i)$$

where m_i is an indicator variable denoting a successful recapture of smolt i at BON, the final reach after which smolt i was successfully recaptured by f_i, and $\boldsymbol{d}_i$ is the single-entry unit-vector $[r_1 \quad \dots \quad r_C \quad t_B \quad t_E \quad u]^T$ associated with smolt i, with $r_c = 1$ indicating a recovery from Caspian tern colony c which forages upstream of MCN, $t_B = 1$ indicating recovery from all other colonies upstream BON, $t_E = 1$ indicating recovery from any colony downstream of BON or recapture downstream of BON and $u = 1$ indicates an unrecovered/unrecaptured tag following f_i.

For recoveries on tern colonies upstream of MCN, if $f_i = 1$,

$$\text{prob}(r_c = 1 \mid f_i = 1) = \theta_{c,i}\boldsymbol{\rho}_{\mathrm{i}}^T\boldsymbol{\gamma}_c,$$

if $f_i = 0$,

$$\text{prob}(r_c = 1 \mid f_i = 0) = S_0 * (1 - p_0) * \text{prob}(r_c = 1 \mid f_i = 1),$$

and, otherwise,

$$\text{prob}(r_c = 1 \mid f_i > 1) = 0.$$

For recoveries on other colonies upstream of BON, if $f_i = 1$,

$$p(t_B = 1 \mid f_i = 1) = \left(1 - s_{1,i}\right) * \nu_{1,i},$$

if $f_i = 0$,

$$p(t_B = 1 \mid f_i = 0) = s_{0,i} * \left(1 - p_{0,i}\right) * p(t_B = 1 \mid f_i = j + 1)$$

and, otherwise,

$$\text{prob}(t_B = 1 \mid f_i > 1) = 0.$$

For recovery/recapture (at the estuary trawl or as an adult return), if $f_i = 2$,

$$p(t_E = 1 \mid f_i = 2) = \nu_{2,i}$$

and for $f_i < 2$,

$$p(t_E = 1 \mid f_i = j) = s_{j,i} * \left(1 - p_{j,i}\right) * p(t_E = 1 \mid f_i = j + 1).$$

For fish that were not recovered following their last recapture,

$$p(u = 1 \mid f_i = 2) = 1 - \nu_{2,i}$$

and for $f_i < 2$,

$$p(u = 1 \mid f_i = j) = s_{j,i} * \left(1 - p_{j,i}\right) * p(u = 1 \mid f_i = j + 1).$$

Priors assigned to all parameters were generally constructed to be uninformative. However, recapture and recovery information among years was shared to increase precision in the estimates of μ; for example, we assumed $\mu_{p_1} \sim \text{logistic}\left(\mu_{p_1}^*, \sigma_{\mathrm{p}_1}^*\right)$. Priors for each μ^* were assumed to be logistic(0,1). Priors for the vector $[\mu_{\theta_1} \quad \cdots \quad \mu_{\theta_C} \quad \mu_M \quad 1 - \sum_c \mu_c - \mu_M]^T$ were assigned a collective prior of Dirichlet($\mathbf{1}$). Priors for all $\vec{\beta}$'s were assigned to be $\mathrm{mvnormal}(\vec{0}, \mathbf{I})$. Priors for all σ's were assigned to be $\mathrm{normal}(0,1.5)$.

The choice of which covariates to include in a model comes with the inherent trade-off between a potential bias from the exclusion of impactful covariates and the unnecessary reduction in precision associated with including irrelevant covariates. With this in mind, for the estimates of the system parameters of greatest concern (i.e. survival and tern predation), we removed covariate parameters that lacked evidence. That is, we removed covariates parameters whose posterior distributions significantly overlapped 0 (i.e., greater than 5% of the posterior distribution lay above and below 0). For all system parameters at or downstream of BON, we simply assumed a saturated model, including all possible covariates; rear-type, failed/successful at RIS for the first seen at RRJ smolts, and failed/successful detection at BON.

All models were implemented using the software STAN accessed through R version 4.1.0 (RCDT 2021) using the rstan package (version 2.30 SDT 2022). 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. Chain convergence was verified using the Gelman-Rubin statistic (Gelman et al. 2013) and all accepted chains reported zero divergent transitions.

Finally, to evaluate whether differences in weekly survival and tern predation probabilities between cohorts (RRJ, RIS) were statistically significant, we compared the ratio of the odds of survival and tern predation within each week and across all years of available data (see *Appendix B, Methods* sections above for details regarding odds ratios and odds ratio tests).

Results & Discussion**:** The vast majority of steelhead passively detected at RRJ were hatchery-reared, representing 89.3% to 96.1% of all smolts each year. By comparison, 72.2% to 78.7% of steelhead smolts that were randomly selected for tagging at RIS were hatchery-reared (see also *Appendix B, Table B1* above). Although information on the size (fork length) and external condition (signs of injuries and disease) of RIS tagged smolts were available, there was no information for RRJ smolts because fish were passively detected (i.e., not handled), so the influence of these factors on relative differences in predation and survival could not be explored as part of this analysis.

Of the steelhead smolts first seen at RRJ, only 1.4% to 2.2% were subsequently recaptured at the RIS fish trap each year. For those steelhead seen at RRJ, but not at RIS, the time in which they passed RIS was inferred from the travel time model. Results indicated that each year, between 73.9% and 90.9% of all steelhead seen at RRJ were never seen again during outmigration. For these fish, the probability of passage day at RIS was estimated directly using the values of $\widehat{\mathbf{M}}(0)$, the state-transition matrix associated with the RRJ to RIS river reach. Of the RRJ steelhead that were seen downstream of RIS but not at RIS, up to 10.5%, 13.6% and 14.0% were seen each year at MCN, JDA, and BON, respectively. For these fish, the added information of when they were seen downstream of RIS provided additional information with which to estimate their day (and consequently week) of passage at RIS.

Each year, we estimated that the average travel time from RRJ to RIS, MCN, JDA, and BON ranged annually from lows of 0.8, 7.3, 9.6, and 10.4 days to highs of 3.1, 11.0, 14.4, and 15.8 days, respectively (*Figure C.1*). The relative odds of a wild fish seen at RRJ being recaptured at RIS versus a hatchery fish varied to a statistically significant degree from year to year. On average, for steelhead passing RRJ within the same week, the odds of a wild fish being recaptured at the RIS fish trap were 1.1 (95% credible interval = 0.9–1.3) times greater than those of a hatchery fish. After accounting for rear-type in recapture probabilities at RIS, we found no evidence of a difference in survival probabilities among rear-type cohorts from RRJ to RIS. However, due to the considerably low recapture rates at RIS, and the imperfect information related to when a fish arrived at RIS, estimates of smolt survival from RRJ to RIS were only weakly identifiable. Furthermore, the overall lack of certainty in these initial river-reach survival estimates impacts the precision of all estimates of survival from RIS for fish first seen at RRJ. Therefore, to compare estimates most effectively between cohorts, we compared estimates from their first (initial) sighting (either at RRJ or at RIS). We kept our estimation of passage timing at RIS in the model, however, in order to compare estimates of survival and tern predation among fish arriving at RIS within the same week.

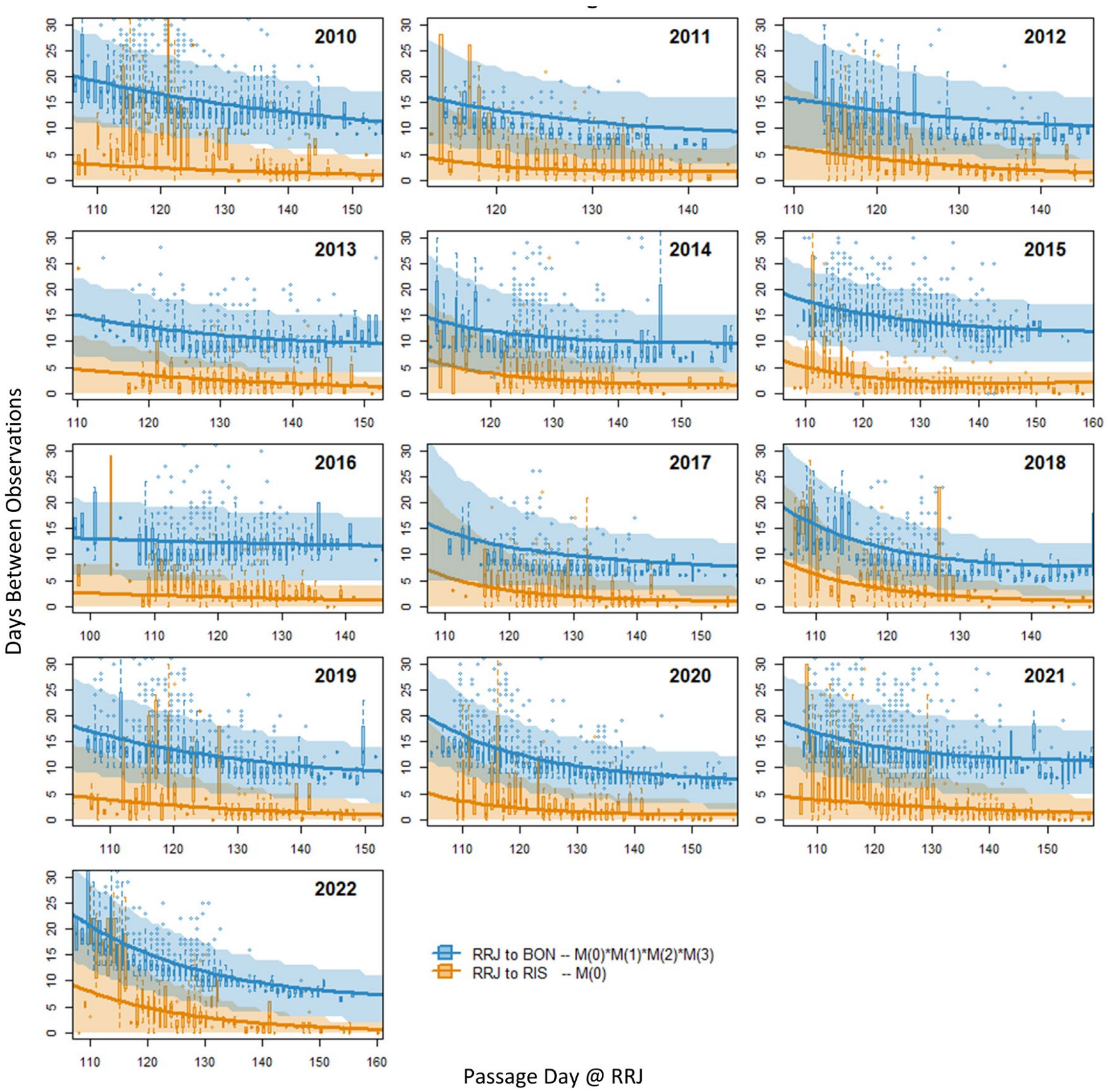

Figure C.1: Boxplots representing a subset of all observed travel times of PIT-tagged steelhead smolts detected at Rocky Reach Dam (RRJ) and downstream recapture sites (orange for RRJ to Rock Island Dam [RIS]; blue for RRJ to Bonneville Dam [BON]) by day of passage at RRJ. Shaded regions represent the best model fit of the travel time each year.

We evaluated possible associations between rear-type and each system-parameter separately for each cohort. For those fish first seen at RRJ, we additionally assessed the potential covariate impact of being handled at RIS for the system parameters downstream of RIS. For all evaluated system parameters, we found no evidence of an association with being handled at RIS by year or, on average, across all years. In evaluating predation by terns, we found a significant association between rear-type and predation probabilities for both cohorts, with hatchery fish more likely to be consumed than wild fish, but no significant difference among years. For fish first seen at RRJ and passing RIS within the same week, we estimated the odds of a hatchery fish being consumed by terns were 1.37 (1.18–1.69) times greater than those of wild fish. For fish tagged and released from RIS within the same week, we observed a similar association, with the odds of a hatchery fish being consumed by terns estimated to be 1.36 (1.14–2.49) times greater those of wild fish. Similarly, for both cohorts, we found a significant association between rear-type and the probabilities of mortality due to other, non-tern causes (e.g., gull predation), but not a significant difference in this association among years.

After accounting for the estimated travel time of steelhead smolts from RRJ to RIS, the disparity between the run-timing of smolts initially seen at RRJ versus those tagged at RIS was still apparent (*Figure C.2*). Results indicated that the run-timing of RRJ detected smolts were often left-shifted (early) compared with smolts tagged at RIS and that the temporal distribution of the run-at-large (as approximated by the Steelhead Passage Index, which includes all tagged and untagged fish passing RIS; FPC 2021) more closely resembles smolts tagged at RIS than smolts first seen passing RRJ. In many, but not all years, survival probabilities were also higher for fish that migrate earlier in the run (*Figure C.2*). Given the early detection of hatchery steelhead initially seen at RRJ, it follows that the overall (annual) proportion of these smolts that survive to BON would be greater than that of steelhead tagged at RIS. However, we might expect that survival probabilities for steelhead smolts should be relatively similar for cohorts migrating within the same time period (i.e. comparing weekly rates versus annual rates). *Figure C.2* illustrates that indeed, the weekly estimated survival probabilities for the RRJ and RIS smolts were similar, with very similar patterns of variation in survival probabilities across each year.

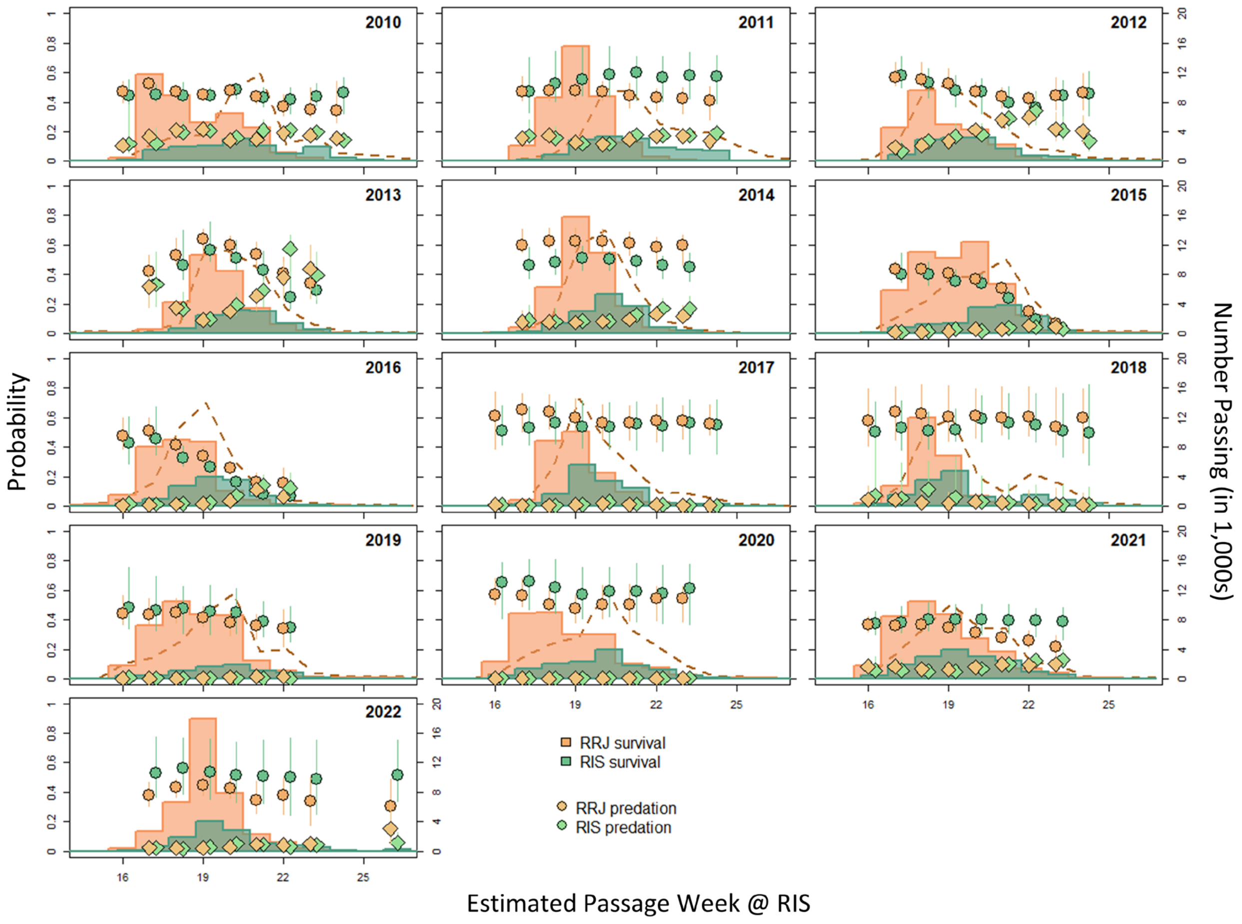

Figure C.2: Weekly probabilities of steelhead smolt survival (circles) and predation by Caspian terns (diamonds) for fish passively detected at Rocky Reach Dam (RRJ) and those tagging at Rock Island (RIS). Polygons represent histograms of the estimated weekly proportion of all steelhead first seen at RRJ and the observed weekly proportion of all steelhead at RIS within each year. The brown dotted line represents the steelhead smolt index as measured at RIS (data not available in 2022).

By comparing the estimated odds of survival to BON we can better assess relative estimated differences in rates. The estimated odds of survival to BON for each cohort were also similar on a weekly basis, with no statistical difference in the odds of survival from RRJ to BON and RIS to BON in 96 of the 102 weekly-comparisons (*Figure C.3*). There was also no evidence of a difference when data from all weeks were considered (*Figure C.3*). It should be noted that given the longer migration distance from RRJ to BON (528 Rkm), estimates of smolt survival would presumably be lower than those of smolts during passage from RIS to BON (495 Rkm). This trend was observed in five of the 13 study years, with the weekly odds of survival for RIS tagged smolts slightly higher than those of smolts last seen at RRJ. In some years (2013-2016, 2018), however, estimates of weekly survival were consistently higher from RRJ to BON (*Figure C.3*) and it is certainly possible that these relative differences were associated with handling effects at RIS. It is unclear, however, why there would be presumed handling effects in some weeks and some years and not others. Regardless, relative differences in smolt survival were small and were not statistically significant for either cohort when data from all weeks and years were considered.

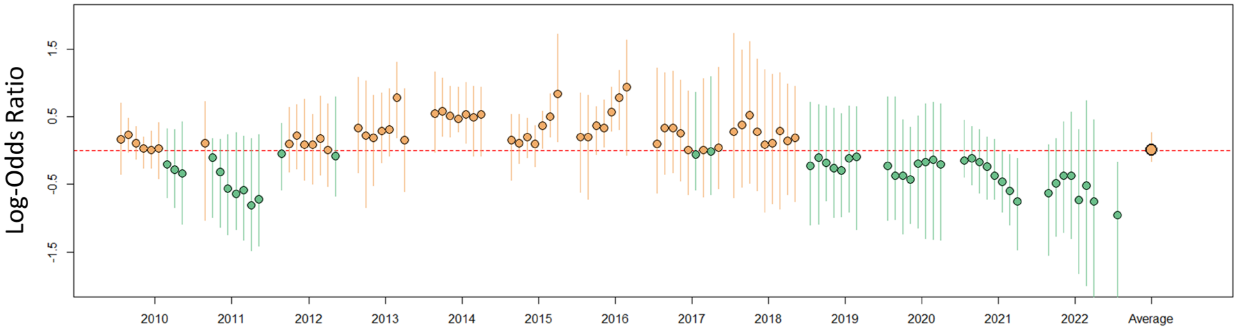

Figure C.3: Weekly estimates of the log odds of smolt survival for steelhead passively detected at Rocky Reach Dam versus those tagged at Rock Island Dam to Bonneville Dam. Vertical line segments represent 95% credible intervals. Orange-colored points above the red dashed line at 0 represent weeks in which the odds of survival were greater for the Rocky Reach Dam cohort, while green-colored points below the red dashed line represent weeks in which the odds of survival were greater for the Rock Island Dam cohort. Credible intervals extending across this line indicate there was no statistically signficant difference.

The FPC memos motivating this analysis evaluated tern predation effects based on the proportion of available PIT tags from each cohort (RRJ, RIS) that were recovered on the Goose Island tern colony in Potholes Reservoir (i.e. recovery rates). Recovery rates, however, grossly underestimate predation rates because they do not account for the proportion of consumed tags that were deposited by birds on their breeding colony and the proportion of deposited tags that were subsequently recovered by researchers after the breeding season (see *Methods* section above). Detection probabilities of PIT tags on bird colonies vary by colony and week, generally with the more time a tag remains on-colony, the lower the probability of recovering it after the breeding season (Evans et al. 2012, Hostetter et al. 2015, Payton et al. 2019). This is a notable consideration as, proportionally, RRJ detected smolts migrated earlier than RIS tagged smolts and consequently were depredated earlier in the season. *Figure C.4* illustrates that the weekly estimated tern predation probabilities for both cohorts, those that account for detection and deposition probabilities, were very similar to one another and demonstrate similar patterns of variation in predation across each year. For instance, in 2013 and 2014, years with high levels of tern predation, as tern predation rates on smolts increased, smolt survival rates decreased for both cohorts. Conversely, in years with low levels of tern predation, like in 2017-2020, smolt survival to Bonneville Dam was generally higher and less variable for both cohorts.

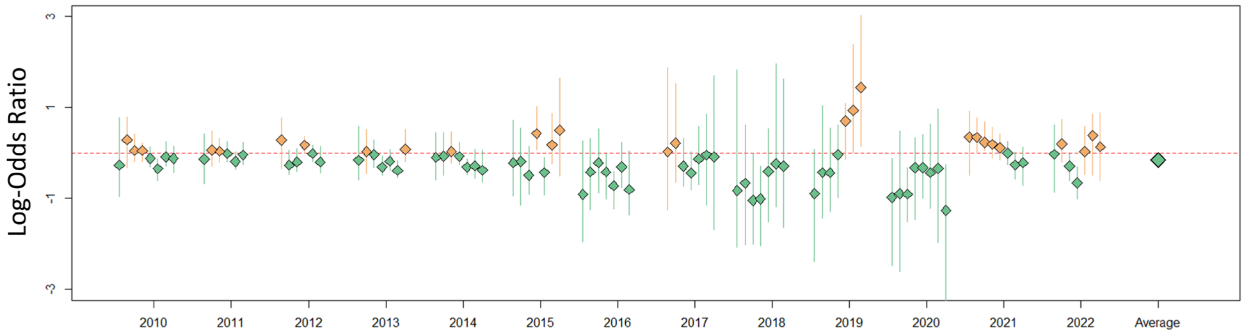

Figure C.4. Weekly estimates of the log odds of predation by terns for steelhead smolts passively detected at Rocky Reach Dam versus those tagged at Rock Island Dam. Vertical line segments represent 95% credible intervals. Orange-colored points above the red dashed line at 0 represent weeks in which the odds of predation were greater for the Rocky Reach cohort, while green-colored points below the red dashed line reprensent weeks in which the odds of predation were greater for the Rock Island Dam cohort. Credible intervals extending across this line indicate there was no statistically signficant difference.

Results of odd ratio tests indicated that the estimated odds of tern predation between cohorts were very similar on a weekly basis, with no statistical difference in the odds of tern predation in 94 of the 102 weekly-comparisons (see *Figure C.4* above). Across all years, RIS tagged steelhead smolts were slightly (1.16 [1.01–1.32] times) more likely to be consumed than their RRJ counterparts, a somewhat expected finding given the RRJ estimates of availability were not adjusted for smolt survival to RIS.

In summary, we found little to no evidence of handling effects or other factors that influenced the relative fitness of fish tagged at RIS versus those passively detected at RRJ when comparing survival and tern predation within the same week and among similar rear-types. This is not to say that handling effects at RIS (or at any other tagging site) are not real but survival and tern predation probabilities of handled (RIS) and unhandled (RRJ) smolts were similar, with very similar trends within and across years. Although estimates of smolt survival and tern predation probabilities were similar, they are not necessarily equivalent, and there is no way to say how much of the residual differences were due to random chance (e.g., sampling error) versus inherent differences between these two groups of tagged fish. As a random sample of all available steelhead smolts (fish from all major spawning tributaries, of all sizes, rear-types, and conditions), however, fish tagged at RIS can be used to make credible inferences about the UCR steelhead population at-large (tagged and untagged). In contrast, steelhead detected at RRJ were disproportionately early-run hatchery fish that notably excluded smolts from the Wenatchee River, a major component of the UCR steelhead population. As such, smolts passively detected at RRJ were not representative of the UCR steelhead population at-large because they do not include all elements or characteristics of the population at-large. Finally, future studies that estimate smolt survival from RRJ to RIS will presumably result in more accurate relative comparisons of survival and avian predation, but such studies are currently limited by extremely low recapture rates of tagged fish during outmigration from RRJ to Bonneville Dam.