AVIAN PREDATION IN THE COLUMBIA RIVER BASIN

2023 Final Annual Report

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EXECUTIVE SUMMARY

There are three 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). The third plan targets 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 unmanaged 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 predation on salmonid mortality.

A total of 35 active breeding colonies of piscivorous waterbirds were detected in the CRB during the 2023 breeding season. Of those, cormorant and gull colonies were the most prevalent (14 and 11 colonies, respectively), followed by terns (8 colonies), and pelicans (2 colonies). Most of these breeding colonies (23) were in the CPR, with 8 and 3 colonies located in the CRE and lower Columbia River, respectively. Tern and cormorant colonies continue to be managed as part of the three 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. 40,000 individuals) of all the piscivorous colonial waterbirds in the CRB, followed by cormorants (ca. 8,100 breeding pairs), pelicans (ca. 3,900 individuals), and terns (ca. 1,000 breeding pairs). Recent data suggest that the numbers of terns and cormorants nesting in the CRB have declined, which was the objective of management, but has also resulted in declines in the Pacific Flyway breeding populations of both species. This raises concerns about the conservation status of these populations, especially for the rapidly declining Pacific Flyway breeding population of Caspian terns.

The designated tern colony on ESI declined to the lowest level ever recorded (524 breeding pairs) and once again failed to produce any young in 2023, with complete nesting failure being observed at that colony in six out of the last eight years. The reason(s) for the complete colony failures at the ESI tern colony is likely due to gull predation on tern eggs, especially during disturbances of the colony by bald eagles (*Haliaeetus leucocephalus)*. Adding to the growing conservation concern for the Pacific Flyway population of terns is the recent emergence of highly pathogenic avian influenza (HPAI), which caused mortality of over 1,200 terns in the Pacific Flyway population in 2023, including in the CRE. Concurrent with the decline in colony size and nesting failure at the ESI tern colony in recent years, there has been a large influx of terns attempting to nest on Rice Island, where the per capita (per breeding pair) impacts of terns on smolt survival are known to be appreciably higher than those of terns nesting on ESI. Adaptive management conducted by the U.S. Army Corps of Engineers (USACE) to prevent nesting by terns on Rice Island in 2023 was eventually successful in dispersing the terns from that site, but not until after the peak of the smolt outmigration period, during which time there were up to ca. 1,400 terns 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 12 breeding pairs were observed on Goose Island in 2023. Management to actively dissuade terns from nesting on Crescent Island in McNary Reservoir ceased in 2021, however, and terns re-established a colony in 2022 (149 breeding pairs) and 2023 (88 breeding pairs). While placement of large woody debris prior to the 2023 breeding season prevented terns from nesting on the historical colony site on Crescent Island, terns nested in new areas of open habitat without woody debris. Starting in 2015, most of the terns that were dissuaded from nesting on Crescent Island had relocated to nest on the Blalock Islands in John Day Reservoir. Beginning in 2021, however, adaptive management to raise the water elevation of John Day Reservoir was implemented to inundate tern nesting habitat in the Blalock Islands and in 2021–2023 no nesting by terns was observed at that site. This adaptive management apparently resulted in a shift of terns back to Crescent Island, where active tern dissuasion efforts had ceased, and to Badger Island, located on the Columbia River just upstream of Crescent Island. Despite adaptive management implemented in 2022–2023 to reduce tern nesting habitat on Badger Island (via placement of large woody debris), the largest tern colony in the CPR during 2022 and 2023 was on Badger Island (267 and 274 breeding pairs in 2022 and 2023, respectively). A total of 472 breeding pairs of terns nested at all colonies in the CPR combined in 2023, exceeding the management goal of less than 200 pairs. The number of tern breeding pairs in 2023 was slightly lower than the number of terns that nested in 2022 (510 breeding pairs) and an over-all 46% reduction in the total number of pairs in the CPR since management actions commenced in 2014. Due to high fidelity to historical nesting sites, adaptive management will continue to be important to meet the goals and objectives of management plans for terns in both CRE and CPR.

One of the primary objectives of the CPR tern management plan is to reduce predation rates on smolts by terns to less than 2% per ESA-listed salmonid population (Evolutionarily Significant

Unit [ESU] or Distinct Population Segment [DPS]), per colony. In 2023, this objective was met for the first time since management actions commenced in 2014. Predation rates were highest by terns breeding on Badger Island in McNary Reservoir, where an estimated 1.4% (95% credible interval = 0.9–2.5%) and 1.9% (1.2–3.2%) of Upper Columbia River (UCR) steelhead and Snake River (SR) steelhead, respectively, were consumed by terns in 2023. Estimated predation rates by terns nesting on Goose Island in Potholes Reservoir and on Shoal Island in Lenore Lake were both less than 1.0% per salmonid ESU/DPS. Due to adaptive management actions that resulted in higher reservoir elevations in John Day Reservoir, no terns have nested on the Blalock Islands during 2021–2023, a nesting site where predation rates exceeded the 2% threshold for several ESUs/DPSs in previous years (2015–2020). Since management commenced, average annual predation rates on UCR steelhead by terns nesting on Goose Island and elsewhere in Potholes Reservoir have been reduced from 15.7% (14.1–18.9%) prior to management (2007–2013) to 1.0% (0.7–1.7%) during the management period (2014–2023). 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 management plan. Due to increases in predation on smolts by terns that have re-nested on Crescent Island and terns that are now nesting on Badger Island and Lenore Lake, adaptive management actions will likely be necessary to prevent regression of smolt survival gains associated with tern management in the CPR.

Predation rates by terns nesting on ESI have been greatly reduced since management actions were first implemented in 2008, with predation rates on steelhead DPSs reduced by 65% to 76%, reductions that met or exceeded those specified by the tern management plan for the CRE. In recent years, however, large numbers of terns have attempted to nest on Rice Island in the upper CRE and terns continue to attempt to nest outside of the designated colony area on ESI. Over the past two years, estimated predation rates by terns that attempted to nest on Rice Island have been as high as 1.1% (0.8–1.8%) for UCR steelhead and 2.9% (1.6–5.1%) for Middle Columbia River steelhead. Collectively, results of tern management actions in the CRE and CPR indicate that adaptive management will be necessary to prevent terns from relocating to nest at other sites in the CRB where predation rates on steelhead smolts remain high. Due to reductions in colony size and the lack of productivity at the ESI tern colony, adaptive management is also necessary to ensure the long-term viability of the Pacific Flyway population of Caspian terns, including providing new and improved nesting opportunities for terns outside of the basin.

Management at the ESI cormorant colony has also resulted in the dispersal of birds away from ESI and 2023 marks the fifth consecutive year with little to no cormorant nesting on ESI. An unintended consequence of management actions on ESI was the subsequent rapid expansion of cormorant colonies on the Astoria-Megler Bridge (AMB), Longview Bridge (Lewis and Clark Bridge), channel markers, and transmission towers (TRT) located in upper CRE and LCR. Cormorant colony size on the AMB increased dramatically during implementation of the cormorant management plan on ESI during 2015–2019, culminating in 5,153 breeding pairs nesting on the bridge in 2023. The AMB is located upstream of ESI in the freshwater mixing zone of the CRE and the three other colony locations (Longview Bridge, channel markers, and

TRT) are located in the strictly freshwater zone of LCR, aquatic environments where fewer alternative, non-salmonid prey fish are available. Thus, smolts are consumed in greater proportion on a per capita basis by cormorants nesting at these locations. Estimates of per capita predation rates on salmonid smolts by cormorants nesting on the AMB and TRT were 4–10 times greater (depending on salmonid ESU/DPS and colony) than average annual per capita predation rates by cormorants that nested on ESI. Estimates of colony-wide (all breeding pairs) predation rates by cormorants nesting on the AMB in 2023 ranged from 3.8% (1.1–7.9%) for SR fall Chinook salmon to 14.3% (6.3–24.4%) for SR sockeye salmon, estimates that were higher than those of cormorants on ESI in years past. Estimates of predation rates by cormorants on TRT in 2023 ranged from 0.5% (0.1–1.9%) for SR fall Chinook to 3.7% (1.3–12.6%) for Middle Columbia River steelhead. Collectively, these results indicate that colony location is closely related to the magnitude of cormorant predation on smolts in the CRE and LCR. While predation estimates for cormorant colonies at Longview Bridge and channel markers were unavailable (i.e. sites preclude PIT tag recovery), per capita impacts may be similar to cormorants nesting on the AMB and TRT due to their location in the mixing and freshwater zones of the CRE. Due to the movement of cormorants from the marine-zone to the mixing- and freshwater-zones of the CRE and LCR, the impact of cormorant predation on smolt survival has increased since cormorant management actions were first implemented on ESI in 2015. Adaptive management to prevent cormorants from nesting in the upper CRE and LCR and to attract cormorants to colony locations where their impacts on smolt survival would be less (including ESI) will be necessary to achieve the goals and objectives of the cormorant management plan.

An investigation of predation on smolts by piscivorous birds nesting at other unmanaged colonies indicated that smolt consumption rates by gulls nesting in the CPR were often higher than those of managed tern colonies in 2023. For instance, estimates of gull consumption rates as high as 5.8% (2.1–9.9%) for UCR steelhead and 6.7% (3.5–12.4%) for SR steelhead were documented for gulls nesting at Badger Island in McNary Reservoir and Miller Rocks in The Dalles Reservoir, respectively. Unlike terns and cormorants, however, gulls are scavengers and are known to consume dead or moribund fish and to steal (kleptoparasitize) fish from other waterbirds. Consequently, we use the term “gull consumption rates” rather than “gull predation rates.” Estimates of cormorant predation rates in the CPR as high as 2.1% (1.1–4.6%) and 3.1% (1.6–6.4%) for SR spring/summer Chinook and SR steelhead, respectively, were documented at the Foundation Island colony in McNary Reservoir. Despite the relatively high predation/consumption rates on smolts observed at some unmanaged gull and cormorant colonies in 2023, impacts from piscivorous waterbirds nesting at several other colonies were low to non-existent. For example, estimates of predation 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 < 0.5% per salmonid ESU/DPS, per colony. Results of this and several other studies indicate that predation/consumption rates on salmonid smolts by piscivorous colonial waterbirds are highly variable and that not all colonies pose a threat to smolt survival in the CRB.

Based on an investigation of the cumulative effects of avian predation/consumption (predation by all colonies combined), an estimated 23.2% (19.7–27.9%) of SR steelhead and 28.6% (24.0–34.2%) of UCR steelhead were consumed by piscivorous colonial waterbirds during smolt outmigration from the middle Columbia River (for UCR smolts) and the lower Snake River (for SR smolts) to the Pacific Ocean in 2023. Cumulative predation/consumption rates were significantly lower for Chinook salmon ESUs, with an estimated 8.2% (6.2–10.6%) of SR sub-yearling Chinook and 8.7% (6.9–11.7%) of SR yearling Chinook consumed. Predation/consumption on SR sockeye salmon, however, was similar to that of steelhead, with an estimated 19.4% (14.8–24.3%) of SR sockeye consumed by birds in 2023, among the highest estimates of cumulative predation on sockeye smolts observed to-date. By river reach, predation/consumption rates were highest for smolts between Bonneville Dam and the Pacific Ocean, followed by the river reach from 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 2023. Reach-specific predation/consumption rates in 2023 were similar to those of previous years, except for predation in the CRE, where predation impacts were above average due to cormorants nesting on the AMB. Collectively, results indicate that the cumulative effects of avian predation/consumption remain a substantial source of smolt mortality in the CRB, particularly for steelhead and sockeye populations.

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 Columbia River Power System. Over the last two decades, numerous research, monitoring, and evaluation (RM&E) studies of avian predation have been conducted to assess the impacts on smolt survival of consumption by Caspian terns (*Hydroprogne caspia*; hereafter referred to as “tern” or “terns”), double-crested cormorants (*Nannopterum auritum*; hereafter referred to as “cormorant” or “cormorants”), California and ring-billed gulls (*Larus californicus* and *L. delawarensis*; hereafter referred to as “gull” or “gulls”), and American white pelicans (*Pelecanus erythrorhynchos*; hereafter referred to as “pelican” or “pelicans”) in the CRB.

To address concerns about the impact of avian predation on the survival of salmonid smolts originating from the CRB, the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps of Engineers (USACE), and their management partners developed and implemented three separate management plans to reduce predation rates on smolts by piscivorous waterbirds nesting at four breeding colonies in the CRB (USFWS 2005, USACE 2014, USACE 2015): the tern and cormorant breeding colonies on East Sand Island (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) with limited egg collection under permit 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 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 or near complete colony failure/abandonment which is contributing to the movement of birds from managed to unmanaged sites within the CRB and may be negatively affecting the conservation status of these bird 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. 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 2023 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 2023 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, to include survival rates for Upper Columbia River (UCR) steelhead, Snake River (SR) steelhead, SR yearling Chinook, SR sub-yearly Chinook, and SR sockeye smolts; (d) estimated the additive effects of tern consumption on UCR steelhead 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

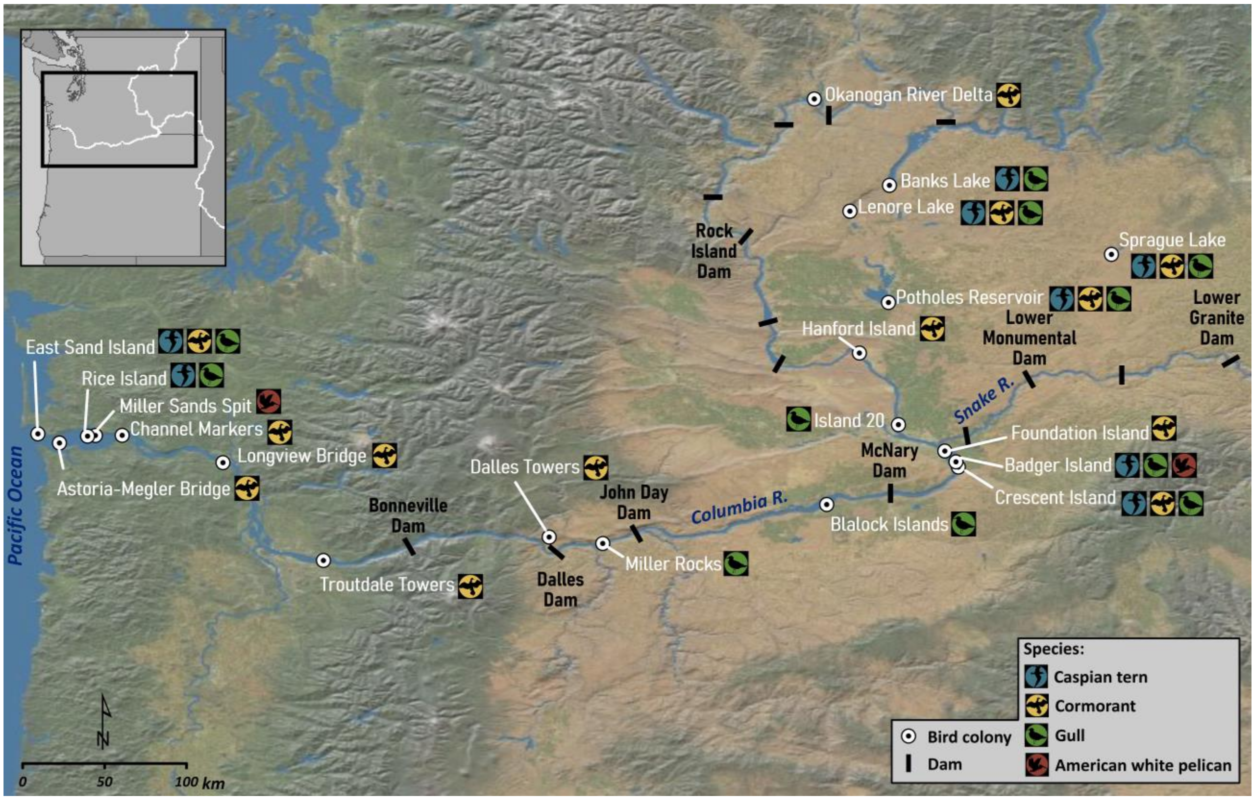
This work is part of a comprehensive program to implement, monitor, and evaluate avian predation management plans (including adaptive management) to reduce the impacts of avian predators on the survival of ESA-listed juvenile salmonids in the CRB. Action effectiveness monitoring was conducted in the 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 2023 (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 2023, nest dissuasion activities on Goose Island and elsewhere in Potholes Reservoir were conducted by the Bureau of Reclamation (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 2023). The Fisheries Field Unit (FFU) at the USACE carried out nest dissuasion efforts outside the designated tern colony area on ESI, and as an adaptive management component of the plan, on Rice Island in 2023, a description of those activities was summarized in two reports produced by that agency (Roberts et al. 2024; Blair et al. 2024). Nest dissuasion also occurred on sections of the Astoria-Megler-Bridge (AMB) that were slated for repair/painting in 2023 (M. Alex, USDA-APHIS-WS, personal communication). Nest dissuasion activities were also performed as an adaptive management component to the Inland Avian Predation Management Plan (IAPMP; USACE 2014) at several tern colonies in the CPR, including the managed site on Crescent Island (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm) and unmanaged sites on Badger Island (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm) and the Blalock Islands (Brandtner and Tidwell 2021; Strong and Tidwell 2022b; Blair et al. 2024; see *below* for more detail). Finally, the gull colony on Miller Rocks was adaptively managed to prevent gull nesting and reduce the size of the colony in 2023 (B. Sharp, Yakama Nation, pers. comm.; see *below* for more details).

NESTING DISTRIBUTION & COLONY SIZE

Monitoring of piscivorous waterbird colonies in the CRB (*Map 1*) were conducted from the air (fixed-wing aircraft), the water (boat-based surveys), and/or from shore, with precautions taken to minimize disturbance to actively nesting 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 2023 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; Evans et al. 2023).



Map 1. Study area in the Columbia River basin in 2023.

Periodic (weekly to monthly) ground- and boat-based surveys were conducted throughout the breeding season to identify where piscivorous colonial waterbirds were nesting in the CRB. Three fixed-winged aerial surveys (13–15 May, 3–5 June, and 15–16 June) 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 high-resolution aerial imagery corresponding with the peak in nesting activity at each colony and enumerating visible birds using ArcGIS (ESRI 2020. ArcGIS Desktop: Release 10.8.1. Redlands, CA) and DotDotGoose (Ersts, P.J. Version 1.7.0. American Museum of Natural History, Center for Biodiversity and Conservation). 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.

At Goose Island, and other suitable nesting colony sites in northern Potholes Reservoir, we monitored the activities of terns and gulls weekly or biweekly 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 AMB in the Columbia River estuary was monitored by boat on three occasions (16–17 May, 31 May–1 June, and 13 June) to determine nesting chronology and peak colony size. Finally, the size of the tern and cormorant colonies on ESI and the tern colony on Rice Island were estimated by the USACE-FFU in 2023 (Roberts et al. 2024; Blair et al. 2024).

AVIAN PREDATION RATES

Using previously developed protocols and methods, we analyzed smolt passive integrated transponder (PIT) tags collected on bird colonies to estimate predation rates on ESA-listed salmonid ESUs/DPSs and to assess relative differences in predation rates prior to and during management actions, with a focus on data collected in 2023. Comparisons between current (2023) and previous predation rates (2007–2022) were made in the context of management initiatives for terns in CPR relative to the IAPMP 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 all piscivorous colonial waterbird species and colonies combined) on salmonid 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 (terns, cormorants, gulls, pelicans) and colonies posed the greatest potential threat to smolt survival in 2023. 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 in 2023, smolts were intentionally captured, PIT-tagged, and released into the tailrace of Rock Island Dam (RIS) on the middle Columbia River as part of this study. 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. 2012, Evans et al. 2019). 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, rear-type; 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 peak smolt out-migration period of April to June. Steelhead were randomly selected for tagging (i.e. tagged regardless of condition, rear-type, or 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 (see also *Appendix B* and *C* for additional details). The target sample size goal was to PIT-tag approximately 7,000 steelhead smolts for use in predation and survival analyses in 2023. 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 rate 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 occurred 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. Furthermore, in some cases, tagging of ESA-listed fish at collection sites at mainstem dams has been discontinued (e.g., spring Chinook at RIS), so adequate sample sizes for use in this and other studies no longer exist.

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: logit^{-1} $(\mu_\theta) \sim uniform(0,1)$ and $\sigma_\theta^2 \sim uniform(0,20)$. This allows each week (*i*) to have a unique predation probability (θ_i), while still sharing information among weeks improving precision.

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

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

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

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

For some colonies and years 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). 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:

$$\dot{\theta_p} = \sum_p \sum\nolimits_i(\theta_{ip} * n_i)/\sum\nolimits_i(n_i) \Big/ C_p$$

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:

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

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 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 – 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 (2014–2023) 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). Since 2019 it has been the responsibility of the USACE and its contractors to recover smolt PIT tags and to estimate ESU/DPS-specific predation rates and per capita predation rates by terns and cormorants on East Sand Island. In 2023, the USACE recovered smolt PIT tags from both the tern and cormorant colonies on East Sand Island (Blair et al. 2024) but they have not estimated predation rates.

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 2023. Estimates of cumulative predation effects, however, rely on a different analytical framework (see *Cumulative Predation and Survival* section) than those of colony- and ESU/DPS-specific predation rates (see *Predation Rate Estimates* section), so the two estimates are not directly comparably to one another, although the two methods can generate similar estimates of predation at the colony 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 2023. 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), a net trawl detection system and at antennas located on pile dikes 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 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, Evans et al. 2022a). As such, estimates of avian predation were compared to estimates of total smolt 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 2023 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 BON, 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 $1 - \sum_{j \le k} \sum_d \Theta_{j,d}$.
>
> **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

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

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

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

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 across 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 2015–2023, 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 (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: Understanding the degree to which avian predation limits fish survival is paramount to evaluating the efficacy of management actions aimed at increasing fish survival. In particular, 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 on UCR steelhead 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 (BON) as an adult in the absence of tern predation (Payton et al. 2020, Payton et al. 2021). For these initial studies, results were derived from data associated with smolts tagged/recaptured at RIS (see *PIT-tagging of Upper Columbia River Steelhead above*). The process of capturing, handling, and PIT-tagging steelhead smolts at RIS, however, could negatively influence fish survival, fish susceptibility to avian predation, or otherwise bias the results of additive or other related analyses. To this end, in 2023, we investigated whether the additive relationship observed in smolts from RIS was also apparent and similar in magnitude for steelhead smolts last detected at the Rock Reach Dam juvenile bypass facility (RRJ) on the middle Columbia River. Results may support the hypothesis that the additive effects of predation on steelhead are similar within and across the same salmonid population, regardless of where smolts are sampled in space and time.

Additionally, Payton et al. (2020) reparametrized the additive/compensatory model of Sandercock et al. (2011) 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; Payton et al. 2020). For this analysis we disentangled 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. With the results from these analyses herein and those previous based on RIS sampling, we can compare closely related but disparate groups of smolts over different spatial scales and potentially better elucidate the nature of any additive relationship between avian predation and smolt survival.

Weekly and annual estimates of predation and survival probabilities on PIT-tagged UCR steelhead smolts that were tagged/recaptured at RIS and RRJ were used to investigate to what degree tern predation was an additive source of mortality during smolt outmigration to BON from 2010–2023 (years when both RRJ bypass was operational and RIS tagging studies occurred). We used the mark-recapture-recovery model of Payton et al. (2020) to assess the strength, magnitude, and direction of the relationship between tern predation on steelhead smolt survival based on smolt from each location (RIS, RRJ) passing each river reach or segment. 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,

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

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 avian 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

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

We model the assumed similarity in baseline survival among years as $s_y^0 \sim normal(\mu_{s^0}, \sigma_{s^0})$ for each year y. We further 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.

RESULTS & DISCUSSION

NEST DISSUASION ACTIVITIES

Columbia Plateau Region

As was the case in 2020–2022, the U.S. Bureau of Reclamation and its contractor (USDA-APHIS-WS) carried out all nest dissuasion activities of terns on Goose Island and elsewhere in Potholes Reservoir in 2023. 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 2023 breeding season. 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 terns prospecting for nest sites 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, 2023).

The growth of willows planted on Crescent Island in 2016 had mostly eliminated all upland habitat suitable for tern nesting on Crescent Island. Consequently, 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) and have nested there every year since. 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 roughly 2,000 individuals in recent years has 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). Prior to the 2023 breeding season, agency personnel placed large woody debris in areas of unvegetated habitat on Crescent Island (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm.) to prevent tern nesting. No active hazing of prospecting terns was conducted on Crescent Island in 2023, 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 2021-2023, 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, 2023). In 2022 and 2023, agency personnel placed large woody debris on one of two nesting areas used by terns on Badger Island in 2021 (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm.). 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.

Over concerns of its impact to the survival of juvenile salmonids, the gull colony on Miller Rocks was adaptively managed (under permit) to prevent gull nesting and reduce the size of the colony in 2023. A variety of nest dissuasion methods were used by the Yakama Nation and its contractors in 2023 including active hazing by humans and dogs, air cannons, falconry, and pyrotechnics (B. Sharp, Yakama Nation, pers. comm.).

Columbia River Estuary

As was the case in 2000–2022, the USACE-FFU conducted tern monitoring and tern nest dissuasion efforts on ESI outside the 1-acre designated colony area and on Rice Island in 2023 (Blair et al. 2024; Roberts et al. 2024). 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 double-crested cormorant colony on ESI in 2023 (Blair and Tidwell 2024). For further details on avian predation monitoring and management efforts on ESI and Rice Island in 2023, see Roberts et al. (2024) and Blair et al. (2024).

Active hazing of cormorants was conducted by USDA-APHIS-WS on two spans (166–167) of the AMB slated for maintenance and painting in 2023 (M. Alex, USDA-APHIS-WS, pers. comm.). Because nesting habitat for cormorants on the bridge is currently not limiting, it is presumed that cormorants displaced from these spans would move to renest on other areas of the bridge where dissuasion efforts were not occurring. Without continued nest dissuasion activities in future years, areas where dissuasion efforts occurred in 2023 are expected to be available again for cormorants to nest during the 2024 breeding season.

NESTING DISTRIBUTION & COLONY SIZE

A total of 35 active colonies of piscivorous waterbirds were detected in the CRB during the 2023 breeding season. Of those, cormorant and gull colonies were the most prevalent (14 and 11 colonies, respectively), followed by tern and pelican colonies (8 and 2 colonies, respectively; *Table 1* and *Table 2*). Most of these breeding colonies (23) were in the CPR, with 8 and 3 colonies located in the CRE and LCR, respectively. Some tern colonies continue to be managed as part of ongoing management plans and management has resulted in a shift in the nesting distribution of terns, prompting adaptive management at several colony sites (i.e. Rice Island, the Blalock Islands, Crescent Island, and Badger Island; see *below*). As has been the case in the past, gulls were the most numerous (ca. 40,000 individuals) of all the piscivorous colonial waterbirds in the CRB, followed by cormorants (ca. 8,100 breeding pairs), pelicans (ca. 3,900 individuals), and terns (ca. 1,000 breeding pairs). Recent data suggest that the numbers of terns and cormorants nesting in the CRB have declined, the objective of management, but has apparently resulted in declines in the Pacific Flyway breeding populations for the species. This raises concerns about the conservation status of these populations, especially for the rapidly declining Pacific Flyway population of Caspian terns.

Table 1. Annual colony size (peak number of breeding pairs) for Caspian terns in the Columbia River basin during 2007–2023. "Active" denotes a colony that was active but not counted to determine peak number of breeding pairs. Estimates of colony size for Caspian terns on East Sand Island and Rice Island in 2020–2023 were provided by the U.S. Army Corps of Engineers (see USACE-FFU 2020, Brandtner and Tidwell 2021; Roberts et al. 2024; Blair et al. 2024 for more details).

Colony	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023
Columbia River Estuary																	
East Sand Is. (Columbia River)	9,623	10,668	9,854	8,283	6,969	6,416	7,387	6,269	6,240	5,915	3,500	4,959	3,861	2,387	2,050	1,725	524
Rice Is. (Columbia River)[1]	0	0	0	0	0	0	0	0	0	0	0	0	0	0	Active	Active	Active
Columbia Plateau Region																	
Goose Is. (Potholes Reservoir)	282	293	487	416	422	463	340	159	2	0	0	0	0	6	22	16	12
Crescent Is. (Columbia River)	355	388	349	375	419	422	393	474	0	0	0	0	0	0	1	149	88
Blalock Islands (Columbia River)	43	104	79	136	20	6	26	45	677	483	449	313	379	150	0	0	0
Badger Is. (Columbia River)	0	0	0	0	33	60	0	0	0	0	41	8	0	0	231	267	274
Twinning & Goose Is. (Banks Lake)	31	27	61	34	19	22	13	67	64	6	0	0	0	0	0	0	7
Harper Is. (Sprague Lake)	0	11	4	4	4	30	1	8	10	3	92	79	18	0	85	2	10
North Rocks & Shoal Is. (Lenore Lake)	0	0	0	0	0	0	0	0	0	0	123	91	48	53	61	76	81
Total (Columbia River Estuary)	**9,623**	**10,668**	**9,854**	**8,283**	**6,969**	**6,416**	**7,387**	**6,269**	**6,240**	**5,915**	**3,500**	**4,959**	**3,861**	**2,387**	**na**	**na**	**na**
Total (Columbia Plateau Region)	**711**	**823**	**980**	**965**	**917**	**1,003**	**773**	**753**	**753**	**492**	**705**	**491**	**445**	**209**	**400**	**510**	**472**
Grand Total (Columbia River Basin)	**10,334**	**11,491**	**10,834**	**9,248**	**7,886**	**7,419**	**8,160**	**7,022**	**6,993**	**6,407**	**4,205**	**5,450**	**4,306**	**2,596**	**na**	**na**	**na**

[1] *Although Caspian terns were successfully dissuaded from establishing a colony on Rice Island in most years, roosting and loafing terns were resident on the island every year. Colony size on Rice Island in 2021–2023 was not estimated.*

[2] *Data provided by USDA-APHIS-WS (2023).*

Table 2. Size of California/ring-billed gull (LAXX), double-crested cormorant (DCCO), and American white pelican (AWPE) breeding colonies in the Columbia River basin in 2023. "Active" denotes a colony that was active but not counted in 2023.

Colony	LAXX [1]	DCCO [2]	AWPE [1]
Columbia River Estuary			
East Sand Is.	Active [3]	241 [4]	
Astoria-Megler Bridge		5,153	
Rice Island	Active [3]		
Channel Markers [5]		360	
Miller Sands Spit			1,272
Lower Columbia River			
Channel Markers [6]		132	
Longview Bridge		284	
Troutdale Transmission Towers		401	
Columbia Plateau Region			
The Dalles Transmission Towers (Columbia River)		46	
Miller Rocks (Columbia River)	3,951		
Blalock Is. (Columbia River)	2,005		
Crescent Is. (Columbia River)	3,892	199	
Badger Is. (Columbia River)	2,816		2,593
Foundation Is. (Columbia River)		380	
Island 20 (Columbia River)	6,488		
Hanford Reach (Columbia River)		109	
Okanogan River Delta (Columbia River)		62	
Goose Is. (Potholes Reservoir)	11,642		
North Potholes Reservoir (Potholes Reservoir)		65	
North Rocks, Shoal Is., and unnamed island (Lenore Lake)	1,423	165	
Harper Is. (Sprague Lake)	3,084	528	
Twinning Is. and Goose Is. (Banks Lake)	4,575		

[1] *Number of individuals.*

[2] *Number of breeding pairs.*

[3] *Includes glaucous-winged/western and a large number of ring-billed gulls on Rice Island. Data provided by the U.S. Army Corps of Engineers – Fisheries Field Unit (see Roberts et al. 2024).*

[4] *Data provided by the U.S. Army Corps of Engineers – Fisheries Field Unit. Reported as number of nests (see Blair et al. 2024)*

[5] *Channel Markers located from river kilometer 0 to river kilometer 53.*

[6] *Channel Markers located from river kilometer 127 to river kilometer 218.*

Columbia Plateau Region

Caspian Tern Colonies: In 2023, terns nested at five locations in the CPR, including Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir (both "managed colonies" as part of the IAPMP; USACE 2014), Badger Island (where adaptive management was implemented in 2023 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; *Table 1*). As in 2021 and 2022, adaptive management prevented terns from nesting at the Blalock Islands in John Day Reservoir by raising reservoir elevation to inundate all previously used nesting habitat in 2023 (USACE 2021, 2022, 2023).

Fewer terns were observed roosting and nesting along the shoreline and visible upland areas of Goose Island in Potholes Reservoir during 2023 compared with previous years (*Figure 1*). This was likely due to continued passive and active nest dissuasion efforts along the shoreline and higher reservoir water levels through early June (USDA-APHIS-WS 2023), which flooded most of the shoreline habitat where terns have previously nested. From 1 April to 14 June, reservoir water levels remained above 318 m (1,042 ft), like in 2020 and 2022 during this same period (USDA-APHIS-WS 2023). While shoreline nesting habitat was mostly unavailable during the 2023 breeding season, terns did nest on exposed shoreline habitat on the south spit of the main island. Terns also attempted to nest at one upland location on the main island. These habitats were mostly unvegetated areas that either lacked passive nest dissuasion (stakes, cable, and flagging) or where openings in passive nest dissuasion were observed. Terns were observed nesting in isolated groups in the same locations during multiple aerial surveys.

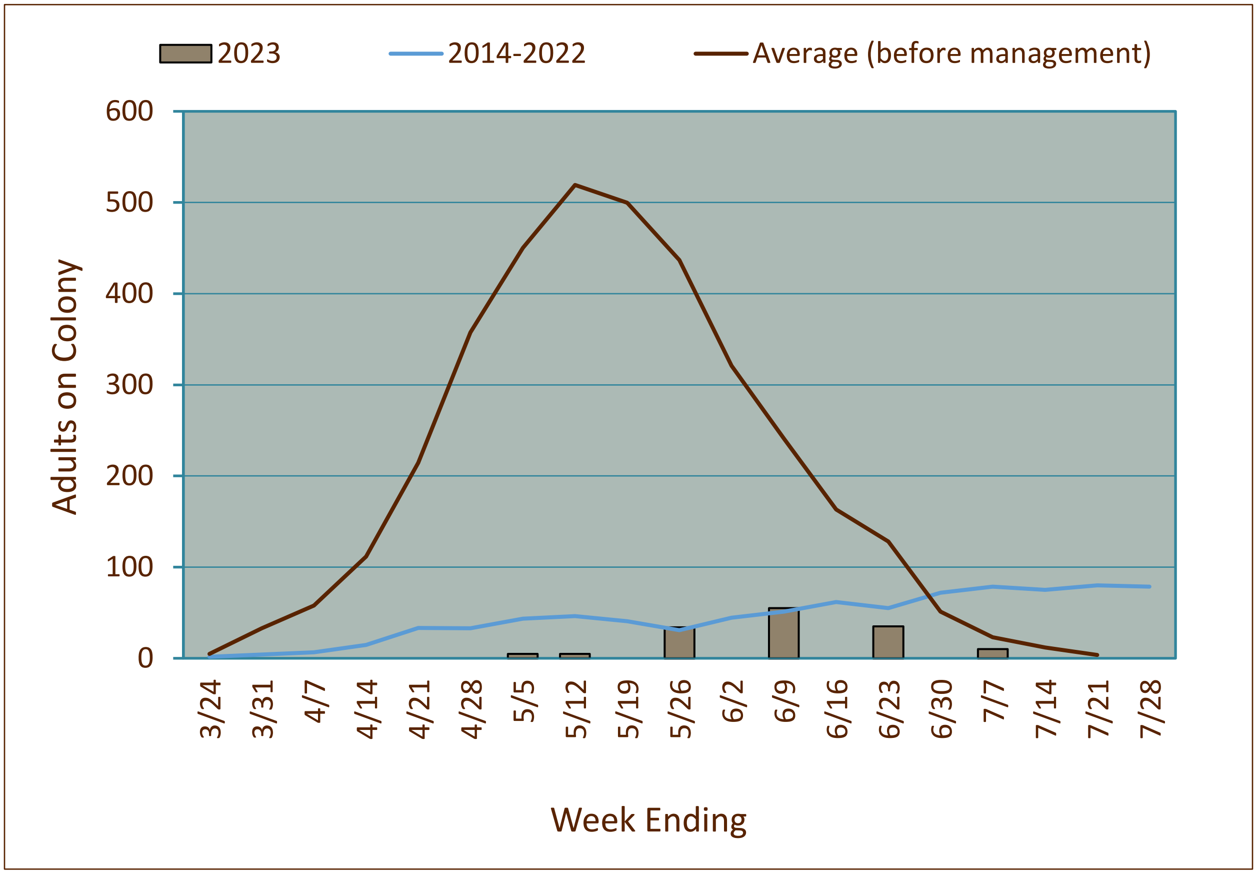


Figure 1. Weekly estimates of colony attendance by adult Caspian terns on Goose Island and surrounding islets in Potholes Reservoir. Counts are from boat-based surveys and show the average number of adults on colony throughout the breeding season prior to tern management (2010–2013), during tern management (2014–2022), and this past breeding season (2023). 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 fixed-wing surveys and are reported elsewhere (see below).

In 2023, passive and active nest dissuasion on Goose Island, along with the collection of tern eggs (n = 74, USDA-APHIS-WS 2023) under BOR's depredation permit, were effective in preventing terns from reaching the threshold colony size stipulated in the *IAPMP* (40 breeding pairs). We estimate the peak colony size at Goose Island in 2023 was 12 breeding pairs (*Figure 2*). This is based on a count of incubating terns from aerial photography collected in early June. Most of these nesting attempts occurred on the southern end of the island both along the shoreline and on the adjacent slope. No tern fledglings were observed on Goose Island by our crew in 2023, suggesting that the tern nesting attempts on Goose Island in 2023 were unsuccessful. After the final aerial survey in mid-June, additional tern nesting attempts (<12 breeding pairs) were observed and documented through late July by USDA-APHIS-WS (2023).

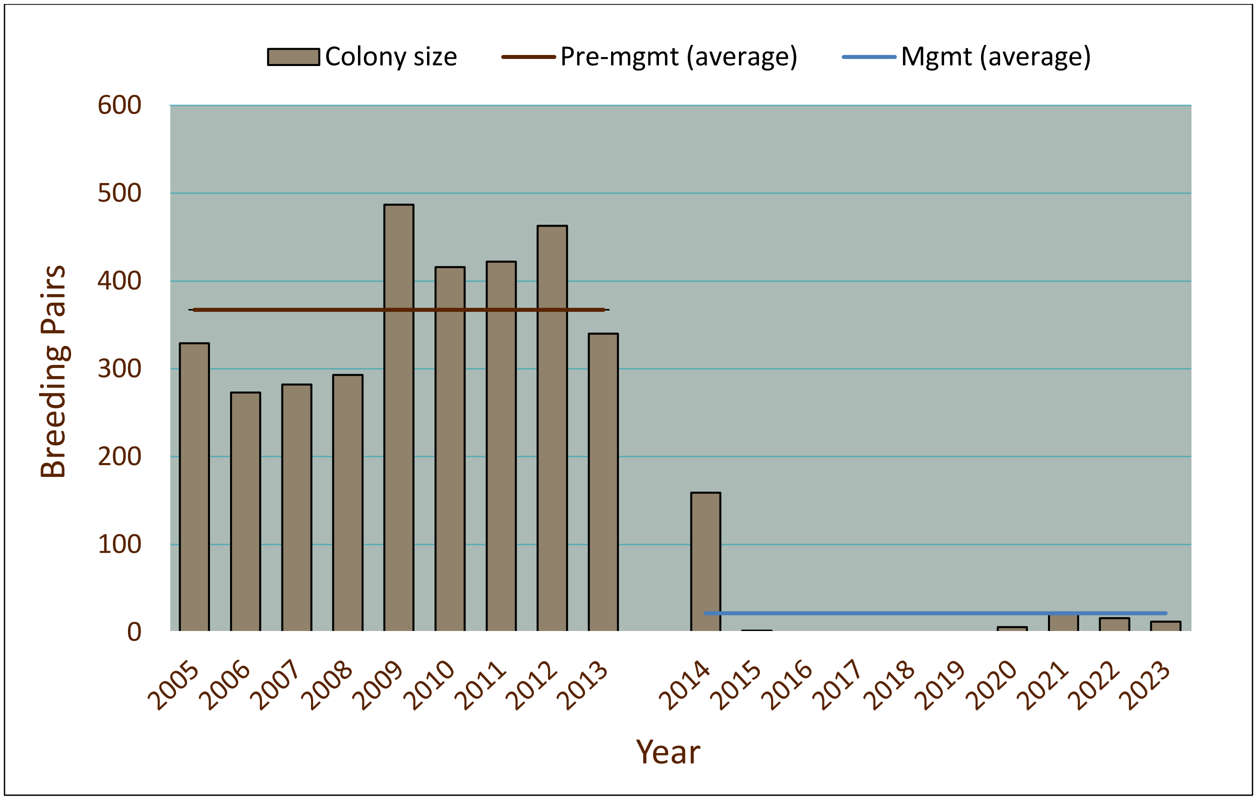


Figure 2. Annual colony size (number of breeding pairs) of Caspian terns at Goose Island and surrounding islets in Potholes Reservoir before tern management (Mgmt; 2005–2013) and during tern management (2014–2023). No Caspian tern nesting occurred at Goose Island or surrounding islets during 2016–2019.

Prospecting terns were documented at five different sites in northern Potholes Reservoir during the 2023 breeding season (USDA-APHIS-WS 2023), down from seven sites used by prospecting terns in 2022 (USDA-APHIS-WS 2022). Active hazing at all five sites and egg collection (under permit) at one site were successful in preventing tern colony formation at prospective colony sites (USDA-APHIS-WS 2023). While these results are encouraging, terns continued to show strong fidelity to Goose Island and other islands in Potholes Reservoir. This suggests that, without ongoing adaptive management, terns could reestablish colonies at potential colony sites in Potholes Reservoir.

An unexpectedly large colony of terns reformed on Crescent Island in McNary Reservoir in 2022 and 2023. The re-establishment of a tern colony on Crescent Island was facilitated by the recent reduction in vegetative cover on the island. The vegetation loss is likely 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.

In 2021, a single breeding pair of Caspian terns nested at the historical tern colony site on Crescent Island, the first successful tern nesting attempt on Crescent Island since 2014. In 2022 and 2023, the tern colony increased in size to 149 and 88 breeding pairs, respectively (*Figure 3,*

Table 1). 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 to pre-management conditions. Gulls may provide social attraction to nesting terns and also cause a die-back in willows and other native vegetation that dissuades terns from nesting there.

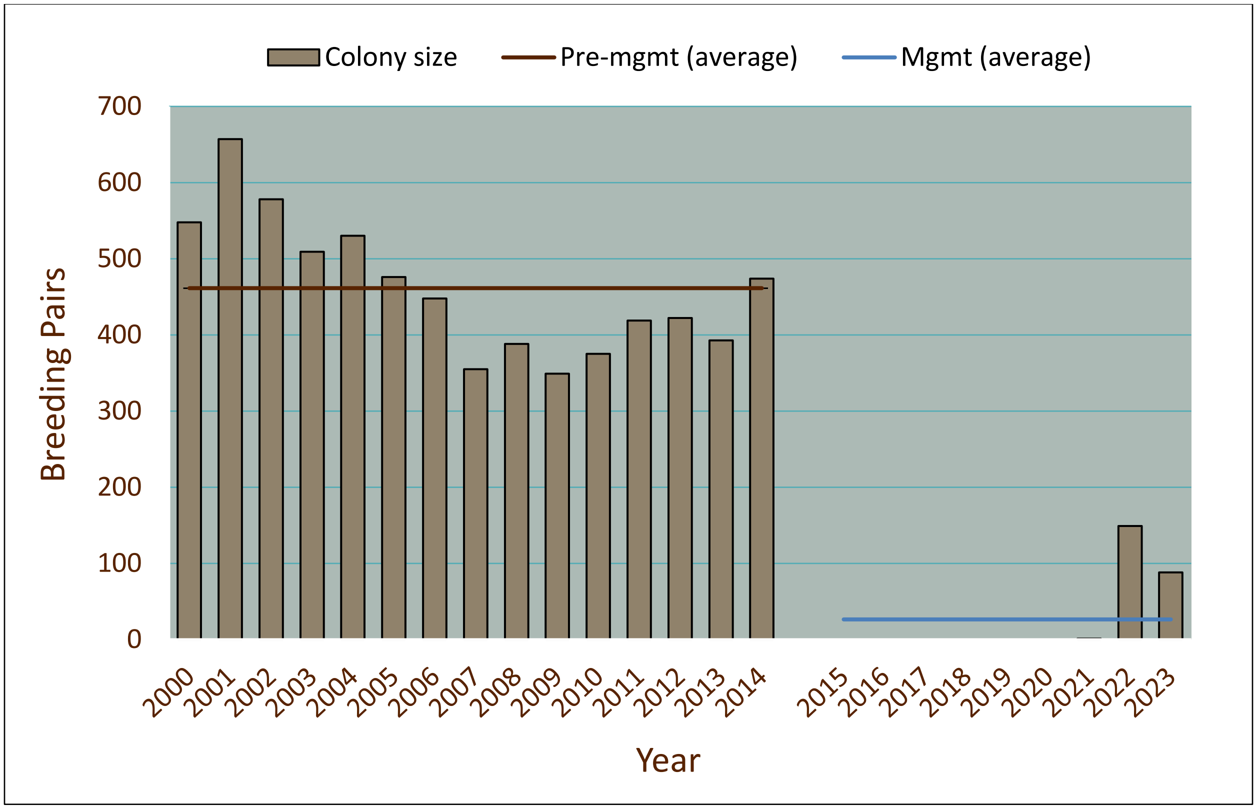


Figure 3. Annual colony size (number of breeding pairs) of Caspian terns at Crescent Island in the mid-Columbia River before tern management (2005–2014) and during tern management (2015–2023). No terns nested at Crescent Island during 2015–2020; one tern breeding pair successfully nested on Crescent 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 in 2015, most of the terns prevented from nesting at that site relocated to the nearby Blalock Islands (*Table 1*). During the 2021–2023 breeding 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, 2023). 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 was implemented to prevent/reduce tern nesting on Badger Island in 2022 and 2023 (i.e. placement of large woody debris on nesting areas used by terns in 2021), colony size has not decreased. While the placement of woody debris at an interior colony site on Badger Island was successful in preventing tern nesting, another colony area located along the northeastern shoreline expanded to ultimately support the largest tern colony in the CPR in 2022 and 2023 (267 and 274 breeding pairs, respectively; *Table 1*). Woody debris placement on the northeastern shoreline colony prior to the 2023 nesting season was not effective in dissuading terns from nesting at this location. A combined total of 362 breeding pairs of terns nested at Crescent and Badger Islands in 2023, representing 78% of the total breeding pairs in the CPR during that year.

A total of 472 breeding pairs of terns nested in the CPR in 2023 (*Table 1* and *Figure 4*), slightly lower than the number of terns that nested in the CPR in 2022 (510 breeding pairs), and more than double the maximum regional population size stipulated in the *IAPMP* (200 breeding pairs; USACE 2014). The regional population size in 2023 represented a 46% reduction in the size of the regional tern breeding population compared to the pre-management average (875 breeding pairs; *Figure 4*), but the population has increased over the past three years. 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 increased numbers of nesting terns in other regions of the Pacific Flyway (Peck-Richardson et al. 2019, Lawes et al. 2022).

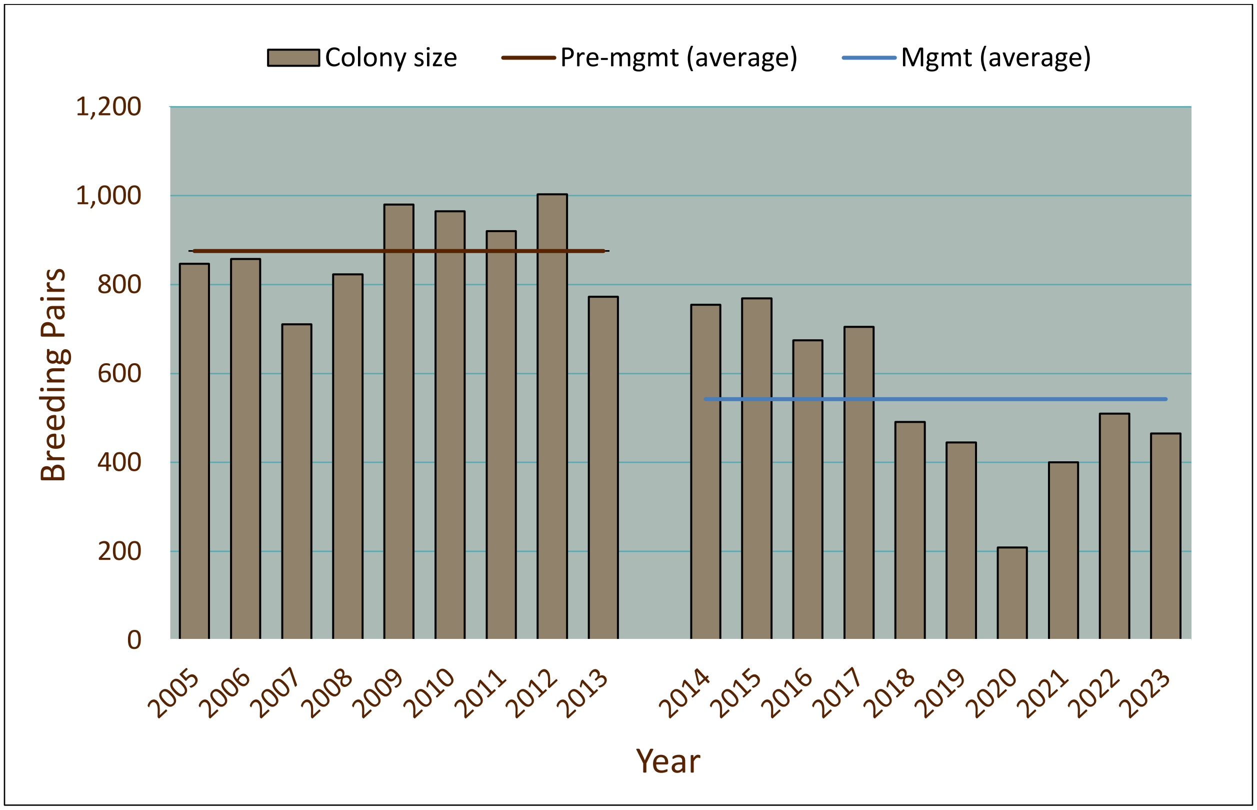


Figure 4. Annual and average number of Caspian tern breeding pairs nesting at all known colonies before tern management (2005–2013) and during tern management (2014–2023) in the Columbia Plateau region.

Other Piscivorous Waterbird Colonies: A total of nine active California and ring-billed gull colonies were detected in the CPR region in 2023, ranging in size from ca. 1,400 individuals (Lenore Lake) to nearly 12,000 individuals (Goose Island in Potholes Reservoir; *Table 2*). Nesting double-crested cormorants were confirmed at eight colonies in the CPR in 2023, with colony sizes ranging from 46 breeding pairs (The Dalles transmission towers) to 528 breeding pairs (Harper Island in Sprague Lake; *Table 2*). 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 ca. 2,600 individuals in 2023 (*Table 2*), a reduction in size relative to 2022 and 2021 (3,486 and 3,624 individuals, respectively; Evans et al. 2023).

Lower Columbia River

In 2023, two active double-crested cormorant nesting colonies were identified below Bonneville Dam on the Lower Columbia River: the Longview Bridge (also referred to as the Lewis and Clark Bridge; 284 breeding pairs) and the and transmission towers near Troutdale, Oregon (401 breeding pairs; *Table 2*). Additionally, channel markers between river kilometer 127 and 218 supported an additional 132 breeding pairs. All nesting sites increased in size compared to 2022 (Evans et al. 2023), which may coincide with the dispersal of cormorants away from ESI as more cormorants are nesting in the freshwater zone of the Lower Columbia River.

Columbia River Estuary

Caspian Tern Colonies: The USACE-FFU estimated that 524 breeding pairs of terns nested on the prepared 1-acre colony site on ESI in 2023 (Blair et al. 2024). This is the smallest colony size ever recorded on ESI (Roby et al. 2021b and *Table 1*). Efforts to prevent terns from nesting outside the designated 1-acre colony area were successful in 2023 using passive and active nest dissuasion and egg collection (under permit). However, the tern colony on the designated site on ESI once again failed to produce any young. Complete nesting failure has been observed at that colony in 6 out of the last 8 years. The factor(s) causing colony failure in recent years are not clearly understood because close monitoring of the colony ceased after the 2019 breeding 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. Recent colony failures at ESI may also be exacerbated by the ongoing threat of HPAI to the Pacific Flyway population, which negatively impacted tern colonies in the CRE in 2023. In late May, the designated tern colony area on ESI was mostly abandoned. During the same time period up to ca. 1,400 terns were counted on Rice Island, in the freshwater zone of the estuary (Roberts et al. 2024). When terns attempted to nest on Rice Island, active and passive dissuasion of the incipient tern colony was used to prevent colony formation. This adaptive management was eventually successful in dispersing the terns from the incipient colony site, but only after 2.5 weeks, which corresponded with the peak of the steelhead run in the CRE (see Roberts et al. 2024 and Blair et al. 2024 for further details). Hundreds of terns were also observed at Port of Astoria Piers 1, 2, and 3, and at a pier at Tongue Point, although no nesting was confirmed at these locations.

Double-crested Cormorant Colonies: There were several active double-crested cormorant colonies in the CRE in 2023, including ESI, AMB, and numerous channel markers located between river kilometers 0 and 53 (*Table 2*). In 2023, the USACE-FFU reported a peak colony size for cormorants on ESI of 241 breeding pairs with no fledged chicks (Blair et al. 2024). This marks the fifth year in a row when little to no cormorant nesting and no cormorant nesting success occurred at the colony (USACE-FFU 2020, 2021, 2022a; Blair et al. 2024). Management and other factors (e.g., colony disturbance and nest predation) at the ESI cormorant colony has resulted in the dispersal of cormorants away from ESI, with most nesting cormorants in the CRE now located in the mixing zone of the CRE on the AMB (*Table 2*). The cormorant colony on the AMB has increased dramatically from 333 breeding pairs in 2014 (year prior to management at the ESI cormorant colony) to 5,123 breeding pairs in 2023 (*Table 2*), the largest number observed to-date (Roby et al. 2021a). Without efforts to both restore the double-crested cormorant colony on ESI and dissuade cormorants from nesting on the bridge, we expect the AMB colony to persist, and perhaps grow, as there is additional available nesting habitat on the bridge. This is a concern to fisheries managers because the per capita smolt impacts of cormorants nesting on the AMB is 2–4 times higher than those of cormorants nesting on ESI (Cramer et al. 2021b; Evans et al. 2022b; Evans et al. 2023, and *below*).

Other Piscivorous Colonial Waterbird Colonies: Both glaucous-winged/western gulls and ring-billed gulls nested in the Columbia River estuary in 2023. The first confirmed nesting by ring-billed gulls in the Columbia River estuary was on ESI in 2004 (Roby et al. 2021a), where a colony persisted in each year up until this year. That colony was abandoned in April–May of 2023 and those birds presumably relocated to nest on Rice Island where a sizable colony was counted (>3,000 individuals; Blair et al. 2024). As in previous years, glaucous-winged/western gulls nested at dispersed colonies on ESI and Rice Island. Although estimates of colony size in 2023 are unavailable, in past years these colonies numbered in the 1,000s (Roby et al. 2021a).

There was one active American white pelican colony in the CRE in 2023 on Miller Sands Spit (*Table 2*). The Miller Sands Spit pelican colony is one of only two pelican colonies in the CRB; the other colony is on Badger Island in McNary Reservoir in the CPR (*Table 2*). The Miller Sands Spit pelican colony was established in 2010, when 42 individuals attempted to nest there, although it has relocated to nearby Rice Island in some years (Cramer et al. 2021a). In 2023, the colony on Miller Sands Spit was estimated at 1,272 individuals (*Table 2*).

In addition to double-crested cormorants nesting on the AMB, Brandt's cormorants (BRAC) and pelagic cormorants (*Phalacrocorax pelagicus*; PECO) also nested on the bridge in 2023. Peak counts for BRAC were 1,224 breeding pairs and for PECO were 116 breeding pairs, increases for both species compared with 2022 (Evans et al. 2023).

AVIAN PREDATION RATES

PIT-tagging of Upper Columbia River Steelhead

A total of 8,377 UCR steelhead were sampled and available for predation and survival analyses in 2023, which includes 7,237 newly tagged smolts and 1,140 recaptured smolts (i.e., previously tagged). Of these, 6,172 and 2,205 were classified as hatchery and wild, respectively (see also *Appendix B*). Steelhead were tagged and released from 12 April to 12 June, a period which accounted for > 98% of all steelhead encountered in the trap in 2023. Mean fork length was 197 mm (standard deviation [SD] = 21 mm; range = 82–300 mm). An evaluation of external fish condition indicated that most steelhead were in good over-all external condition in 2023, with 4.8% 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 classified as being in compromised conditions (Evans et al. 2014, Evans et al. 2023). The most common type of anomaly in 2023 was severe descaling, followed by body injuries and fin damage.

In 2023, in an attempt to more accurately classify the rear-type (hatchery, wild) of steelhead sampled at RIS, non-adipose clipped smolts with no signs of fin erosion or other marks/tags indicative of a hatchery environment – fish that met the visual classification criteria of a wild smolt (Evans et al. 2014) – were scanned for the presence of a coded wire tag (CWT), an indication that the fish was actually hatchery, not wild. In total, only three (3) of the 2,152 presumed wild steelhead sampled at RIS had a CWT. To ensure the CWT detector/antenna was properly functioning, adipose clipped steelhead (i.e. known hatchery fish) were periodically checked for the presence of a CWT during the study period and these fish were routinely found to have a detectible CWT (T. Jackson, Chelan PUD, pers. com.).

PIT Tag Recovery

A total of 25 genera-specific avian colonies in the CRB were scanned for fish PIT tags following the 2023 breeding season, a record number since scanning commenced in 1996 (Roby et al. 2021, Evans et al. 2023). Colonies included 6 tern colonies, 5 gull colonies, 8 cormorant colonies, 2 great blue heron (*Ardea herodias)* colonies, 1 great egret (*Ardea alba*) colony, 2 pelican colonies, and 1 Brandt's cormorant colony (*Table 3*). Scanning was also conducted at 13 avian loafing/roosting sites, areas where large numbers of piscivorous colonial waterbirds were observed during the smolt out-migration period. A total of 33,867 PIT tags from 2023 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead combined) were recovered in 2023 (*Table 3*). Of these, more than 90% were from tags recovered from breeding sites, with the largest number recovered on mixed gull and pelican colony sites on Badger Island (n = 8,720), followed by the tern colony on ESI (n = 3,221), and the cormorant colony on the AMB (n = 3,022). Scanning at AMB was limited to areas accessible to researchers following the breeding season, resulting in a subsample of all available nests (*Table 3*; see also *below* for additional details). Relatively large numbers of tags were also recovered from the gull colony at Miller Rocks (n = 2,419), the cormorant colony on the TRT (n = 1,747), and the incipient tern colony on Rice Island (n = 1,205; *Table 3*). A relatively small number of smolt PIT tags were recovered on the ESI cormorant colony (n = 50), the Miller Sands Spit pelican colony (n = 72), and the AMB Brandt's cormorant colony (n = 118; *Table 3).*

In addition to PIT tags from 2023 migration year smolts, a total of 93 tags from other salmonid species and age-classes were recovered on avian colonies in 2023. This included PIT tags from 46 adult sockeye salmon, 4 adult steelhead, 3 Pacific Lamprey (*Entosphenus tridentatus*), 1 adult jack Chinook salmon, 1 bull trout (*Salvelinus confluentus),* 1 white sturgeon *(Acipenser transmontanus),* and 1 Northern pikeminnow (*Ptychocheilus oregonensis*) from colonies in the CPR (*Table 3*). A total of 36 tags implanted in sea-run cutthroat trout (*O. clarkii clarkii*) were also recovered on avian colonies in the CRE (*Table 3*).

Finally, for the first time since the BPA-funded avian predation project commenced in 1996, we attempted to recover PIT tag from three heron colonies or rookeries in the CRE and LCR. The primary impetus for scanning these sites were to increase sample sizes of tagged fish known to have survived passage to Bonneville Dam for use in mark-recapture-recovery models (see *Smolt Survival to Bonneville Dam* section *below*) and to learn if appreciable numbers of smolts were being consumed by herons. In total, just 50 smolt PIT-tags were recovered from all three heron colonies combined in 2023 (*Table 3*).

Table 3. Numbers of 2023 migration year smolt PIT tags (Chinook salmon, sockeye salmon, coho salmon, and steelhead combined) and life stages and species 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), great egret (GREG), great blue heron (GBHE), Brandt's cormorants (BRAC), and American white pelicans (AWPE). Mixed-species sites represent 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 piscivorous birds.

Location	Rkm	Bird Species	Area Use	Smolt PIT Tags	Other PIT Tags[1]
Okanogan	Off-river	DCCO	Breeding	433	
Goose Island (Potholes R.)	Off-river	CATE	Breeding	111	
		Mixed	Loafing	16	
Lenore Lake	Off-river	CATE	Breeding	252	
Hanford Island	592	DCCO	Breeding	298	
		Mix	Loafing	91	
Island 20	549	LAXX	Breeding	738	
		Mix	Loafing	32	
Foundation Island	518	DCCO	Breeding	1,159	
		Mixed	Loafing	476	1
Badger Island	512	LAXX	Breeding	443	
		AWPE	Breeding	256	7
		CATE	Breeding	2,173	0
		Mixed	Breeding	8,720	45
		Mixed	Loafing	962	1
Burbank Slough	Off-river	Mixed	Loafing	194	
Crescent Island	510	CATE	Breeding	841	
		LAXX	Breeding	570	
		DCCO	Breeding	1,209	
		Mixed	Loafing	13	
Blalock Islands	441-439	LAXX	Breeding	983	
		Mixed	Loafing	92	
Miller Rocks	331	LAXX	Breeding	2,419	3
		Mixed	Loafing	77	
Troutdale Towers	189	DCCO	Breeding	1,747	1
		Mixed	Loafing	5	
Port of Woodland	137	GREG	Breeding	4	
Port of Longview	107	GBHE	Breeding	40	5
Miller Sands Spit	38	AWPE	Breeding	72	
		Mixed	Loafing	6	
Rice Island	34	CATE [2]	Breeding	1,205	10
Rice Island Channel Markers	33-50	DCCO	Breeding	452	
Astoria-Megler Bridge	23	DCCO [3]	Breeding	3,022	2
		BRAC [3]	Breeding	118	
		Mixed [3]	Breeding	308	
		Mixed [3]	Loafing	731	
Piers 1, 3, & 5	21-28	Mixed	Loafing	322	
Lois Island	30	GBHE	Breeding	6	
East Sand Island [4]	8	CATE	Breeding	3,221	18
		DCCO	Breeding	50	
Total				33,867	93

[1] Includes adult sockeye (n=46; Badger Island), adult steelhead (n=4; Badger Island), adult Chinook (n=1; Badger Island), white sturgeon (n=1 badger Island), pikeminnow (n=1; Badger Island); bull trout (n=1; Foundation Island), Pacific Lamprey (n=3; Miller Rocks), and sea-run cutthroat (n=36; East Sand Island, Astoria-Megler Bridge, Rice Island, Port of Longview, and Troutdale Towers).

[2] Some tags could have potentially been deposited by ring-billed gulls that also nested on Rice Island.

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

[4] Tags recovered by the USACE.

Smolt Survival to Bonneville Dam

In addition to providing information on predation, recoveries of smolt PIT tags on bird 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 2023, we expanded efforts to recover smolt PIT tags from additional 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 8,155 PIT tags from 2023 migration year smolts that survived outmigration to below Bonneville Dam prior to being consumed by a bird. This information, coupled with 2023 migration year smolt tags recovered by the USACE on the ESI tern and cormorant colonies (n = 3,271), tags in live fish detected at antennas on pile dikes in the CRE (n = 14,294), and tags of live fish detected by the National Marine Fisheries Service's net trawl in LCR (n = 11,100), were used as part of this study and other federally-funded projects to estimate smolt survival to Bonneville Dam in 2023.

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 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. 2015a, Evans et al. 2022c). Detection probabilities were directly measured in 2023 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. 2015a 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 (*Table 4*).

No information on deposition probabilities were available for the heron colonies scanned for smolt PIT tags in the CRE (*Table 3*) and, as such, predation rate estimate were not generated for these colonies in 2023. Due to the paucity of tags recovered (n = 50 from all three colonies combined), however, predation impacts were presumably very low, if not non-existent on most ESA-listed salmonid ESUs/DPSs in 2023.

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 2023. Estimates were used to generate predation rates based on the number of smolt PIT tags recovered following the breeding season (Table 3). 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	0.60–0.68	0.71 (0.51–0.89)
Goose Island	Off-river	CATE [2]	0.98	0.71 (0.51–0.89)
Hanford Island	592	DCCO	0.12–0.48	0.51 (0.34–0.70)
Island 20	549	LAXX	0.92–0.96	0.15 (0.11–0.21)
Foundation Island	549	DCCO	0.16–0.22	0.51 (0.34–0.70)
Badger Island	512	AWPE	0.84–0.94	0.47 (0.24–0.73)
		LAXX	0.84–0.96	0.15 (0.11–0.21)
		CATE [2]	0.80	0.71 (0.51–0.89)
Crescent Island	510	LAXX	0.64–0.76	0.15 (0.11–0.21)
		CATE [2]	0.72	0.71 (0.51–0.89)
		DCCO	0.66–0.99	0.51 (0.34–0.70)
Blalock Islands	441-439	LAXX	0.98–0.99	0.15 (0.11–0.21)
Miller Rocks	331	LAXX	0.76–0.96	0.15 (0.11–0.21)
Troutdale Towers	189	DCCO	0.08–0.72	0.51 (0.34–0.70)
Miller Sands Spit	38	AWPE [2]	0.86	0.47 (0.24–0.73)
Rice Island	34	CATE	0.73–0.94	0.71 (0.51–0.89)
Astoria-Megler Br.	23	DCCO	0.36–0.75	0.51 (0.34–0.70)
		BRAC [3]	0.12–0.99	0.51 (0.34–0.70)
East Sand Island	8	CATE	0.61–0.96	0.71 (0.51–0.89)
		DCCO [2]	0.85	0.51 (0.34–0.70)

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

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

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

Efficacy of Avian Predation Management Plans

Inland Avian Predation Management Plan (IAPMP): Estimates of tern predation rates on the ESA-listed salmonid ESUs/DPSs evaluated in 2023 were below the 2% threshold identified in IAPMP (*Table 5*). This represents the first year since management was initiated in 2014 that colony-specific estimates from all tern colonies in the CPR were below the target threshold. The highest estimates in 2023 were from Badger Island terns on Snake River steelhead at 1.9% (1.2–3.2%) and Badger Island terns on Upper Columbia River steelhead at 1.4% (0.9–2.5%; *Table 5*). Estimates on all other ESUs/DPSs were less than 1% per ESU/DPS, per colony (*Table 5*).

Table 5. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (i.e. sockeye, Chinook, and steelhead), 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 2023. 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%	1.0% (0.4–2.1)	0.5% (0.2–1.3)
SR Steelhead	< 0.1%	< 0.1%	1.9% (1.2–3.2)	0.7% (0.4–1.4)
UCR Steelhead	0.6% (0.3–1.2)	0.4% (0.2–0.8)	1.4% (0.9–2.5)	0.8% (0.4–1.6)

Decreases in colony-specific tern predation rates in 2023 compared with 2022 were observed at most colonies, with the largest reduction observed at the Crescent Island tern in McNary Reservoir. Terns, however, have just recently (2021) returned to Crescent Island following management actions that eliminated the colony during 2015–2020. The largest tern colony in CPR continues to reside on Badger Island in McNary Reservoir, where predation rates in 2021 and 2022 were above the 2% threshold and were near the threshold on Snake River steelhead at 1.9% (1.2–3.2%) in 2023 (*Table 5*). Predation rates by terns on Goose Island in Potholes Reservoir in 2023 (0.6% on Upper Columbia River [UCR] steelhead; *Table 5*) were lower than those observed in 2021 (ca. 3.9%) but higher than those in 2022 (0.1%; *Appendix A*).

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 (*Appendix A*), impacts that prompted adaptive management actions at this and other surrounding islands in northern Potholes Reservoir during 2017–2023. Since then, active and passive dissuasion techniques have been successful at preventing terns from nesting on islands in northern Potholes Reservoir, thereby reducing impacts from tern predation. Due to the relatively large number of terns that continue to prospect for nest sites on Goose Island (see *Figure 1*) and the re-establishment of the Goose Island tern colony in recent years (2020–2023), 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 (2014-2023 for Goose Island and North Potholes; 2015–2023 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 (2021–2023). 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 2014–2023 [1]	Last 3-years 2021–2023	Pre-Management 2007–2013	Management 2016 [2]	Last 3-years 2021–2023	Pre-Management 2007–2014	Management 2015–2023	Last 3-years 2021–2023 [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% (0.1–0.2)	0.2% (0.1–0.5)
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	NC	4.5% (4.1–5.1)	0.2% (0.1–0.3)	0.6% (0.4–0.9)
UCR Steelhead	15.7% (14.1–18.9)	1.0% (0.7–1.7)	1.5% (1.0–2.3)	NC	4.1% (2.9–6.3)	NC	2.5% (2.2–2.9)	0.1% (0.1–0.2)	0.4% (0.3–0.7)

[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 third consecutive year, terns nested on Crescent Island in McNary Reservoir (*Table 1*). In 2021, just one pair of terns nested, with 149 and 88 pairs in 2022 and 2023, respectively. Predation rates in 2023 were less than 1.0% per ESU/DPS, with the highest estimates observed on SR steelhead (0.7%) and UCR steelhead (0.8%; *Table 5*). 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 smolts 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 (*Table 6*). 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%; *Table 6*). 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 (*Table 6*; 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 2022–2023, 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 in 2023 were less than 1% on all ESU/DPS evaluated, with the highest estimated observed on UCR steelhead at 0.4% (0.2–0.8%; *Table 5*). In 2022, estimates on UCR were above the target threshold at 2.1% (1.4–3.4%; *Appendix A*). Variation in UCR steelhead predation rates does not appear to be closely related to variation in colony size. For instance, in 2022 the colony was roughly the same size at 76 pairs as it was in 2023 at 81 pairs. An estimated 123 pairs 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 are related to predation effects by terns nesting in Lenore Lake. Results suggest that future monitoring and possible management of the tern colony in Lenore Lake may be necessary to achieve the goals of IAPMP.

There was no evidence of sustained colony of terns in Banks Lake, WA (*Table 1 above*), so PIT tag recovery was not conducted in 2023. The last time a sustained 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 (e.g., Goose Island) in Banks Lake to detect tern nesting and to estimate the impacts of nesting terns on smolt survival is warranted.

There was a small tern colony on Harper Island in Sprague Lake, WA in 2023, with just 10 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 pairs were documented (Evans et al. 2022b). Because the island is privately owned and we have not been able to contact the landowner, we were not able to scan the island for PIT tags in 2023. 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 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 to the small size of the colony in 2012 (30 breeding pairs; Collis et al. 2021b). 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 2023, ranged from 0.2% (0.1–0.4%) on SR spring/summer Chinook to 1.9% (2.0–4.4%) on SR steelhead (*Table 5*). Estimates were 1.4% (0.9–2.5%) on UCR steelhead (*Table 5*). Estimates of predation rates were consistent with what might be expected based on a colony size of 274 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 (*Table 1* and *Appendix A*). Since the Badger Island tern colony became re-established in 2021, average annual predation rates on SR steelhead have been 2.1% (1.6–2.8%; *Table 7*). Adaptive management efforts (e.g., placement of woody debris or other passive dissuasion techniques) on the Badger Island tern colony will likely be necessary in the future to achieve the goals of the IAPMP. Badger Island, however, is large (> 7 ha), with numerous interior and shoreline areas where terns could potentially nest, factors that could make it challenging to prevent nesting in the future. Finally, American white pelicans (a species of conservation concern in the State of Washington) also nest on Badger Island, which has hampered tern nest dissuasion activities at that site in previous years.

For the third consecutive year since adaptive management was implemented (2021–2023), 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 (*Table 1*). 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 (*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 this 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. 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). As such, increases in predation rates on smolts by terns nesting on the Blalock Islands had initially offset the benefits achieved by the elimination of the tern colonies on Crescent and Goose islands due to management. Since implementation of adaptive management actions to inundate (flood) the Blalock Islands, predation rates have been greatly reduced (if not eliminated) during 2021–2023. 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.

In summary, predation rates by terns nesting at colonies in the CPR in 2023 were amongst the lowest observed since management actions associated with the IAPMP commenced in 2014, with the target goal of predation rates less than 2% per ESU/DPS, per colony, achieved for the first time. Throughout the course of the management period (2014–2023), 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). Adaptive management actions that raised water levels in the John Day Reservoir successfully prevented tern nesting on the Blalock Islands in all years (2021–2023). Some proportion of terns that formerly nested at the Blalock Islands, however, have presumably re-nested at both Badger Island and Crescent Island in recent years. Thus, continued adaptive management at these tern nesting sites will likely be needed in 2024 and beyond to achieve the goals and objectives of the IAPMP.

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 (2014–2023). 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 (2021–2023). 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 2021–2023	Pre-Management 2007–2013 [1]	Management 2014-2023[2]	Last 3-years 2021-2023[2]	Pre-Management 2007–2013	Management 2014–2020[3]	Last 3-years 2021–2023
SR Sockeye	< 0.1%	0.1% (0.0–0.5)	NC/NA	NC/NA	NA	NA	0.2% (0.1–0.4)	1.6% (1.0–2.5)	NC
SR Sp/Su Chinook	< 0.1%	< 0.1%	NC/NA	NC/NA	0.1% (0.1–0.3)	0.4% (0.2–0.9)	0.1% (0.1–0.2)	0.6% (0.5–0.8)	NC
UCR Sp Chinook	< 0.1%	0.2% (0.0–0.7)	NC/NA	NC/NA	< 0.1%	< 0.1%	< 0.1%	0.6% (0.4–0.8)	NC
SR Fall Chinook	< 0.1%	< 0.1%	NC/NA	NC/NA	0.2% (0.1–0.3)	0.3% (0.6–1.0)	< 0.1%	0.7% (0.5–0.9)	NC
SR Steelhead	< 0.1%	< 0.1%	NC/NA	NC/NA	0.7% (0.5–0.9)	2.1% (1.6–2.8)	0.5% (0.4–0.9)	3.5% (2.9–4.3)	NC
UCR Steelhead	0.1%	1.1% (0.8–1.6)	NC/NA	NC/NA	0.4% (0.3-0.5)	1.2% (0.9–1.6)	0.5% (0.3–0.7)	4.0% (3.2–5.0)	NC

	Lenore Lake Islands			Harper Island		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2023	Last 3-years 2021–2023	Pre-Management 2012 [4]	Management 2014–2023	Last 3-years 2020–2023
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	0.2% (0.1–0.4)	0.2% (0.1–0.9)	< 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	0.7% (0.6–0.9)	1.1% (0.8–1.4)	< 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 2021–2023.*

[3] *No established tern nesting colony was present in the Blalock islands during 2021–2023.*

[4] *Small colony existed in several other years but was not scanned for smolt PIT tags due to lack of landowner permission.*

Estuary Caspian Tern & Double-crested Cormorant Management Plans: Salmonid ESU/DPS-specific estimates of predation by terns nesting on ESI in 2023 were not available (estimates generated with funding from USACE in years past), although smolt PIT tags were recovered on ESI by the USACE-FFU following the 2023 breeding season (*Table 3*). Although ESU/DPS-specific predation rates by terns on ESI in 2023 were not available, ESU/DPS-specific predation rates were generated by terns that attempted, but ultimately failed, to nest on Rice Island in 2023 as part of this study. Results of predation analyses indicated that predation rates ranged from 0.1% (<0.1–0.3%) on SR Fall Chinook to 1.1% (0.8–1.8%) on UCR steelhead (*Table 8*). Terns also attempted but ultimately failed to successfully nest on Rice Island in 2022, where predation rates were as high 2.9% (1.6–5.1%) on Middle Columbia River steelhead (Evans et al. 2023; see also *Appendix A*). 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 as part of estuary tern management plan are not offset by increased tern predation rates at other sites in the CRE (see also Collis et al. 2024 for a more detailed discussion of the efficacy of tern management 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 2023. 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 2023 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	0.6% (0.3–1.2)	14.3% (6.3–24.4)	NA	NA
SR Sp/Su Chinook	0.1% (0.1–0.5)	7.1% (4.1–11.3)	NA	NA
UCR Sp Chinook	0.2% (0.1–0.8)	8.2%(4.2–13.8)	NA	NA
SR Fall Chinook	0.1% (<0.1–0.3)	3.8% (1.1–7.9)	NA	NA
SR Steelhead	0.8% (0.5–1.2)	10.9% (6.2–17.4)	NA	NA
UCR Steelhead	1.1% (0.8–1.8)	10.4% (5.1–17.7)	NA	NA
MCR Steelhead	0.9% (0.5–1.8)	6.8% (1.5–14.4)	NA	NA

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

For the first year since 2020, there was some evidence of sustained, late-season (August - October) breeding by cormorants on ESI in 2023 (241 pairs; Blair et al. 2024). Like 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. It should be noted, however, that due to late formation of the colony and the small number of smolt PIT tags recovered (n = 50) following the breeding season (*Table 3*), predation rates were presumably low (e.g., < 0.1% per salmonid ESU/DPS). In 2022, when cormorants also attempted to nest, just 99 smolts PIT tag were recovered by USACE following the breeding season, again suggesting that predation rates by cormorants on ESI have been low in recent years (Evans et al. 2023).

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 fish, 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). Starting in 2021, to estimate predation rates on salmonid smolts by cormorants breeding on the AMB, 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 PIT tags could be recovered by researchers following the breeding season (see also Evans et al. 2022b). Estimates of per capita (per breeding pair) predation rates derived from cormorants nesting on the North Crib footing (n=817 pairs in 2023) were then extrapolated to account for all cormorants nesting on the bridge (n=5,153 pairs in 2023) to generate colony-wide estimates of predation in 2023. Results of predation analyses indicated that per capita predation rates of cormorants nesting on the AMB ranged from 0.0007% (0.0002–0.0015%) on SR Fall Chinook to 0.0027% (0.0012–0.0047%) on SR sockeye salmon. Colony-wide estimates of predation, those extrapolated to all birds, ranged from 3.8% (1.1–7.9%) on SR Fall Chinook to 14.3% (6.3–24.4%) on SR sockeye (*Table 8*). 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 (*Table 8*). For example, estimates of colony-wide predation rates on SR steelhead were 10.9% (6.2–17.4%) of available fish, an estimate that was similar to and not significantly different than that observed UCR steelhead at 10.4% (5.1–17.7%) and UCR spring Chinook at 8.2% (4.2–11.3%; *Table 8*). Estimates of predation, however, were often imprecise, in part due to the process of extrapolating estimates from a subsample of breeding pairs on the North Crib footing to all breeding pairs on the AMB (see *Methods & Analysis* section for details).

Estimates of colony-wide predation rates on salmonid smolts by cormorants nesting on the AMB in 2023 were the highest observed since research started on the bridge in 2021 (Evans et al. 2022a). For instance, predation rates on Snake River sockeye were 6.6% (1.7–14.7%) in 2022 but were 14.3% (6.3–24.4%) in 2023. Increases in predation rates in 2023 compared with years past were observed in all ESA-listed ESU/DPSs evaluated (*Appendix A*). Increases in predation in 2023 were presumably associated, in part, with the larger size of the cormorant colony on the AMB, with the colony increasing from 4,054 in 2022 to 5,153 pairs in 2023 (*Table 1 above*). Estimates of per capita predation by cormorants on AMB in 2021–2023 were approximately 2–5 times greater (depending on salmonid 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 Evans et al. 2023). For example, per capita predation rates on steelhead 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 at 0.0021% (0.0012 – 0.0034%). As such, despite the much smaller size of the double-crested cormorant colony on the AMB (4,453 breeding pairs on average during 2021–2023) 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 greater than those of cormorants on ESI. For instance, average annual colony-wide 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 to 6.1% (4.3–9.1%) and 9.2% (6.4–13.9%), respectively, by cormorants on the AMB during 2022–2023 (*Table 8* and *Appendix A*). Collectively, results indicate that colony location is a key factor associated with cormorant predation on salmonid smolts, and that currently, predation rates by cormorants nesting in the CRE are higher than the rates observed prior to management when cormorants nested on ESI (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 size of PIT-tagged smolts tagged and released from Lower Columbia River (LCR) ESUs/DPSs in recent years (since 2020), estimates of predation effects by cormorants breeding on the AMB are lacking, but impacts 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 *Appendix A*.

Finally, cormorants nested on channel markers (492 pairs), the Longview Bridge (284 pairs), and Transmission towers (TRT; 401 pairs) in the CRE and LCR in 2023 (*Map 1 and Table 2*), all increased compared with counts from 2022 (Evans et al. 2023). Smolt PIT tags were recovered from channel markers in CRE and the TRT colony but not from Longview Bridge due to unsafe sampling conditions. Results from channel markers in CRE and the TRT are provided *below*.

Predation Rates by Other Piscivorous Colonial Waterbirds

Predation/consumption rates of juvenile salmonids by California and ring-billed gulls, double-crested cormorants, and American white pelicans nesting at some colonies in the CPR were similar to or greater than those of Caspian terns nesting at colonies in the CPR, with estimates upwards of 10.0% of available smolts observed in some of the ESUs/DPSs evaluated in 2023. Similar to years past, 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). Terns, cormorants, and pelicans are strictly piscivorous and are believed to rarely consume dead fish in the wild (Evans et al. 2022a). As such, we refer to gull estimates of fish loss as “consumption rates” and losses to terns, cormorants, and pelicans as “predation rates” (see also Evans et al. 2022a).

ESU/DPS-specific predation/consumption rates by gulls, cormorants, and pelicans nesting at select colonies – those scanned for smolt PIT tags (*Table 5*) – in 2023 are provided *below*.

California and Ring-billed Gulls - Island 20: Consumption rates of smolts by gulls nesting on Island 20 ranged from 0.2% (0.1–0.5%) for SR Spring/Summer Chinook to 2.8% (1.8–4.6%) for UCR steelhead (*Table 9*). Similar to years past, 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). The years with the highest estimates of consumption of steelhead 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). Over-all, estimates of consumption by Island 20 gulls on smolts in 2023 were similar to or slightly lower (depending on the ESU/DPS) than those observed in years past (Appendix A).

California and Ring-billed Gulls - Badger Island: Consumption rates on salmonid smolts for gulls nesting at Badger Island ranged from 0.6% (0.1–3.2%) for SR Fall Chinook to 5.8% (2.1–9.9%) for UCR steelhead. Consumption rates by gulls nesting at Badger Island were also high for SR steelhead with an estimated 2.6% (1.0–4.9%) of available fish consumed in 2023 (*Table 9*). Estimates of consumption in 2023 were similar to those in 2022 and most years dating back to 2015, when the gull colony become established on Badger Island. 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 (*Table 9* and *Appendix A*). Unlike the gull colony at Island 20, however, terns were co-nesting with gulls on Badger Island, so it's possible that some of the smolt PIT tags recovered on the Badger Island gull colony in 2023 were from smolts initially captured by terns and subsequently kleptoparasitized by gulls.

California and Ring-billed Gulls - Crescent Island: Consumption rates on salmonid smolts for gulls nesting on Crescent Island, a colony that reformed in 2020 following management action to dissuade terns, ranged from 0.2% (0.1–0.%) for SR Spring/Summer Chinook to 1.8% (1.2–4.3%) for UCR steelhead in 2023 (*Table 9*). Estimates of consumption were 1.3% (0.6–2.5%) on SR steelhead and 1.0% (0.2–3.7%) for SR Fall Chinook (*Table 9*). Consumption estimates by gulls on Crescent Island in 2023 were similar to those observed in 2021 and 2022 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 3,681 adults counted during 2021–2023, 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 2021–2023 due to the smaller size of the tern colony on Crescent in these years (*Table 1 above*).

California and Ring-billed Gulls - Blalock Islands: In several years dating back to 2012 there were 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 and 2023, a gull colony did not form on Straight Six Island, but gulls did nest on Anvil Island in both years. In 2023, consumption rate estimates from Anvil Island ranged from 0.4% (0.1–1.2%) for SR Sp/Su Chinook to 3.6% (2.8–10.2%) for SR steelhead (*Table 9*). Similar to Island 20, Badger Island, and the Crecent Island gull colonies, consumption estimates by Blalock Island gulls were highest on steelhead DPSs compared with salmon ESUs. Estimates of consumption rates for gulls nesting at the Blalock Islands in 2023 were also similar to 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*).

California and Ring-billed Gulls - Miller Rocks: Of the five individual gull colonies included in the study, estimates of smolt consumption rates by gulls breeding on Miller Rocks in The Dalles Reservoir were consistently the highest, with estimates ranging from 2.0% (1.0–3.5%) for SR spring/summer Chinook to 9.9% (4.9–18.7%) for UCR steelhead (*Table 9*). An estimated 4.7% (2.4–8.9%) of SR Fall Chinook were also consumed by Miller Rocks gulls, the highest estimate by a gull colony on a salmon ESU in 2023 (*Table 9*). 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;* see also *Cumulative Predation and Survival* section *below*).

Estimates of smolt consumption rates by gulls nesting at Miller Rocks in 2023 were slightly higher than those observed in 2022 and similar to those in several other years dating back to 2007 (*Appendix A*). In 2022 and 2023, the Confederated Tribes and Bands of the Yakama Nation used human hazing and falconry to dissuade gulls from breeding on Miller Rocks during early portion of the breeding season (April; B. Sharp, Yakama Nation, personal communication). Although efforts were successful at preventing colony formation while dissuasion occurred, a colony eventually formed. There was evidence that peak size of the gull colony on Miller Rocks was smaller in 2023 (3,951 adults) compared with 2022 (5,832 adults). The average annual peak count was 4,466 adult gulls on Miller Rocks during 2007–2021 (Cramer et al. 2021a, Evans et al. 2022b), suggesting that management actions during 2022–2023 have not dramatically changed the size of the colony or predation rates to-date.

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. Addressing to what degree consumption rates by gulls and other piscivorous waterbirds reduces smolt survival during outmigration is paramount to understanding to what degree avian management will benefit smolt and smolt-to-adult survival rates in the CRB (see *Additive Effects of Predation* section for a detailed analysis).

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 2023. 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	Island 20	Badger Is.	Crescent Is.	Blalock Is.	Miller Rocks Is.
SR Sockeye	NA	NA	NA	NA	NA
SR Sp/Su Chinook	0.2% (0.1–0.5)	0.9% (0.3–1.7)	0.2% (0.1–0.6)	0.4% (0.1–1.7)	2.0% (1.0–3.5)
UCR Sp Chinook	NA	NA	NA	1.0% (0.1–4.9)	2.6% (1.1–5.4)
SR Fall Chinook	0.7% (0.1–2.7)	0.6% (0.1–3.2)	1.0% (0.2–3.7)	0.5% (0.1–1.8)	4.7% (2.4–8.9)
SR Steelhead	0.7% (0.3–1.3)	2.6% (1.0–4.9)	1.3% (0.6–2.5)	3.6% (1.6–7.4)	6.7% (3.5–12.4)
UCR Steelhead	2.8% (1.8–4.6)	5.8% (2.1–9.9)	1.8% (1.0–3.4)	3.2% (1.0–8.2)	9.9% (4.9–18.7)

Double-crested Cormorants - Hanford Island: The cormorant colony on Hanford Island in Hanford Reach on the middle Columbia River has periodically been scanned for smolt PIT tags since nesting cormorants were first documented in 2007 (Cramer et al. 2021a), including scanning in both 2022 and 2023. Estimates of predation rates on the ESA-listed ESUs/DPSs in both years were low, with less than 0.6% of available fish consumed per ESU/DPS in 2023 (*Table 10*) and 2022 (*Appendix A*), with highest estimates observed on UCR steelhead in both years. The smaller size of this cormorant colony (109 pairs; *Table 2*) is presumably responsible, in part, for the low predation rates on ESA-listed smolts. 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) and 61% (182 out of 298 recovered tags) of the 2022 and 2023, respectively, migration year smolt tags recovered. 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 and 2023.

Double-crested Cormorants - Foundation Island: Estimates of predation rates for cormorants nesting at the colony on Foundation Island ranged from 0.3% (0.1–0.9%) on UCR steelhead to 3.1% (1.6–6.4%) on SR steelhead in 2023 (*Table 10*). Predation rates on SR Spring/Summer Chinook were also substantial at 2.1% (1.1–4.6%) of available tagged smolts. Estimates of predation on UCR Spring Chinook in 2023 were not available due to inadequate sample sizes of tagged smolts for analysis but estimate in years with adequate sample sizes were consistently less than 0.5% of available smolts (Cramer et al. 2021a). Predation rates by Foundation Island cormorants on UCR smolts were substantially lower than those of SR smolts in 2023 (*Table 10*) and years past (*Appendix A*). 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 on SR smolts have been similar to or greater than those of nearby tern colonies (e.g., Crescent Island and the Blalock Islands), including predation effects prior to implantation of management actions. 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 cormorant colonies (see also Cramer et al. 2021a).

Double-crested Cormorants – Crescent Island: For the first time since the island was created with dredged spoil materials in the 1980's, a larger-sized colony (199 breeding pairs) of cormorants was located on Crescent Island in 2023 (*Table 1*). Cormorants had periodically been documented nesting on Crescent Island in years past, but less than 50 pairs were observed in all other years (Cramer et al. 2021a, Evans et al. 2023). Despite a relatively larger-sized cormorant colony, estimates of predation rates in 2023 were low, ranging from 0.1% (<0.1–0.6%) for SR Fall Chinook to 0.4% (0.2–0.8%) for SR Spring/Summer Chinook (*Table 10*). Predation and per capita predation rates by Crescent Island cormorants, particularly on SR smolts, may be lower than those of cormorants breeding on Foundation Island due to the closer proximity of Foundation Island to the Snake River confluence. At its current size, results suggested that Crescent Island cormorant colony posed a much lower risk to smolt survival than the Foundation Island cormorant colony, but results were limited to only one year of data and several of the arboreal nests on Crescent Island were located over water, areas where deposited tags may have been lost to environment.

Double-crested Cormorants - Troutdale Transmission Towers: For the second consecutive year since the colony site was discovered in 2012, smolt PIT tags were recovered from a cormorant colony located on transmission towers (TRT) near the town of Troutdale, OR, approximately 45 Rkm downstream of Bonneville Dam. Smolt PIT tags were recovered in the area underneath all five transmission towers with nesting cormorants on them. Predation rate estimates, which were based on the number of smolts last detected alive passing Booneville Dam (see *Methods & Analysis* section for details), ranged from 0.5% (0.1–1.9%) for SR Fall Chinook to 3.7% (1.3–12.6%) for Middle Columbia River steelhead (*Table 10*). Analogous to predation by cormorants nesting on the AMB colony site, predation rates by cormorants breeding at the TRT were more similar across steelhead and salmon ESUs/DPSs (*Table 10*), indicating similar levels of susceptibility across salmonid ESUs/DPSs, especially compared with the higher susceptibility of steelhead smolts to tern predation and gull consumption (see also Hostetter et al. 2023 for a more detailed discussion).

Double-crested Cormorants - Channel Markers: Cormorants nesting on channel markers located in the CRE (Rkm 33–50) were scanned for smolt PIT tags in 2023 for just the second time over the course of the last decade (Cramer et al. 2021b). The primary purpose of scanning in 2023 was to increase sample sizes of tagged fish known to have survived passage to Bonneville Dam for use in mark-recapture-recovery models by this and other BPA funded projects (see also *Smolt Survival to Bonneville Dam* section). In total, 452 smolt tags were recovered from channel markers in upper CRE (*Table 3*). Due to a lack of information on PIT-tag detection probabilities on channel markers, however, estimates of predation rates could not be generated. The salmonid species composition of tags recovered from cormorant nests on channel markers (46% Chinook, 41% steelhead, 6% coho, 7% sockeye) was similar to the composition of tags recovered on the nearby AMB cormorant colony (54% Chinook, 35% steelhead, 6% coho, 5% sockeye), suggesting that the foraging behavior of cormorants nesting on channel markers was similar to those of cormorants nesting on the AMB. As such, it's reasonable to assume that per capita predation rates by cormorants nesting at channel markers were similar to cormorants nesting on the AMB, although empirical data to support or refute this assumption is lacking.

With this caveat in mind, based on per capita predation rates observed by cormorants nesting on the AMB in 2023, cormorants nesting on nearby channel markers could have potentially consumed 0.4% of SR Fall Chinook to 1.4% of SR sockeye. Estimates in other groups of PIT-tagged smolts were 0.7% and 1.0% of SR Spring/Summer Chinook and SR steelhead, respectively. Research to quantify detection probabilities of PIT tags deposited by cormorants on channel markers will be necessary to generate unbiased and accurate estimates of predation rates in the future.

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 Hanford Island in the middle Columbia River, Foundation Island and Crescent Island in McNary Reservoir, and the Troutdale Transmission Towers in the lower Columbia River during 2023. 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	Hanford Island	Foundation Island	Crescent Island	Troutdale Towers
SR Sockeye	NA	NA	NA	2.1% (0.8–8.1)
SR Sp/Su Chinook	< 0.1%	2.1% (1.1–4.6)	0.4% (0.2–0.8)	2.9% (1.2–8.4)
UCR Sp Chinook	NA	NA	NA	1.5% (0.6–4.6)
SR Fall Chinook	< 0.1%	1.3% (0.2–4.9)	0.1% (<0.1–0.6)	0.5% (0.1–1.9)
SR Steelhead	< 0.1%	3.1% (1.6–6.4)	0.4% (0.2–0.8)	3.1% (1.3–9.4)
UCR Steelhead	0.5% (0.1–1.9)	0.3% (0.1–0.9)	0.2% (0.1–0.4)	2.5% (1.1–6.7)
MCR Steelhead	-	-	-	3.7% (1.3–12.6)

Brandt's Cormorants - Astoria-Megler Bridge: The same methods used for double-cormorants on the AMB were used for Brandt's cormorants (BRAC) nesting on the AMB, whereby smolt PIT tags were recovered from a concrete footing where a known number of BRAC nested (215 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 in 2023 (1,224 pairs). The AMB nesting habitat near the concrete footings, which was near the WA shoreline of the AMB, were complex (e.g., cormorants of different species nested on different vertical levels of the bridge in and around each footing). Although we limited PIT-tag recovery efforts to footings where only BRAC nested, a small number of recovered tags may have been deposited by double-crested cormorants nesting on beams above the footings, tags that could bias BRAC predation estimates high to an unknown degree. It should also be noted that there were small numbers of smolt PIT tags recovered from BRAC nests in 2023 (n = 118; *Table 3*) which resulted in imprecise estimates of per capita predation rates. With these caveats in mind, results indicated that even with some double-crested cormorants likely contributing tags to the scanned areas 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 2023, with per capita predation rates ranging from 0.0002% (0.0001–0.0006%) on UCR Spring Chinook to 0.0010% (0.0003–0.0036%) on SR sockeye. Colony-wide predation rates ranged from 0.1% (< 0.1–0.8%) in UCR Spring Chinook to

0.9% (0.5–4.3%) in SR sockeye (*Table 12*). Results suggest that despite a relatively large colony on BRAC on AMB in 2023 (1,224 pairs), they posed little threat to smolt survival in the CRE.

Although estimate of predation rates by BRAC nesting on the AMB were low, especially compared with double-crested cormorants on the AMB, estimates in 2023 were slightly higher than those in 2022, the first year the BRAC colony on the AMB was scanned for smolt PIT tags (Evans et al. 2023). Similar to increases observed in predation rates by double-crested cormorants on the AMB from 2022 to 2023, increases in predation rates by BRAC may be due to, in part, to increases in the size of BRAC colony (935 pairs in 2022 compared with 1,224 pairs in 2023). Changes in forage fish availability (e.g., fewer alternative prey) along with other biotic and abiotic factors could also explain increases predation and per capita predation by BRAC in the CRE in 2023 compared with 2022 (see Cramer et al. 2021b for a more detailed discussion of BRAC foraging behavior and smolt predation impacts in CRE).

Table 11. 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 Brandt's cormorants nesting on Astoria-Megler Bridge in the Columbia River Estuary during 2023. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods).

ESU/DPS	Predation Rate
SR Sockeye	0.9% (0.5–4.3)
SR Sp/Su Chinook	0.3% (0.1–0.7)
UCR Sp Chinook	0.1% (<0.1–0.8)
SR Fall Chinook	0.7% (0.3–2.1)
SR Steelhead	0.4% (0.1–1.2)
UCR Steelhead	0.5% (0.2–1.6)
MCR Steelhead	0.5% (0.1–2.4)

American White 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 0.1% (<0.1–0.3%) for SR Spring/Summer Chinook to 0.7% (0.2–2.7%) for UCR steelhead (*Table 12*). Results from 2023 where very similar to those in 2020, 2021, and 2022 (*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 non-ESA-listed sub-yearling Chinook are migrating (Stinson 2016; Payton et al. 2023). For instance, Payton et al. (2023) estimated that predation rates on sub-yearling Chinook from the Upriver Bright stock in the Hanford Reach of the middle Columbia River by pelicans nesting on Badger Island were substantial, with upwards of 25% of wild smolts consumed by pelicans in some years. Wild sub-yearling Chinook rear and reside in the middle Columbia River prior to outmigration, behavior that is believed to increase their susceptibility to pelicans breeding on nearby Badger Island (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 and studies to estimate predation rates on Yakima River origin smolts are currently ongoing (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 2023, 46 tags from adult sockeye that were tagged at the Bonneville Dam adult fishway during May–July were recovered on the Badger Island pelican colony (*Table 3*), amongst the largest number to-date (Cramer et al. 2021a). Studies are currently ongoing to quantify predation rates and consumption (number of adults consumed) on adult sockeye by Badger Island pelicans. Preliminary results indicate that predation rates and consumption are highly variable depending on the year and the size of the adult sockeye run, with predation rates ranging from ~ 1% to 8% and consumption from ~ 8,000 to 38,000 adult sockeye annually during 2014–2022 (authors, unpublished data). In addition to adult sockeye salmon, tags from other fish species have also been recovered on Badger Island, including white sturgeon, bull trout, pikeminnow, 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.

American White Pelicans - Miller Sands Spit: Estimates of predation rates by pelicans breeding on Miller Sands Spit in the CRE were the amongst the lowest of all colonies evaluated in CRB in 2023 (*Table 13*) and in years past (*Appendix A*). Despite an estimated colony size of 1,272 individuals in 2023 (the largest number counted since the colony formed in 2010; Roby et al. 2021a), only 72 tags from juvenile salmonids were recovered on the colony following the breeding season (*Table 3*), resulting in estimates of predation rates of < 0.1% per salmonid ESU/DPS (*Table 13*). Estimates of predation rates by pelicans on Miller Sands Spit in 2023 were very similar to those in 2021 and 2022, with predation rates of < 0.1% per salmonid ESU/DPS (Evans et al. 2023). Similar to the 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 72 smolt PIT tags recovered on the Miller Sands Spit pelican colony, 32 or 44% 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, were too small to generate reliable estimates of pelican predation. 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.

Table 12. 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 2023. 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	0.1% (<0.1–0.3)	< 0.1%
UCR Sp Chinook	NA	< 0.1%
SR Fall Chinook	0.2% (0.1–1.1)	< 0.1%
SR Steelhead	0.6% (0.1–1.6)	< 0.1%
UCR Steelhead	0.7% (0.2–2.7)	< 0.1%
MCR Steelhead	-	< 0.1%

Cumulative Predation & Survival

Based on recoveries of smolt tags from the bird colonies included in this study, avian predation effects were first observed in the river reach located downstream of RIS and downstream of Lower Monumental Dam (LMN). As such, for SR smolts, results indicating there was no measurable consumption of tagged fish between Lower Granite Dam (LGR) and LMN associated with the bird colonies evaluated in this study. For UCR smolts, however, predation occurred immediately following release/detection at RIS. Estimates of avian predation/consumption and total mortality (1 - survival) of smolts downstream of RIS and LMN varied considerably by bird species, colony location, and river reach, and are provided *below* by salmonid species and age-class (yearling, sub-yearling; Chinook only). Results from 2023 are then compared to results in years past to identify trends in predation and survival across river reaches and years dating back to 2008.

Upper Columbia River Steelhead: The cumulative effects of avian predation/consumption (predation by all avian predator species and colonies evaluated in the study combined) on UCR steelhead smolts during passage from RIS to the Pacific Ocean were substantial in 2023, with an estimated 28.6% (24.0–34.2%) of smolts consumed (*Figure 5*). Of the avian predator species evaluated, estimated consumption by gull colonies was the greatest at 17.1% (13.1–21.6%), followed by tern predation at 5.2% (4.2–6.4%), cormorant predation at 5.1% (3.1–9.2%), and pelican predation at 0.5% (0.1–2.1%). It is important to note that cumulative estimates are based on smolt availability at RIS or LMN (for SR migrants) 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 2023 were very similar to those in 2022, but lower than those observed in 2021, and lower than those observed in several, but not all, years dating back to 2008 (*Figure 5*). 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 2016–2020, 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 estimates were minimum estimates of predation on UCR and SR ESUs/DPSs in several, but not all, years past (as noted in figure descriptions, e.g., *Figure 5*).

Comparisons of avian predation/consumption to total mortality, which were available during smolt passage from RIS to Bonneville Dam (BON) or LMN to BON (for SR migrants) but not downstream of BON due to lack of smolt survival estimates to the CRE (see *Methods & Analysis*), indicated that avian predation accounted 54.1% (37.7–65.4%) of all UCR steelhead mortality sources during smolt passage to BON in 2023. Avian predation/consumption has accounting for > 50% of all UCR steelhead smolt mortality sources during outmigration from RIS to BON in 13 of the last 16 years (*Figure 6*; 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 has been the dominate or single greatest source of mortality for UCR steelhead smolts over the past 16 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) 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)

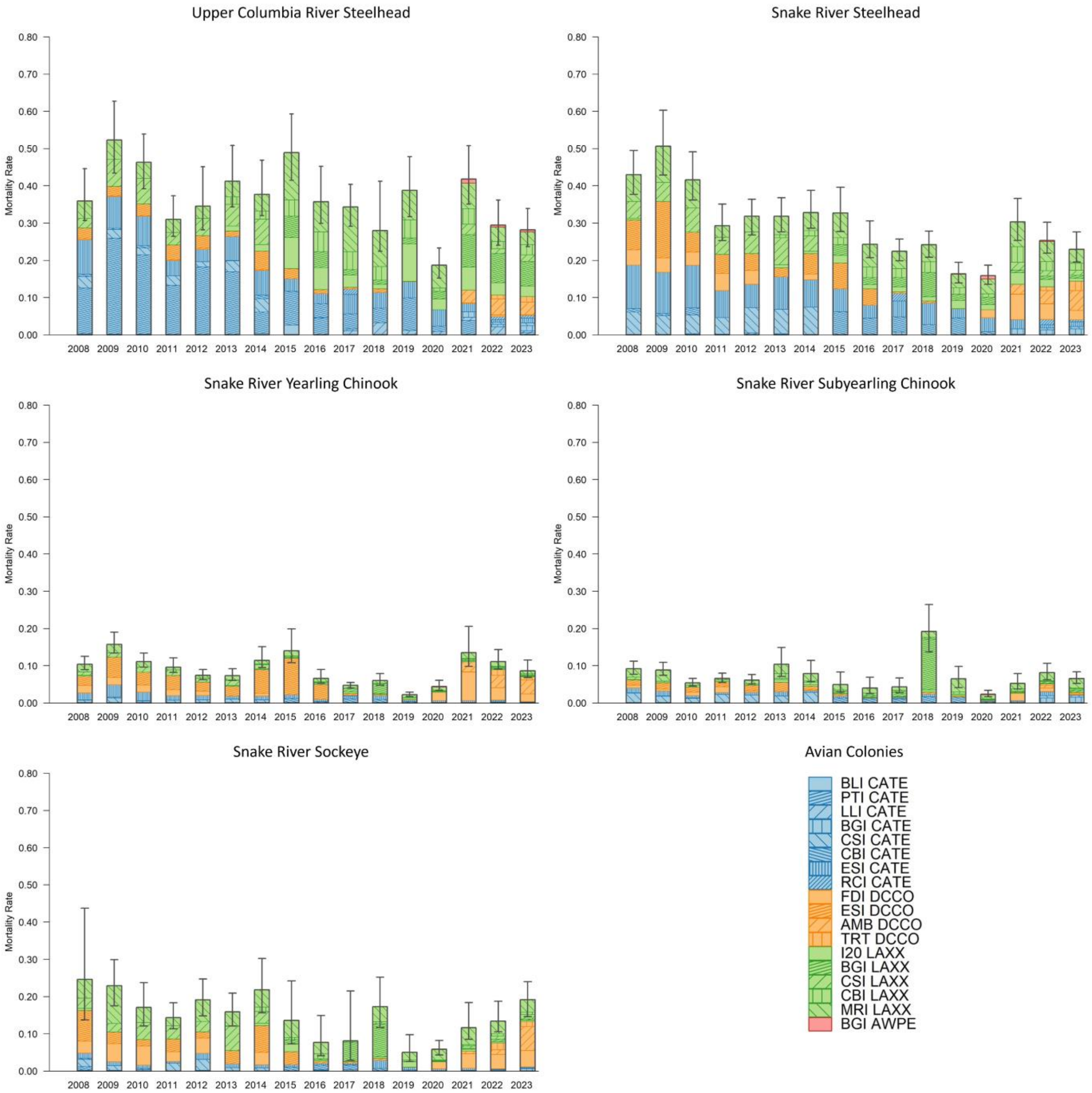


Figure 5. Estimated cumulative predation/consumption for Upper Columbia River steelhead smolts during passage from Rock Island Dam to the Pacific Ocean and for Snake River steelhead, yearling Chinook, subyearling Chinook, and sockeye smolts during passage from Lower Monumental Dam to the Pacific Ocean. 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).

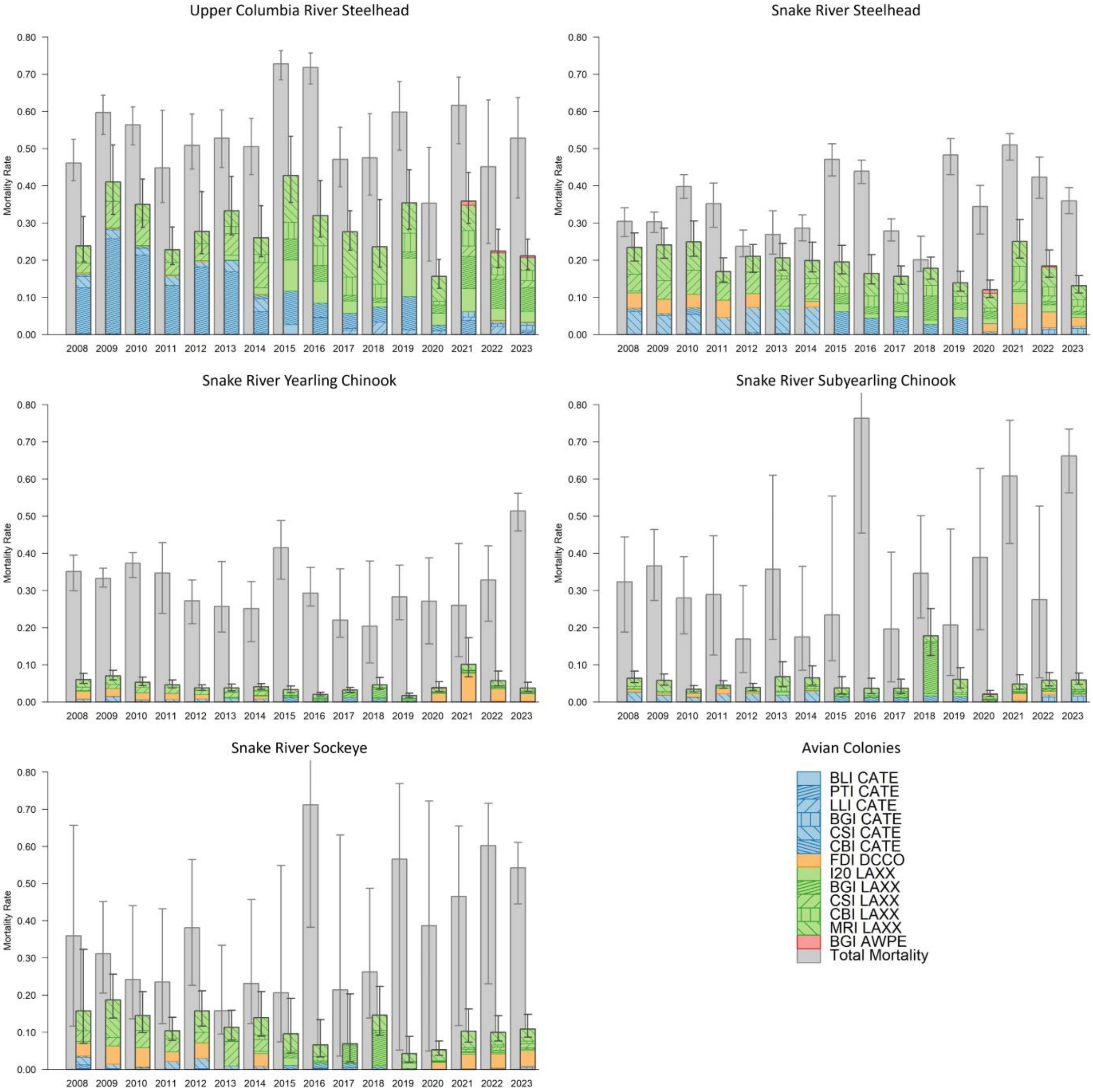


Figure 6. Estimated total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Upper Columbia River steelhead during passage from Rock Island Dam to Bonneville Dam or Snake River steelhead, yearling Chinook, sub-yearling Chinook, and sockeye during passage 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 2023 were similar to those of UCR steelhead, with an estimated 23.2% (19.7–27.9%) of SR steelhead smolts consumed (*Figure 5*). Of the avian predator species evaluated, estimated consumption by cormorant colonies was the greatest at 10.5% (7.4–14.8%), followed by gulls at 8.4% (7.0–10.1%), terns at 4.0% (3.3–4.9%), and pelicans at 0.2% (<0.1–0.6%; *Figure 5*). Predator species-specific aggregate predation effects in SR steelhead were similar, but lower, to those of UCR steelhead. One exception was 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 migrants compared with those originating from the UCR (see also *Foundation Island Cormorants* section). The cumulative effects of avian predation/consumptions on SR steelhead smolts in 2023 was slightly lower than that observed in 2022 and lower than those in several, but not all, previous years (*Figure 5*).

Estimated total mortality (1 - survival) of SR steelhead during passage from LMN to BON was 35.9% (32.5–39.5%) in 2023. Estimates of total mortality in 2023 were relatively low compared to most, but not all, years past (*Figure 6*). Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 45.5% (40.7–55.5%) of all SR smolt mortality sources during passage from LMN to BON in 2023 (*Figure 6*). Analogous to results on UCR steelhead, avian predation/consumption accounting for > 50% of all SR steelhead smolt mortality sources in 11 of the last 16 years (*Figure 6*; Evans et al. 2022a), with avian predation/consumption the dominate source of mortality during smolt passage to Bonneville Dam.

Snake River Yearling Chinook: Similar to data in previous years, the cumulative effects of avian predation/consumption on SR yearling Chinook during passage from LMN to the Pacific Ocean in 2023 were significantly lower than those of SR and UCR steelhead, with estimated 8.7% (6.9–11.7%) of smolts consumed (*Figure 5*). Large numbers of PIT-tagged yearling Chinook were detected at LGR (n = 77,662) which resulted in precise estimates of both predation and survival downstream of LGR. Of the predator species evaluated, estimated predation/consumption was the highest by cormorant colonies at 6.6% (4.9–9.5%; *Figure 5*). Predation/consumption by gulls at 1.5% (1.2–1.9%), terns at 0.4% (0.3–0.6%), and pelicans at 0.1% (< 0.1–0.2%) was significantly lower than that of cormorants (*Figure 5)*. Although the cumulative effects of avian predation/consumptions on SR yearling Chinook smolts in 2023 was very similar to estimates in previous years, 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.

Survival of SR yearling Chinook from LGR to LMN was high (99.2%) but then dropped significantly during passage from LMN to BON in 2023 (48.6%; *Figure 6*). Comparisons of avian predation/consumption to total mortality (1 - survival) indicated that avian predation/consumption accounted 7.2% (5.0–11.4%) of all SR yearling Chinook smolt mortality sources during passage from LMN to BON in 2023 (*Figure 6*). Results are consistent with those in the past and 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 of steelhead, albeit avian predation was a substantial source of SR yearling Chinook smolt mortality in some river reaches and years, particularly cormorant predation in the CRE (see *Reach-specific Predation and Survival* section *below* and Evans et al. 2022b for a more detailed discussion).

Snake River Sub-yearling Chinook: An estimated 8.2% (6.2–10.6%) of SR sub-yearling Chinook smolts were consumed by the avian predator species and colonies included in the study during smolt passage from LMN to the Pacific Ocean in 2023 (*Figure 5*). Similar to SR yearling Chinook, of the avian predator species evaluated, estimated predation/consumption on SR sub-yearling Chinook were the greatest by cormorant colonies at 3.8% (1.1–6.2%), following by gulls at 3.6% (2.7–4.9%), terns at 2.2% (1.6–3.1%) and then pelicans at 0.2% (0.1–0.8%; *Figure 5*). The cumulative effects of avian predation/consumptions on SR sub-yearling Chinook smolts in 2023 were similar to those observed in 2022 and previous years (*Figure 5*; Evans et al. 2022a).

Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 14.4% (8.8–22.7%) of all SR sub-yearling Chinook smolt mortality sources during passage from LMN to BON in 2023 (*Figure 6*). Estimates of predation and survival in 2023 were relatively precise compared with estimates in several, but not all, years past due to fluctuations in sample sizes of PIT-tagged smolts, with as few as 2,546 tagged smolts to as many as 34,742 tagged smolt available for use in predation analyses in years past (Evans et al. 2022a). It is 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. Finally, results from 2023 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 (*Figure 6*; 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 19.4% (14.8–24.3%) of smolts in 2023 (*Figure 5*). Of the avian predator species evaluated, estimated consumption was the greatest by cormorant colonies at 12.4% (8.1–16.9%), followed by gulls at 5.7% (4.5–7.5%). Predation estimates on sockeye were significantly lower by tern and pelican colonies at just 1.1% (0.8–1.5%) and 0.1% (<0.1–0.8%), respectively (*Figure 5*). The cumulative effects of avian predation/consumptions on SR sockeye in 2023 were amongst the highest observed since system-wide studies of predation commenced in 2008 (range = 5% to 24% annually; *Figure 5*). Despite the fact that cumulative estimates were based on availability at LMN, predation was the highest by cormorant colonies in the CRE, over 500 Rkm downstream (see *Reach-specific Predation and Survival* section *below* for estimates of predation after accounting for survival to each river reach).

Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 20.0% (14.2–33.2%) of all SR sockeye smolt mortality sources during passage from

LMN to BON in 2022 (*Figure 6*). Similar to results from SR sub-yearling Chinook, fluctuations in small samples size of PIT-tagged SR sockeye resulted in imprecise estimate of predation and, especially, survival in some but not all previous years (*Figure 6*). 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). Again, however, the highest reach-specific estimates of predation on SR sockeye were downstream of BON in the CRE, a segment of river where estimates of total smolt mortality were unavailable (see *Methods & Analysis* section for details).

Collectively, results of this study indicate that the cumulative effects of avian predation/consumption on salmonid smolts and the proportion of all sources of smolt mortality that were associated with 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 CRB 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. For instance, 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 all upstream hydroelectric dams, predation from piscivorous fish, predation by predator species and colonies that were not included in the study, mortality from disease, and all other mortality factors combined. Our results over the course of last 16 years of research provide strong evidence that avian predation/consumption is 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: Reach-specific predation/consumption effects, those that account for the survival of smolts to each downstream dam with PIT tag detection capabilities, indicated that colonial waterbirds were consuming smolts in all river-reaches downstream of RIS and LMN but that predation/consumption varied by reach, salmonid species, and bird colony. Results indicated that predation effects for all five salmonids species/age-classes included in the study were the greatest during smolt passage from BON to the Pacific Ocean, the furthest downstream reach evaluated. By salmonid species, predation rates in this reach were the highest for SR sockeye at 17.9% (11.1–28.1%), followed by SR steelhead at 15.5% (11.4–22.1%), UCR steelhead at 15.2% (9.6–25.1%), SR yearling Chinook at

10.2% (7.2–15.3%), and SR sub-yearling Chinook at 2.3% (1.0–4.1%) in 2023 (*Figure 7*). 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 through the hydrosystem, the benefits of managing avian predators downstream of BON may be greater on a per fish basis than managing inland avian predators (Roby et al. 2002, Payton et al. 2020).

Predation/consumption effects, particularly for UCR steelhead, were also substantial during smolt passage through the middle Columbia and lower Snake rivers. For instance, an estimated 14.3% (10.8–18.6%) of UCR steelhead were consumed during passage from RIS to McNary Dam (MCN) in 2023 (*Figure 7*). Predation/consumption of SR steelhead and SR sockeye was also substantial in this river reach at 7.5% (6.1–9.8%) and 6.2% (4.4–10.0%), respectively, in 2023 (*Figure 7*). Predation/consumption on smolts during passage from MCN to John Day Dam (JDA) and JDA to BON was almost entirely due to gulls in 2023, with similar levels of predation/consumption observed within each of these two river reaches. Unlike trends in other river reaches, however, predation was the highest on sub-yearling Chinook. 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) 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).

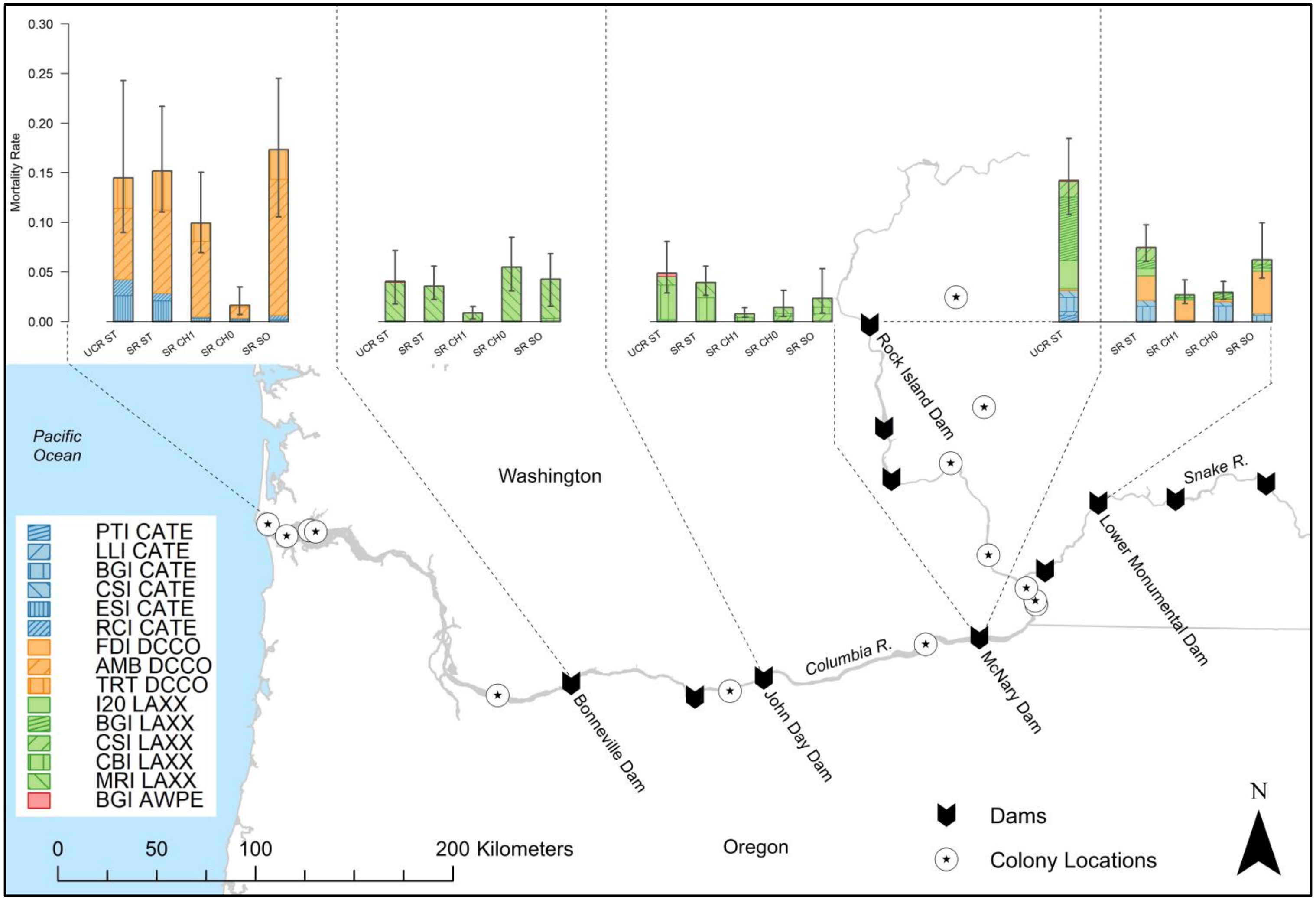


Figure 7. Reach-specific predation/consumption for Upper Columbia River (UCR) steelhead smolts during passage from Rock Island Dam to the Pacific Ocean and for Snake River (SR) steelhead, yearling Chinook (CH1), sub-yearling Chinook (CH0), and sockeye (SO) during passage from Lower Monumental Dam to the Pacific Ocean in 2023. 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.

In 2020 and 2023, record low estimates of cumulative avian predation/consumption rates on UCR steelhead coincided with record high estimates of smolt survival from RIS to MCN (*Figure 8*). Reductions in predation/consumption rates in these years were associated with the complete (in 2020) or near complete (in 2023) elimination of the Goose Island tern colony in Potholes Reservoir and Crescent Island tern colony 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 and 2023. 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 8*). On average, over the course of the entire management period (2014–2023), however, tern predation rates on UCR steelhead smolts have significantly declined following management actions at Goose and Crescent islands (*Figure 8*). For instance, on average, tern predation rates on UCR steelhead have decreased from 19.6% (16.5–24.1%) to 3.4% (2.7-4.4%) following management while survival has increased from 59.8% (56.6-63.3%) to 70.4% (63.8-76.2%; *Figure 8*). Our results provide strong evidence that management efforts reduce tern predation rates have increased smolt survival rates within the area where management actions have occurred, but that if tern predation rates increase, smolt survival will decrease, perhaps to levels observed prior to management.

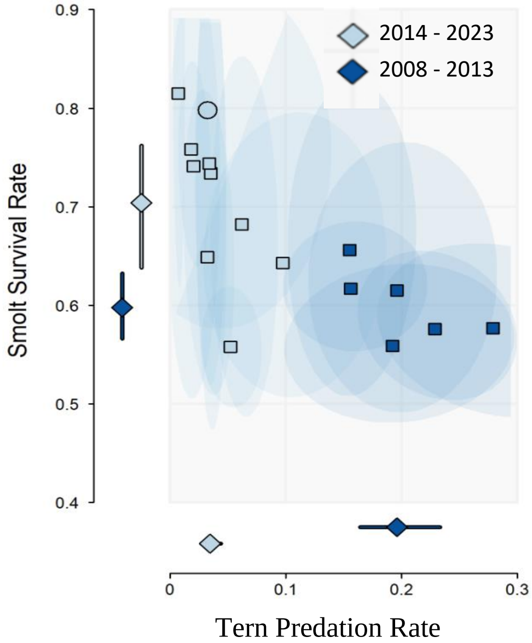


Figure 8. Caspian tern predation rates and survival rates of Upper Columbia River steelhead smolts during passage from Rock Island Dam to McNary Dam during (dark blue) and following (light blue) management actions that reduced the size of tern colonies at Goose Island and Crescent Island. Median annual 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 2023.

In summary, the reach-specific effects of avian predation/consumption on the survival of smolts, particularly UCR steelhead, SR steelhead, and SR sockeye, were substantial and continued to be one of the greatest sources of smolt mortality during outmigration in 2023. Avian predation/consumption rates by colonies residing downstream of BON were consistently the highest, primarily due to predation by cormorants breeding on the AMB in 2023. Upstream of BON, however, reductions in cumulative avian predation/consumption rates, in particular tern predation rates, were coincident with increases in UCR steelhead smolt survival during the management period (2014–2023) as a whole (*Figure 8 above*). Despite evidence that UCR salmonids have benefited from management actions associated with tern management, there is less 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. This is due, in part, to the total number of terns foraging on SR smolts being similar in all years since 2008 (*Table 2*), 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.

Consumption rates by gulls nesting at colonies in the CPR were often greater than those of nearby tern and cormorant colonies. Because gulls are known to consume dead, moribund, and injured smolts and are known to kleptoparasitize smolts that have been depredated by other piscivorous waterbirds, like terns, the actual impact of gull consumption on smolt survival is largely unknown (Evans et al. 2022a). Recent data, however, suggests that at least some of the smolts consumed by gulls would have survived outmigration to Bonneville Dam in the absence of gull consumption upstream of Bonneville Dam (Evans et al. 2023). For instance, higher levels of gull consumption of both UCR and SR steelhead smolts during passage from McNary Dam to BON were, on average, associated with lower levels of steelhead survival in this river reach (Evans et al. 2023). The relationship was statistically significant for both steelhead DPSs when data from all years were considered (Evans et al. 2023).

In the CRE, there is evidence that predation by tern colonies has substantially 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 (e.g., Rice Island) 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, on average, greater than those observed prior to implementation of management actions on ESI due to the rapid expansion of cormorant colonies on the AMB and TRT. There are also cormorants nesting on channel markers and on the Longview or Lewis and Clark Bridge in the CRE, where predation estimates were not available, but individuals nesting here maybe consuming appreciable numbers of juvenile salmonids based on results from the AMB and TRT colonies in 2022 and 2023.

Finally, estimates of reach-specific predation and survival were also generated on a weekly basis, estimates that were used to investigate the additive effects of predation on smolt survival (see *Additive Effects of Predation* section *below*). Weekly estimates of predation and survival for UCR and SR smolts in 2023 are provided in *Appendix D*.

Additive Effects of Predation

Results indicated that increases in tern predation on steelhead smolts during passage from RRJ to BON were associated with statistically significant decreases in smolt survival (*Table 13* and *Figure 8*). The value of a, ostensibly representing the proportion of depredated fish that would have survived in the absence of tern predation, was estimated to be 1.28 (0.70–2.08). Estimates from RRJ smolts were similar to, and not significantly different from, estimates of a in steelhead smolts tagged downstream at RIS, which were estimated to be 1.59 (1.06–1.99; *Table 13)*. As such, these results from RRJ steelhead smolts were consistent with findings of previously published data from RIS steelhead smolts, providing additional evidence that significantly more steelhead smolts would, on average, survive outmigration from the middle Columbia River to lower Columbia River in the absence of tern predation upstream of BON (Payton et al. 2020, Payton et al. 2022).

Table 13 Average annual predation and survival probabilities of steelhead smolts during outmigration from Rocky Reach Dam (RRJ) and Rock Island Dam (RIS) to Bonneville Dam during 2010–2023. Estimates of the magnitude of the association between predation probabilities and survival probabilities ($\boldsymbol{a}$, additivity) and the difference in survival probabilities from estimated baseline survival probabilities ($\boldsymbol{\Phi}^{\Delta}$) are provided. Values are reported as medians with 95% credible intervals.

Group	Reach	Survival	Predation	a	Φ^{Δ}
RRJ	RRJ to BON	0.48 (0.47–0.50)	0.08 (0.07–0.09)	1.28 (0.70–2.08)	0.06 (0.02–0.10)
RIS	RIS to BON	0.45 (0.43–0.48)	0.10 (0.09–0.11)	1.59 (1.06–1.99)	0.14 (0.09–0.19)

For the opportunistic sample of steelhead passive detected (previously tagged and not handled) at RRJ, the estimated difference between observed survival (i.e. survival with tern predation) and baseline survival (i.e. survival in the absence of tern predation) was 0.06 (0.02–0.10) which was comparable to, but lower than, the estimated difference of 0.14 (0.09–0.19) associated with steelhead from RIS (*Table 13*). The relative differences in these estimates may reflect the differences in the spatial scale being evaluated (RRJ to BON compared to RIS to BON), in addition to the discrepancies in the demographics and run-timing between these two groups of smolts (see *Appendix C* for a detailed analysis of these inherent difference). For example, the average annual steelhead survival probability from RRJ to BON during 2010–2023 was estimated to be 0.48 (0.47–0.50) with annual steelhead survival probability from RIS to BON estimated to be slightly lower at 0.45 (0.43–0.48) during those same years. Results suggest that, following the assumptions of the additive relationship modelled herein, survival probabilities to BON in the absence of all tern predation may have been 0.56 (0.51–0.61) and 0.60 (0.53–0.67) for these two groups of smolts respectively. Since RRJ was not operational until 2010, relative comparisons between RRJ and RIS smolts prior to 2010 were not possible, years when tern predation probabilities were the highest within the middle Columbia River due to predation by terns nesting on Goose Island in Potholes Reservoir and years in which the measured additive relationship was the most acute (Payton et al. 2020).

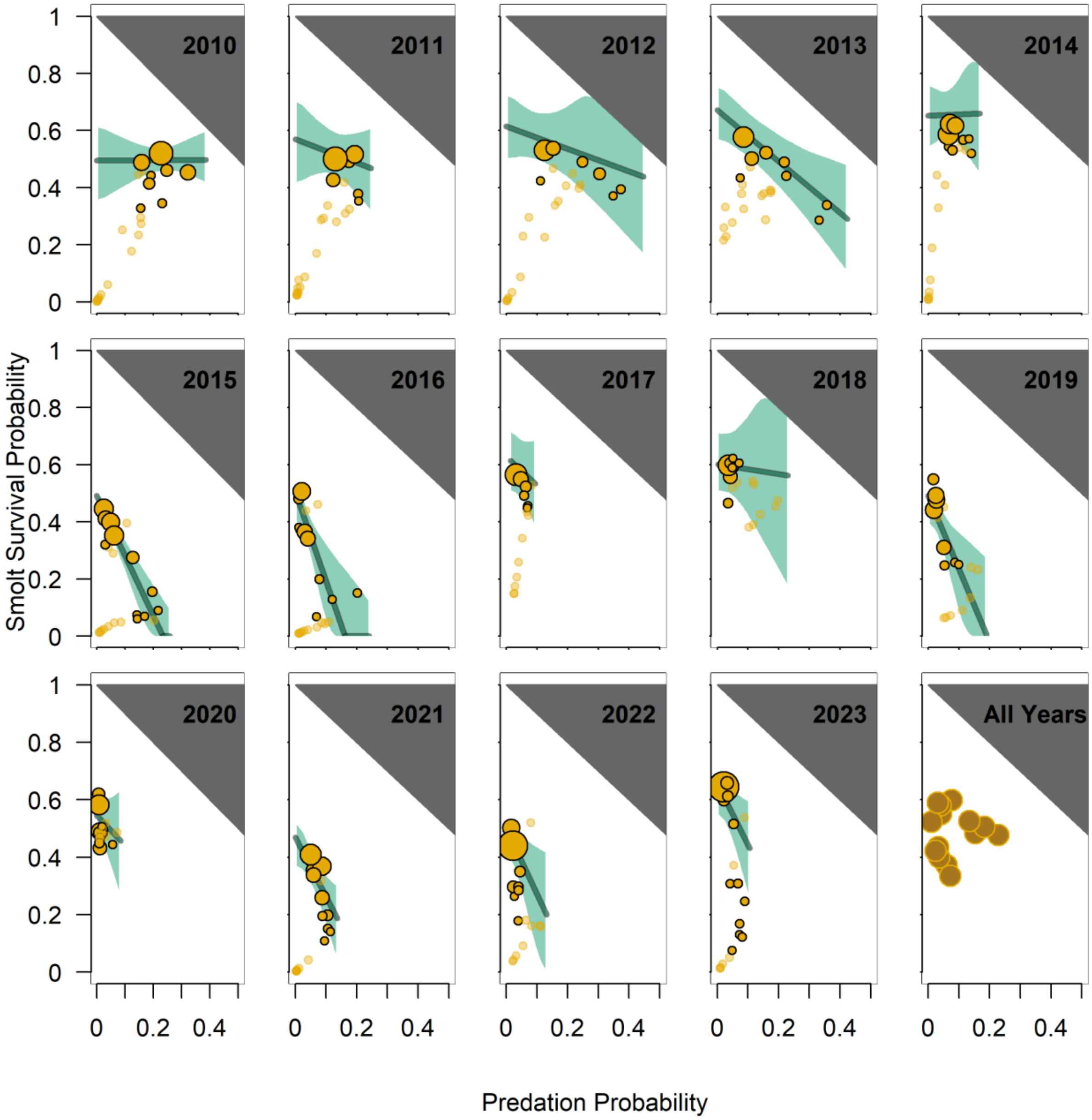


Figure 8. Weekly smolt survival probabilities for Upper Columbia River steelhead as a function of Caspian tern predation probabilities during smolt out-migration from Rocky Reach Dam (RRJ) to Bonneville Dam during 2012–2023. The size of circles depicts the relative number of PIT-tagged smolts detected at RRJ each week. Lines represent the best fit estimate of the relationship between predation and survival and shading denotes 95% credible intervals around the best fit. Circles lacking a black outline indicate weekly samples sizes less than 100. The grey triangle represents the excluded portion of the parameter space where predation cannot be greater than survival (see also Payton et al. 2020).

Difference between baseline survival estimates may also explain the nominal difference in estimates of additivity (a) between RRJ and RIS smolts. For example, with respect to all steelhead smolts passively recaptured at RRJ, estimates suggest that, on average, for every 10 additional smolts eaten by a tern, 12.8 died prior to reaching BON. From this result alone, the α parameter does not accurately measure its originally intended purpose, isolating the effects of tern predation relative to other mortality sources. However, further inference from our model can be made from the estimates of baseline survival which imply that for any hypothetical 10 smolts, 5.6 would be expected to survival and the other 4.4 would be expected to die during passage from RRJ and arrival at BON, irrespective of tern predation. We must interpret the failed arrival of 12.8 smolts for every 10 consumed with respective to this estimate that 4.4 smolts would have died otherwise. It then follows that, as the values of baseline mortality will vary due to differences in the run-timing, demographics, and the spatial scales evaluated, comparisons of estimates of α among release groups (RRJ, RIS) of smolts become complicated. It is with the considerations that the original Burnham model was parameterized and, as such, that original parameterization will be more appropriate for future relative comparisons like these.

Finally, with our continued analyses of additivity it has become increasingly apparent that our current parameterization, with the focus on α and Φ^{Δ}, may be an overly-broad summary of the super additive effects of tern predation across large spatial-scales or river reaches and multiple years. The analysis is further complicated by the shifting and changing location(s) of tern predation due to management actions at Goose and Crescent islands during the study period (see *Efficacy of Avian Predation Management Plans above*). For instance, in the earlier years of the analysis, a large proportion of tern predation was located closer to the release sites (RRJ, RIS) due to terns nesting on Goose Island in Potholes Reservoir. Following the dissuasion of terns from Goose Island and Crescent Island in McNary Reservoir, we saw increased colony sizes at the Blalock Islands in John Day Reservoir, over 100 Rkm downstream of the foraging range of terns on Goose Island and over 50 Rkm downstream of the foraging range of terns on Crescent Island. The relative impact of tern mortality is thus diminished when assessed from RRJ or RIS as a significant amount of unrelated mortality has taken place prior to smolts survival to the foraging range of terns nesting at the Blalock Islands. For example, the birds could be consuming 20% of the smolts passing their colony but if 50% of the smolts released from upstream died prior to being exposed to these colonies, the observed predation rate associated with these colonies (as measured from RIS or RRJ) will be significantly reduced. Graphically, this can be observed in the steep slope of the relationship depicted in *Figure 8* during the later years of the study, primarily in 2015 and 2016 when larger numbers of terns nested on the Blalock Islands.

Finally, estimates of α have been greater than 1.0 in several studies of tern predation (Payton et al. 2020, Payton et al. 2021, Evans et al. 2022), estimates that are indicative of super-additive effects. These effects have been hypothesized to be due to latent mortality from unsuccessful tern foraging attempts that sub-lethally injury (i.e., latent mortality or crippling losses) smolts or due to kleptoparasitisim of smolts by gulls and potentially other piscivorous waterbirds species (see also Payton et al. 2020). Levels of latent mortality and levels of kleptoparasitisim from individual tern colonies were presumably consistent across release groups of smolts (RRJ, RIS) but may be different among the array of colonies and colony locations included in each study. With all these considerations in mind, we recommend implementing an interpretation of the parameters of the Burnham model, which disentangles the estimate of baseline survival or mortality from the inherent additivity/super-additivity of the mortality cause (in this case predation) under consideration. We further suggest allowing baseline survival to vary by smolt release group, year, and river reach; a much more complex model but one which could provide insights into the level of super-additivity for predators at certain colonies across smolt cohorts of disparate origins. These and other modelling considerations will be considered as part of our on-going efforts to investigate the relationship between avian predation and fish survival in the CRB.

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 2023 and in previous years (Collis et al. 2021b, Roby et al. 2021b, Evans et al. 2022b, Evans et al. 2023, Collis et al. 2024), 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 occurred in 2021 and 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 2023 the colony consisted of just 524 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. Adding to the growing conservation concern for the Pacific Flyway population of Caspian terns is the recent emergence of highly pathogenic avian influenza (HPAI), which has caused mortality of over 1,000 Caspian terns in the Pacific Flyway population, including in the Columbia River estuary, by time of writing (M. J. Lawonn, Oregon Department of Fish and Wildlife, pers. comm.). HPAI is becoming 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. 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 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; Collis et al. 2024).

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 to the CPR in general. This is evidenced by the re-establishment of a tern colony on Crescent Island in 2022–2023, 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 also likely to become re-established. Finally, the shift in the distribution of nesting terns from managed colonies to unmanaged colony sites in the CPR (i.e. Badger Island, Shoal Island, and Goose Island at Banks Lake) 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 to assure lasting success, as suggested by the adaptive management provisions included in the IAPMP (USACE 2014, Section 3). 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 2021–2023, per capita predation rates by cormorants nesting on the Astoria-Megler Bridge were 2–5 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 now higher than those of cormorants that nested 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 elsewhere in upper CRE and Lower Columbia River 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 (Evans et al. 2023), the proportion of smolts consumed by gulls at dams that were dead or moribund is unknown and warrants additional investigation. Based on the perceived impacts of gulls nesting on Miller Rocks on smolt survival, nest dissuasion activities were implemented at that colony (see *Nest Dissuasion Activities* section above). Although there is some evidence that dissuasion activities have reduced the size of the Miller Rocks gull colony, predation rates have remained unchanged since management actions commenced in 2022. Continuation of nest dissuasion activities at select gull colonies may bring smolt survival benefits, especially if the smolt mortality caused by gulls is at least partially additive.

Results of this and other studies indicated that 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 also indicate that predation by Brandt’s cormorants nesting in the CRE and double-crested cormorants nesting on in Potholes Reservoir and Shoal Island in Lenore Lake (both 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 on Badger Island in McNary Reservoir and Miller Sands Spit in the Columbia River estuary remain low, 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 2007–2023 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 2007–2022 are those previously reported by Evans et al. (2023).

			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
2023	0.1% (<0.1–0.2)	0.5% (0.2–1.3)	-	-	0.7% (0.4–1.4)	0.8% (0.4–1.6)
N	12,498	1,558			8,809	8,377

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
2023	0.2% (0.1–0.4)	1.0% (0.4–2.1)	-	-	1.9% (1.2–3.2)	1.4% (0.9–2.5)
N	12,498	1,558			8,809	8,377

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
2023	<0.1%	<0.1%	-	-	<0.1%	0.6% (0.3–1.2)
N	12,498	1,558			8,809	8,377

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
2023	<0.1%	<0.1%	-	-	<0.1%	0.4% (0.2–0.8)

N	12,498	1,558			8,809	8,377
			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 2021–2023, 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								
2023	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 2020–2023. 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								
2023	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 2007-2023 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 2007–2022 are those previously reported by Evans et al. (2023). Results from Goose Island gulls were available in select years only (2012, 2020, 2022) but gulls nested on Goose Island in all years during 2007–2023.

	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
2023	0.2% (0.1–0.6)	1.0% (0.2–3.7)	-	-	1.3% (0.6–2.5)	1.8% (1.0–3.4)
N	12,498	1,558			8,809	8,377

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
2023	0.9% (0.3–1.7)	0.6% (0.1–3.2)	-	-	2.6% (1.0–4.9)	5.8% (2.1–9.9)
N	12,498	1,558			8,809	8,377

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	0.5% (0.2–1.3)	0.1% (0.1–0.6)	-	-	1.0% (0.4–2.1)	3.4% (2.1–5.6)
N	6,492	4,643			4,626	6,214
2023	0.2% (0.1–0.5)	0.7% (0.1–2.7)	-	-	0.7% (0.3–1.3)	2.8% (1.8–4.6)
N	12,498	1,558			8,809	8,377

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
2023	2.0% (1.0–3.5)	4.7% (2.4–8.9)	2.6% (1.1-5.4)	-	6.7% (3.5–12.4)	9.9% (4.9–18.7)
N	5,701	2,182	1,603		1,468	819

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
2023	0.4% (0.1–1.7)	0.5% (0.1–1.8)	1.0% (0.1–4.9)	-	3.6% (1.6–7.4)	3.2% (1.0–8.2)
N	5,701	2,182	1,603		1,468	819

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 2007–2023 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 2007–2022 are those previously reported by Evans et al. (2023).

	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
2023	2.1% (1.1–4.6)	1.3% (0.2–4.9)	-	-	3.1% (1.6–6.4)	0.3% (0.1–0.9)

N	12,498	1,558			8,809	8,377

Hanford Island, Middle Columbia River

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2018	<0.1%	<0.1%	0.2% (0.1-0.7)	<0.1%	<0.1%	0.2% (0.1-0.4)
N	19,986	8,753	2,090	1,443	19,632	7,511
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
2023	<0.1%	<0.1%	-	-	<0.1%	0.5% (0.1–1.9)
N	12,498	1,558			8,809	8,377

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 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), and middle Columbia River (MCR; based on detection at Bonneville Dam) during 2022–2023 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. Although the Astoria-Megler Bridge was scanned for smolt PIT tags in 2021, ESU/DPS-specific estimates were not available due to small sample sizes of known ESA-listed smolts within experimental plots on the bridge (see Evans et al. 2022b). Estimates by salmonids species, however, were reported (see Evans et al. 2022b, 2023).

	Astoria-Megler Bridge, Columbia River Estuary						
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2022	4.9% (2.6-8.1)	3.1% (2.1-7.9)	5.2%(2.0-10.3)	6.6% (1.7-14.7)	7.4% (2.1-15.5)	7.2% (3.5-12.0)	8.6% (3.2-15.1)
N	14,345	2,069	3,390	1,677	1,791	7,935	3,294
2023	7.1% (4.1–11.3)	3.8% (1.1–7.9)	8.2%(4.2–13.8)	14.3% (6.3–24.4)	6.8% (1.5–14.4)	10.9% (6.2–17.4)	10.4% (5.1–17.7)
N	17,539	3,738	6,238	2,904	2,245	10,670	5,272

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 2020–2023 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 2020–2022 are those previously reported by Evans et al. (2023).

	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
2023	0.1% (<0.1–0.3)	0.2% (0.1–1.1)	-	-	0.6% (0.1–1.6)	0.7% (0.2–2.7)
N	12,498	1,558			8,809	8,377

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 2021–2023 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 2020–2022 are those previously reported by Evans et al. (2023).

	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
2021	< 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
2022	< 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
2023	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
N	17,539	3,738	6,238	2,904	2,245	10,670	5,272

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

The analysis presented herein was originally included in the 2022 Avian Predation Annual Report (Evans et al. 2023) and has been updated with data and results from 2023.

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: Steelhead smolts PIT-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 erosion of pectoral, pelvic, or dorsal fins or a PIT-tag where the original tagger classified the fish as hatchery or a coded wire tag (2013–2016, 2023) that indicated the fish was of hatchery origin (see *PIT-tagging of Upper Columbia River Steelhead* section for additional details). Wild fish were classified by the presence of fully intact fins, with no tags indicating hatchery origin. It is possible, however, that some hatchery fish also met the criteria of being a wild fish and were thus misclassified as wild.

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). In brief, for use in this analysis, we defined cohort specific parameters for survival, predation, and recapture, with,

> $\boldsymbol{\Theta}^c$**,** defined to be a DxJ matrix where $\Theta^c_{d,j}$ represented the probability (from release) that a fish from release cohort r succumbed to depredation by colony d for $d \in \{1,2,\dots,D-1\}$ 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^c_j 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 $d \in \{1,2,\dots,D-1\}$, 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,

$$L = \prod_r \left(\prod_{j \le m} \left({p^c_j}^{y_j} * \left(1 - p^c_j\right)^{(1-y_j)} \right) * \prod_d {\chi^c_{m+1,d}}^{r_d} \right);$$

where,

$$\chi^c_{j,d} = \begin{cases} \theta^c_{j,d} * \gamma_d + \left(1 - p^c_{j+1}\right) * \chi^c_{j+1,d}; & for\ d \in 1,\dots,D-1 \\ \displaystyle\sum_{i=1}^{D} \theta^c_{j,i} * (1 - \gamma_i) + \left(1 - p^c_{j+1}\right) * \chi^c_{j+1,i}; & for\ d = D \end{cases}$$

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).

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 2008-2023 (*Table B.1*). Ratios of hatchery to wild fish were also relatively consistent across the study period (*Table B.1*).

Table B.1. Rear-type (hatchery, wild) of PIT-tagged Upper Columbia River steelhead smolts at Rock Island Dam during 2008–2023.

	Rear-type	
Year	No. (%) Hatchery	No. (%) 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 (28.4%)
2021	6,366 (78.7%)	1,724 (21.3%)
2022	4,579 (73.7%)	1,638 (26.3%)
2023	6,172 (73.7%)	2,205 (26.3%)

Predation/consumption probabilities by rear-type varied by avian predator species, river reach, and year (2008–2023; *Figure B.1-B.3*). Results indicated that hatchery fish were more likely to be consumed than their wild counterparts by CATE and LAXX breeding at colonies that foraged on smolts between RIS and MCN (*Figure B.1*). 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 pronounced in LAXX compared with CATE. Taken together, predation/consumption probabilities by both CATE and LAXX (i.e. all birds combined) on hatchery steelhead averaged 19.2% (17.5–20.9%) compared with 12.0% (10.5–13.1%) on presumed wild steelhead. There was also some evidence that wild fish were more likely, on average, to survive 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.

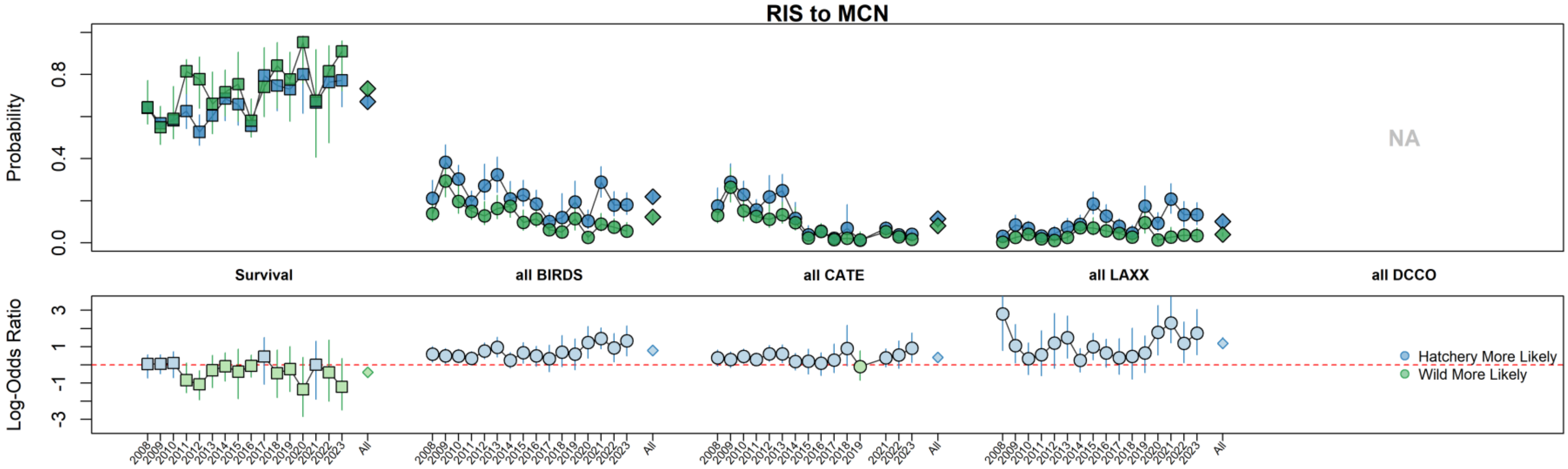

Figure B.1. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from Rock Island Dam (RIS) to McNary Dam (MCN) during 2008–2023 (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.

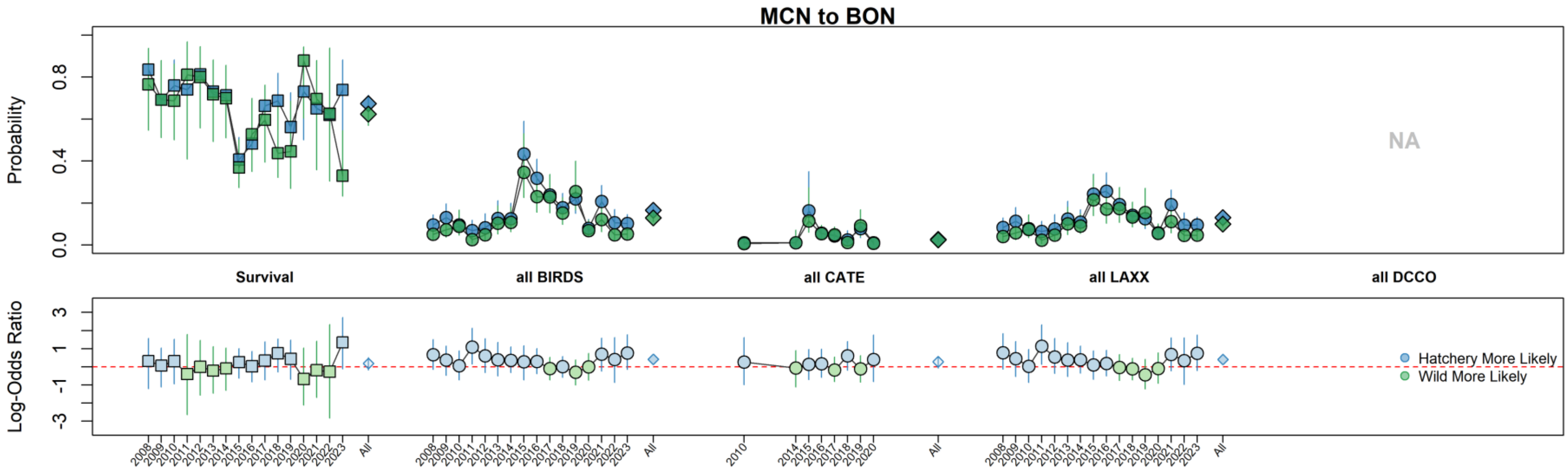

Figure B2. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from McNary Dam (MCN) to Bonneville Dam (BON) during 2008–2023 (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.

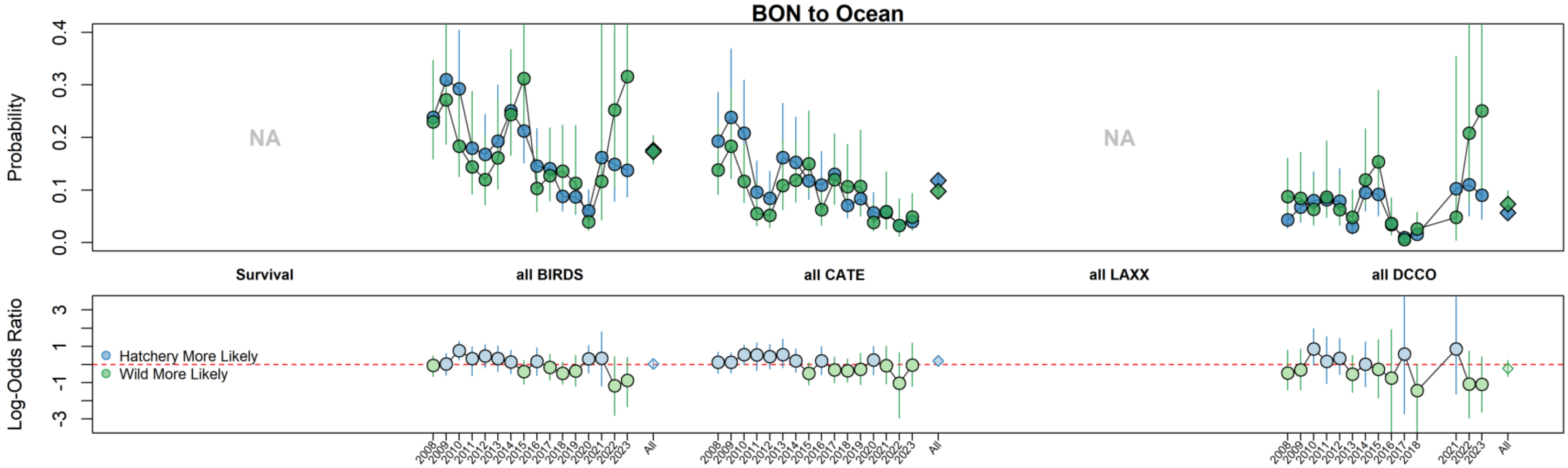

Figure B3. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from Bonneville Dam (BON) to the Pacific Ocean during 2008–2023 (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.

Although there were active colonies of DCCO and AWPE 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 in this river reach 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 (*Figure B2*). 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 16.2% (14.7–17.9%) and 12.7% (11.1–14.7%) 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 consistent 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 (*Figure B3*). There was limited evidence that DCCO disproportionately consumed wild smolts in some years (e.g., 2018, 2022–2023) but results were not statistically significant when data from all years are considered. Taken together (all birds) the odds of predation by rear-type were nearly indistinguishable between hatchery and wild steelhead downstream of BON. For instance, average annual predation/consumption probabilities by all birds were 17.7% (15.9–19.8%) and 18.2% (15.7–21.8%) for hatchery and wild smolts, respectively. Annual trends in CATE and DCCO predation also followed very similar patterns across the 16-year study period (*Figure B3*). 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 on influenced the size distribution and subsequent smolt and smolt-to-adult survival of UCR steelhead.

Unlike predation by CATE and LAXX, 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, intricacies that should be considered when evaluating the over-all effects of predation on prey populations (see also Hostetter et al. 2023 for a more detailed discussion).

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

The analysis presented herein was originally included in the 2022 Avian Predation Annual Report (Evans et al. 2023) and has been updated with data and results from 2023.

As part of the Avian Predation Project, the PIT-tagging of juvenile (smolt) 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 *PIT-tagging of Upper Columbia River Steelhead* section 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 smolt 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 by terns 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 *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. 2015a). 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 and in Hostetter et al. 2015a). 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 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). 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* for colony locations). 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 $\left(d_j, d_{j+1}\right)^{\text{th}}$ 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 – $\mathrm{prob}\left(\beta_{j.} > 0\right) \in [0.05,0.95]$ – were dropped. We assign priors of $\boldsymbol{\beta}_j \sim \mathrm{mvnorm}(0, \mathbf{I})$ for all j and $\sigma \sim \mathrm{half} - \mathrm{normal}(0, 10)$. 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 individual 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 below*) and 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 (*Appendix B,* see also Hostetter et al. 2023). 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

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

where $\boldsymbol{x}_{s_0}$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 $\boldsymbol{\epsilon}_{p_0} \sim \mathrm{normal}(0, \sigma_{\mathrm{p}_0})$.

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$^{\text{th}}$ 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$^{\text{th}}$ 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 $\boldsymbol{\epsilon}_{\theta_c} \sim \mathrm{normal}(0, \sigma_{\theta_c})\ \forall c$, 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 $(C + 1)^{\text{th}}$ 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 $f_i = 1$,

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

if $f_i = 0$,

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

and, otherwise,

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

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,

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

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 $\mu_{p_1} \sim \text{logistic}(\mu^*_{p_1}, \sigma^*_{p_1})$. Priors for each μ^* were assumed to be $\text{logistic}(0,1)$. 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 $\text{mvnormal}(\vec{0}, \mathbf{I})$. Priors for all σ's were assigned to be $\text{normal}(0,1.5)$.

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 (RDCT 2014) 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* and *Methods & Analysis* for details regarding odds ratios and odds ratio tests).

Results & Discussion**:** Most 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*). 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 3.5% 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 1.2 (95% credible interval = 0.9–1.4) 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.

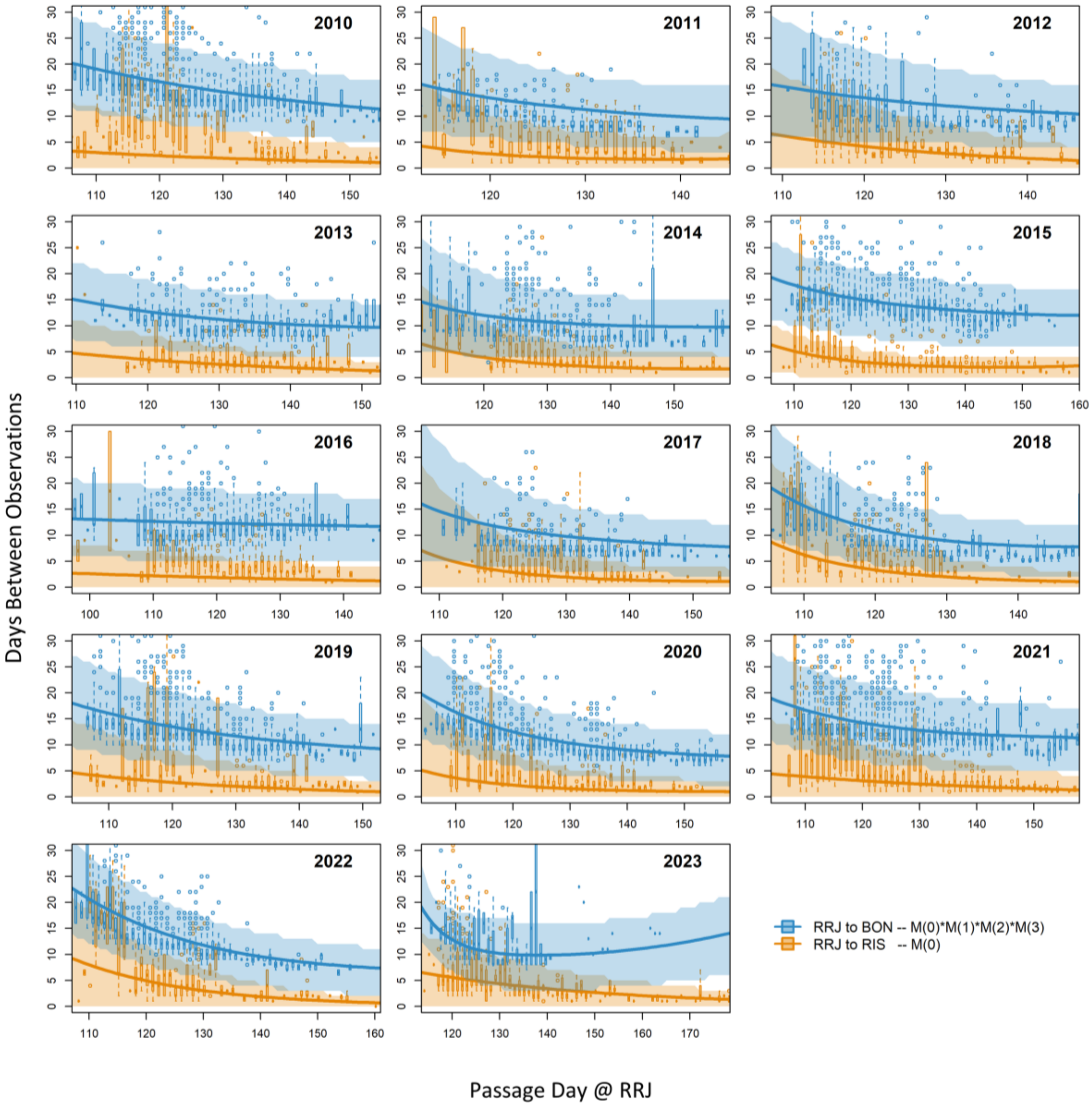

Figure C.1: Boxplots representing a subset of all observed travel times of PIT-tagged steelhead smolts detected at Rocky Reach Dam (RRJ) and downstream recapture sites (orange for RRJ to Rock Island Dam [RIS]; blue for RRJ to Bonneville Dam [BON]) by day of passage at RRJ. Shaded regions represent the 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 1.49 (1.31–1.71) 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 1.42 (1.30–1.55) 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) 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.

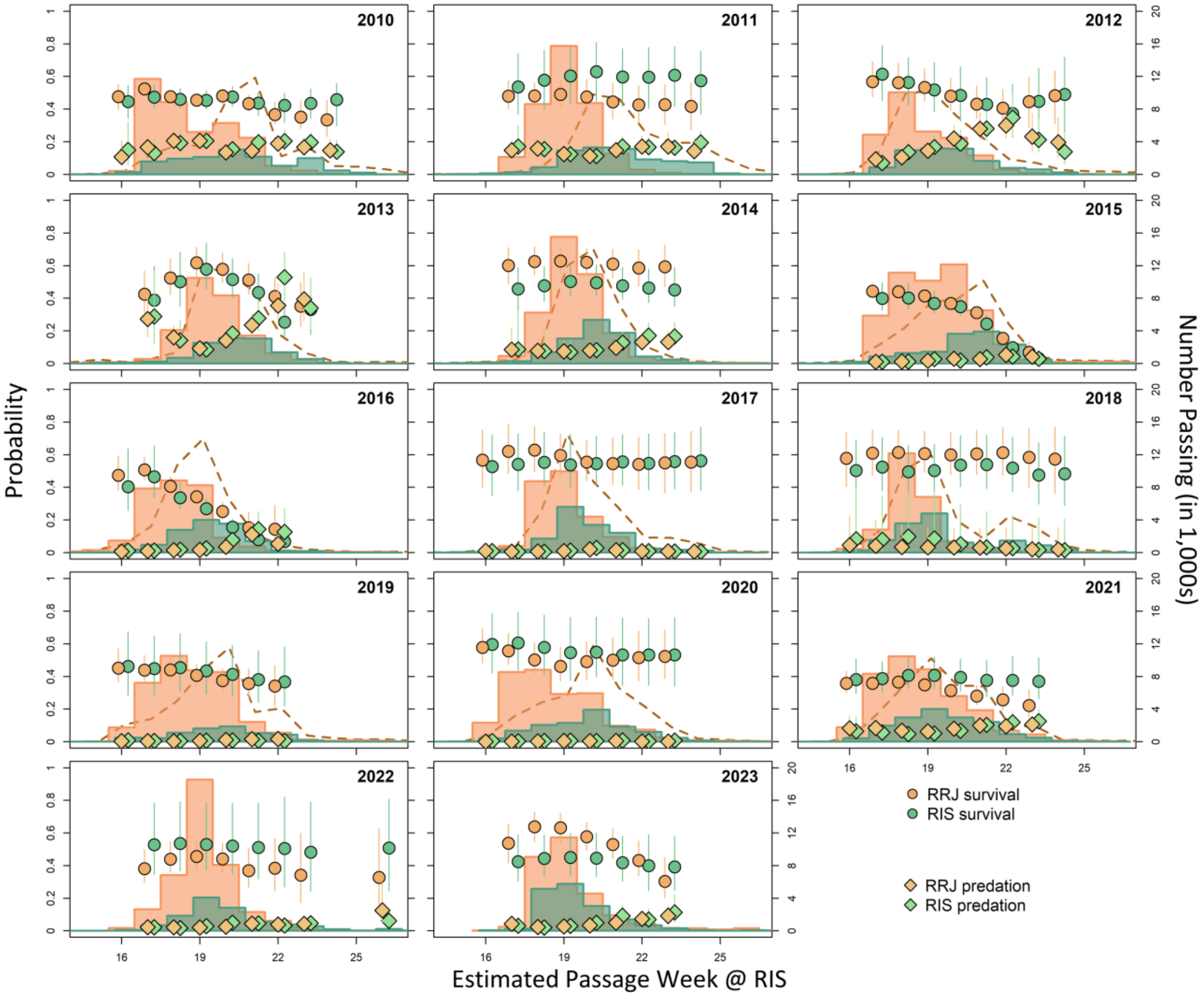

Figure C.2: Weekly probabilities of steelhead smolt survival (circles) and predation by Caspian terns (diamonds) for fish passively detected at Rocky Reach Dam (RRJ) and those tagging at Rock Island (RIS). Polygons represent histograms of the estimated weekly proportion of all steelhead first seen at 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 and 2023).

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 102 of the 109 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 14 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.

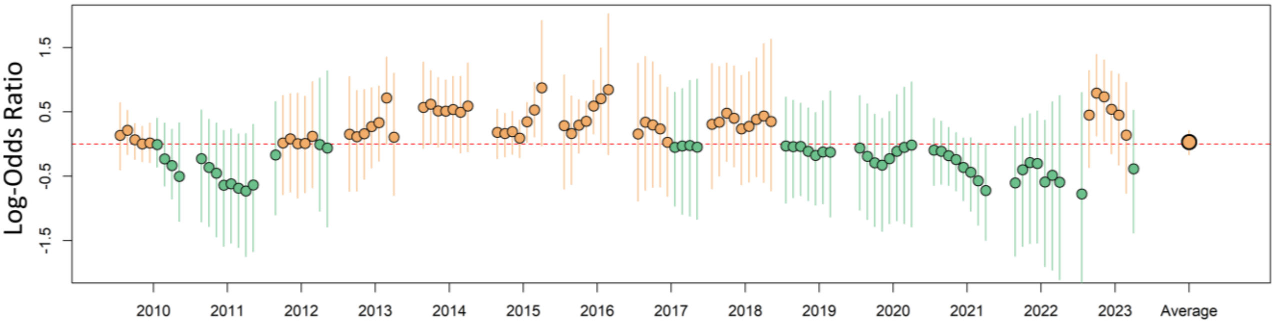

Figure C.3: Weekly estimates of the log odds of smolt survival for steelhead passively detected at Rocky Reach Dam versus those tagged at Rock Island Dam to Bonneville Dam. Vertical line segments represent 95% credible intervals. Orange-colored points above the red dashed line at 0 represent weeks in 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 significant 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 & Analysis* section). 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. 2015a, 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, 2023, smolt survival to Bonneville Dam was generally higher and less variable for both cohorts.

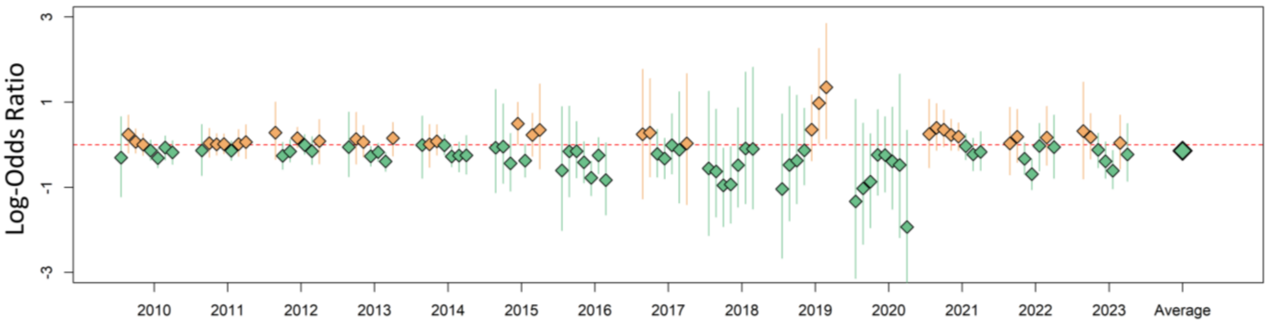

Figure C.4. Weekly estimates of the log odds of predation by terns for steelhead smolts passively detected at Rocky Reach Dam versus those tagged at Rock Island Dam. Vertical line segments represent 95% credible intervals. Orange-colored points above the red dashed line at 0 represent weeks in which the odds of predation were greater for the Rocky Reach cohort, while green-colored points below the red dashed line represent 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 significant difference.

Results of odd ratio tests indicated that the estimated odds of tern predation between cohorts were similar on a weekly basis, with no statistical difference in the odds of predation in 97 of the 109 weekly-comparisons (*Figure C.4*). Across all years, RIS tagged steelhead smolts were slightly (1.13 [0.97–1.29] times) more likely to be consumed than their RRJ counterparts, an expected finding given the RRJ estimates of availability were not adjusted for smolt survival to RIS.

In summary, there was 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, but such studies are currently limited by extremely low recapture rates of tagged fish during outmigration from RRJ to Bonneville Dam.

APPENDIX D: REACH-SPECIFIC WEEKLY PREDATION AND SURVIVAL OF UPPER COLUMBIA RIVER AND SNAKE RIVER SMOLTS

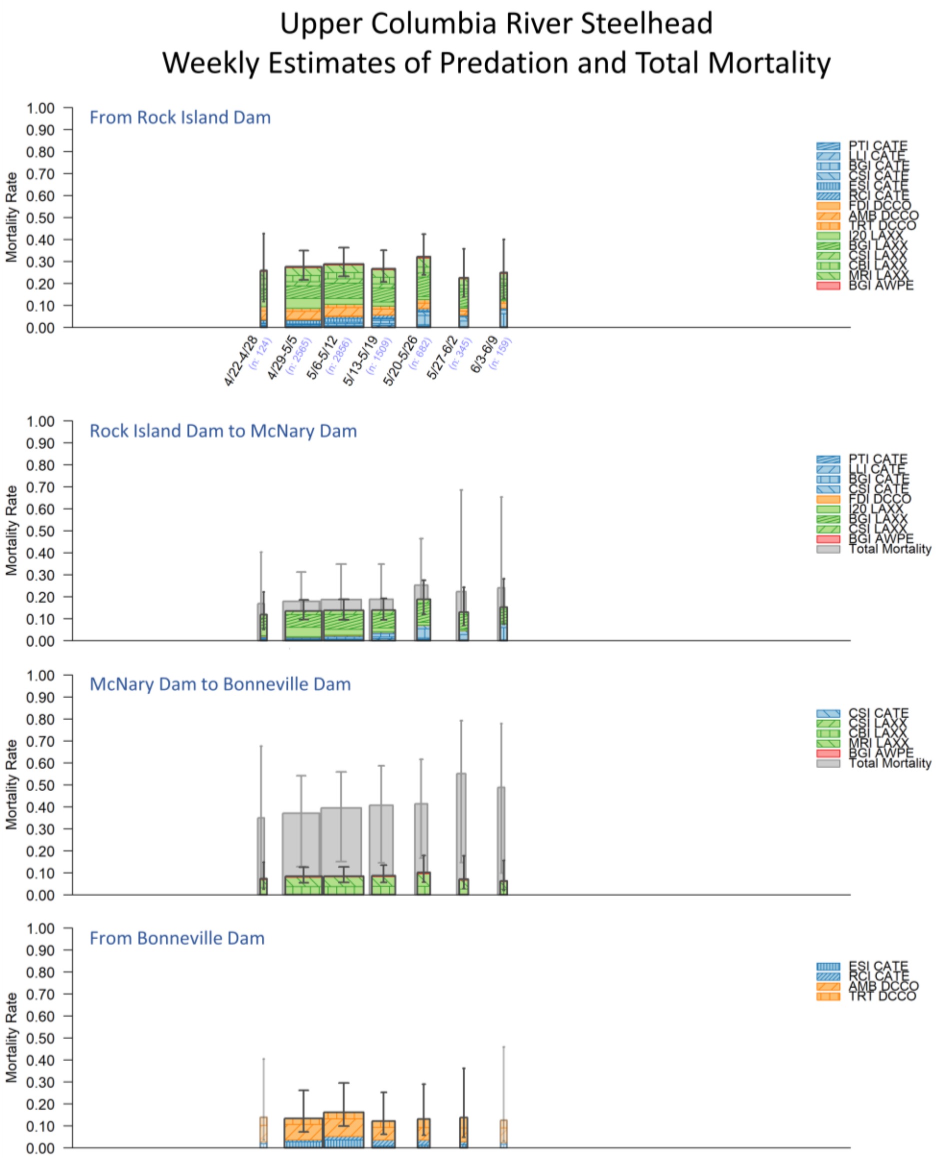

Figure D1. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Upper Columbia River steelhead during smolt passage from Rock Island Dam to the Pacific Ocean (top), Rock Island Dam to McNary Dam, McNary Dam to Bonneville Dam, and following passage at Bonneville Dam. Bar widths are proportional to sample abundance at Rock Island Dam and only weeks with > 99 PIT-tagged smolts were included. Bars are transparent for reach/weeks combinations for which the JMS model estimated less than < 100 surviving smolts entered the reach downstream of Rock Island Dam. Estimates are proportions with error bars representing 95% credible intervals. 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).

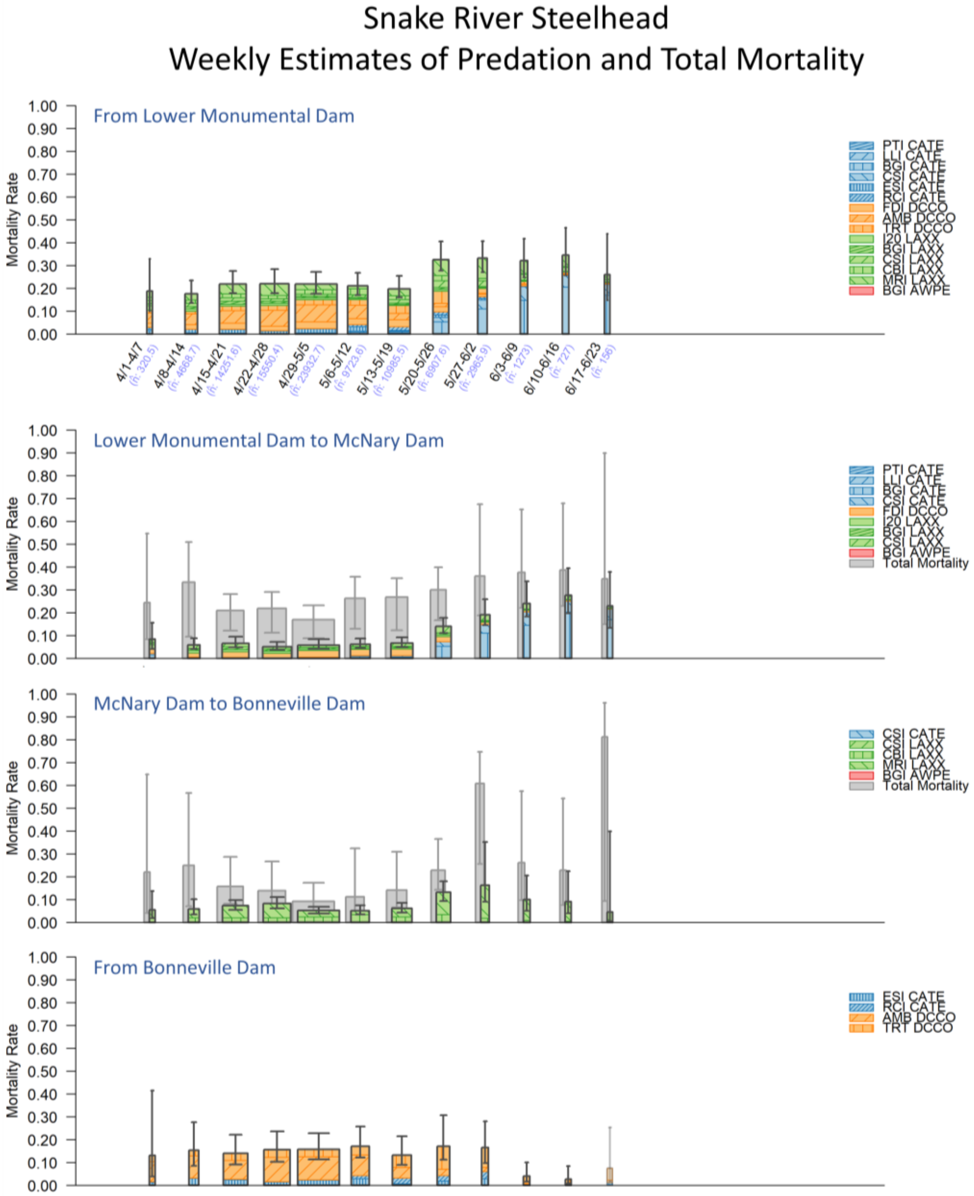

Figure D2. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River steelhead during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and following passage at Bonneville Dam. Bar widths are proportional to estimated sample abundance at Lower Monumental Dam and only weeks with > 99 PIT-tagged smolts were included. Bars are transparent for reach/weeks combinations for which the JMS model estimated less than < 100 surviving smolts entered the reach downstream of Lower Monumental Dam. Estimates are proportions with error bars representing 95% credible intervals. 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).

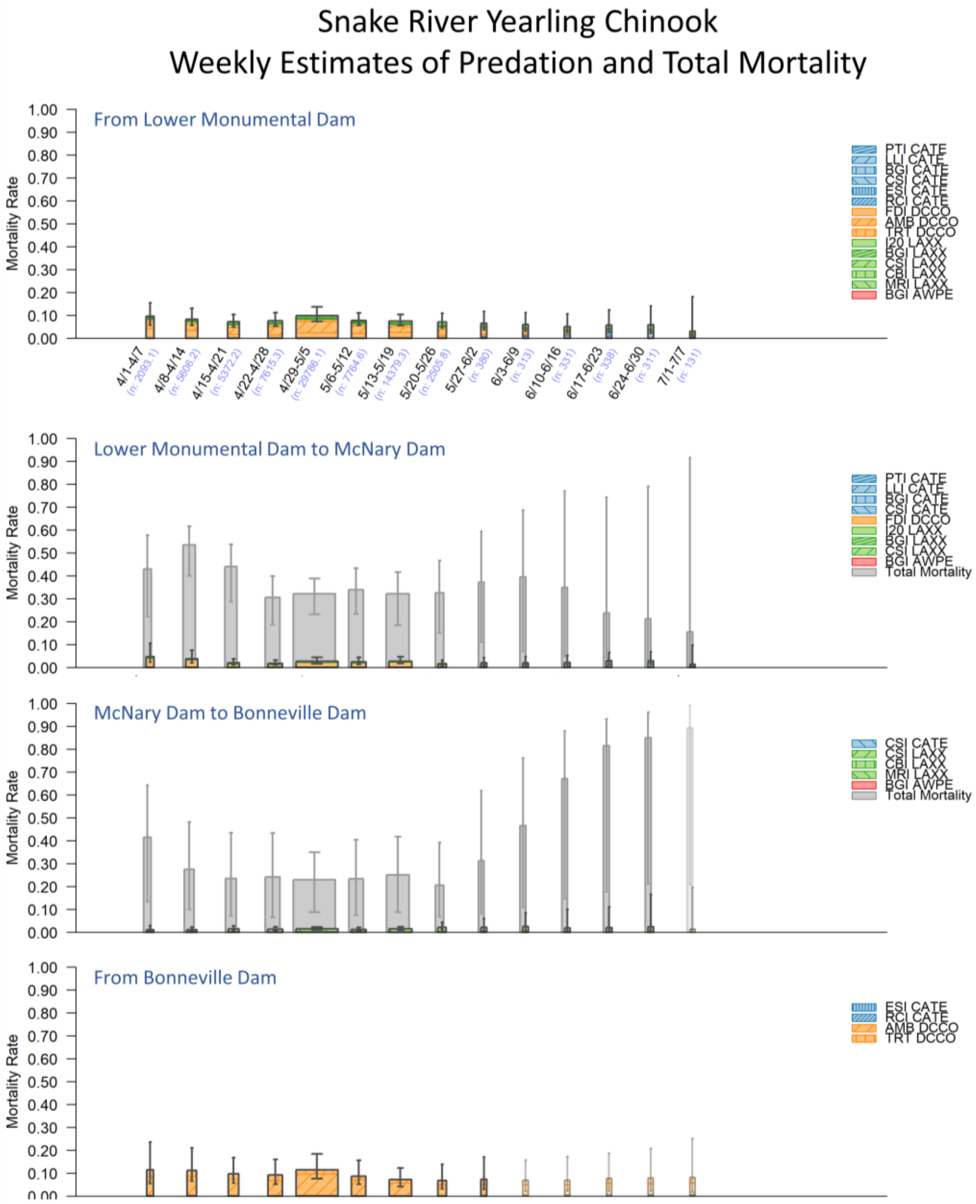

Figure D3. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River yearling Chinook during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and following passage at Bonneville Dam. Bar widths are proportional to estimated sample abundance at Lower Monumental Dam and only weeks with > 99 PIT-tagged smolts were included. Bars are transparent for reach/weeks combinations for which the JMS model estimated less than < 100 surviving smolts entered the reach downstream of Lower Monumental 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).

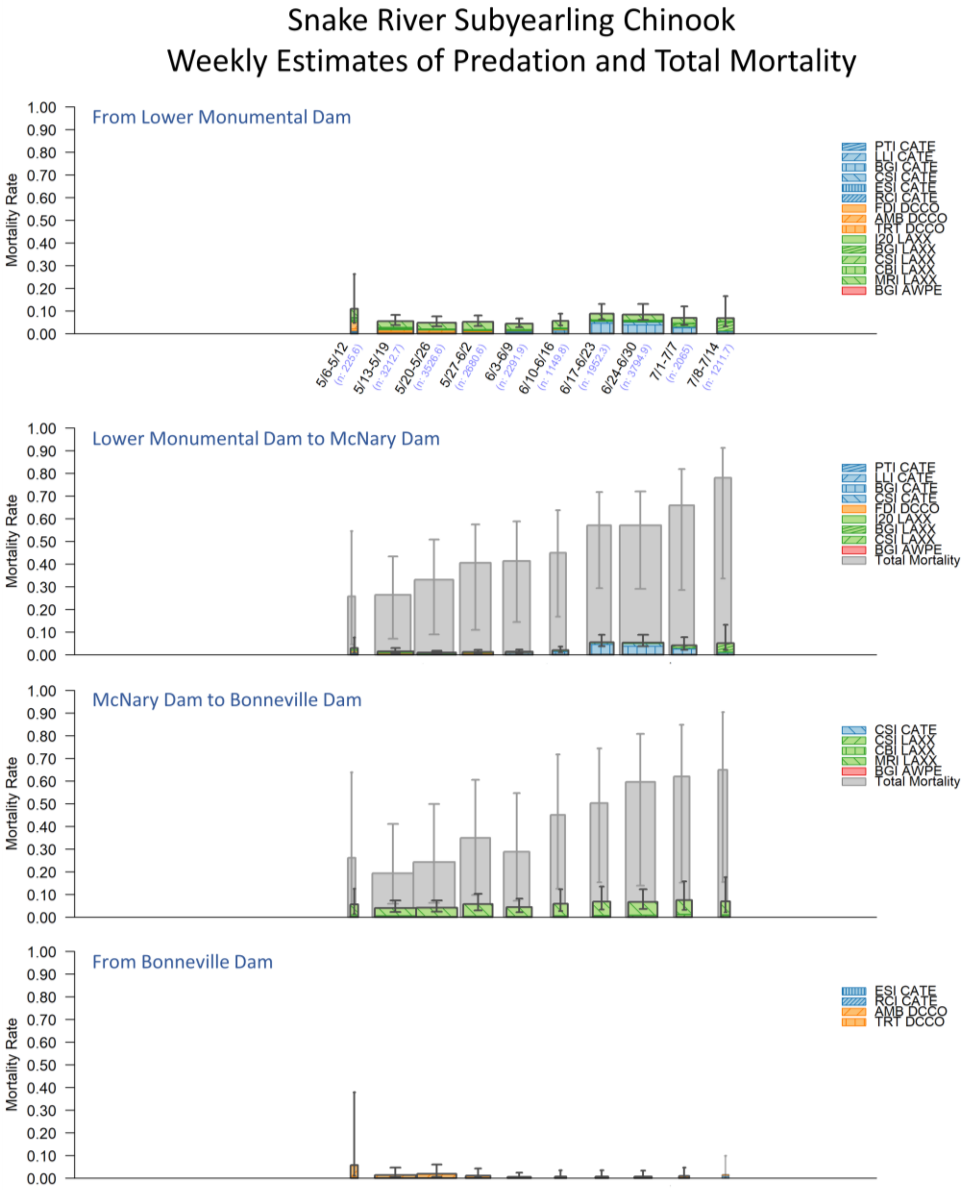

Figure D4. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River sub-yearling Chinook during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and following passage at Bonneville Dam. Bar widths are proportional to estimated sample abundance at Lower Monumental Dam and only weeks with > 99 PIT-tagged smolts were included. Bars are transparent for reach/weeks combinations for which the JMS model estimated less than < 100 surviving smolts entered the reach downstream of Lower Monumental 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).

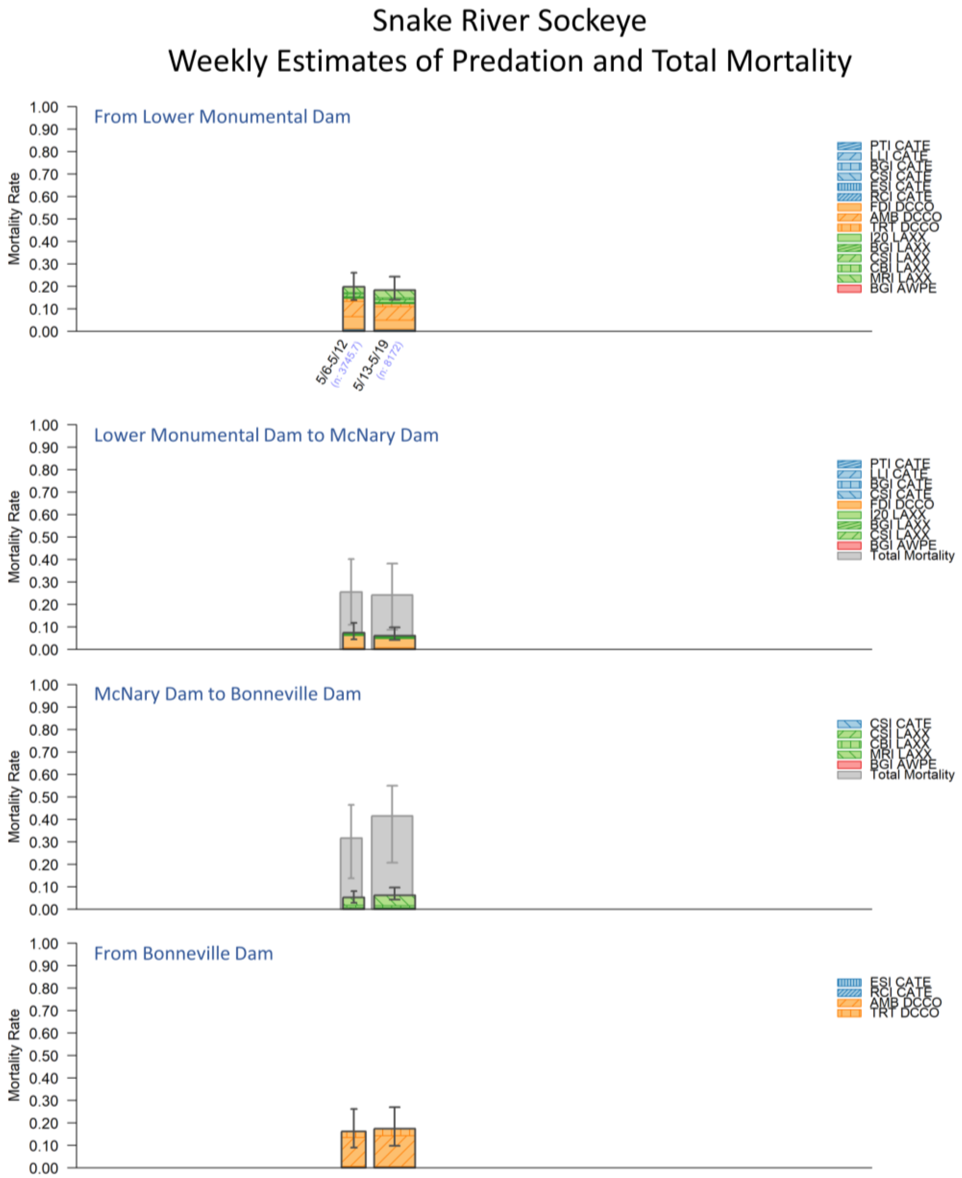

Figure D5. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River sockeye during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and following passage at Bonneville Dam. Bar widths are proportional to estimated sample abundance at Lower Monumental Dam and only weeks with > 99 PIT-tagged smolts were included. Bars are transparent for reach/weeks combinations for which the JMS model estimated less than < 100 surviving smolts entered the reach downstream of Lower Monumental 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).