AVIAN PREDATION IN THE COLUMBIA RIVER BASIN

2024 Final Annual Report

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Title: Avian Predation in the Columbia River Basin: 2024 Final Annual Report

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EXECUTIVE SUMMARY

The primary objective of this study was to investigate piscivorous colonial waterbird predation on juvenile salmonids (smolts; *Oncorhynchus* spp.) in the Columbia River basin (CRB) in 2024. Piscivorous colonial waterbird species investigated included Caspian terns (*Hydroprogne caspia;* hereafter "terns"), double-crested cormorants (*Nannopterum auritum;* hereafter "cormorants"), California gulls (*Larus californicus*) and ring-billed gulls (*L. delawarensis*; hereafter collectively as "gulls"), and American white pelicans (*Pelecanus erythrorhynchos*; hereafter "pelicans"). Managed predator species included terns and cormorants from colonies in the Columbia Plateau region (CPR) and the Columbia River estuary (CRE). The study sought to (1) estimate colony sizes for colonies within foraging range of smolts in the middle Columbia River, lower Snake River, lower Columbia River (LCR), and the CRE, (2) estimate colony-specific predation rates (percentage of fish consumed) on Endangered Species Act (ESA) listed smolts, (3) estimate the cumulative or system-wide effects of predation from all colonies combined, and (4) to evaluate the efficacy of on-going management plans aimed at reducing predation on smolts.

A total of 37 active breeding colonies of piscivorous waterbirds were documented within the study area in 2024. Of those, cormorant and gull colonies were the most prevalent (14 and 12 colonies, respectively), followed by terns (8 colonies), and pelicans (3 colonies). A total of 26 colonies were in the CPR and 11 colonies located in the lower Columbia River and CRE. 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. 7,979 breeding pairs), pelicans (ca. 4,253 individuals), and terns (ca. 1,963 breeding pairs). The number of active breeding colonies observed in 2024 was similar to that in many, but not all, years past. The size and location of breeding colonies, especially tern and cormorant colonies, however, have changed over the course of the last seven years following management actions in the CPR and CRE.

Tern and cormorant colonies continue to be managed as part of three separate management plans in the CRB (*Estuary Tern Management Plan*, *Estuary Cormorant Management Plan*, and the *Inland Avian Predation Management Plan*). Results of this study indicated that the numbers of terns nesting in the CRE have declined, which was the objective of management. However, management actions have also contributed to declines in the Pacific Flyway breeding population of terns, which raises concerns about their conservation status. The designated tern colony on East Sand Island (ESI) has rapidly declined and productivity (number of fledglings produced) has been low to non-existent, with complete nesting failure observed in six out of the last nine years (2016–2024). Concurrent with the decline in colony size and nesting failure at the ESI tern colony, there has been a large influx of terns attempting to nest on Rice Island in the upper estuary, where the per capita (per breeding pair) impacts of terns on smolt survival are 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 2024 was successful in dispersing terns from that site. However, the number of terns on the ESI remains lower than the target colony size identified in the management plan (3,125–4,375 pairs), a number, that if achieved, would help to maintain the Pacific Flyway breeding population of terns.

Predation rates on smolts by terns nesting on ESI have been greatly reduced since management actions were first implemented in 2008, with predation rates on ESA-listed steelhead reduced by upwards of 75%, reductions that exceed those specified in the management plan. The large numbers of terns attempting to nest on Rice Island, however, have partially offset the lower predation impacts by terns on ESI to some degree. For instance, in 2022, estimated predation rates by terns that attempted to nest on Rice Island were as high as 2.9% (95% credible interval = 1.6–5.1%) of Middle Columbia River (MCR) steelhead. Collectively, results of tern management actions in the CRE indicate that adaptive management will continue to be necessary to encourage nesting at ESI and to prevent terns from relocating to nest at other sites in the CRE, especially at sites in the upper estuary where predation rates on steelhead smolts are significantly higher on a per capita basis.

Management at the ESI cormorant colony has resulted in the dispersal of birds away from ESI, and 2024 marked the sixth consecutive year with little to no cormorant nesting on ESI. The failure of the ESI colony resulted in a rapid expansion of cormorant colonies on the Astoria-Megler Bridge (AMB), Lewis and Clark Bridge, channel markers, and transmission towers (TRT) located in the upper CRE and LCR. For instance, cormorant colony size on the AMB increased dramatically during implementation of the cormorant management plan on ESI during 2015–2019, supporting 5,348 breeding pairs in 2024, surpassing the estimate of 5,153 pairs in 2023. The AMB is located upstream of ESI in the freshwater mixing zone of the CRE and the other colony locations (Lewis and Clark Bridge, channel markers, and TRT) are in the freshwater zone of the estuary, 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 that nest at colonies in the mixing and freshwater zones of the CRE. Estimates of per capita predation rates on salmonid smolts by cormorants nesting on the AMB and TRT were 2–10 times greater (depending on salmonid ESU/DPS and colony) than per capita predation rates by cormorants that formerly nested on ESI. Estimates of predation rates by cormorants nesting on the AMB in 2024 ranged from 2.8% (1.8–7.6%) on Snake River (SR) Fall Chinook Salmon to 10.9% (5.0–20.8%) on MCR steelhead, estimates that were higher than those of cormorants on ESI in years past. Estimates of predation by cormorants on TRT in 2024 ranged from 2.4% (1.0–5.7%) on SR Fall Chinook to 5.9% (3.2–12.1%) for Upper Columbia River (UCR) steelhead. Predation on SR Sockeye Salmon has also dramatically increased in recent years due to cormorants breeding on the AMB and TRT, with estimates greater than 10% of available smolts in recent years (2023–2024). Taken together, results indicate that predation by cormorants in the CRE are now greater than those observed prior to implementation of management actions on ESI. As such, adaptive management to prevent cormorants from nesting in the upper CRE and LCR and to re-attract cormorants back to ESI are necessary to achieve the goals of the *Estuary Cormorant Management Plan*.

Management implemented at tern colonies in the CPR as part of the *Inland Avian Predation Management Plan* has resulted in a shift in the number and location of tern colonies in the region. A total of 439 breeding pairs of terns nested at colonies in the CPR in 2024, a 50% reduction compared to the pre-management period (average of 875 pairs during 2007–2013) but still higher than target colony size goal of 200 pairs. Efforts to dissuade terns from nesting on Goose Island in Potholes Reservoir in 2024 were largely successful. Twenty pairs nested on the island in 2024, reduced from an average of 367 pairs prior to management. However, an incipient tern colony formed on an unnamed island in northern Potholes Reservoir in 2024 with 126 pairs. Adaptive management techniques (i.e., human hazing, fencing, and predator effigies) were able to successfully prevent nesting shortly after the colony site had been identified in an aerial survey. Although management actions at Crescent Island in McNary Reservoir were initially successful at preventing colony formation during 2014–2021, active management ceased in 2021, and terns subsequently re-established a colony in 2022 (149 breeding pairs), 2023 (88 breeding pairs), and 2024 (186 pairs). Adaptive management to raise the water elevation of John Day Reservoir was implemented to inundate tern nesting habitat in the Blalock Islands in John Day Reservoir during 2021–2024 and no nesting by terns was observed at that site during this period. Due to high fidelity to historical nesting sites, adaptive management will continue to be important, especially at Crescent Island and islands in Potholes Reservoir, to meet the stated goals of the management plan.

One of the primary goals of the *Inland Avian Predation Management Plan* is to reduce predation rates on smolts by terns to less than 2% per ESA-listed salmonid population, per colony. In 2024, this objective was met for the second time since management actions commenced in 2014. Predation rates were the highest on UCR steelhead at 1.7% (1.0–3.2%), 1.5% (0.9–2.6%), 1.3% (0.7–2.6%) by terns breeding on an island in Lenore Lake, the incipient colony in northern Potholes Reservoir, and Goose Island in Potholes Reservoir, respectively. Estimates of predation rates on all other ESA-listed salmonid populations were less than 1% per salmonid population, per colony, in 2024. 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 1.7% (1.0–2.8%) during the management period (2014–2024). There was also evidence that the 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 in larger numbers at other sites (e.g., Lenore Lake, northern Potholes Reservoir), however, adaptive management actions will continue to necessary to achieve the goals of the management plan in 2025 and beyond.

An investigation of predation on smolts by other piscivorous waterbird species indicated that smolt consumption rates were highly variable depending on the predator species, colony location, colony size, and salmonid species. Predation by gulls nesting in the CPR were often higher than those of managed tern colonies in 2024. For instance, estimates of gull consumption rates as high as 5.2% (2.2–10.8%) for SR steelhead and 8.5% (5.6–13.2%) for UCR steelhead were documented for gulls nesting at Miller Rocks in The Dalles Reservoir and Island

20 in the middle Columbia River, respectively. Unlike terns and cormorants, gulls are scavengers and are known to consume dead or moribund fish and to steal (kleptoparasitize) fish from other waterbirds. Consequently, the term "gull consumption rates" is used, rather than "gull predation rates." Estimates of consumption at several other large (several thousand individuals) gull colonies in the CRP region were low, with consumption rates less than 1.0% per ESA-listed salmonid population, indicating that not all gull colonies posed a potential threat to smolt survival. Estimates of cormorant predation in the CPR as high as 2.5% (1.4–4.8%) and 2.6% (1.1–6.4%) for SR spring/summer Chinook and SR steelhead, respectively, were observed for cormorants breeding on Crescent Island in McNary Reservoir, a colony that rapidly increased from less than 40 pairs in 2021 to more than 629 pairs in 2024, making it one of the largest cormorant colonies in the CPR. Two new, but smaller, cormorant colonies formed at the Murdock Towers (37 pairs) in the Bonneville Reservoir and on Island 20 (44 pairs) in the middle Columbia River in 2024, with predation rates of < 1.0% per salmonid population. Estimates of predation by American white pelicans nesting on Badger Island in McNary Reservoir, by Brandt's cormorants (*Phalacrocorax penicillatus*) nesting on the AMB in the CRE, and gulls nesting on Rice Island in CRE were generally < 0.5% per salmonid population, per colony, indicating these colonies posed little threat to ESA-listed smolts. Pelicans, however, are capable of consuming both juvenile and adult salmonids, and other studies have documented substantial predation effects on adult sockeye and juvenile salmonids from specific stocks (e.g., non-listed Chinook from the Upriver Bright population).

An investigation of the cumulative effects of avian predation/consumption (predation by all colonies combined) indicated that predation was a substantial source of smolt mortality during outmigration to the Pacific Ocean. Predation was consistently the highest on steelhead populations compared to salmon populations. For instance, in 2024, the cumulative effects of avian predation were estimated at 7.2% (5.0–10.8%) on SR Chinook but were 28.2% (23.6–35.3%) and 31.4% (26.5–37.9%) on UCR and SR steelhead, respectively. Predation on SR sockeye salmon, however, was similar to that of steelhead, with an estimated 29.0% (21.9–40.1%) of SR sockeye consumed by birds in 2024, the highest cumulative estimate on sockeye observed since studies commenced in 2008. Comparisons of total mortality (1 – survival) to mortality due to avian predation/consumption indicated that avian predation by the colonies included in the study accounted for between 8.8% (5.0–16.9%) and 53.4% (37.2–82.3%) of all smolt mortality sources during outmigration to Bonneville Dam, depending on the salmonid population. By river reach, predation/consumption rates were highest for SR smolts between Bonneville Dam and the Pacific Ocean and for UCR steelhead between Rock Island Dam and McNary Dam in 2024. Reach-specific predation/consumption rates in 2024 were similar to those of previous years, except for predation in the CRE, where predation impacts were above average, especially on SR sockeye due to cormorants breeding on the AMB and TRT. Collectively, results indicate that the cumulative effects of avian predation/consumption remain a substantial source of smolt mortality in the CRB, particularly for ESA-listed steelhead and sockeye populations.

Finally, results of this and other published studies have found a statistically significant relationship between avian predation and fish survival, indicating that avian predation was an additive source of mortality during the smolt life stage and a partially additive source of mortality to the adult life stage. The additive relationship was the most pronounced in SR and UCR steelhead due to high levels of tern predation in the CPR and CRE, where reductions in predation were associated was significantly higher estimates of smolt survival and significantly higher smolt-to-adult returns to Bonneville Dam. There was evidence of a partially additive relationship in UCR and SR steelhead due to predation/consumption by gull colonies. Estimates of gull additivity, however, were less precise and pronounced than those of terns (due to lower levels of weekly and annual variation in predation). Further, gulls likely consume stunned, dead, and moribund fish in the tailrace of dams, so some proportion of gull predation/consumption is compensatory. For cormorants, there was evidence of an additive relationship in the CRE for both steelhead and salmon populations but, like gulls, low levels of variation in cormorant predation probabilities resulted in imprecise estimates of additivity. Furthermore, in cases where avian predation impacts on smolts from a specific predator species and colony were low (e.g., less than 5%), reductions in avian predation would not dramatically increase fish survival due to the paucity of avian predation effects relative to other, non-avian sources of mortality during smolt outmigration. Research on the degree to which pelican predation limits fish survival is currently lacking. In the case of predation on adult sockeye, however, pelican predation is presumably at least partially additive, but additional research may be warranted.

BACKGROUND

Predation by piscivorous colonial waterbirds on out-migrating juvenile salmonids (smolts; *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 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 studies of avian predation have been conducted to assess the impacts on smolt survival from 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 smolts originating from the CRB, the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps of Engineers (USACE), U.S. Bureau of Reclamation (BOR), 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 for some predator species and colonies. 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 on smolt survival are as high or higher than at 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 research, monitoring and evaluation (RM&E) 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 salmonid smolts in the CRB in 2024 were to assess the distribution and size of piscivorous waterbird colonies and determine the colony-specific and cumulative effects of avian predation on smolt survival. Results from this project were also used to investigate the efficacy of on-going management plans for piscivorous colonial waterbirds, plans that aim to reduce the size of managed tern and cormorant colonies to increase survival of juvenile salmonids in the CRB. As part of these objectives, this study (a) located and estimated peak colony size at all piscivorous waterbird colonies within foraging range of smolts in the middle Columbia River, lower Snake River, lower Columbia River, and Columbia River estuary, including information on any new or incipient colonies; (b) estimated colony-specific predation/consumption rates by multiple piscivorous waterbird species; and (c) measured the cumulative or system-wide effects of avian predation on salmonid smolt survival, including an investigation of factors that influence smolt susceptibility to avian predation and the additive effects of predation on smolt survival. RM&E conducted in 2024 was a continuation of work that commenced in 2020.

METHODS & ANALYSES

This work is part of an ongoing, 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. Similar to years past, 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 management initiatives to reduce avian predation on smolts and to help identify new and emerging avian predation concerns that may warrant future management consideration. The primary methods used in 2024 (see *below*) were the same as those used in previous years by this project, ensuring that results were comparable across years, both before and during the implementation of avian predation management actions in the CRB.

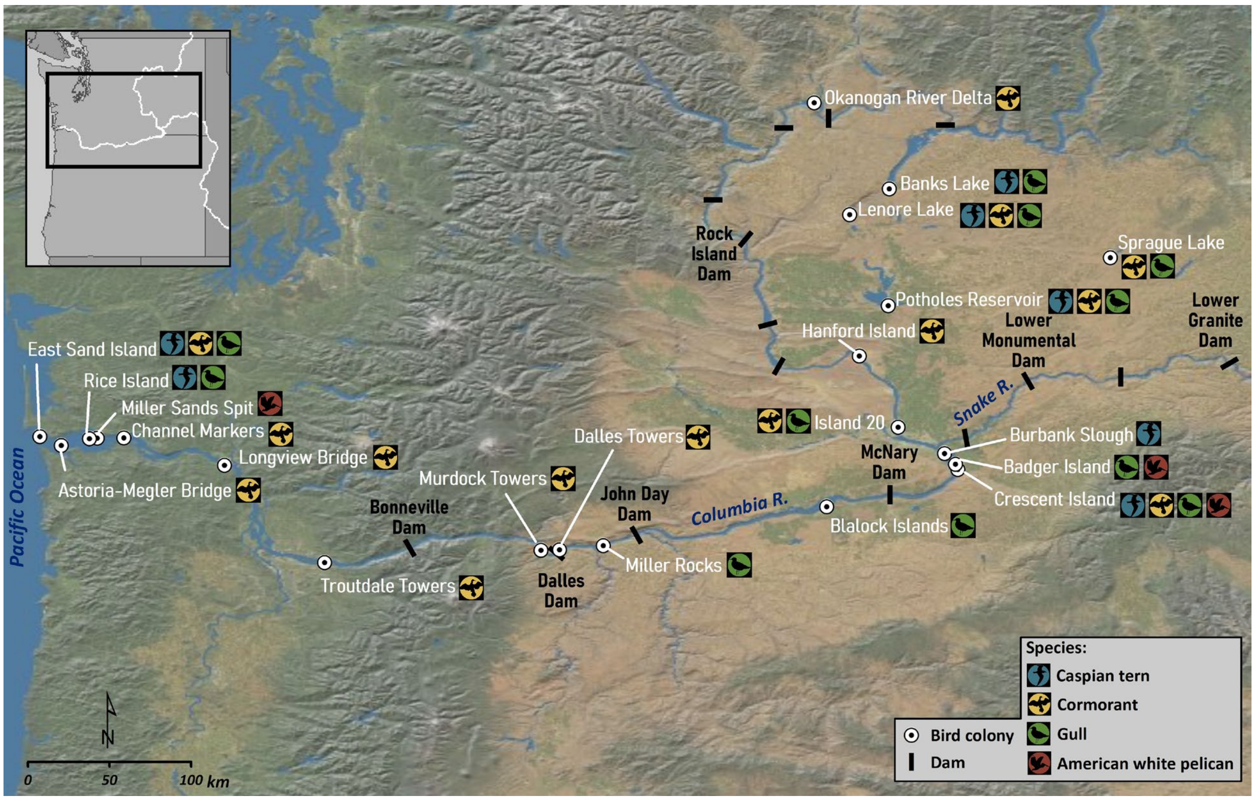
NEST DISSUASION ACTIVITIES

As part of the Inland Avian Predation Management Plan (IAPMP), nest dissuasion activities on Goose Island in Potholes Reservoir were conducted by the BOR and its contractor, U.S. Department of Agriculture-Animal and Plant Health Inspection Service-Wildlife Services (USDA-APHIS-WS). These activities were summarized in an annual report completed by the USDA (USDA-APHIS-WS 2024). The Fisheries Field Unit (FFU) at the USACE carried out nest dissuasion efforts associated with the Caspian Tern Management Plan in the Columbia River Estuary on ESI and as an adaptive management component on Rice Island. A description of those activities was summarized in reports produced by USACE (Roberts et al. 2025, Blair et al. 2025). Nest dissuasion activities were also performed as an adaptive management component to the IAPMP 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 an unnamed island in North Potholes Reservoir (USDA-APHIS-WS 2024) and Badger Island (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm). Finally, the gull colony on Miller Rocks was adaptively managed to prevent gull nesting and reduce the size of the colony over the course of the last three breeding seasons (T. De Boer, Yakama Nation, pers. comm.; see *below* for more details).

NESTING DISTRIBUTION & COLONY SIZE

Monitoring of piscivorous waterbird colonies in the CRB (*Map 1*) was conducted from the air (fixed-wing aircraft and unmanned aerial vehicles), 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 2024 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, Evans et al. 2024a).



Map 1. Study area in the Columbia River basin in 2024.

Periodic (bi-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 (17–18 May, 31 May – 1 June, 15–19 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 using high-resolution digital aerial imagery corresponding with the peak in nesting activity at each colony and enumerating visible birds using ArcGIS (ArcGIS Desktop: Release 10.8.1. and ArcGIS Pro 3.3.0, Redlands, CA). Colony size is reported as the number of birds on-colony, or, 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, the activities of terns and gulls were monitored biweekly throughout the breeding season. The remaining piscivorous waterbird colonies (*Map 1*) in the CPR were monitored one to two times per month throughout the breeding season. The cormorant colony on the AMB in the Columbia River estuary was monitored by boat on two occasions (29–30 May and 13–14 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 2024 (Roberts et al. 2025; Blair et al. 2025).

AVIAN PREDATION RATES

Using previously established methods, smolt passive integrated transponder (PIT) tags recovered (detected) on bird colonies were analyzed to estimate colony-specific predation rates (percentage of available tagged fish consumed by birds) on ESA-listed salmonid ESUs/DPSs. The study also investigated the cumulative effects of avian predation (predation from all colonies combined) on salmonid smolts and compared mortality due to avian predation to total mortality (1 - survival). Results provide information on the system-wide effects of avian predation and identify which predator species (terns, cormorants, gulls, pelicans) and colonies posed the greatest potential threat to smolt survival.

PIT-tagging of Upper Columbia River Steelhead

To help ensure adequate numbers of ESA-listed UCR steelhead were available for predation analyses, 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. Previous 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 provides 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).

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 (tagged and untagged passing RIS). All juvenile steelhead captured at RIS were part of the ESA-listed UCR steelhead DPS, as all hatchery and wild steelhead originating from tributaries upstream of RIS are part of the ESA-listed population (NOAA 2021). The sampling approach used at RIS as part of this study ensures that (1) steelhead from all 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, and (3) that smolts are tagged in-concert with the run at-large; criteria that allow us to make credible inference about the entire ESA-listed UCR steelhead population (see also Payton et al. 2020 and Evans et al. 2024a). The target sample size goal was to PIT-tag approximately 7,000 steelhead smolts for use in predation and survival analyses in 2024. 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, such as 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. (2015a) 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. This study employed 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:

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

and $\varepsilon_w \sim normal(0,\ \sigma_\theta^2)\ \forall\ w$. Non-informative priors were placed 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, or pelican) and are assumed to be mutually independent from colony to colony. For terns, α = 16.20 and β = 6.55 was assumed, for cormorants α = 15.98 and β = 15.29 was assumed, for gulls α = 33.71 and β = 183.61 was assumed, and for pelicans α = 6.70 and β = 7.37 was assumed.

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

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

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

For some colonies 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:

$$\dot{\theta_p} = \sum_p \frac{\sum_i (\theta_{ip} * n_i) / \sum_i (n_i)}{C_p}$$

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 for research conducted in 2024:

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 (2014–2024) at both managed and unmanaged tern colonies in the CPR. The IAPMP also uses the most recent 3-year average during the management period (currently 2022–2024) to further evaluate if management goals are being met or whether additional adaptive management is necessary (USACE 2014).

Estuary Caspian Tern and Double-crested Cormorant Management Plans – A stated goal of estuary tern and cormorant management plans was 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. During 2021–2024, the USACE Fisheries Field Unit (FFU) recovered smolt PIT tags from the tern and cormorant colonies on East Sand Island (Roberts et al. 2025) but estimates of ESU/DPS-specific predation rates and per capita predation rates have not been generated since 2020 (see also Appendix A).

As part of this study, smolt PIT tags recovered by the USACE-FFU on East Sand Island were incorporated to evaluate the cumulative effects of avian predation on select salmonid species and age-classes, those that must migrate through the forage range of multiple colonies, in 2024. Estimates of cumulative predation effects rely on a different analytical framework (see *Cumulative Predation and Survival* section) than those of colony-specific predation rates (see *Predation Rate Estimates* section), so the two estimates are not directly comparable to one another (Payton et al. 2019, Evans et al. 2021). Furthermore, several other ESA-listed salmonid ESUs/DPSs (e.g., those that originate from the middle and lower Columbia rivers) are susceptible to predation by East Sand Island terns and cormorants, but those impacts have not been documented since 2020.

Cumulative Predation and Survival: This study 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, subyearling Chinook, and sockeye that were tagged/detected at Lower Granite Dam (LGR) in 2024. Downstream river reaches were defined by locations where PIT-tagged smolts were redetected 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 is 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 2024 were compared with those from years past, including the results of previously published studies (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-class 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 naturally emigrating, in-river smolts within each river reach were included in analyses, 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, population, and age-class evaluated. As described above, 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 $j \in \{1,2, \dots , J - 1\}$ 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 $d \in \{1,2, \dots , D - 1 = 14\}$, 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 $d \in \{1,2, \dots , D - 1\}$ 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. This study defines $p_J = 0$, 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 $d \in \{1,2, \dots , D - 1\}$, 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 m is defined as 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

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

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:

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

Two further modelling considerations beyond those of Payton et al. (2019) were included to better inform the 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 2015–2024, with portions of each genus' colonies overlapping spatially creating a “mixed” or co-nesting area on BGI. This study 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, it was possible 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 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.

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, four Hamiltonian Monte Carlo (HMC) Markov Chain processes were run. 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. Specifically, 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 were 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 associated with this and other predation studies in CRB indicated that predation by terns was largely an additive source of 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). The finding that tern predation was an additive source of smolt mortality was consistent in both SR and UCR steelhead and from fish that were initially tagged/recaptures at RIS and Rocky Reach Dam on the middle Columbia River and Lower Granite Dam and Lower Monumental Dam on the lower Snake River (Payton et al. 2021, Evans et al. 2023). In 2024, the time series of UCR steelhead analyses was updated, with the focus on the efficacy of tern management actions to decrease predation and increase fish survival. Weekly and annual estimates of predation and survival probabilities on PIT-tagged UCR steelhead smolts that were tagged/recaptured at RIS were used to investigate to what degree tern predation was an additive source of mortality during smolt outmigration to BON from 2008–2024. The mark-recapture-recovery model of Payton et al. (2020) was used 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, letting

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

and

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

Therefore, within each year, simplex weekly rates of survival and aggregated predation are constructed as,

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

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

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

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

Independent of the variation described *above*, any further decrease/increase observed in survival is assumed to be proportional to the level of 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 $\theta_w^{\{PREDATOR\}}$ accounts for all unmeasured mortality, the relationship between survival and predation necessarily becomes directly inverse. In cases in which survival is zero, there is necessarily no relationship between predation and survival. Therefore, "observed" weekly survival, s_w^*, can be expressed as

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

The assumed similarity in baseline survival among years is modeled as $s_y^0 \sim normal(\mu_{s^0}, \sigma_{s^0})$ for each year y. Further, b is also assumed 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.

Finally, as part of this report, additive/compensatory effects from other predator species (cormorants and gulls) were briefly summarized based on previously published data from this and other studies (Payton et al. 2021, Evans et al. 2022b).

RESULTS & DISCUSSION

NEST DISSUASION ACTIVITIES

Columbia Plateau Region

The BOR and its contractor (USDA-APHIS-WS) installed passive nest dissuasion arrays (i.e., stakes, cable, and flagging) on upland habitat that was suitable for tern nesting on Goose Island in Potholes Reservoir prior to the onset of the 2024 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 necessary repairs to the passive nest dissuasion array as needed. USDA-APHIS-WS also used boats to patrol islands in northern Potholes Reservoir for terns and to use passive and active nest dissuasion techniques to prevent tern nesting in these areas, as warranted. Monitoring efforts in the present study discovered a tern and gull colony on an unnamed island in northern Potholes Reservoir that was beyond the area surveyed by the USDA-APHIS-WS in 2024. The research team located the colony during an aerial survey on May 18, 2024, and immediately notified USDA-APHIS-WS of its existence. Within 48 hours of notification, USDA-APHIS-WS deployed their team to remove tern eggs (under permit) and install passive dissuasion (i.e., flagging, fencing, and a coyote effigy) on the island, with no (zero) breeding terns observed at the site for the remainder of the breeding season. The gull colony, which surrounded the tern colony, continued to nest on the island following installation of tern dissuasion arrays. 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 2022, 2023, 2024).

The growth of willows planted on Crescent Island in 2016 had mostly eliminated 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 dieback of willows and other native vegetation on Crescent Island, leaving large areas unvegetated, and hence, suitable for tern nesting (see *below* for further discussion). Prior to the 2023 and 2024 breeding seasons, 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 try to prevent tern nesting. In both years, however, terns successfully nested in areas adjacent to woody debris. No active hazing of prospecting terns during the breeding season has been conducted on Crescent Island since 2015, the first year of management actions on Crescent Island.

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 2021–2024, 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. In 2022–2024, agency personnel placed large woody debris on nesting areas used by terns on Badger Island (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm.).

Over concerns of its impact to a cultural site of the Yakama people and the survival of juvenile salmonids, the gull colony on Miller Rocks was adaptively managed to prevent gull nesting and reduce the size of the colony in 2022–2024. A variety of nest dissuasion methods were used by the Yakama Nation and its contractors, including active hazing, air cannons, and pyrotechnics. Limited lethal take (under permit) was implemented in 2024 to reinforce non-lethal techniques (T. De Boer, Yakama Nation, pers. comm).

Columbia River Estuary

As was the case in 2020–2023, 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 2024 (Roberts et al. 2025 and Blair et al. 2025). 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 have been performed at the cormorant colony on ESI since 2018 (Blair and Tidwell 2024). For further details on avian predation monitoring and management efforts on ESI and Rice Island in 2024, see Roberts et al. (2025) and Blair et al. (2025).

While active hazing of cormorants was conducted by USDA-APHIS-WS on two spans (166–167) of the AMB for maintenance and painting in 2023, there was no hazing of cormorants on the AMB in 2024 (M. Alex, USDA-APHIS-WS, pers. comm.). Cormorants reestablished nesting sites on locations where hazing occurred in 2023 during the 2024 breeding season. Nesting habitat for cormorants on the bridge is currently not limiting, as there continues to be more available habitat that the birds are not using.

NESTING DISTRIBUTION & COLONY SIZE

A total of 37 active tern, cormorant, pelican and gull colonies were detected in the CRB during the 2024 breeding season (*Map 1*). Of those, cormorant and gull colonies were the most prevalent (14 and 12 colonies, respectively), followed by tern and pelican colonies (8 and 3 colonies, respectively; *Table 1* and *Table 2*). Most of these breeding colonies (26) were in the CPR, with 8 and 3 colonies located in the CRE and LCR, respectively (*Map 1*). 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. 7,979 breeding pairs), pelicans (ca. 4,253 individuals), and terns (ca. 1,963 breeding pairs). Aerial imagery and counts from this study have been published (Banet et al. 2025) and are available from Dryad at https://doi.org/10.5061/dryad.k98sf7mht.

Colony size estimates suggest that while management actions have achieved their objective of reducing Caspian tern and double-crested cormorant nesting populations in the Columbia River Basin, these reductions appear to have contributed to broader declines in breeding populations of both species throughout the Pacific Flyway. This raises concerns about the conservation status of these populations, especially for the rapidly declining Pacific Flyway population of terns. Whether terns and cormorants from the CRB have dispersed to locations outside of the CRB cannot be fully assessed within the scope of current monitoring efforts.

Table 1. Annual colony size (peak number of breeding pairs) for Caspian terns in the Columbia River basin during 2008–2024. "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 2020–2024 were provided by the U.S. Army Corps of Engineers (see also Roberts et al. 2025; Blair et al. 2025 for more details).

Colony	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Columbia River Estuary																	
East Sand Is. (Columbia River)	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	1,524
Rice Is. (Columbia River)[1]	0	0	0	0	0	0	0	0	0	0	0	0	0	Active	Active	Active	Active
Columbia Plateau Region																	
Goose Is. (Potholes Reservoir)	293	487	416	422	463	340	159	2	0	0	0	0	6	22	16	12	20
Unnamed Is. (North Potholes Res.)	0	0	0	0	0	0	0	0	144	0	0	0	0	0	0	0	126
Crescent Is. (Columbia River)	388	349	375	419	422	393	474	0	0	0	0	0	0	1	149	88	186
Burbank Slough (Columbia River)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	23
Blalock Islands (Columbia River)	104	79	136	20	6	26	45	677	483	449	313	379	150	0	0	0	0
Badger Is. (Columbia River)	0	0	0	33	60	0	0	0	0	41	8	0	0	231	267	274	0
Twinning & Goose Is. (Banks Lake)	27	61	34	19	22	13	67	64	6	0	0	0	0	0	0	7	15
Harper Is. (Sprague Lake)	11	4	4	4	30	1	8	10	3	92	79	18	0	85	2	10	0
North Rock/Shoal Is. (Lenore Lake)	0	0	0	0	0	0	0	0	0	123	91	48	53	61	76	81	69
Total (Columbia River Estuary)	**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**	**1,524**
Total (Columbia Plateau Region)	**823**	**980**	**965**	**917**	**1,003**	**773**	**753**	**753**	**636**	**705**	**491**	**445**	**209**	**400**	**510**	**472**	**439**
Grand Total (Columbia River Basin)	**11,491**	**10,834**	**9,248**	**7,886**	**7,419**	**8,160**	**7,022**	**6,993**	**6,551**	**4,205**	**5,450**	**4,306**	**2,596**	**2,450**	**2,235**	**996**	**1,963**

[1]Although Caspian terns were successfully dissuaded from establishing a colony on Rice Island in most years, roosting, loafing, and attempted nesting by terns were observed during 2021–2024 but accurate estimates of the number of breeding pairs were not available.

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 2024. "Active" denotes a colony that was active but not counted in 2024.

Colony	LAXX [1]	DCCO [2]	AWPE [1]
Columbia River Estuary			
East Sand Is.	Active [3]	Active [4]	
Astoria-Megler Bridge		5,348	
Rice Is.	Active [3]		
Channel Markers [5]		337	
Miller Sands Spit			997
Lower Columbia River			
Channel Markers [6]		124	
Longview Bridge		334	
Troutdale Transmission Towers		371	
Columbia Plateau Region			
Murdock Towers (Columbia River)		37	
The Dalles Transmission Towers (Columbia River)		36	
Miller Rocks (Columbia River)	3,708		
Blalock Is. (Columbia River)	1,192		
Crescent Is. (Columbia River)	3,739	629	51
Badger Is. (Columbia River)	907		3,205
Island 20 (Columbia River)	7,559	44	
Hanford Reach (Columbia River)		99	
Okanogan River Delta (Columbia River)		54	
Goose Is. (Potholes Reservoir)	13,642	74	
North Potholes Reservoir (Potholes Reservoir)	336	35	
North Rocks, Shoal Is., and unnamed island (Lenore Lake)	1,159	155	
Harper Is. (Sprague Lake)	3,059	302	
Twinning Is. and Goose Is. (Banks Lake)	5,001		

[1] *Number of individuals.*
[2] *Number of breeding pairs.*
[3] *Includes glaucous-winged/western and a large but unquantified number of ring-billed gulls on Rice Island. Data provided by the U.S. Army Corps of Engineers – Fisheries Field Unit (see Blair et al. 2025 and Roberts et al. 2025).*
[4] *Data provided by the U.S. Army Corps of Engineers – Fisheries Field Unit. Reported as number of nests (see Roberts et al. 2025)*
[5] *Channel Markers located from river kilometer 0 to river kilometer 53; see Appendix E for counts by channel marker.*
[6] *Channel Markers located from river kilometer 127 to river kilometer 218; see Appendix E for counts by channel marker.*

Columbia Plateau Region

Caspian Tern Colonies: In 2024, terns nested at six locations in the CPR, including Goose Island in Potholes Reservoir, an unnamed island in northern Potholes Reservoir, Crescent Island in McNary Reservoir ("managed colonies" as part of the IAPMP; USACE 2014) and at three other unmanaged colony sites (Burbank Slough in McNary National Wildlife Refuge, Shoal Island in Lenore Lake, and Goose Island in Banks Lake; *Table 1*). Adaptive management prevented terns from nesting at Badger Island in McNary Reservoir by placing woody debris on tern nesting habitat and at the Blalock Islands in John Day Reservoir by raising reservoir elevation to inundate all previously used nesting habitat (B. Parker, Columbia River Inter-Tribal Fish Commission, pers. comm. and J. Macdonald, USACE, pers. comm.).

A relatively small colony of terns nested on the upper portion of Goose Island and an adjacent islet to Goose Island in Potholes Reservoir with a peak count of 20 pairs in June of 2024 (*Table 1* and *Figure 1*). Smaller tern colonies attempted nesting at multiple sites on Goose Island, although none were successful due to active and passive management. While terns were observed roosting and loafing along the shoreline and other visible upland areas of Goose Island in Potholes Reservoir, lower numbers were observed in 2024 compared to previous years during monitoring in May and June (Evans et al. 2024a). This was consistent with observations of another contractor that observed the highest daily counts of terns on Goose Island in July of 2024 (USDA-APHIS-WS 2024). Declines in Potholes Reservoir water levels were similar to previous years, exposing more shoreline habitat as the breeding season continued. However, terns did not attempt to nest on shoreline habitat of Goose Island in 2024 and no additional passive dissuasion was added to these areas (USDA-APHIS-WS 2024). Lower nest counts on Goose Island in 2024 were likely due to continued passive and active nest dissuasion efforts (USDA-APHIS-WS 2024) and the establishment of a northern Potholes Reservoir colony, where most tern nesting attempts in Potholes Reservoir were observed (126 pairs) prior to dissuasion efforts that eliminated the colony in mid-May (see *below* for details). Since the IAPMP commenced in 2014, tern colonies on islands in northern Potholes Reservoir have been observed twice, in 2016 and in 2024 (*Figure 1*).

The total of 146 pairs of terns nesting at Goose Island and the unnamed island in northern Potholes Reservoir in 2024 exceeded the threshold colony size stipulated in the IAPMP of 40 breeding pairs (USACE 2014). There is a possibility that some terns dissuaded from the unnamed island in northern Potholes Reservoir re-nested at Goose Island in June or July. In 2024, passive and active nest dissuasion occurred on Goose Island throughout the breeding season and an unnamed island in northern Potholes Reservoir, once it was identified, along with the collection of tern eggs (n = 63) under BOR's depredation permit (USDA-APHIS-WS 2024). No tern fledglings were observed on Goose Island by the monitoring team from the present study in 2024, suggesting that the tern nesting attempts on Goose Island during the historic breeding season were unsuccessful. However, after the final aerial survey in late-June, some additional tern nesting attempts were observed and documented through July of 2024 by USDA-APHIS-WS (2024).

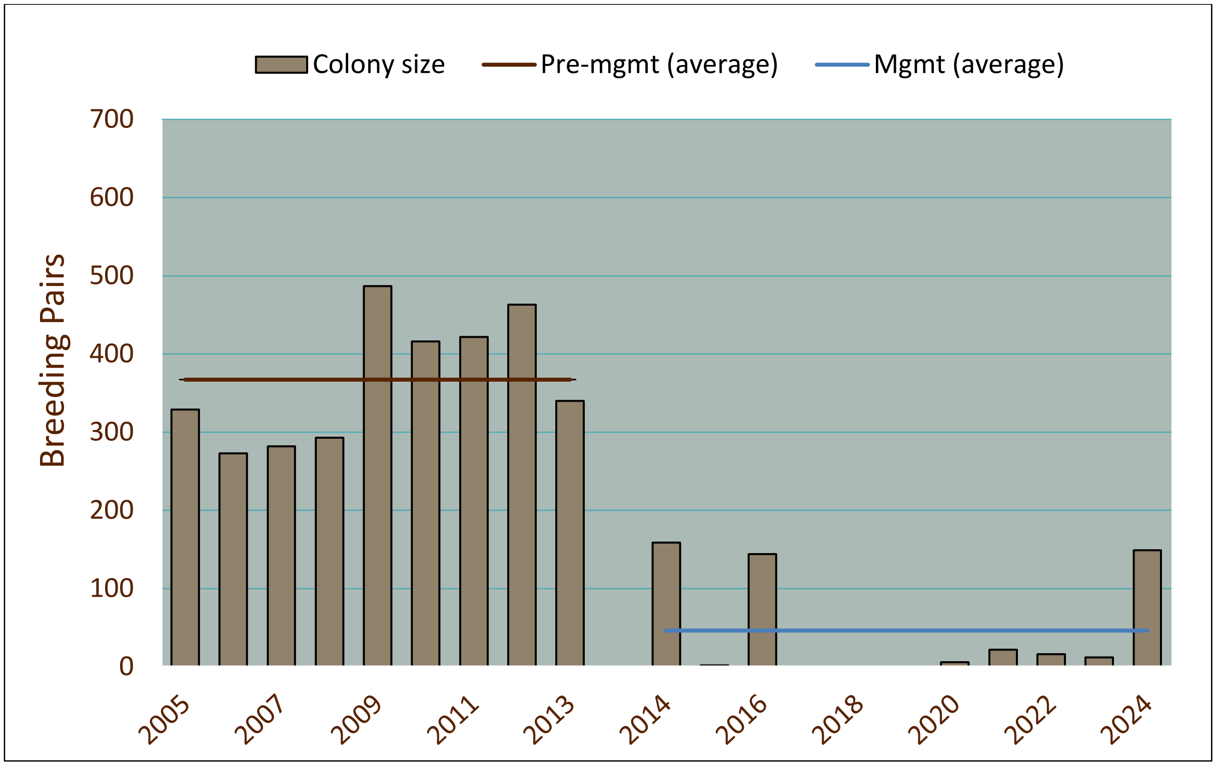


Figure 1. Annual colony size (number of breeding pairs) of Caspian terns at Goose Island and other islands in Potholes Reservoir before tern management (Mgmt; 2005–2013) and during tern management (2014–2024). In 2016 and 2024, terns nested on unnamed islands in northern Potholes Reservoir. Terns nested exclusively on Goose Island or surrounding islets in all other years.

Prospecting terns were documented at five different sites, four of which were in northern Potholes Reservoir, during the 2024 breeding season (USDA-APHIS-WS 2024), the same number of sites used by prospecting terns in 2023 (USDA-APHIS-WS 2024). Active hazing, passive dissuasion, and egg collection (under permit) was required at one site in northern Potholes Reservoir since a tern colony of 126 pairs was established. Terns continued to show strong fidelity to Goose Island and other islands in Potholes Reservoir, seeking out nesting habitat without dissuasion. This suggests that, without ongoing adaptive management, terns could continue to reestablish colonies at potential colony sites throughout Potholes Reservoir in the future.

An unexpectedly large colony of terns reformed on Crescent Island in McNary Reservoir during 2022–2024. The re-establishment of a tern colony on Crescent Island was facilitated by the recent reduction in vegetative cover on the island. Gulls may also provide social attraction to nesting terns. 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 are suitable for tern nesting (see also Methods & Analysis, Nest Dissuasion *above*).

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, 2023, and 2024, the tern colony increased in size to 149, 88, and 186 breeding pairs, respectively (*Figure 2*). In the absence of active nest dissuasion and larger areas of non-vegetated habitat, the tern colony on Crescent Island is expected to continue to grow in the future, perhaps to levels observed prior to management.

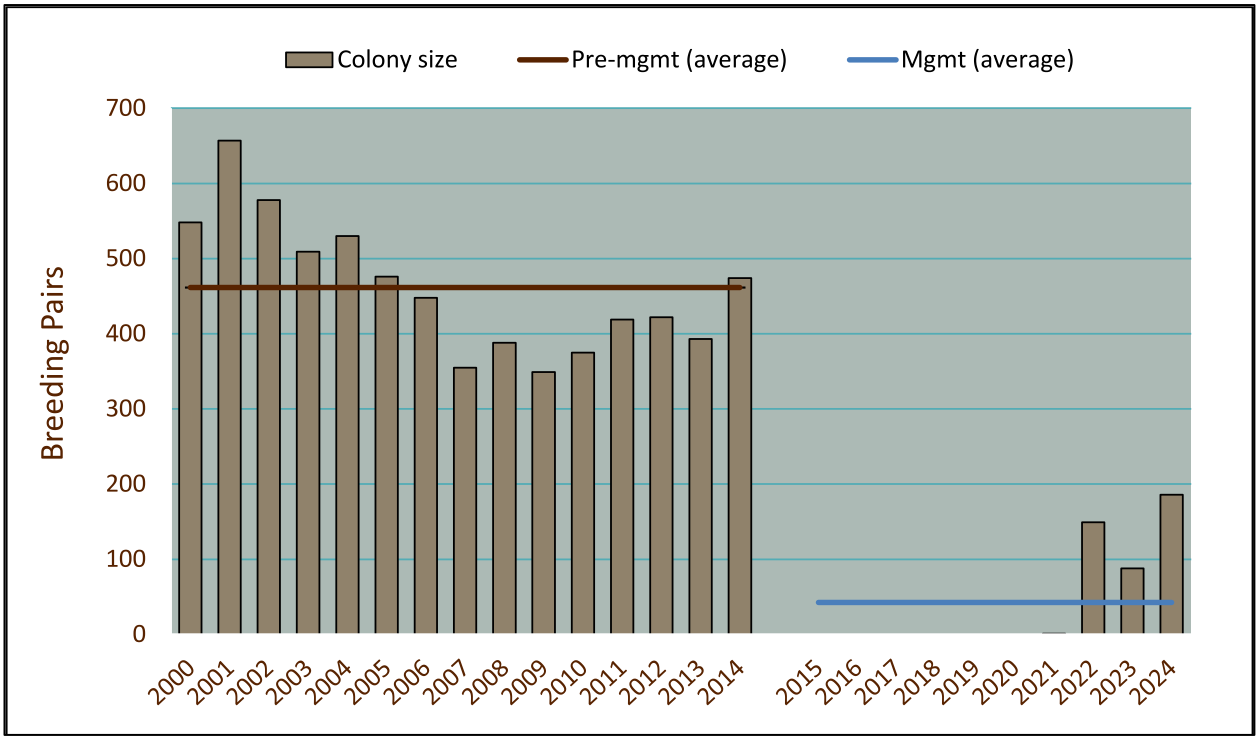


Figure 2. Annual colony size (number of breeding pairs) of Caspian terns at Crescent Island in the mid-Columbia River before tern management (2005–2014) and during tern management (2015–2023). No terns nested at Crescent Island during 2015–2020; one tern breeding pair successfully nested on Crescent 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. During the 2021–2024 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. 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*), to Badger Island in 2021–2023, and most recently (2024) to Burbank Slough in McNary National Wildlife Refuge where terns briefly attempted to nest. Adaptive management to prevent/reduce tern nesting on Badger Island was implemented 2022–2024 by placing woody debris on areas used by nesting terns. That effort was successful in preventing tern nesting on Badger Island in 2024, but not in 2022 and 2023, where colonies persisted despite adaptive management efforts.

A total of 439 breeding pairs of terns nested in the CPR in 2024 (*Table 1*, *Figure 3*), slightly lower than the number of terns that nested in the CPR in 2023 (472 breeding pairs), but more than double the maximum regional population size stipulated in the IAPMP (200 breeding pairs; USACE 2014). The regional population size in 2024 represented a 50% reduction in the size of the regional tern breeding population compared to the pre-management average (875 breeding pairs), but the population has remained between 400–510 breeding pairs over the past four years (*Figure 3*). While there is evidence that some terns displaced from the CPR have relocated to colonies outside of the basin (Lawes et al. 2021a, Banet et al. 2024), 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).

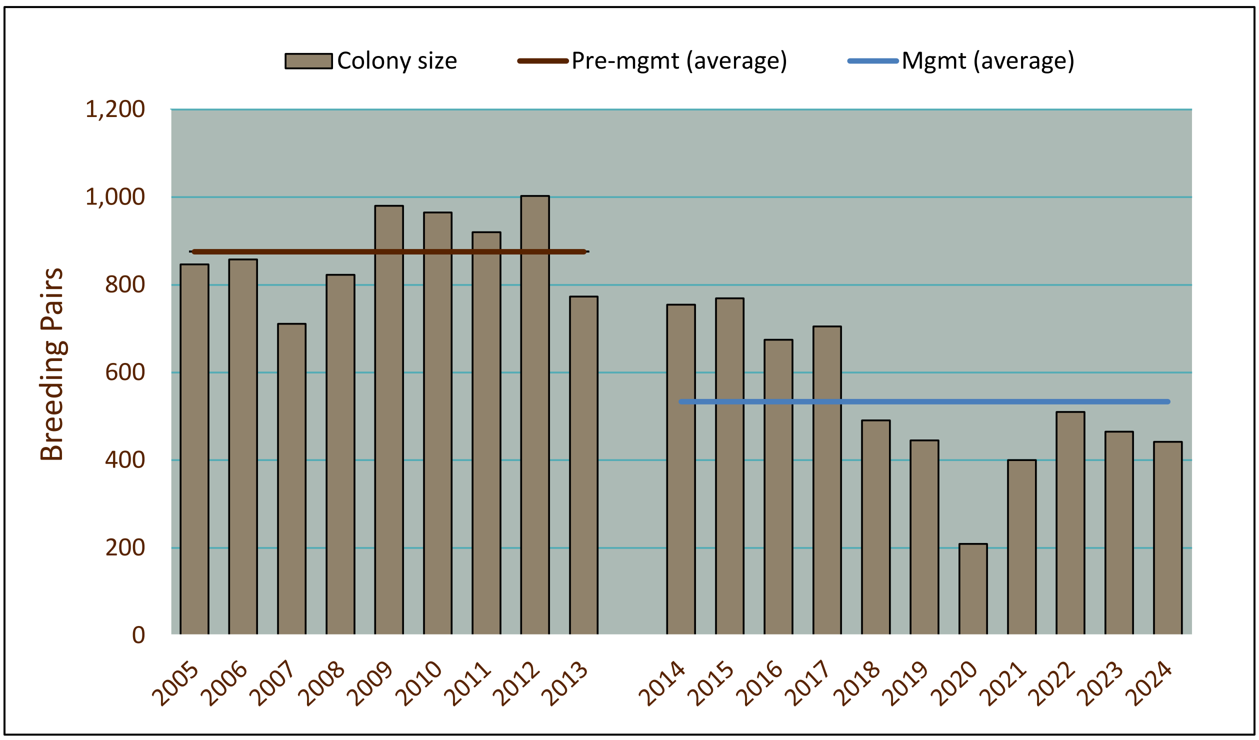


Figure 3. Annual and average number of Caspian tern breeding pairs nesting at all known colonies before tern management (2005–2013) and during tern management (2014–2024) in the Columbia Plateau region.

Other Piscivorous Waterbird Colonies: A total of ten active California and ring-billed gull colonies were detected in the CPR region in 2024, ranging in size from ca. 300 individuals (North Potholes Reservoir) to over 14,000 individuals (Goose Island in Potholes Reservoir; *Table 2*). Nesting double-crested cormorants were confirmed at ten colonies in the CPR in 2024, with colony sizes ranging from 36 breeding pairs (The Dalles transmission towers) to 629 breeding pairs (Crescent Island; *Table 2*). Finally, the size of the American white pelican colony on Badger Island was estimated to be ca. 3,200 individuals in 2024 (*Table 2*), an increase in size relative to 2023 (2,593), but lower than colony sizes in 2022 and 2021 (3,486 and 3,624 individuals, respectively; Evans et al. 2023). A small colony (51 individuals) of pelicans also formed on Crescent Island in 2024 (*Table 2*), a historical colony site that has not been active since 2010 (Roby et al. 2021)

Lower Columbia River

In 2024, active double-crested cormorant nesting colonies were identified below Bonneville Dam on the LCR, including the Longview Bridge (also referred to as the Lewis and Clark Bridge; 334 breeding pairs), transmission towers near Troutdale, Oregon (371 breeding pairs), and numerous channel markers located between river kilometer 127–218 (124 pairs; *Table 2* and *Appendix E*). The Lewis and Clark Bridge apparently increased in size compared to 2023 (Evans et al. 2023). Boat counts proved to be more accurate than aerial counts at that site since cormorants were observed nesting immediately below the road deck—nests not visible from aerial imagery. This new counting method likely contributed to higher counts at Lewis and Clark Bridge in 2024. All sites in the LCR continue to support nesting habitat for the dispersal of cormorants away from ESI and additional nesting habitat is available on Lewis and Clark Bridge, Troutdale Transmission Towers, and channel markers in the freshwater zone of the LCR.

Columbia River Estuary

Caspian Tern Colonies: The USACE-FFU estimated that 1,524 breeding pairs of terns nested on the prepared 1-acre colony site on ESI in 2024 (*Table 1*; see also Roberts et al. 2025). Efforts to prevent terns from nesting outside the designated 1-acre colony area on ESI were successful in 2024 (Robert et al. 2025). However, the tern colony on ESI was well below the target colony size identified in *Estuary Tern Management Plan* of 3,125–4,275 pairs (USFWS 2008), raising concern regarding the status and sustainability of the Pacific Flyway breeding population of terns. For the first time since 2019, there was evidence of tern productivity (fledglings) on ESI but counts were less than 200 chicks, with an unknown number of fledglings produced in 2024 (Roberts et al. 2025). Nesting phenology was also delayed in 2024 by approximately 2 months, with the largest number of terns observed in late July as opposed to the traditional peak in late May (Roby et al. 2021). Complete nesting failure has been observed at the ESI colony in six out of the last nine years (2016 –2024). 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. During the same period, terns were counted on Rice Island, which is located in the freshwater zone of the estuary (Blair et al. 2025). 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 successful in dispersing the terns from the incipient colony site on Rice Island in 2024 (see Roberts et al. 2024 and Blair et al. 2024 for further details). Terns were also observed at Port of Astoria Piers and at piers at Tongue Point, although no nesting was confirmed at these locations.

Double-crested Cormorant Colonies: Double-crested cormorant colonies in the CRE in 2024 included the AMB and birds breeding on numerous channel markers (n=10) located between river kilometers 0 and 53 (*Table 2 and Appendix E*). Although double-crested cormorants were observed on the historical colony site on ESI during the breeding season, the USACE-FFU reported no (zero) breeding pairs of cormorants in 2024 (Roberts et al. 2025). This marks the sixth year with little or no cormorant nesting on ESI (USACE-FFU 2020, 2021, 2022a; Blair et al. 2024; Roberts et al. 2025). 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 5,348 breeding pairs in 2024 (*Table 2*), the largest number observed to-date (Evans et al. 2024a). Without efforts to both restore the double-crested cormorant colony on ESI and dissuade cormorants from nesting on the bridge, it is expected that the AMB colony will 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 2–5 times higher than those of cormorants nesting on ESI (Cramer et al. 2021b; Evans et al. 2022b; Evans et al. 2024a, and *below*).

Other Piscivorous Colonial Waterbird Colonies: Both glaucous-winged/western gulls and ring-billed gulls nested in the Columbia River estuary in 2024. 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 in 2023 (>3,000 individuals; Blair et al. 2024). A large colony of ring-billed gulls were again present on Rice Island in 2024, but the status of the colony on ESI in 2024 is unknown. As in previous years, glaucous-winged/western gulls nested at dispersed colonies on ESI and Rice Island. Although estimates of colony size in 2024 are unavailable, in past years these colonies numbered in the 1,000s (Roby et al. 2021a).

In addition to double-crested cormorants (DCCO) nesting on the AMB, Brandt's cormorants (BRAC) and pelagic cormorants (*Phalacrocorax pelagicus*; PECO) also nested on the bridge in 2024. Peak counts for BRAC were 1,214 breeding pairs and for PECO were 106 breeding pairs, similar counts for both species compared with 2023 (Evans et al. 2024a). The BRAC colony in the AMB is the largest in the CRE and amongst the largest in the Pacific Northwest. Brandt's cormorants also nested on ESI in 2024, which was formerly the largest colony site in the CRE. Estimates of the number of BRAC pairs on ESI in 2024, however, were not available due to the inability to distinguish BRAC from DCCO in aerial imagery acquired and analyzed by the USACE-FFU (Robert et al. 2025).

There was one active American white pelican colony in the CRE in 2024 on Miller Sands Spit (*Table 2*). The Miller Sands Spit pelican colony is one of only three pelican colonies in the CRB; the other two were on Badger Island and Crescent 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 2024, the colony on Miller Sands Spit was estimated at 997 individuals (*Table 2*), a decrease of several hundred individuals compared to 2023 (Evans et al. 2024a).

AVIAN PREDATION RATES

PIT-tagging of Upper Columbia River Steelhead

A total of 3,800 UCR steelhead were sampled at RIS and available for predation and survival analyses in 2024, which includes 3,472 newly tagged smolts and 328 recaptured smolts (i.e., previously tagged). The number of steelhead smolts available for predation analyses in 2024 was well below the target sample size goal of approximately 7,000. This was due to a record low number of steelhead smolts being collected at the RIS fish trap in 2024 (n=3,884) compared with the annual average of 13,050 smolts (annual range=7,168 to 30,368) during 1997–2023 (Columbia Basin Research 2024). Of the tagged smolts from 2024, 3,038 and 762 were classified as hatchery and wild, respectively (see also *below* and *Appendix B*). Steelhead were tagged and released from 12 April to 13 June, a period which accounted for >98% of all steelhead encountered in the trap in 2024. Mean fork length was 197 mm (standard deviation [SD]=25 mm; range=90–303 mm). An evaluation of external fish condition indicated that most steelhead were in good over-all external condition in 2024, with 5.4% 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. 2024a). The most common type of anomaly in 2024 was descaling, followed by body injuries. Previously published studies have investigated whether steelhead smolts with signs of external injuries/damage/disease were more susceptibility to avian predation than seemly heathy or uncompromised smolts. Results indicated that compromised smolts were more likely to be consumed by terns and gulls than uncompromised smolts. Differences, however, were often small, with compromised smolts only slightly more likely to be consumed (e.g., 1.1–1.3 times more likely) than compromised smolts and that large numbers of uncompromised smolts were also being consumed by avian predators (Hostetter et al. 2012, Evans et al. 2019, and Hostetter et al. 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. In total, 4 (or 0.5%) of the 766 presumed wild steelhead sampled at RIS had a CWT, fish that were reclassified as hatchery. In 2023, 3 (or 0.1%) of the 2,155 presumed wild steelhead sampled at RIS had a CWT. To ensure the CWT detector/antenna was functioning, a subsample of 50 hatchery steelhead were checked each week for the presence of a CWT. In total, 60 (or 17.6%) of the 340 hatchery steelhead scanned for a CWT had a detectible CWT, indicating the antenna was functioning. Of the 3,038 fish classified as hatchery, 2,465 had their adipose fin clipped and 573 had fin erosion or other signs indicative of a hatchery environment. Collectively, results provide evidence that steelhead classified as wild in the RIS trap were likely wild, but the possibility remains that some hatchery fish could have been misclassified as wild (see also *Appendix B*).

PIT Tag Recovery

A total of 32 genera-specific avian colonies in the CRB were scanned for fish PIT tags following the 2024 breeding season. Colonies included 7 tern colonies, 6 gull colonies, 11 cormorant colonies, 4 great blue heron (*Ardea herodias)* and/or great egret (*Ardea alba*) colonies, 2 Brandt's cormorant colonies, and 2 pelican colonies (*Table 3*). Scanning was also conducted at multiple avian loafing/roosting sites, areas where large numbers of piscivorous colonial waterbirds were observed during the smolt out-migration period. A total of 30,286 PIT tags from 2024 migration year smolts (Chinook Salmon, Coho Salmon, Sockeye Salmon, and steelhead combined) were recovered in 2024 (*Table 3*). Of these, more than 90% were tags recovered from breeding sites, with the largest number recovered on a mixed gull and pelican colony sites on Badger Island (n=6,508), followed by cormorant colonies on Crescent Island (n=3,662), Troutdale Towers (n=3,004) and the Astoria-Megler Bridge (n=2,253). Scanning at AMB was limited to areas accessible to researchers following the breeding season (see Methods & Analysis section), resulting in a subsample of all available nests at that site (*Table 3*). Relatively large numbers of tags were also recovered from the gull colony at Miller Rocks (n=2,166) and Island 20 (n=1,921; *Table 3*). Small numbers of smolt PIT tags were recovered on the Harper Island cormorant colony (n=44), a mixed double-crested and Brandt's cormorant colony on ESI (n=17), the Rice Island gull colony (n=33), along with several other, smaller-sized colonies in 2024 (*Table 2, Table 3).*

A total of 60 tags from other fish species and salmonid age-classes were recovered on avian colonies in 2024. This included PIT tags from 19 adult Sockeye Salmon, 5 adult steelhead, 5 adult Chinook Salmon, 7 Northern Pikeminnow (*Ptychocheilus oregonensis*), 3 Pacific Lamprey (*Entosphenus tridentatus*), 1 Bull Trout (*Salvelinus confluentus) and 1* White Sturgeon *(Acipenser transmontanus)* from colonies in the CPR (*Table 3*). A total of 19 tags from Sea-run Cutthroat Trout (*O. clarkii clarkii*) were also recovered on avian colonies in the CRE (*Table 3*).

Finally, for the second consecutive year, PIT tags were recovered from select heron colonies or rookeries in the CRB. The primary impetus for scanning these sites was 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, a less studied avian predator species (Cramer et al. 2021a). A lack of information on PIT tag detection and deposition probabilities at these sites, however, precluded an analysis of predation rates by heron colonies in both 2023 and 2024 (see also PIT Tag Detection and Deposition Probabilities section). With this caveat in mind, just 23 smolt PIT-tags were recovered from heron colonies downstream of Bonneville Dam near the Port of Longview but a relatively large number of tags (n=453) were recovered from a heron colony on Island 20 in the middle Columbia River (*Table 3*). Interestingly, the majority (n=308 or 68%) of the tags recovered on the Island 20 heron colony were from fish that originated from the Yakima River, which enters the middle Columbia River downstream of Island 20. By comparison, of the tags recovered on the gull colony on Island 20, 159 or 8% of all tags (n=1,921) were from Yakima River smolts. Results suggest that herons were disproportionately foraging on smolts from the Yakima River relative to gulls that also nested on Island 20.

Table 3. Numbers of 2024 migration year smolt PIT tags (Chinook Salmon, Sockeye Salmon, Coho Salmon, and steelhead combined) and tags from other fish species recovered at avian breeding/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 egrets (GREG), great blue herons (GBHE), Brandt's cormorants (BRAC), and American white pelicans (AWPE). 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]
Harper Island (Sprague L.)	Off-river	DCCO	Breeding	44	
Goose Island (Banks L.)	Off-river	CATE	Breeding	30	
Okanogan River Mouth	859	DCCO	Breeding	519	4
Goose Island (Potholes R.)	Off-river	CATE	Breeding	134	
		DCCO/GREG	Breeding	21	
		Mixed	Loafing	9	
Unnamed Island (Potholes R.)	Off-river	CATE	Breeding	620	
Lenore Lake	Off-river	CATE	Breeding	321	
Hanford Island	592	DCCO	Breeding	203	
		Mix	Loafing	68	
Island 20	549	LAXX	Breeding	1,921	
		DCCO	Breeding	313	
		GREG/GBHE	Breeding	453	
		Mix	Loafing	8	
Foundation Island	518	Mixed	Loafing	36	
Badger Island	512	LAXX	Breeding	156	
		AWPE/LAXX	Breeding	6,508	30
		Mixed	Loafing	153	
Burbank Slough	Off-river	CATE	Breeding	182	
Crescent Island	510	CATE	Breeding	887	
		LAXX	Breeding	1,356	
		DCCO	Breeding	3,662	1
		AWPE	Breeding	24	
Blalock Islands	441-439	LAXX	Breeding	740	
		Mixed	Loafing	132	
Miller Rocks	331	LAXX	Breeding	2,166	
		Mixed	Loafing	23	
Murdock Towers	299	DCCO	Breeding	536	1
		Mixed	Loafing	257	
Troutdale Towers [2]	189	DCCO	Breeding	3,206	4
Lewis & Clark Bridge [3]	106	DCCO	Breeding	195	1
Port of Longview	107	GBHE	Breeding	23	3
Rice Island	34	LAXX	Breeding	33	
		LAXX/CATE	Breeding	75	1
Channel Markers	33-218	DCCO	Breeding	163	
Astoria-Megler Bridge [3]	23	DCCO [3]	Breeding	2,253	9
		BRAC [3]	Breeding	78	
		Mixed [3]	Breeding	280	
		Mixed [3]	Loafing	1,413	
Piers 1, 3, & 5	21-28	Mixed	Loafing	98	
East Sand Island [4]	8	CATE	Breeding	1,172	6
		DCCO/BRAC	Breeding	17	
Total				30,488	60

[1] Includes adult Sockeye (n=19; Badger Island), steelhead (n=5; Badger Island), and Chinook (n=5; Badger Island); bull trout (n=1; Badger Island); Northern Pikeminnow (n=7; Astoria-Megler Bridge, Lewis & Clark Bridge; Troutdale and Murdock towers), Pacific Lamprey (n=3; Okanogan), White Sturgeon (n=1; Okanogan), and sea-run cutthroat (n=19; East Sand Island, Astoria-Megler Bridge, Rice Island, Port of Longview, Troutdale Towers, and Crescent Island).

[3] Includes tags (n=202) recovered by BPA in tower nests. 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 additional information for use in smolt survival models, we recovered smolt PIT tags from several additional avian breeding and loafing sites that were not included in the original scope of work for this study. In 2024, scanning associated with this additional effort resulted in the detection of 8,408 PIT tags from 2024 migration year smolts that survived outmigration to below Bonneville Dam prior to being consumed by a bird. This information, coupled with 2024 smolt tags recovered by the USACE-FFU on the ESI tern and cormorant colonies (n=1,189), tags in live fish detected at antennas on pile dikes in the CRE (n=11,815), and tags of live fish detected by the National Marine Fisheries Service's net trawl in CRE (n=12,882), were used by this project and other projects (e.g., NOAA Fisheries Survival Study) to generate more accurate and precise estimates of smolt survival to Bonneville Dam (see also Hostetter et al. 2018, Payton et al. 2019, Payton et al. 2023).

PIT Tag Detection & Deposition Probabilities

Table 4 provides results of the probability that a consumed PIT tag by a breeding bird from a given colony 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 2024 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 deposited during the breeding season (*Table 4*).

No information on deposition probabilities were available for the heron colonies scanned for smolt PIT tags (*Table 3*) and, as such, predation rate estimates were not generated for these colonies. Similarly, detection probabilities were not available for cormorants breeding on channel markers and the Lewis and Clark Bridge in the CRE. Finally, with the exception of the large mixed gull and pelican colony on Badger Island (see Methods), predation rate estimates were not available for mixed species breeding sites due to uncertainty associated with which species consumed the fish and/or lack of information on PIT tag deposition and detection probabilities at these sites.

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 2024. Estimates were used to generate predation rates based on the number of smolt PIT tags recovered on each colony 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]
Sprague Lake	Off-river	DCCO	0.96–0.99	0.51 (0.34–0.70)
Banks Lake	Off-river	CATE [2]	1.0	0.71 (0.51–0.89)
Lenore Lake	Off-river	CATE	0.64–0.76	0.71 (0.51–0.89)
Potholes Reservoir	Off-river			
Goose Island		CATE [2]	0.31–0.96	0.71 (0.51–0.89)
Unnamed Island		CATE	0.68	0.71 (0.51–0.89)
Hanford Island	592	DCCO	0.36–0.60	0.51 (0.34–0.70)
Island 20	549	LAXX	0.88–0.96	0.15 (0.11–0.21)
		DCCO	0.84	0.51 (0.34–0.70)
Burbank Slough	Off-river	CATE	0.99	0.71 (0.51–0.89)
Badger Island	512	AWPE	0.56–0.90	0.47 (0.24–0.73)
		LAXX	0.44–0.92	0.15 (0.11–0.21)
Crescent Island	510	LAXX	0.72–0.80	0.15 (0.11–0.21)
		DCCO	0.24–0.72	0.51 (0.34–0.70)
		AWPE	0.72–0.80	0.47 (0.24–0.73)
		CATE	0.40–0.76	0.71 (0.51–0.89)
Blalock Islands	441-439	LAXX	0.80–0.99	0.15 (0.11–0.21)
Miller Rocks	331	LAXX	0.62–0.80	0.15 (0.11–0.21)
Murdock Towers	308	DCCO	0.32–0.92	0.51 (0.34–0.70)
Troutdale Towers	189	DCCO	0.20–0.56	0.51 (0.34–0.70)
Rice Island	34	LAXX	0.52	0.71 (0.51–0.89)
Astoria-Megler Br.	23	DCCO	0.38–0.88	0.51 (0.34–0.70)
		BRAC [3]	0.60–0.80	0.51 (0.34–0.70)
East Sand Island	8	CATE	0.73–0.96	0.71 (0.51–0.89)
		DCCO/BRAC [2]	0.43–0.93	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): In 2024, estimates of tern predation rates on the ESA-listed salmonid ESUs/DPSs evaluated were below the 2% per colony threshold identified in IAPMP (*Table 5*). This represents the second consecutive 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 2024 were from Lenore Lake terns on Upper Columbia River steelhead at 1.7% (1.0–3.2%), followed by terns on the unnamed, incipient colony site in northern Potholes Reservoir at 1.5% (0.9–2.6%), and terns on Goose Island in Potholes Reservoir at 1.3% (0.7–2.6; *Table 5*). Estimates of predation rates on all other ESUs/DPSs by tern colonies in 2024 were less than 1% per ESU/DPS (*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 and an unnamed island in Potholes Reservoir, Shoal Island in Lenore Lake, Badger Island in McNary Reservoir, Crescent Island in McNary Reservoir, and Burbank Slough in McNary Reservoir during 2024. 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	North Potholes	Lenore Lake	Banks Lake	Crescent Island	Burbank Slough
SR Sockeye	NA	NA	NA	NA	NA	NA
SR Sp/Su Chin	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
UCR Sp Chin	NA	NA	NA	NA	NA	NA
SR Fall Chin	< 0.1%	< 0.1%	< 0.1%	< 0.1%	0.2% (0.1–0.7)	< 0.1%
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	< 0.1%	0.6% (0.2–1.3)	< 0.1%
UCR Steelhead	1.3% (0.7–2.6)	1.7% (1.0–3.2)	1.5% (0.9–2.6)	0.2% (0.1–0.5)	0.9% (0.4–1.7)	< 0.1%

Estimates of predation on UCR steelhead by terns on Goose Island (ca. 1.3%) were not commensurate with the estimated number of tern pairs (n=18) on Goose Island in 2024. A much larger number of terns, however, attempted to nest on Goose Island during the breeding season, but dissuasion activities were successful at preventing colony formation by those birds. As such, predation rates were greater than those implied by a colony size of 20 pairs in 2024. A similar phenomenon has been observed at Goose Island and other actively managed tern colonies in years past, whereby predation estimates were higher than those implied by colony size alone (Collis et al. 2022).

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*). In 2024, an incipient tern colony again formed on a different, unnamed island in northern Potholes Reservoir (126 pairs), but adaptive management actions were initiated in-season and predation rates were significantly lower (1.7% {1.0–3.2}) in 2024 compared with

2016, despite similar colony sizes in both years. Based on tagging data from RIS steelhead in 2024, terns were nesting at the unnamed island in northern Potholes Reservoir as early as April 26th (based on the release dates of tagged fish at RIS). Terns were successfully dissuaded from the unnamed island by the BOR on May 21st. Collectively, results suggest that active and passive dissuasion techniques can successfully prevent or limit 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 in Potholes Reservoir (see *Figure 1*), however, continued management will likely be necessary to achieve the stated goals of the IAPMP in the future.

Despite a relatively large colony (n=186 pairs), predation rates by Crescent Island terns in 2024 were <1.0% per ESU/DPS, with the highest estimates observed on Snake River and Upper Columbia River steelhead at 0.6% (0.2–1.3) and 0.9% (0.4–1.7), respectively (*Table 6*). 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 (*Table 6*; see also Collis et al. 2023). Due to the re-establishment and rapid growth of the tern colony on Crescent Island in recent years, however, future management may be necessary to achieve the goals of the IAPMP. For instance, the adaptive management colony size threshold of 40 pairs has now been exceeded for three consecutive years (2022–2024) at Crescent Island.

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 (2014-2024 for Goose Island and North Potholes; 2015–2024 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 (2022–2024). See Appendix A for annual estimates. NA denotes that predation estimates were not available.NC denotes that no colony existed during that period.

	Goose Is.			North Potholes Is.			Crescent Is.		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2024 [1]	Last 3-years 2022–2024	Pre-Management 2007–2013	Management 2016, 2024 [2]	Last 3-years 2024	Pre-Management 2007–2014	Management 2015–2024	Last 3-years 2022–2024 [3]
SR Sockeye	< 0.1%	< 0.1%	NA	NC	< 0.1%	NA	1.5% (1.2–2.0)	< 0.1%	NA
SR Sp/Su Chin	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	0.8% (0.7–1.0)	< 0.1%	< 0.1%
UCR Sp Chin	2.5% (1.7–3.6)	< 0.1%	< 0.1%	NC	0.2% (0–1.6)	0.2% (0–3.1)	0.5% (0.3–0.9)	< 0.1%	< 0.1%
SR Fall Chin	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	1.0% (0.9–1.2)	0.1% (0–0.2)	0.3% (0.1–0.4)
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	4.5% (4.1–5.1)	0.2% (0.2–0.3)	0.2% (0.7–1.4)
UCR Steelhead	15.7% (14.1–18.9)	1.0% (0.7–1.6)	0.6% (0.4–1.0)	NC	3.0% (2.2–4.5)	1.7% (1.1–3.3)	2.5% (2.2–2.9)	0.1% (0.1–0.2)	0.6% (0.4–1.0)

[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 2024

Estimates of predation rates on smolts at the unmanaged tern colony on Shoal Island in Lenore Lake in 2024 were 1.5% (0.9–2.6%) on UCR steelhead (*Table 5*). Sample sizes of PIT-tagged UCR Spring Chinook Samon were too small (<500) to generate reliable estimates of predation (see Methods and Analysis section). 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 2024 at 69 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 may be related to predation effects by terns nesting in Lenore Lake. Collectively, results suggest that future monitoring and possible management of the tern colony in Lenore Lake may be necessary to achieve the goals of IAPMP.

In 2024, a small tern colony formed on an island in Banks Lake, WA, with 15 breeding pairs observed. The highest predation rate estimate was 0.2% (<0.1–0.5) on UCR steelhead (*Table 5*). In 2023, there was no evidence of a sustained colony of terns in Banks Lake during the smolt outmigration period (Evans et al. 2024a), so PIT tag recovery was not conducted in 2023. Prior to 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 in Banks Lake to detect tern nesting and to estimate predation impacts are warranted.

There was no tern colony on Harper Island in Sprague Lake, WA in 2024. Harper Island has had a relatively small colony (< 40 breeding pairs) in some years (Collis et al. 2021b), except for 2021 when estimated 85 pairs were documented (Evans et al. 2022b). Scanning for PIT tags at the Harper Island tern colony has occurred only once in the past in 2012 (2012; Collis et al. 2021b). Predation rate estimates in 2012 indicated that terns consumed less than 0.3% of available salmonid ESUs/DPSs, with the highest rate observed on SR steelhead at 0.2% (Collis et al. 2021b). Predation rates on smolts were low, in part, due to the small size of the colony in 2012 (30 breeding pairs; Collis et al. 2021b) and the distance from Harper Island to the Snake River (65 km). Future monitoring of predation rates, however, may be warranted if a large tern colony becomes established.

No terns successfully nested on Badger Island in McNary Reservoir in 2024. As part of passive adaptive management, woody debris was placed in the area where terns successfully nested in 2023 to prevent tern nesting in 2024. Estimates of predation rates by terns nesting on Badger Island in 2023 (274 pairs), the largest tern colony in the CPR in 2023, were as high as 1.4% (0.9–2.5%) and 1.9% (2.0–4.4%) on UCR and SR steelhead, respectively (Evans et al. 2024a, see also *Appendix A*). Estimates of predation 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 temporarily re-established in 2021, average annual predation rates on SR steelhead have been 2.1% (1.6–2.8%; *Appendix A*) in years when a colony was present. As was the case in 2024, adaptive management efforts (e.g., placement of woody debris or likely other passive dissuasion techniques) on the Badger Island tern colony may be necessary to achieve the goals of the IAPMP in the future. Badger Island is also large (> 7 ha), with numerous interior and shoreline areas where terns could potentially nest, factors that could make it challenging to prevent nesting in any given year using passive methods.

In 2024, an incipient tern colony formed along the banks of Burbank Slough, part of the McNary Wildlife Refuge Complex, near the town of Burbank, WA. Peak colony size was 23 pairs and terns occupied the land-bridged site starting in late-May, after the peak smolt outmigration period. Relatively small numbers of smolt PIT tags (n= 182) were recovered following the breeding season (*Table 3*) and estimated predation rates were <0.1% for each ESU/DPS evaluated in 2024 (*Table 5*). Since 2021, terns have been observed loafing and roosting on areas of Burbank Slough, but 2024 was the first confirmed colony. As such, future monitoring of the site may be warranted, especially if future dissuasion activities are successful at preventing tern nesting at the Badger and Cresent Island colony sites, which are located less than 12 km from Burbank Slough.

For the fourth consecutive year since adaptive management was implemented (2021–2024), 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 (J. Macdonald, USACE, pers. comm.). During 2015–2020, predation rates by terns nesting in the Blalock Islands in John Day Reservoir had been, on average, significantly higher since management actions on the Crescent Island tern colony were implemented in 2015 (*Appendix A*). 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, 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, however, predation rates have been greatly reduced (if not eliminated) during

2021–2024. Results indicated that the elimination of nesting habitat on the Blalock Islands successfully prevented the formation of a tern colony 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 2024 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 tern colony, achieved. Reductions in tern colony sizes at both Goose Island and, to a lesser extent, Crescent Island, have reduced predation by terns breeding at these sites since management was initiated in 2014 (see also *Cumulative Predation and Survival* section). Adaptive management actions that placed woody debris on Badger Island, used active and passive dissuasion on an unnamed island in northern Potholes Reservoir, and that raised water levels in the John Day Reservoir have successfully prevented tern nesting. Terns, however, continue to demonstrate strong fidelity to CPR and continued adaptive management will likely be needed in 2025 and beyond to achieve the goals 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 (2014–2024). 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 (2022–2024). NA denotes that predation estimates were not available. NC denotes that no colony existed during that period.

	Banks Lake			Badger Island			Blalock Islands		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2016	Last 3-years 2022–2024	Pre-Management 2007–2013 [1]	Management 2014-2024 [2]	Last 3-years 2022-2024[2]	Pre-Management 2007–2013	Management 2014–2020[3]	Last 3-years 2022–2024
SR Sockeye	< 0.1%	0.1% (0.0–0.5)	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%	< 0.1%	NC/NA	0.1% (0.1–0.3)	0.1% (0.1–0.2)	0.1% (0.1–0.2)	0.6% (0.5–0.8)	NC
UCR Sp Chinook	< 0.1%	0.2% (0.0–0.7)	< 0.1%	NC/NA	< 0.1%	< 0.1%	< 0.1%	0.6% (0.4–0.8)	NC
SR Fall Chinook	< 0.1%	< 0.1%	< 0.1%	NC/NA	0.2% (0.1–0.3)	0.5% (0.3–0.8)	< 0.1%	0.7% (0.5–0.9)	NC
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	NC/NA	0.6% (0.5–0.8)	1.6% (1.2–2.2)	0.5% (0.4–0.9)	3.5% (2.9–4.3)	NC
UCR Steelhead	0.1%	1.1% (0.8–1.6)	< 0.1%	NC/NA	0.4% (0.3-0.5)	0.7% (0.5–1.1)	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 2014–2024	Last 3-years 2022–2024	Pre-Management 2012 [4]	Management 2014–2023	Last 3-years 2022–2024
SR Sockeye	NC	< 0.1%	NA	< 0.1%	NA	NA/NC
SR Sp/Su Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA/NC
UCR Sp Chinook	NC	0.2% (0.1–0.5)	0.3% (0–1.4)	< 0.1%	NA	NA/NC
SR Fall Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA/NC
SR Steelhead	NC	< 0.1%	< 0.1%	0.2% (0.1–1.3)	NA	NA/NC
UCR Steelhead	NC	0.8% (0.7–1.0)	1.3% (1.0–1.8)	< 0.1%	NA	NA/NC

[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 and not colony existed in 2024.
[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 2024 were not available, estimates that were generated with funding from USACE in the past. Smolt PIT tags, however, were recovered on ESI by the USACE-FFU following the 2024 breeding season (*Table 3*), so the data necessary to generate predation estimates exists. The number of smolt PIT tags recovered on ESI tern colony in 2024 (n=1,172) was well below the annual average recovered prior to tern management on ESI (26,553; Roby et al. 2021). Peak nesting by terns on ESI in 2024 also occurred in July (Robert et al. 2025) after most salmonids had passed through the estuary. The estimated number of terns breeding on ESI in 2024 was amongst the lowest observed to-date, with an estimated 1,524 pairs (Robert et al. 2025; see also *Table 1*) compared with an average of 9,079 pairs prior to management during 2000–2007 (Collis et al. 2023). Estimates of cumulative predation generated by this study – estimates that use a different analytical structure than the ESU/DPS-specific model (see Methods and Analysis section) – estimated predation by ESI terns at 1.1% (0.4–2.6%) and 0.7% (0.5–1.0%) on UCR and SR steelhead, respectively. These estimates were the lowest observed on steelhead since management was initiated to reduce the size of the colony in 2008. By comparison, prior to management (2000–2007), average annual estimates from the ESU/DPS-specific model were 17.2% (15.2–19.5%) and 25.3% (22.7–28.3%) for UCR and SR steelhead, respectively, but again, relative comparisons presented herein should be interpreted cautiously due to differences in the modelling framework used in 2024 relative to years past. See Collis et al. (2023) for relative comparisons using the same modeling framework during 2000–2022, the last year updated estimates are currently available.

In 2024, terns attempted but failed to successfully nest on Rice Island in the CRE due to active and passive dissuasion implemented by USACE as part of on-going management at that site (Blair et al. 2025). In 2022 and 2023, a tern colony on Rice Island successfully formed and predation rates were as high 2.9% (1.6–5.1%) on MCR steelhead (Evans et al. 2023). Results indicate 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 the *Estuary Tern Management Plan* are not offset by increased tern predation rates at other sites in the CRE, like Rice Island. Collis et al. (2023) provides a detailed assessment of the efficacy of the estuary tern management plan to reduce predation on ESA-listed smolts in the CRE, with data reported through the 2022 breeding season.

There was some evidence of late-season (July - October) breeding attempts by double-crested cormorants on ESI in 2024 but an estimate of number of pairs was not available (Robert et al. 2025). Researchers from USACE-FFU were not able to distinguish double-crested cormorants from Brandt's cormorants, which also nested on ESI in 2024. While USACE-FFU recovered smolt PIT tags following the breeding season, ESU/DPS-specific predation rate estimates were not generated, work that had previously been funded by USACE. It should be noted, however, that due to paucity of smolt PIT tags (n = 17; Table 3) recovered on the ESI cormorant colony (tag deposited by both double-crested and Brandt's cormorants) in 2024, predation rates were presumably very low (e.g., < 0.1%) or non-existent, depending on ESU/DPS. In 2022 and 2023, when small number of cormorants also attempted to nest on ESI, just 99 and 50 smolt PIT tags, respectively, were recovered by USACE following the breeding season, suggesting that predation rates by double-crested cormorants on ESI have been low in recent years, but formal predation rate analyses are lacking (Evans et al. 2024a).

An unintended consequence of management actions at the double-crested cormorant colony on ESI during 2015–2019 was the complete abandonment of the colony site and the subsequent expansion of the cormorant colony on the AMB and elsewhere in the CRE (Lawes et al. 2021b, Evans et al. 2023). For example, in 2014 an estimated 333 pairs of cormorants nested on the AMB—a number that expanded to 5,348 pairs in 2024, the highest count to date. 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, efforts to recover smolt PIT tags deposited by cormorants nesting on a concrete footing of the bridge (referred to as the North Crib) commenced. The North Crib is 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 (771 pairs in 2024) were then extrapolated to account for all cormorants nesting on the AMB (5,348 pairs in 2024) to generate colony-wide estimates of predation in 2024 (and in years past dating back to 2021).

Results of predation analyses in 2024 indicated that per capita (per breeding pair) predation rates of cormorants nesting on the AMB ranged from 0.0005% (0.0001–0.0014%) on SR Fall Chinook to 0.0020% (0.0009–0.0039%) on MCR steelhead. Colony-wide estimates of predation, those extrapolated to all cormorants on AMB, ranged from 2.8% (1.8–7.6%) on SR Fall Chinook to 10.9% (5.0–20.8%) on MCR steelhead (*Table 8*). With the exception of predation on SR Fall Chinook, estimates of predation in other ESUs/DPSs were similar amongst and between salmon and steelhead ESUs/DPSs, with no statistically significant difference detected in relative comparisons of predation (*Table 8*). For example, estimates of colony-wide predation rates on SR steelhead and SR sockeye were 9.9% (4.9–17.3%) and 9.1% (2.9–18.7%), respectively (*Table 8*). It should be noted, however, that estimates of predation were often imprecise due to the process of extrapolating estimates from a subsample of breeding pairs on the North Crib to all breeding pairs on the AMB (see also Methods & Analysis section).

Estimates of colony-wide predation rates on salmonid smolts by cormorants nesting on the AMB in 2024 were similar to those in 2023 and amongst the highest observed since research started on the bridge in 2021 (*Table 8* and *Appendix A*). Increases in both 2023 and 2024 were presumably associated, in part, with the larger size of the cormorant colony on the AMB in these two years relative to years past, with the colony increasing from 4,151 and 4,054 pairs in 2021 and 2022, respectively, to 5,153 and 5,348 pairs in 2023 and 2024, respectively (*Table 1 above*). Estimates of per capita predation by cormorants on AMB in 2021–2024 were approximately 2–5 times greater (depending on ESU/DPS) than average annual per capita predation rates by cormorants that nested on ESI prior to implementation of management actions on ESI (2003–2014; *Appendix A*). For example, per capita predation rates on SR 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 in 2024 at 0.0018% (0.0009 –0.0032%). As such, despite the smaller size of the double-crested cormorant colony on the AMB (4,652 breeding pairs on average during 2021–2024) 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, on average, than those of cormorants that formerly nested 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 5.6% (4.6–9.0%) and 9.3% (6.6–12.8%), respectively, by cormorants on the AMB during 2022–2024 (*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 on AMB and other sites in the upper CRE and LCR (see *below*) are higher than the rates observed prior to management when the vast majority of 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 LCR ESUs/DPSs in recent years (since 2020), estimates of predation effects by cormorants breeding on the AMB and other colonies in CRE and LCR are lacking (Evans et al. 2023). Impacts, however, are likely substantial given predation effects on LCR salmonids by cormorants that formerly nested on ESI. 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 breeding on AMB and at other locations in the CRE in the future (see also Evans et al. 2023).

For the third consecutive year, 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 in LCR. Smolt PIT tags were recovered in the area underneath five transmission towers with breeding cormorants on them. Predation estimates, which were based on the number of PIT-tagged smolts last detected alive passing Bonneville Dam (see *Methods & Analysis* section), ranged from 2.4% (1.0–5.7%) for SR Fall Chinook to 5.9% (3.2–12.1%) for UCR 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 species. Estimates of predation by TRT cormorants in 2024 were similar to those in 2022 and 2023 and were commensurate with the estimated colony size of 371 pairs in 2024 (*Table 2*). The TRT cormorant nesting site is in the freshwater zone of the CRE and results demonstrate that per capita predation rates were the highest of three cormorant colonies evaluated downstream of Bonneville Dam (ESI in the marine-zone, AMB in the mixing-zone, and TRT in the freshwater zone). For instance, estimates of average annual per capita (per breeding pair) predation rates on SR steelhead by cormorant on ESI (during 2003–2014), AMB (during

2022–2024), and TRT (during 2022-2024) were 0.0006% (0.0005–0.0007%), 0.0018% (0.0009 – 0.0032%), and 0.0086% (0.0066-0.0162%), respectively. Collectively, these results provide strong evidence that predation effects were significantly higher on per capita basis the further upstream cormorants nest in the Columbia River and that the goals of the *Estuary Cormorant Management Plan* will not achieved until the colony on ESI is either re-established at levels specific in the Plan (no more than 5,380 to 5,939 pairs; USACE 2015) and until cormorants are prevented from nesting at sites in mixing and freshwater zones of 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 double-crested cormorants (DCCO) nesting on the Astoria-Megler Bridge (AMB) and the Troutdale Towers (TRT) during 2024. See Appendix A for estimates of predation by these colonies in previous years.

ESU/DPS	AMB DCCO [1]	TRT DCCO
SR Sockeye	9.1% (2.8–18.7)	3.0% (1.4–6.4)
SR Sp/Su Chinook	4.4% (2.6–10.8)	3.2% (1.9–6.9)
UCR Sp Chinook	7.1% (2.8–13.3)	2.7% (1.4–6.4)
SR Fall Chinook	2.8% (1.8–7.6)	2.4% (1.0–5.7)
SR Steelhead	9.9% (4.9–17.3)	3.8% (2.2–6.9)
UCR Steelhead	6.3% (1.5–13.8)	5.9% (3.2–12.1)
MCR Steelhead	10.9% (5.0–20.8)	4.8% (2.6–9.3)

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

Cormorants nesting on select channel markers in the CRE and LRC (Rkm 0–218) were also scanned for smolt PIT tags in 2024. The primary purpose of scanning in 2024 and years past 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, 163 smolt tags were recovered from channel markers in 2024 (*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 as part of this study. In 2023, 452 smolt PIT tags were recovered from cormorant nests on channel markers. Research to quantify detection probabilities of PIT tags deposited by cormorants on channel markers will be necessary to generate estimates of predation rates in the future (see also Evans et al. 2024a).

Finally, cormorants also nested on the Lewis and Clark Bridge in 2024 (334 pairs; Table 2) and a portion of the bridge was scanned for smolt tags for the first time. In total, 195 smolt PIT tags were recovered in 2024 (*Table 3*). Similar to cormorants nesting on channel markers, however, there was no information available on PIT tag detection probabilities at the Lewis and Clark Bridge. Furthermore, the area scanned for tags on the Lewis and Clark Bridge was between 20 –

40 meters below cormorant nests (which were located on bridge support beams). As such, an unknown (but likely small) proportion of tags consumed by cormorants were subsequently deposited on the area that was scanned by researchers for smolt tags. This situation differs from the AMB cormorant colony, where researchers were able to scan a nesting site with a known number of nests (a flat, concrete platform; see *above* for details).

Predation Rates by Other Piscivorous Colonial Waterbirds

Predation/consumption rates of ESA-listed salmonid ESUs/DPSs by California and ring-billed gulls, double-crested cormorants, and American white pelicans nesting at colonies in the CPR were often greater than those of Caspian terns nesting at colonies in the CPR. Similar to years past, estimates were particularly high at some of the gull colonies in the CPR, with estimates of predation in excess of 8.0% on some ESUs/DPSs in 2024. 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 2024 are provided *below*.

California and Ring-billed Gulls - Island 20: Consumption rates of smolts by gulls nesting on Island 20 ranged from 0.6% (0.1–1.9%) for SR Fall Chinook to 8.5% (5.6–13.2%) for UCR steelhead (*Table 9*). Similar to trends in 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). To-date, the years with the highest estimates of consumption of steelhead by Island 20 gulls were those of 2015, 2021, and 2024 with estimates of 7.9%, 6.1%, and 8.5%, respectively, were observed on UCR steelhead (*Table 9* and *Appendix A*). These increases may be associated with increases in colony size and/or environmental factors (e.g., river flow, water transit time, turbidity) that increase smolt exposure to gull consumption during outmigration (Hostetter et al. 2012, Roby et al. 2016, Payton et al. 2016, Hostetter et al. 2023). For example, the years with the highest estimates of predation/consumption by gulls on steelhead coincide with years of below average flows (Hostetter et al. 2023, Evans et al. 2024a).

California and Ring-billed Gulls - Badger Island: Consumption rates on salmonid smolts for gulls nesting at Badger Island were amongst the lowest of the gull colonies evaluated in 2024, ranged from 0.1% (0.1–0.6%) for SR Fall Chinook to 1.1% (0.4–3.6%) for SR steelhead. Estimates of consumption in 2024 were significantly lower than those observed in 2023 and most other years dating back to 2015, when the gull colony first became established on Badger Island (Cramer et al. 2021a). In 2023, Badger Island gulls consumed an estimated 2.6% (1.0–4.9%) of available SR steelhead (*Appendix A*). Reductions in Badger Island gull consumption estimates appear commensurate with reduction in the size of gull colony on Badger Island, with an estimated 2,816 breeding individuals in 2023 and 1,026 breeding individuals in 2024. Unlike 2022 and 2023, terns did not co-nest on Badger Island in 2024, and this too could be related to lower predation/consumption, whereby gulls on Badger Island could not kleptoparasitize smolts from terns on Badger Island in 2024.

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 0.7% (0.1–2.5%) for SR Fall Chinook to 2.5% (1.2–4.8%) for UCR steelhead in 2024 (*Table 9*). Estimates of consumption were 2.3% (0.9–5.1%) on SR steelhead (*Table 9*), again demonstrating the greater susceptibility of steelhead to gull predation/consumption relative to salmon species. Consumption estimates by gulls on Crescent Island in 2024 were similar to, but slightly higher, than those observed during 2021–2023 (*Appendix A*). Estimates in recent years, however, have been substantially lower since colony reformation on Crescent Island in 2020 (*Appendix A*). This is likely related, in part, to the smaller size of the gull colony on Crescent Island (average of 3,558 individuals counted during 2021–2024, compared with 7,379 individuals during 2007–2015) and/or the smaller size of the tern colony on Crescent Island, whereby kleptoparasitism rates were also potentially lower during 2021–2024 due to the smaller size of the tern colony in these years (*Table 1*). For instance, if a larger tern colony coincides with higher rates of gull kleptoparasitisim, then the number of tags deposited by gulls would also presumably be higher.

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, smolt consumption rates were 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 were 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 California gulls and Straight Six Island was dominated by ring-billed gulls; the former are known to consume a higher proportion of juvenile salmonids; Collis et al. 2002). Data from Hostetter et al. (2015a) 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).

During 2022–2024 a gull colony did not form on Straight Six Island, but gulls nested on Anvil Island in all three years. In 2024, predation/consumption rate estimates from Anvil Island gulls ranged from 0.5% (0.2–1.1%) for SR Sp/Su Chinook to 1.7% (0.4–4.7%) 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, although estimates on UCR Spring Chinook were elevated in 2024 at 1.5% (0.4–4.1%). Estimates of consumption rates for gulls nesting at the Blalock Islands in 2024 were similar to those observed in most, but not all, previous years dating back to 2013, the first year the gull colony on Anvil Island was scanned for smolt PIT tags (*Appendix A*).

California and Ring-billed Gulls - Miller Rocks: Consumption rates on salmonid smolts for gulls nesting on Miller Rocks Island in The Dalles Reservoir, a colony where management actions (active hazing see *below*) occurred in April, ranged from 0.5% (1.0–3.5%) for SR spring/summer Chinook to 5.2% (2.2–10.8%) for SR steelhead (*Table 9*). Sample sizes of PIT-tagged UCR steelhead and SR sockeye were too small to generate reliable estimates of consumption based on smolts last detected alive passing McNary Dam, the location used to determine smolt availability for gulls on Miller Rocks (see *Methods & Analysis* section). Results from previous years with adequate sample sizes indicate that, among available salmon ESUs, SR sockeye smolts were particularly susceptible to consumption by gulls nesting at Miller Rocks, with estimates often greater than 5% and upwards of 9.1% (5.3–15.1%) of SR sockeye observed in years past (*Appendix A*). Estimates of predation/consumption on both UCR and SR sockeye, however, were available from the JMS model used to estimate cumulative predation rates (see *Cumulative Predation and Survival* section *below*).

Estimates of smolt consumption rates by gulls nesting at Miller Rocks in 2024 were similar to those in 2022 and 2023 but were slightly lower than estimates in other years dating back to 2007 (*Appendix A*). Starting in 2022, the Confederated Tribes and Bands of the Yakama Nation used active hazing techniques to dissuade gulls from breeding on Miller Rocks during the early portion of the breeding season with increased dissuasion efforts in 2024 (T. De Boer, Yakama Nation, pers. comm). Although efforts were successful at preventing colony formation while dissuasion occurred, a colony eventually formed in all three years (2022–2024). There was some evidence that the peak size of the gull colony on Miller Rocks was smaller in 2024 (3,708 individuals, *Table 2*) compared with 2007–2021 (4,466 individuals, Cramer et al. 2021a, Evans et al. 2022b), suggesting that management actions may have changed the size of the colony and consumption rates, but changes have been relatively small (*Appendix A*).

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

California and Ring-billed Gulls - Rice Island: Small numbers of smolt PIT tags (n=33) were recovered from a mixed colony area of California and ring-billed gulls on Rice Island in 2024 (*Table 3*). Due to the paucity of tags recovered, predation/consumption rates were <0.1% for all ESUs/DPSs evaluated and results indicated that gulls nesting on Rice Island posed little threat to smolt survival in the CRE in 2024. Estimates of the number of gulls breeding on Rice Island were not available but periodic, ground-based counts of individuals exceed 2,000 individuals during the traditional peak breeding period of late-May to early-June, with ring-billed gulls the dominate species enumerated (Blair et al. 2025). It should be noted that a slightly larger number of smolt tags (n=75) were recovered on a mixed gull and tern nesting area on Rice Island and presumably some of these tags were deposited by terns or where from gulls that kleptoparasited tagged smolts from terns. Regardless, the small number of tags recovered on Rice Island in 2024 suggest predation rates by breeding birds (both gulls and terns) were low.

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 Island 20, Badger Island, and Crescent Island in McNary Reservoir; Blalock Islands in John Day Reservoir; and Miller Rocks in The Dalles Reservoir during 2024. 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	1.0% (0.2–3.6)	0.4% (0.1–1.7)	1.3% (0.2–4.4)	0.5% (0.2–1.1)	0.5% (0.2–1.2)
UCR Sp Chinook	NA	NA	NA	1.5% (0.4–4.1)	1.4% (0.2–4.7)
SR Fall Chinook	0.6% (0.1–1.9)	0.1% (0.1–0.6)	0.7% (0.1–2.5)	0.7% (0.1–2.7)	1.5% (0.3–4.4)
SR Steelhead	1.0% (0.3–2.8)	1.1% (0.4–3.6)	2.3% (0.9–5.1)	1.7% (0.4–4.7%)	5.2% (2.2–10.8)
UCR Steelhead	8.5% (5.6–13.2)	0.7% (0.3–1.3)	2.5% (1.2–4.8)	NA	NA

Double-crested Cormorants - Okanogan River Mouth: There was no evidence that cormorants nesting at an arboreal colony site at the mouth of the Okanogan River, located 129 Rkm upstream of RIS on the middle Columbia River, consumed PIT-tagged smolts within the study area (i.e., Rock Island Dam to the Pacific Ocean), with no (zero) tags from smolts tagged and released at RIS recovered the cormorant colony in 2024. As such, predation estimates were presumably to be at or near zero for fish that survived passage to RIS. A relatively large number of smolt PIT tags (n=519), however, were recovered on the colony following the breeding season (*Table 3*), indicating cormorants were consuming juvenile salmonids upstream of RIS. Most tags were from steelhead, coho, and Chinook smolts that were released into tributaries of the UCR (e.g., Okanogan and Methow rivers). Tags from other fish species, like juvenile White Sturgeon and Pacific Lamprey were also recovered on the Okanagan cormorant colony (*Table 3).* Additional analysis—analysis that was outside the scope of this study—may be warranted to estimate stock-specific predation impacts by cormorants nesting at the mouth of the Okanogan River, a site that has been active since at least 2004 (Cramer et al. 2021a).

Double-crested Cormorants - Hanford Island: The cormorant colony on Hanford Island in the Hanford Reach of the middle Columbia River has periodically been scanned for smolt PIT tags since nesting cormorants were first documented at the site in 2007 (Cramer et al. 2021a), including scanning during 2022–2024. Estimates of predation rates on the ESA-listed ESUs/DPSs have been low, with less than 0.3% of available fish consumed per ESU/DPS in 2024 (*Table 10*) and less than 0.4% consumed during 2022 and 2023 (*Appendix A*). In 2024, tags were recovered from UCR Spring Chinook, but sample sizes of available tagged smolts at RIS were too small to generate reliable estimates of predation. The smaller size of this cormorant colony (99 pairs; *Table 2*) is presumably responsible, in part, for the lower predation rates on ESA-listed smolts compared with other nearby cormorant colonies like Crescent Island. It's worth noting, however, that most of the smolt PIT tags recovered from the Hanford cormorant colony were from non-listed subyearling Chinook from the Upriver Bright population, which accounted for more 50% of all recovered tags. 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 subyearling Chinook, however, are currently unknown but may be appreciable given the relatively large number of tags recovered on this colony in recent years.

Double-crested Cormorants - Island 20: In 2024, a newly discovered cormorant colony was detected on Island 20 and scanned for smolt PIT tags following the breeding season. No estimate of pre-season PIT tag detection efficiency was available for use in predation models; therefore, predation estimates are minimum estimates—adjusted for deposition and post-season detection probabilities but not pre-season detection probabilities (see Methods & Analysis section). Minimum estimates of predation were <0.1% for all ESU/DPS evaluated in 2024 (*Table 10*). Interestingly, of the 313 smolts tags recovered on this new colony (*Table 13*), 148 or 47% were smolts released into the Yakima River, located 10 Rkm downstream of Island 20, and 88 or 28% were subyearling Chinook from the non-listed Hanford Bright Chinook population. Like data from the cormorant colony at the mouth of the Okanogan River, additional analyses may be warranted to estimate stock-specific predation using availability data from tributary releases—analyses that were outside the scope of this study.

In addition to tags recovered on the Island 20 cormorant colony, tags were also recovered on a newly discovered Great Blue Heron and Great Egret nesting site on Island 20, with a total 453 tags recovered following the breeding season (*Table 3*). No PIT tag detection or deposition probabilities, however, were available for this colony site and avian species, so predation rates could not be generated (see Methods & Analysis section). Similar to tags recovered on the Island 20 cormorant colony, the majority of tags were from fish originating from Yakima River (308 or 68%) or from subyearling Chinook from the non-listed Hanford Bright population (53 or

12%). Results suggest that heron are disproportionately commuting to the lower Yakima River to forage on smolts relative to the middle or lower Columbia rivers. Research to better understand predation by herons at this and other colonies in CPR, including studies to estimate PIT tag detection and deposition probabilities, may be warranted.

Double-crested Cormorants - Foundation Island: For the first time since at least 2003, no (zero) cormorants nested on Foundation Island in McNary Reservoir in 2024. Estimates of predation rates for cormorants nesting at the colony on Foundation Island in years past were appreciable, with estimates of more than 3% on SR steelhead in some years (*Appendix A*). Predation rates on SR Spring/Summer Chinook were also substantial in some years, with estimates of more than 3% of available tagged smolts (*Appendix A*). Predation rates by Foundation Island cormorants on UCR smolts, however, were consistently and substantially lower than those of SR smolts (*Appendix A*). Higher predation rates on SR smolts compared with UCR smolts were 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 were 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). Despite the lack of nesting in 2024, the high rates of predation observed in years past warrant the continued monitoring of the historical colony site on Foundation Island.

Double-crested Cormorants - Crescent Island: A large colony (629 breeding pairs) of cormorants was on Crescent Island in 2024 (*Table 2*). The colony rapidly expanded from just 20 pairs in 2022, presumably due to the failure of the Foundation Island cormorant colony located just 9 Rkm upstream. Cormorants had periodically been documented nesting on Crescent Island in years past, but less than 50 pairs were observed in all years prior to 2023 (Cramer et al. 2021a, Evans et al. 2023). In 2024, predation rates were highly variable based on the salmonid ESU/DPS, with estimates ranging from 0.4% (0.1–1.8%) on SR Fall Chinook to 2.6% (1.1–6.4%) on SR steelhead. Estimates of predation on SR Spring/Summer Chinook were also the highest observed of any colony in the CPR in 2024 at 2.5% (1.4–4.8%; *Table 10*). At its current size, results suggested that the Crescent Island cormorant colony now poses a similar risk to smolt survival as the Foundation Island cormorant colony did in years past (*Appendix A*). Cormorants nesting on Crescent Island in 2024 occupy approximately 20% of available tree-nesting habitat. Similar to 2023, cormorants primarily nested in trees on the water's edge in 2024. However, the colony increased from 199 pairs in 2023 to over 600 pairs and birds expanded to interior nesting areas in 2024.

Double-crested Cormorants - Murdock Towers: In 2024, a newly discovered cormorant colony was detected on transmission towers on Murdock Island in Bonneville Reservoir and the colony was scanned for smolt PIT tags following the breeding season. Cormorants nested in three different towers, with most nests located on the middle or central tower. Loafing cormorants were also observed on the ground underneath the furthest downstream tower, so tags recovered from this location (n=257; *Table 3*) were not included in predation rate estimates. Pre-season PIT tag detection probabilities were also not available for use in the predation estimates, so estimates may be minimums. However, if large numbers of non-breeding cormorants or other avian predators (e.g., gulls) also deposited tags on the upstream and middle towers (birds that were not documented during aerial surveys), estimates of predation could be biased high to an unknown degree. As such, results should be interpreted cautiously. With those caveats in mind, estimates of predation by Murdock Tower cormorants were less than 1.0% for all ESU/DPS evaluated, ranged from 0.3% (0.1–1.2%) on SR Spring Chinook to 0.9% (0.5–5.0%) on UCR Spring Chinook (*Table 10*). Due to the novel nesting location and associated uncertainties regarding avian predator use at the site and a lack of pre-season detection probabilities, additional research is warranted if the site remains active in 2025 and beyond.

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 Murdock Towers in Bonneville Reservoir during 2024. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section).

ESU/DPS	Hanford Island	Island 20	Crescent Island	Murdock Towers
SR Sockeye	NA	NA	NA	NA
SR Sp/Su Chinook	< 0.1%	< 0.1%	2.5% (1.4–4.8%)	0.3% (0.1–1.2%)
UCR Sp Chinook	NA	NA	NA	0.9% (0.5–6.0%)
SR Fall Chinook	< 0.1%	< 0.1%	0.4% (0.1–1.8%)	0.9% (0.3–2.3%)
SR Steelhead	< 0.1%	< 0.1%	2.6% (1.1–6.4%)	0.7% (0.1–3.3%)
UCR Steelhead	0.2% (0.1–0.9%)	< 0.1%	0.7% (0.2–1.9%)	NA

Brandt's Cormorants - Astoria-Megler Bridge: The same methods used to estimate predation by double-cormorants on the AMB were used for Brandt's cormorants (BRAC) nesting on the AMB, whereby smolt PIT tags were recovered from concrete footings where a known number of BRAC nested (168 breeding pairs in 2024). Per capita predation rates were then used to generate colony-wide predation rates based on the total number of BRAC nests on the AMB (1,201 pairs in 2024). Although PIT-tag recovery efforts were limited to footings where only BRAC nested, a small number of recovered tags may have been deposited by double-crested cormorants that were nesting on beams above the footings. These tags could bias BRAC predation estimates high to an unknown degree. It should also be noted that only small numbers of smolt PIT tags were recovered from BRAC nesting areas in 2024 (n=78; *Table 3*) which resulted in imprecise estimates of per capita predation rates, especially in cases where sample sizes of available tagged fish passing BON were also small, like for SR sockeye and UCR spring Chinook (*Appendix A*). With these caveats in mind, results indicated that even with some double-crested cormorants potentially 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 2024 (and in years past; see Evans et al. 2024a), with per capita predation rates ranging from 0.0001% (0.0001–0.0004%) on SR Spring/Summer Chinook to 0.0009% (0.0001–0.0042%) on SR Sockeye. Colony-wide predation rates ranged from 0.1% (<0.1–1.7%) in SR Spring/Summer Chinook to 1.1% (0.1–5.2%) in SR Sockeye (*Table 11*) but results for SR sockeye should be interpreted cautiously due to small sample sizes and potential that estimates included some predation by double-crested cormorants. Colony-wide predation rates on all other ESUs/DPSs evaluated were <1.0% in 2024 (*Table 11*).

Results suggest that despite a relatively large colony on BRAC on AMB in both 2023 (1,224 pairs) and 2024 (1,201 pairs), they posed little threat to smolt survival in the CRE. These results were consistent with those of Cramer et al. (2021b), which observed that predation rates on salmonid smolts by BRAC that formerly nested on ESI were significantly lower than those of double-crested cormorants that nested on ESI. Cramer et al. (2021b) hypothesized that BRAC mainly forage in marine waters that are closer the Columbia River mouth where non-salmonids prey types (e.g., anchovy, herring, smelt) are more common (Couch and Lance 2004, Peck-Richardson 2017), that BRAC cormorant peak nesting period is typically in June after the peak smolt out-migration period, and that BRAC are slightly smaller (by body mass) and have lower daily food requirements compared with double-crested cormorants (Couch and Lance 2004).

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 the Astoria-Megler Bridge in the Columbia River Estuary during 2024.

ESU/DPS	Predation Rate [1]
SR Sockeye	1.1% (0.1–5.2)
SR Sp/Su Chinook	0.1% (<0.1–0.7)
UCR Sp Chinook	0.8% (0.1–2.3)
SR Fall Chinook	0.4% (0.2–4.4)
SR Steelhead	0.5% (0.1–1.5)
UCR Steelhead	0.1% (<0.1–2.8)
MCR Steelhead	0.1% (<0.1–1.6)

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

American White Pelicans - Badger Island: Estimates of predation rates by American white pelicans breeding on Badger Island, the largest pelican colony in the CRB, were low, with estimates ranging from 0.1% (<0.1–0.3%) for SR Spring/Summer Chinook to 1.0% (0.2–3.1%) for SR steelhead (*Table 12*). Results from 2024 were like those during 2021–2023 (*Appendix A*) and suggest that pelicans breeding on Badger Island posed little threat to the actively migrating ESA-listed UCR and SR salmonid smolts. Unlike terns, gulls, and cormorants, pelicans forage in shallow water less than three meters deep by dipping their bills 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, particularly during June when large numbers of subyearling Chinook are migrating (Stinson 2016; Payton et al. 2023). Payton et al. (2023) estimated that predation rates on non-listed subyearling Chinook from the Upriver Bright population by pelicans nesting on Badger Island were substantial, with upwards of 25% of wild smolts consumed by Badger Island pelicans in some years. Wild Handford Bright subyearling 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). Recent completed research in the lower Yakima River, a shallow water system that enters the middle Columbia River 28 Rkm upstream of Badger Island, also indicated that pelicans posed a threat to smolt survival in some years, with predation rates on Yakime River origin steelhead, coho, and subyearling Chinook more than 8% of available fish in some years (Evans et al. 2024b). As such, predation rate on smolts by Badger Island pelicans was highly variable depending on the salmonid stock and year, with impacts highest on smolts in tributary systems of the middle Columbia rivers located near Badger Island (see also Payton et al. 2023 and Evans et al. 2024b).

Unlike gulls and terns, pelicans are also capable of consuming adult-sized salmonids. PIT-tagged 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, Payton et al. in-press). In an investigation of the magnitude of pelican predation on adult sockeye during upstream migration from Bonneville Dam to Priest Rapids (on the middle Columbia River) and Ice Harbor Dam (on the Snake River), Payton et al. (2025) estimated that pelicans breeding on Badger Island consumed between 1,328 and 47,264 adult sockeye annually during 2014–2023. Predation rates on adult sockeye ranged from 1.5% to 8.4% of available sockeye each year, with predation impacts the greatest in years with the largest number of returning sockeye to Bonneville Dam. In addition to adult sockeye, tags from other larger-sized fish species have also been recovered on Badger Island, including white sturgeon, bull trout, 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 - Crescent Island: An incipient colony of pelicans formed on Crescent Island in 2024, with a peak count of 51 individuals (*Table 2*). The colony was first observed in early June and was still active in late June once aerial surveys had concluded. Pelicans attempted to nest on Crescent Island in the past, but the last colony was observed in 2010 when 50 individuals were documented (Cramer et al. 2021b). Consistent with a small, late forming colony and the low predation rates observed at the much larger Badger Island pelican colony (3,205 individuals; Table 2), predation rates by Cresent Island pelicans were <0.1% for all ESUs/DPSs evaluated in 2024 (*Table 12*). Only 24 smolt PIT tags were recovered from the nesting area following the breeding season (*Table 3*). Given the large size of the pelican colony on Badger Island, which is located just 3 Rkm upstream of Crescent Island, future monitoring of the Crescent Island pelican nesting area is warranted.

Table 12. Estimated predation 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 American White Pelicans breeding on Badger Island and Crescent Island during 2024.

ESU/DPS	Badger Island	Crescent Island
SR Sockeye	NA	NA
SR Sp/Su Chinook	0.1% (<0.1-0.3)	< 0.1%
UCR Sp Chinook	NA	NA
SR Fall Chinook	0.2% (0.1-1.1)	< 0.1%
SR Steelhead	1.0% (0.2-3.2)	< 0.1%
UCR Steelhead	0.2% (0.1-0.7)	< 0.1%

Cumulative Predation & Survival

Based on releases/detections of live PIT-tagged smolts at Lower Grante Dam (LGR) and RIS in 2024, avian predation effects were first observed in the river reach located downstream of Lower Monumental Dam (LMN; for SR smolts) and downstream of RIS (for UCR smolts). As such, for SR smolts, results indicating there was no measurable consumption of tagged fish between LGR and LMN associated with the bird colonies evaluated in this study. For UCR smolts, however, predation occurred 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, subyearling; Chinook only). Results from 2024 were then compared to results in years past to identify trends in predation and survival across river reaches and years dating back to 2008. Results help to identify hotspots of predation during outmigration from LGR and RIS to the Pacific Ocean.

Upper Columbia River Steelhead: The cumulative effects of avian predation/consumption (predation by all avian predator species and colonies combined) on UCR steelhead smolts during passage from RIS to the Pacific Ocean were substantial in 2024, with an estimated 28.2% (23.6–35.3%) of smolts consumed (*Figure 5*). Of the avian predator species evaluated, estimated consumption by gull colonies was the greatest at 18.3% (9.7–31.9%), followed by tern predation at 5.6% (3.3–9.5%), cormorant predation at 3.1% (1.2–8.0%), and pelican predation at 0.8% (0.2–1.8%). 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 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 2024 were very similar to those in 2023 and 2022, but 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/consumption 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 a lack of smolt survival estimates in the CRE (see *Methods & Analysis*), indicated that avian predation accounted 34.7% (25.4–55.5%) of all UCR steelhead mortality sources in 2024. Results suggesting factors other than avian predation were the dominant sources of smolt mortality. In most years dating back to 2008, however, avian predation/consumption has been the dominate or single greatest source of mortality for UCR steelhead smolts during outmigration to BON (Evans et al. 2016, Payton et al. 2019, Evans et al. 2019, Evans et al. 2022b. For instance, avian predation/consumption has accounted for >50% of all UCR steelhead smolt mortality sources during 13 of the last 17 years (*Figure 6*; see also Evans et al. 2022b), including in 2022 and 2023.

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). Appendix C* also has a detailed analysis of colony- specific and cumulative predation and survival of UCR steelhead from specific hatchery programs and release sites upstream of Rocky Reach Dam, including a relative comparison to predation and survival of wild steelhead during passage through the same river reaches and years.

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).

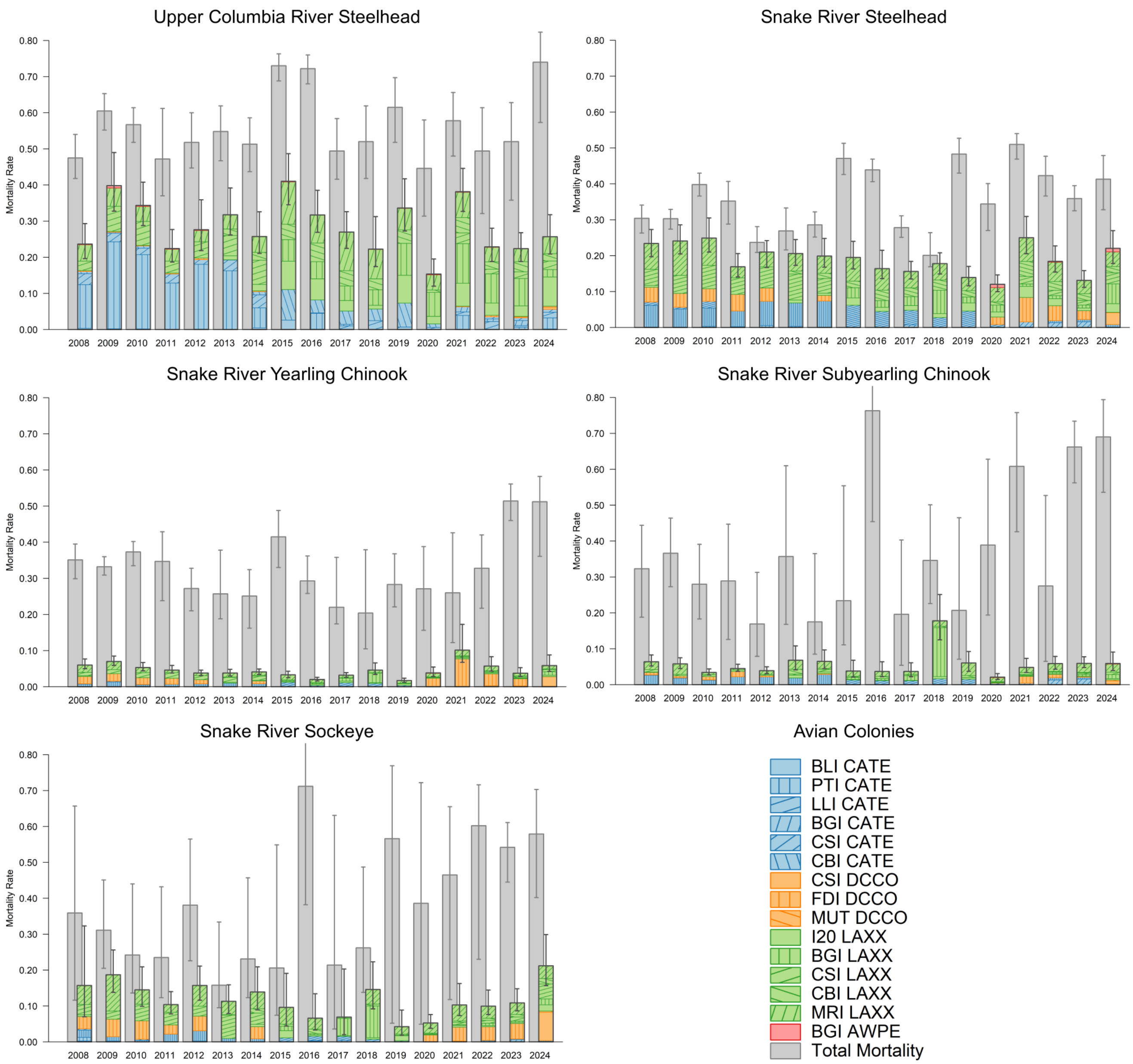


Figure 6. Estimated total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Upper Columbia River steelhead during passage from Rock Island Dam to Bonneville Dam or Snake River steelhead, yearling Chinook, subyearling Chinook, and sockeye during passage 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 2024 were similar to those of UCR steelhead, with an estimated 31.4% (26.5–37.9%) of SR steelhead smolts consumed (*Figure 5*). Of the avian predator species evaluated, estimated consumption by gull colonies was the greatest at 17.5% (9.3–25.1%), followed by cormorants at 11.6% (7.8–19.2%), terns at 1.1% (0.8–1.2%), and pelicans at 0.9% (0.2–2.6%; *Figure 5*). Predation by cormorants on SR steelhead were higher than that of UCR steelhead, especially predation by cormorants that formerly nested on Foundation Island (prior to 2024) and Crescent Island in 2024 (*Figure 5*). The cumulative effects of avian predation/consumptions on SR steelhead smolts in 2024 was slightly higher than that observed during 2021–2023, but consistent with several other years dating back to 2008 (*Figure 5*).

Estimated total mortality (1 - survival) on SR steelhead during passage from LMN to BON was 41.3% (32.8–47.9%) in 2024. Estimates of total mortality in 2024 were slightly higher than those in 2023 and in several, but not all, years past (*Figure 6*). Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 53.4% (37.2–82.3%) of all SR smolt mortality sources during passage from LMN to BON in 2024 (*Figure 6*). Analogous to results on UCR steelhead, avian predation/consumption accounting for > 50% of all SR steelhead smolt mortality sources in 12 of the last 17 years, including 2024 (*Figure 6*; Evans et al. 2022a). As such, avian predation/consumption was the dominant source of mortality during smolt passage from LMN to BON. Estimated total mortality of SR steelhead during passage from LGR to LMN was low at 3.9% (1.1– 11.4%) of smolts in 2024, indicating that the vast majority of mortality occurred downstream of LMN once smolts were exposed to predation by colonial waterbirds and other sources of mortality (see also Evans et al. 2022a).

Snake River Yearling Chinook: Like previous years, the cumulative effects of avian predation/consumption on SR yearling Chinook during passage from LMN to the Pacific Ocean in 2024 were significantly lower than those of SR and UCR steelhead, with estimated 10.2% (8.1–14.6%) of smolts consumed (*Figure 5*). Large numbers of PIT-tagged yearling Chinook were detected at LGR (n=51,862) which resulted in precise estimates of both predation and survival downstream of LGR. Of the predator species evaluated, estimated predation/consumption were the highest by cormorant colonies at 6.7% (4.0–12.2%). Significantly lower levels of predation/consumption were observed in gulls at 2.9% (1.2–5.1%), terns at 0.3% (0.1–0.6%), and pelicans at 0.1% (<0.1–0.2%; *Figure 5)*. The cumulative effects of avian predation/consumptions on SR yearling Chinook smolts in 2024 were very similar to estimates in previous years (*Figure 6*). As noted for UCR and SR steelhead, however, a lack of predation estimates from cormorants on Foundation Island during 2013, 2015–2019, AMB during 2016–2020, and TRT during 2012–2021, resulted in minimum estimates of cumulative predation in those years.

Similar to SR steelhead, total mortality of SR yearling Chinook from LGR to LMN was low (6.2%) but then increased during passage from LMN to BON in 2024 (51.2%; *Figure 6*). Unlike SR steelhead, however, comparisons of avian predation/consumption to total mortality (1 - survival) in SR yearling Chinook indicated that avian predation/consumption accounted for only

11.4% (7.3–24.4%) of all smolt mortality sources during passage from LMN to BON in 2024 (*Figure 6*). Results are consistent with those in the past and suggest that upstream of BON, 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 both upstream and downstream of BON (see *Reach-specific Predation and Survival* and Evans et al. 2022b for a more detailed discussion).

Snake River Subyearling Chinook: Similar to results in SR yearling Chinook, an estimated 7.2% (5.0–10.8%) of SR subyearling 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 2024 (*Figure 5*). Of the avian predator species evaluated, estimated predation/consumption on SR subyearling Chinook were the greatest by gull colonies at 4.2% (1.7–8.4%), followed by cormorant colonies at 2.6% (0.8–4.4%), terns at 0.3% (0.1–0.8%), and pelicans at 0.1% (<0.1–0.8%; *Figure 5*). The cumulative effects of avian predation/consumptions on SR subyearling Chinook smolts in 2024 were similar to those observed in 2023 and in many, but not all, previous years dating back to 2008 (*Figure 5*; Evans et al. 2022a).

Estimates of total mortality in SR subyearling Chinook during passage from LMN to BON were highly variable depending on the year, ranging from 42.8% to 74.8% during 2008–2024 (*Figure 6;* see also Evans et al. 2022a). Comparisons of avian predation/consumption to total mortality indicated that avian predation accounted 8.8% (5.0–16.9%) of all SR subyearling Chinook smolt mortality sources during passage from LMN to BON in 2024 (*Figure 6*). The precision of estimates of predation and survival in SR subyearling Chinook have also be highly variable due to fluctuations in sample sizes of PIT-tagged smolts, with numbers ranging from 2,546 tagged smolts to 34,742 tagged smolts annually (Evans et al. 2022a). It is also worth noting that unlike SR steelhead, SR yearling Chinook, and SR sockeye, a large proportion of SR subyearling Chinook originate downstream of LGR, and results presented herein apply only to those fish that originated upstream of LGR. Finally, results from 2024 were consistent with those of years past, with avian/predation consumption annually accounting for 9% to 39% of all SR subyearling 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 significantly higher than those on Chinook but comparable to those of steelhead, with an estimated at 29.0% (21.9–40.1%) of smolts consumed in 2024 (*Figure 5*). Of the avian predator species evaluated, estimated predation/consumption was the greatest by cormorant colonies at 15.0% (8.1–26.9%), followed closely by gulls at 13.3% (7.0–20.9%). Predation estimates on sockeye were significantly lower by tern and pelican colonies at just 0.4% (0.1–1.1%) and 0.1% (<0.1–0.8%), respectively (*Figure 5*). The cumulative effects of avian predation/consumptions on SR sockeye in 2024 were the highest observed since system-wide studies of predation commenced in 2008 (*Figure 5*). There is also evidence that predation effects have steadily increased since 2019 and that much of this increase is attributed to predation by cormorants. Even though cumulative estimates were based on availability at LMN, predation was the highest by cormorant colonies in the CRE, over 500 Rkm downstream of LMN (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 36.6% (22.3–74.3%) of all SR sockeye smolt mortality sources during passage from LMN to BON in 2024 (*Figure 6*). 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). Like results from SR subyearling Chinook, fluctuations in samples size of PIT-tagged SR sockeye resulted in imprecise estimate of predation and, especially, survival in years past (*Figure 6*; see also Evans et al. 2022a for a more detailed discussion).

Collectively, results of this study indicated 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. The approach of this study to investigate avian predation on multiple salmonid species that share a common migration corridor revealed several important findings, including that (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 subyearling 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).

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 both UCR and SR steelhead smolts associated with predation/consumption by piscivorous colonial waterbirds was greater than that from all other mortality sources combined during outmigration to BON 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. Results from 2024 also indicate the predation/consumption of SR sockeye has increased over the course of the last 5 years (2020–2024), with the highest cumulative estimates observed to-date in 2024. Increased predation on sockeye corresponds with increases in cormorant predation, particularly in the CRE due to cormorants breeding on the AMB and Troutdale Towers (see also Reach-specific Predation/Consumption *below*). These results over the course of last 17 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 UCR and SR steelhead and now SR sockeye (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. In 2024, avian predation effects on SR salmonids 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 18.4% (10.8–31.0%), followed by SR steelhead at 16.0% (12.2–21.1%), SR yearling Chinook at 9.1% (6.7–12.8%), and SR subyearling Chinook at 3.9% (1.8–7.8%) in 2024 (*Figure 7*). Predation in this reach was also substantial for UCR steelhead at 11.4% (5.4–23.8%). Since smolts that have survived outmigration through the Columbia River Power System to the CRE are, on average, more likely to survive to adulthood than those that are yet to complete outmigration, the benefits of managing avian predators in the CRE may be greater on a per fish basis than managing inland avian predators (Roby et al. 2002, Payton et al. 2020, Evans et al. 2022c).

Predation/consumption effects, particularly for UCR and SR steelhead and SR sockeye, were also substantial (>10%) during smolt passage through the middle Columbia and lower Snake rivers to McNary Dam in 2024 (*Figure 7*). For instance, an estimated 18.3% (14.3–23.7%) of UCR steelhead were consumed during passage from RIS to McNary Dam (MCN) in 2024, amongst the highest reach-specific estimates observed (*Figure 7*). Predation/consumption of SR steelhead and SR sockeye was also substantial in this river reach at 15.8% (11.8–20.5%) and 13.3% (8.5–20.8%), respectively, in 2024 (*Figure 7*). Similar to trends in years past, predation/consumption on smolts during passage from MCN to John Day Dam (JDA) and JDA to BON was almost entirely due to gulls in 2024, with similar levels of predation/consumption observed within each of these two rivers reaches (*Figure 7*). For instance, gulls consumed between 2 – 5% of the SR smolts during passage through both MCN to John Day Dam (JDA) and JDA to BON reaches. Predation on UCR steelhead, however, was higher during passage from JDA to BON in 2024 (ca. 9.4%) due to gulls nesting on Miller Rocks located downstream of JDA.

It is 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).

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 2024 are provided in *Appendix D*.

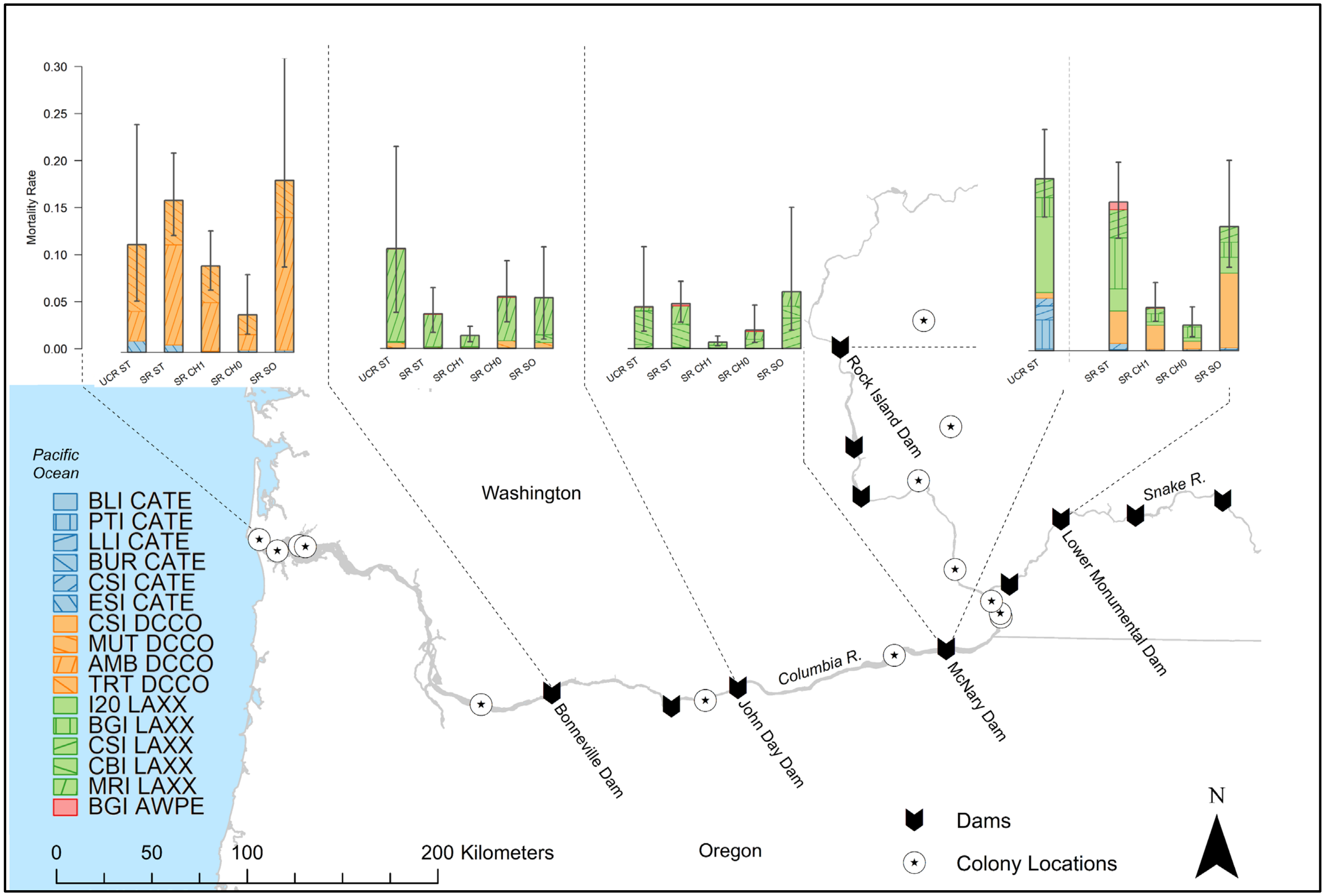


Figure 7. Reach-specific predation/consumption for Upper Columbia River (UCR) steelhead smolts during passage from Rock Island Dam to the Pacific Ocean and for Snake River (SR) steelhead, yearling Chinook (CH1), subyearling Chinook (CH0), and sockeye (SO) during passage from Lower Monumental Dam to the Pacific Ocean in 2024. 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.

Additive Effects of Predation

Caspian Tern Predation: Results of this study continue to reinforce previous studies (Payton et al. 2020, Payton et al. 2021, Evans et al. 2022) that indicate increases in tern predation on steelhead smolts during passage from RIS to BON were associated with statistically significant decreases in smolt survival (*Table 13* and *Figure 8*). This relationship is measured between increases in tern predation and reductions in steelhead smolts, with a, which represents the annual proportion of depredated fish that would have survived in the absence of tern predation. Over the 17 years assessed (2008–2024), a was estimated, on average, to be 1.35 (0.91–1.82; *Table 13*). The apparent reductions in survival that can be attributed to predation are measured with Φ^{Δ}, which represents the difference in observed survival compared to baseline survival, the expected survival rate in the hypothetical absence of tern predation (Payton et al. 2020). Estimates of Φ^{Δ} suggest that annual steelhead survival probabilities from RIS to BON would have been, on average, 0.13 (0.08–0.17) higher in the absence of tern predation. In other words, the observed average annual survival probability of UCR steelhead during passage to BON of 0.45 would have been 0.58 in the hypothetical absence of tern predation (Table 13). Results provide strong evidence that significantly more steelhead smolts would survive outmigration in the absence of tern predation.

The relationship between tern predation on UCR steelhead and steelhead survival was also found to be statistically significant in smolts last detected at Rocky Reach Dam (RRJ; *Table 13*), located 40 Rkm upstream of RIS (Evans et al. 2024s), during passage to BON, which demonstrates the relationship was independent of the sampling location in the middle Columbia River (see also Evans et al. 2024a). The relationship was also assessed using a completely different group of steelhead from the Snake River during passage from LMN to BON and again the results were statistically significant with similar levels of additivity (*Table 13;* Payton et al. 2021). Collectively, results indicate similar levels of additivity across steelhead sampling locations (RRJ, RIS, LGR {the sampling site for fish passing LMN), steelhead DPSs (UCR, SR), and years. See also Payton et al. (2020, 2021) for a more detailed discussion of the relationship between tern predation on UCR and SR steelhead and steelhead survival in the CRB.

Estimates of a during the smolt life stage 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 injured (i.e., 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 are presumably consistent across years but may be different between colonies. It is important to note, however, that the focus on a 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 (see Efficacy of Avian Predation Management Plans section *above*). For instance, in the earlier years of the analysis (2008–2014), a large proportion of tern predation was located closer to 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, however, there were 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, when measured from smolts released at RIS, is thus diminished 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. 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.

Table 13. Average annual Caspian tern predation probabilities and survival probabilities of Upper Columbia River steelhead smolts during outmigration from Rock Island Dam (RIS; during 2008–2024) and Rock Reach Dam (RRJ during 2010–2023) to Bonneville Dam and Snake River steelhead during passage from Lower Monumental Dam (LMN during 2008–2018) to Bonneville Dam. Estimates of the magnitude of the association between tern 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. Estimates are those previously reported by Payton et al. (2020), Payton et al. (2021), and Evans et al. (2024), updated with data from RIS in 2024.

Group	Reach	Survival	Predation	a	Φ^{Δ}
UCR	RIS to BON	0.45 (0.42–0.49)	0.10 (0.09–0.12)	1.35 (0.91–1.82)	0.13 (0.08–0.17)
UCR	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)
SR	LMN to BON	0.62 (0.60–0.64)	0.04 (0.04–0.05)	1.47 (1.04–1.95)	0.05 (0.03–0.07)

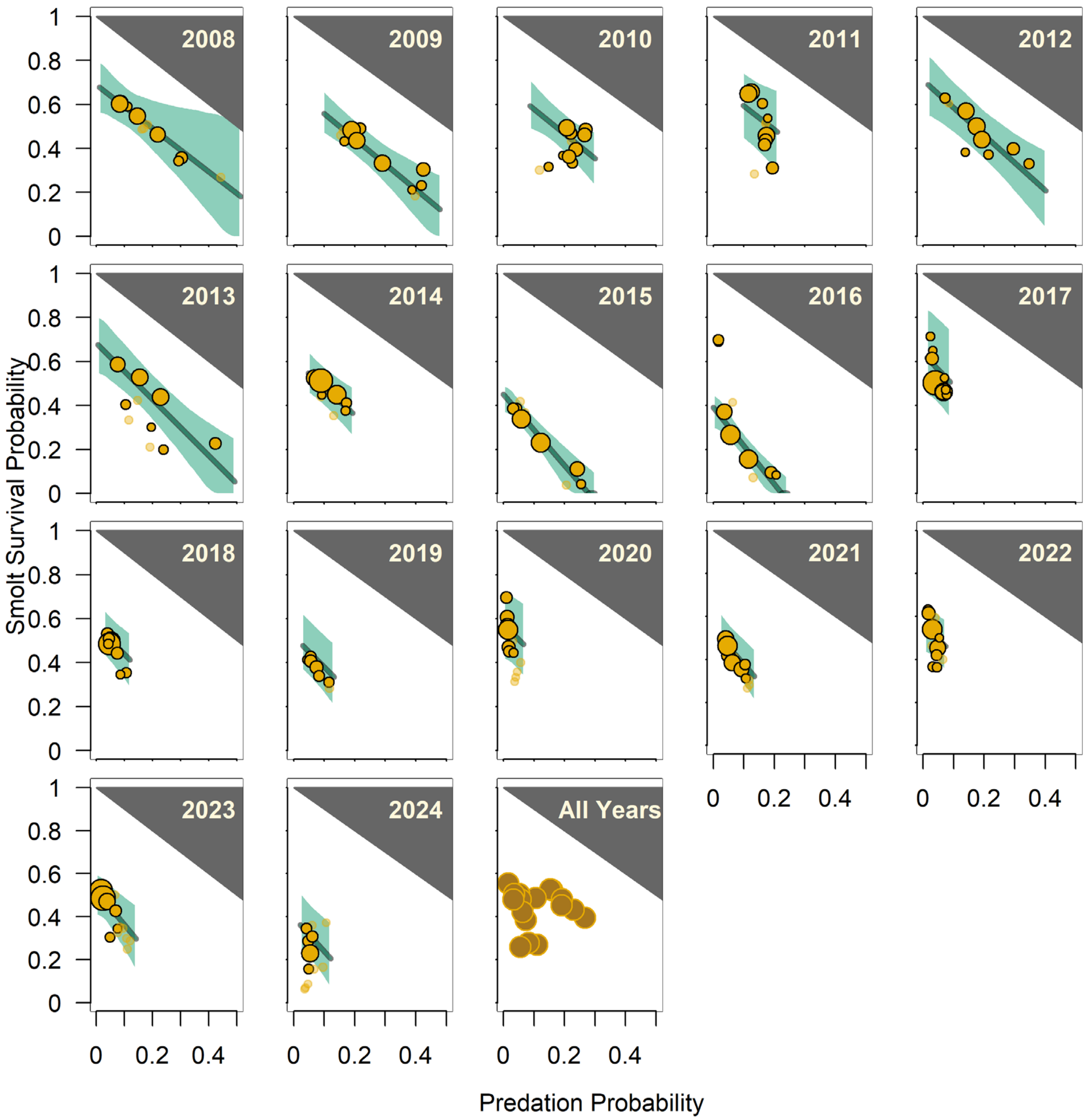


Figure 8. Weekly smolt survival probabilities for Upper Columbia River steelhead as a function of Caspian tern predation probabilities during smolt out-migration from Rock Island Dam (RIS) to Bonneville Dam during 2008–2024. The size of circles depicts the relative number of PIT-tagged smolts detected at RIS 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).

More importantly from a steelhead conservation and tern management perspective, studies have documented a statistically significant relationship between tern predation on UCR and SR steelhead smolts and smolt-to-adult returns, suggesting more adult steelhead would have returned to the Columbia River in the absence of tern predation (*Table 14*; see also Payton et al. 2020, Payton et al. 2021). Results provide evidence that the terns consumed significant portions of SR and UCR steelhead smolts that would have survived to adult return – that is, tern predation was a partially additive source of mortality to the adult life stage – despite predation occurring only during the smolt life stage. As expected, however, given low smolt-to-adult return rates in the CRB, estimates of additivity to the adult life stage suggest that most, but not all, smolts consumed by terns would have died prior to returning to BON regardless of tern predation (Payton et al. 2020). None-the-less, due the large number and proportion of available steelhead smolt consumed by terns and the additive relationship between predation and survival, results suggest that significantly more adult steelhead would return to BON in the absence of tern predation, with estimates of Φ^{Δ} suggesting SARs would have been, on average, 0.02 (0.01–0.03) and 0.01 (<0.01–0.01) higher in the absence of tern predation on UCR and SR steelhead, respectively (*Table 14*). This represents a 33% to 67% increase in the proportion of adult UCR and SR steelhead, respectively, that would have returned BON in the hypothetical absence of tern predation (see also Payton et al. 2020 and Payton et al. 2021 for a more detailed discussion of the additive relationship between tern predation on smolts and smolt-to-adult survival).

Table 14. Average annual predation probabilities by Caspian terns on smolts and smolt-to-adult survival probabilities for Upper Columbia River (UCR) and Snake River (SR) steelhead (ST) during passage from Bonneville Dam (as smolts) back to Bonneville Dam (as adults) during 2008–2017. Estimates of the magnitude of the association between predation probabilities on smolts and smolt-to-adult survival ($\boldsymbol{a}$ *, additivity) and the difference in smolt-to-adult survival probabilities from estimated baseline smolt-to-adult survival probabilities (* $\boldsymbol{\Phi}^{\Delta}$ *) are also provided. Results are those previously reported by Payton et al. (2021).*

Group	Reach	Survival	Predation	a	Φ^{Δ}
UCR	BON to BON	0.03 (0.03–0.03)	0.11 (0.10–0.13)	0.14 (0.06–0.23)	0.02 (0.01–0.03)
SR	BON to BON	0.03 (0.02–0.03)	0.11 (0.10–0.12)	0.06 (0.03–0.09)	0.01 (<0.01–0.01)

As it relates to management actions aimed at reducing tern predation, reductions in predation rates by terns on UCR steelhead are associated with reduction of the size of the Goose Island tern colony in Potholes Reservoir and the Crescent Island tern colony in McNary Reservoir, the two tern colonies managed as part of the IAPMP (USACE 2014, Collis et al. 2021a). In 2020 and 2023, record low tern predation rates were commensurate with record high estimates of UCR steelhead survival in the river reach (RIS to MCN) where terns were managed (Figure 9). In 2021, 2022, and 2024, however, increases in predation rates coincided with lower estimates of UCR steelhead survival. 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. On average, however, over the course of the entire *Inland Avian Predation*

Management Plan period (2014–2024), tern predation rates on UCR steelhead smolts have significantly declined following management actions at Goose and Crescent islands (*Figure 9*). For instance, on average, tern predation rates on UCR steelhead have decreased from 19.6% (16.5–24.1%) to 3.6% (2.8–4.6%) following management while survival has increased from 59.8% (56.6–63.3%) to 70.7% (63.7–76.6%; *Figure 9*). These results provide evidence that management efforts to reduce tern predation have demonstrably improved smolt survival in managed areas; however, if tern colonies reform or grow, smolt survival could decrease to pre-management levels.

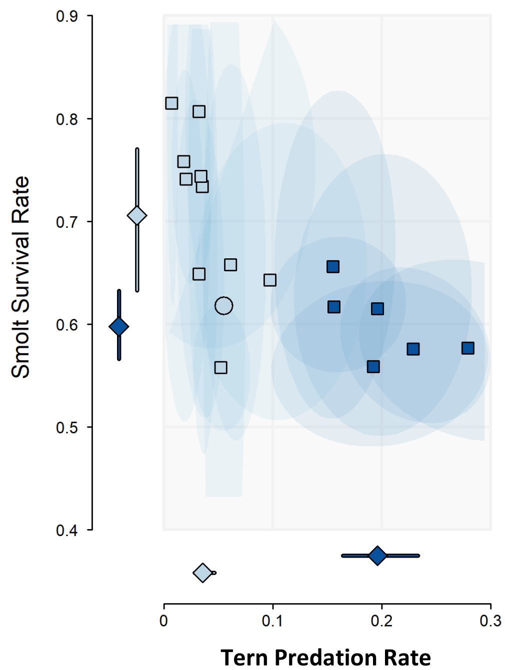


Figure 9. Caspian tern predation rates and survival rates of Upper Columbia River steelhead smolts during passage from Rock Island Dam to McNary Dam during (dark blue) and following (light blue) management actions that reduced the size of tern colonies at Goose Island and Crescent Island. Median annual 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). The circle indicates estimate from 2024.

Despite evidence that UCR salmonids have benefited from management actions associated with tern management in CPR, 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 in 2014. This is due, in part, to the total number of terns foraging on SR smolts being similar in all years since 2008, 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 during 2022–2024. All these breeding sites are located downstream of the confluence of the Snake and Columbia Rivers where SR smolts are susceptibility to predation by terns and other piscivorous waterbirds. In the CRE, however, there is evidence that predation by terns on both SR and UCR steelhead has substantially decreased following management actions that successfully reduced the size of the colony on ESI (Collis et al. 2024). Recent attempts by terns to nest at sites in the upper estuary (e.g., Rice Island), however, are offsetting those benefits to some degree due to the higher per capita predation rates of terns breeding at colonies in the upper versus lower CRE (see also Collis et al. 2024 and Management Recommendations *below*).

Double-crested Cormorants: There was some evidence of a relationship between cormorant predation on smolts and smolt survival, depending on salmonid ESU/DPS, the cormorant colony, and river reach. For instance, there was evidence that cormorant predation in the CRE on UCR steelhead smolts during 2008–2015 was partially additive to the adult life-stage, with increasing cormorant predation probabilities associated with decreasing adult returns (Payton et al. 2021). Estimates of cormorant additivity in the CRE, however, were imprecise (uncertain) due to low levels of weekly and annual variation in estimates of predation (Payton et al. 2021). There was also some evidence of a significant relationship between cormorant predation on SR steelhead upstream BON and smolt survival to BON, but here too estimates were imprecise and varied considerably by year (Payton et al. 2021). Furthermore, only a small or negligible proportion of available smolts (2– 4% per ESU/DPS, per year) were consumed by cormorants nesting at colonies upstream of BON during 2008–2012 and 2014 (years previously investigated), so avian predation was a relatively minor source of fish mortality and efforts to reduce predation would only result in minor increase in smolt survival. See Payton et al. (2021) for more details, including tabular and graphical results of the relationship between cormorant predation and smolt survival. An updated analysis involving increasing levels of predation by cormorants on the AMB and TRT in the CRE during 2021–2024 should be conducted as soon complete adult returns to BON are available (1 – 3 years following ocean residency).

California and Ring-billed Gulls: An analysis of the additive effects of gull predation on steelhead during smolt passage from McNary Dam to Bonneville Dam – a river segment with several larger-sized gull colonies (e.g., Miller Rocks in The Dalles Reservoir) and high level of predation (> 5%) on SR and UCR steelhead – indicated there was a relationship between gull predation and smolt survival during 2013–2022 (Evans et al. 2023). Results suggest that significantly more steelhead smolts would, on average, have survived outmigration to BON in the absence of gull consumption upstream of BON, with an estimated Φ^{Δ} at 0.17 (0.01–0.31) and 0.11 (0.03–0.18) for UCR and SR steelhead, respectively. For example, the average annual SR steelhead survival probability from MCN to BON during 2013–2022 was 0.73 (0.71–0.75) and results suggest that survival would have been 0.84 (0.76–0.91) in the absence of all gull predation in this river reach (see Evans et al. {2023}) for both tabular and graphical results). Similar to estimates of additivity for cormorants, estimates of gull additivity were imprecise (uncertain) due to lower levels of weekly and annual variation in estimates of predation and a shorter time-series of data (Evans et al. 2023). Furthermore, 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 less understood than that of terns (Evans et al. 2023).

MANAGEMENT RECOMMENDATIONS

Based on the results of this study from 2024 and in previous years (Collis et al. 2021b, Roby et al. 2021b, Evans et al. 2022b, Evans et al. 2023, Collis et al. 2024, Evans et al. 2024a), the following management recommendations are offered 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 plan to reduce the size (number of breeding pairs) of the Caspian tern colony on East Sand Island (ESI) to reduce predation rates on juvenile salmonids has been successful, with predation on SR and UCR steelhead reduced by upwards of 75% (Collis et al. 2024). The number of terns nesting on ESI, however, has declined dramatically over the course of the last seven years and is now well below the target colony size identified in the management plan (3,125–4,375 pairs). The latest census of the Pacific Flyway breeding population of terns occurred in 2024 and indicates that the tern population has declined by more than 70% since the population peak in 2009, one year after management began in 2008 (McGuire et al. 2025). A decline in the Pacific Flyway population of 50% 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 tern population at risk (USFWS 2005, Section 4.2.1.3). Unfortunately, this trigger point was reached following the Pacific Flyway census in 2021 (Lawes et al. 2022), indicating the need for adaptive management to halt the decline. In keeping with the goals and objectives of the *Estuary Tern Management Plan*, adaptive management to restore the tern colony at ESI to the size range identified in the plan should be considered. The colony on ESI has not consistently fledged chicks since 2019, which is likely contributing to the overall population decline. We also recommend 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 or enhanced for terns at interior sites and in San Francisco Bay. 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). Please see Collis et al. (2024) for a more detailed review of the history and efficacy of the *Estuary Tern Management Plan*, including adaptive management recommendations.

There is evidence that tern predation rates on UCR steelhead and, to a lesser degree, SR steelhead, have been reduced and survival has increased as a result of the *Inland Avian Predation Management Plan*. There continues, however, to be high fidelity of terns to managed and unmanaged sites in the CPR. This is evident by the re-establishment of a tern colony on Crescent Island, after management actions that had prevented tern nesting on the site since 2015 were discontinued in 2021. The tern colony on Crescent Island has now exceeded the target colony size of 40 pairs for three consecutive years, with an average of 141 pairs during 2022–2024. Adaptive management to eliminate suitable tern nesting habitat from Crescent Island should be implemented or the colony is likely to increase to its pre-management size in the coming years. Terns also demonstrate high fidelity to Goose Island and surrounding islands in Potholes Reservoir, but adaptive management efforts have been largely successful in preventing or quickly dissuading new or incipient colonies. Without continued adaptive management in Potholes Reservoir, large and sustained tern colony(s) are likely to be re-established in the coming years. Finally, the shift in the distribution of terns from managed colonies to unmanaged colony sites in the CPR (i.e., Badger Island, Shoal Island in Lenore Lake, Burbank Slough) 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).

DOUBLE-CRESTED CORMORANTS

Cormorants have largely abandoned ESI as a nesting site and thousands of birds are now nesting further upriver on the Astoria-Megler Bridge, an unintended consequence 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. 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 ESI prior to implementation of the plan, with estimates now as high as 9.8% and 13.4% on SR steelhead and SR sockeye, respectively. The number of cormorants nesting at other sites in upper CRE and Lower Columbia River (LCR) has also increased in recent years, with predation estimates by cormorants nesting at the Troutdale Towers in LCR as high as 4.5% and 3.6% of SR steelhead and SR sockeye, respectively. Given these findings, management of cormorants in the CRE has not improved smolt survival, as intended, and adaptive management to dissuade cormorants from nesting on Astoria-Megler Bridge and other sites in upper CRE and LCR are needed to meet the management objectives of the plan. Adaptive management aimed at restoring the cormorant colony on ESI to the target colony size specified in the plan (5,400–5,900 pairs), while greatly reducing or eliminating cormorant colonies upstream of ESI, is recommended to achieve the original goals of the management plan.

OTHER PISCIVOROUS COLONIAL WATERBIRDS

Predation/consumption on juvenile salmonids by California gulls, ring-billed gulls, Brandt's cormorants, and American white pelicans was highly variable depending on the predator species, salmonid species, colony, and year. Several of the colonies investigated as part of this study posed little threat to smolt survival, while predation/consumption by other colonies, particularly larger-sized gull colonies located on islands in middle and lower Columbia rivers, consumed an appreciable proportion of available smolts. Based on the impacts of gulls nesting on Miller Rocks in The Dalles Reservoir on smolt survival, nest dissuasion activities were implemented at that colony (see *Nest Dissuasion Activities*) and there was some evidence that dissuasion activities have slightly reduced the size of the colony and predation/consumption rates. Although there is evidence that major reductions in the size of certain gull colonies (i.e., large colonies on or near the Columbia River) would increase smolt survival (Evans et al. 2023), the proportion of smolts consumed by gulls at dams that were dead or moribund is unknown and warrants additional research. Results of this and other studies indicated that consumption of smolts by gulls from several colonies in the region, particularly those on islands at considerable distances 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, ring-billed gulls nesting in the CRE, and double-crested cormorants nesting on in Potholes Reservoir and Shoal Island in Lenore Lake (both off-river cormorant colony locations) pose little threat to the smolt survival; these colonies should not be of concern to fisheries managers.

There are currently no management plans aimed at reducing the size of pelican colonies in the Columbia River basin (CRB). Predation rates on SR and UCR smolts by American white pelicans on Badger Island, the largest pelican colony in CRB, were low (< 1% per ESA-listed salmonid ESU/DPS). Recent research, however, indicates pelican predation on specific salmonid stocks from the Yakima River and possibly other large tributaries (e.g., Umatilla River) were substantial in some years (Evans et al. 2024b). Pelicans are also capable of consuming adult-sized fishes, including adult sockeye salmon, and predation effects can be substantial in some years, with upwards of 8.4% of the adult sockeye run passing Bonneville Dam consumed in some years. Management aimed at reducing pelican predation at specific hot spots (e.g., fishways, weirs, shallow water habitat, and other pinch points were smolt are more susceptible to predation) could be effective. A better understanding of how biotic (e.g., prey density, fish travel times, and pelican foraging behavior) and abiotic (temperature, flow) factors influence adult sockeye susceptibility to pelican predation are also warranted and could provide important insight into developing effective management actions to reduce pelican predation in the CPR.

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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 northern 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 2007–2024 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 2007–2023 are those previously reported by Evans et al. (2024).

	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
2024	<0.1%	0.2% (0.1–0.7)	-	-	0.6% (0.2–1.3)	0.9% (0.4–1.7)
N	1,231	2,198			2,611	3,800

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
2024	<0.1%	<0.1%	-	-	<0.1%	1.3% (0.7–2.6)
N	1,231	2,198			2,611	3,800

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
2024	<0.1%	<0.1%	-	-	<0.1%	1.7% (1.0–3.2)
N	1,231	2,198			2,611	3,800

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
2024	<0.1%	<0.1%	-	-	<0.1%	1.5% (0.9–2.6)
N	1,231	2,198			2,611	3,800
			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
2024	<0.1%	<0.1%	-	-	<0.1%	0.2% (0.1–0.5)
N	1,231	2,198			2,611	3,800

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).

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

Table 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 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2020	0.7% (0.5-1.1)	0.3% (0.1-0.7)	0.4% (0.2-0.9)	-	1.1% (0.6-2.2)	5.4% (3.8-7.9)	5.9% (4.5-8.1)	4.5% (3.3-6.4)
N	20,246	3,389	4,895		2,122	3,157	11,868	5,894
2021 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2022 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2023 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2024 *N*	NA	NA	NA	NA	NA	NA	NA	NA

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 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2020 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2021 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2022 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2023 *N*	NA	NA	NA	NA	NA	NA	NA	NA
2024 *N*	NA	NA	NA	NA	NA	NA	NA	NA

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

2024	1.3% (0.2–4.4)	0.7% (0.1–2.5)	-	-	2.3% (0.9–5.1)	2.5% (1.2–4.8)
N	1,231	2,198			2,611	3,800

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
2024	0.4% (0.1–1.7)	0.1% (0.1–0.6)	-	-	1.1% (0.4–3.6)	0.7% (0.3–1.3)
N	1,231	2,198			2,611	3,800

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
2024	1.0% (0.2–3.6)	0.6% (0.1–1.9)	-	-	1.0% (0.3–2.8)	8.5% (5.6–13.2)
N	1,231	2,198			2,611	3,800
			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
2024	0.5% (0.2–1.2)	1.5% (0.3–4.4)	1.4% (0.2–4.7)	-	5.2% (2.2–10.8)	-
N	9,074	1,771	1,246		1,802	
			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
2024	0.5% (0.2–1.1)	0.7% (0.1–2.7)	1.5% (0.4–4.1)	-	1.7% (0.4–4.7%)	-
N	9,074	1,771	1,246		1,802	

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
2024	<0.1%	<0.1%	-	-	<0.1%	0.2% (0.1–0.9)
N	1,231	2,198			2,611	3,800

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 Murdock Towers in Bonneville Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detections at John Day Dam) and Upper Columbia River (UCR; based on detections at John Day Dam) for spring (Sp) summer (Su) and Fall run fish. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable estimates.

	Murdock Towers, Bonneville Reservoir					
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2024	0.3% (0.1–1.2%)	0.9% (0.3–2.3%)	0.9% (0.5-5.0)	-	0.7% (0.1–3.3%)	-
N	3,408	2,284	933		1,191	

Table A9. 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
2024	4.4% (2.6–10.8)	2.8% (1.8–7.6)	7.1% (2.8–13.3)	9.1% (2.8–18.7)	10.9% (5.0–20.8)	9.9% (4.9–17.3)	6.3% (1.5–13.8)
N	14,171	2,462	4,874	2,117	4,333	7,623	2,427
Troutdale Towers, Lower Columbia River							
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2022	2.4% (1.3–4.3)	0.7% (0.1–2.0)	1.7% (0.4–3.6)	4.4% (1.2–9.5)	3.2% (0.8–7.0)	2.8% (1.2–5.0)	3.2% (1.2–6.1)
N	14,345	2,069	3,390	1,677	1,791	7,935	3,294
2023	2.9% (1.2–8.4)	0.5% (0.1–1.9)	1.5% (0.6–4.6)	2.1% (0.8–8.1)	3.7% (1.3–12.6)	3.1% (1.3–9.4)	2.5% (1.1–6.7)
N	17,539	3,738	6,283	2,904	2,245	10,670	5,272
2024	3.2% (1.9–6.9)	2.4% (1.0–5.7)	2.7% (1.4–6.4)	3.0% (1.4–6.4)	4.8% (2.6–9.3)	3.8% (2.2–6.9)	5.9% (3.2–12.1)
N	14,171	2,462	4,874	2,117	4,333	7,623	2,427

Table A10. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by American White Pelicans on Badger Island and Crescent 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
2024						
N						

			Cresent Island, McNary Reservoir			
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2024	< 0.1%	< 0.1%	-	-	< 0.1%	< 0.1%
N	1,231	2,198			2,611	3,800

Table A11. 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 was updated with data and results from 2023 and 2024.

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 was 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. This approach 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, 2023–2024) 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. As such, a likely small but unknown number of tagged fish could have been 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, cohort was defined with specific parameters for survival, predation, and recapture, with,

> $\mathbf{\Theta}^c$**,** 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 $d \in \{1,2,\ldots,D-1\}$ or some other cause of mortality for $d = D$, is the reach of river immediately preceding recapture opportunity j

and

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

and a common probability of tag recovery among cohorts was assumed 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,

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

where,

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

To evaluate if survival and predation probabilities of tagged steelhead differed by rear-type, the odds of predation was compared 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 71.6–79.9% of all tagged fish sampled during 2008-2024 (*Table B.1*). Ratios of hatchery to wild fish were also relatively consistent throughout 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%)
2024	3,038 (79.9%)	762 (20.1%)

Predation/consumption probabilities by rear-type varied by avian predator species, river reach, and year (2008–2024; *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 22.2% (21.0–24.1%) compared with 12.3% (11.1–13.6%) 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*). Differences 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.

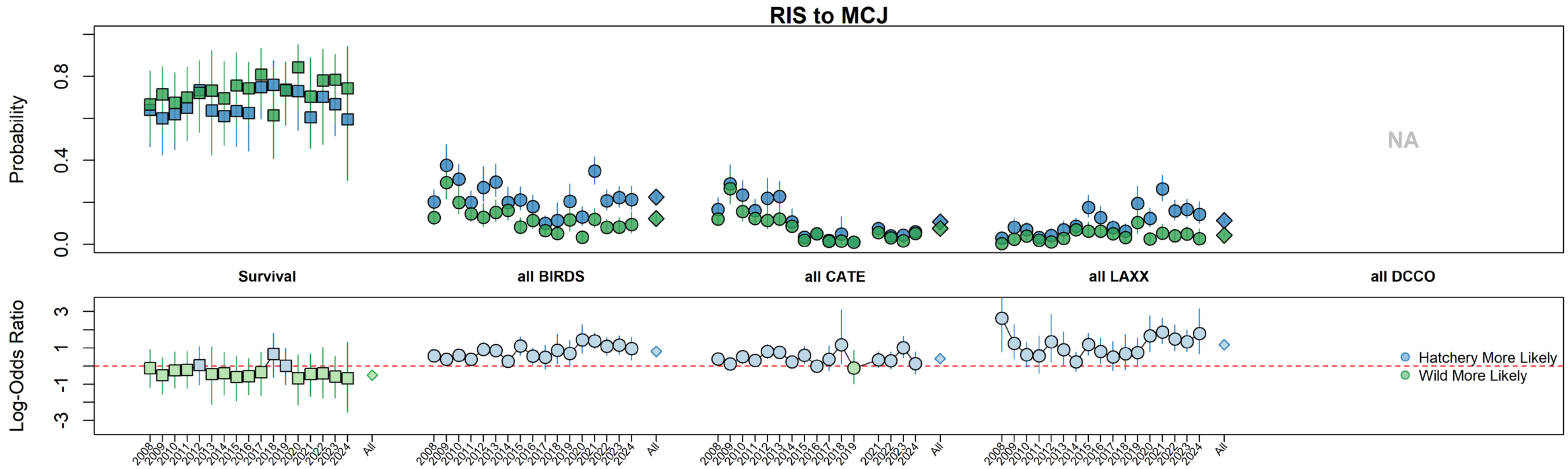

Figure B.1. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from Rock Island Dam (RIS) to McNary Dam (MCN) during 2008–2024 (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 overlapping 0 associated with relative differences that were not statistically significant. NA denotes that predation/consumption or survival probabilities were not available for relative comparisons.

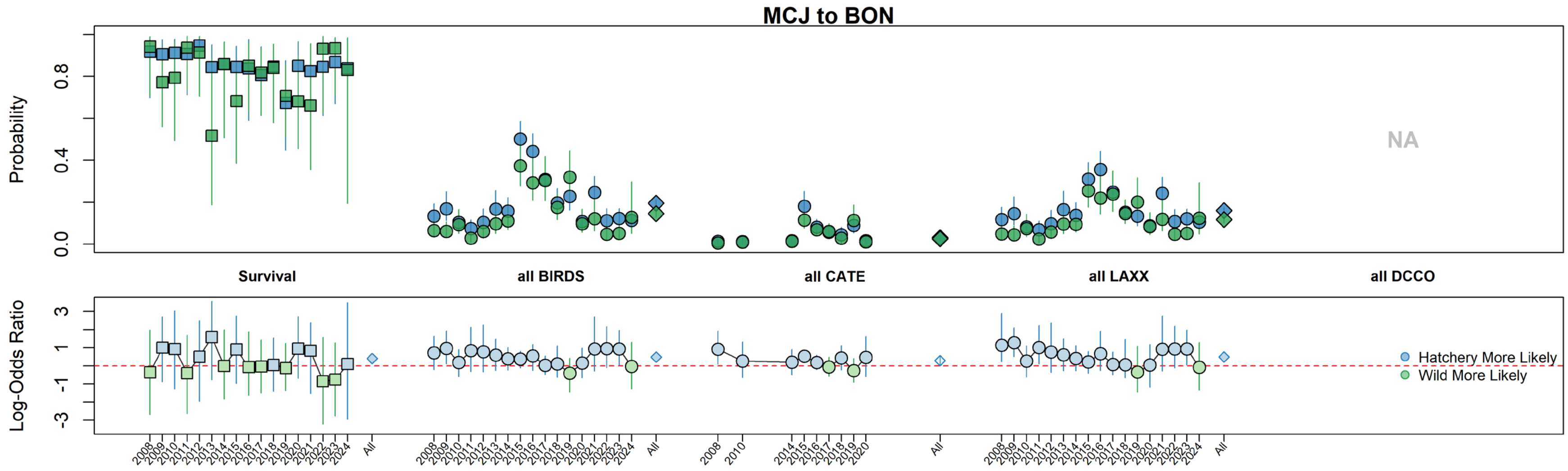

Figure B2. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from McNary Dam (MCN) to Bonneville Dam (BON) during 2008–2024 (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 overlapping 0 associated with relative differences that were not statistically significant. NA denotes that predation/consumption or survival probabilities were not available for relative comparisons.

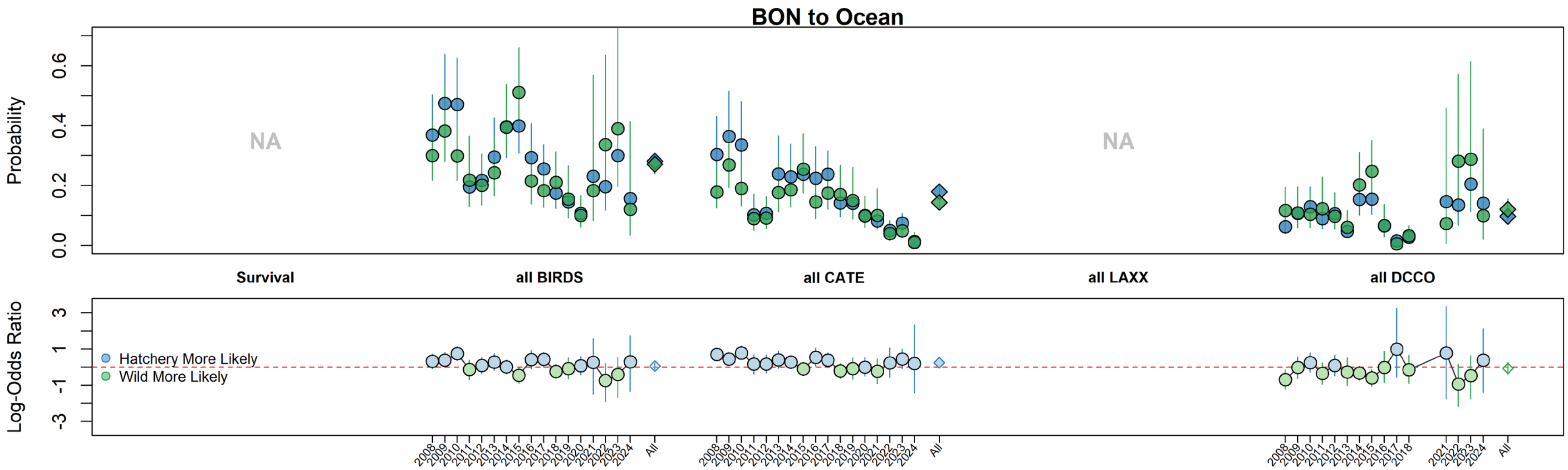

Figure B3. Relative comparisons of annual survival and predation/consumption probabilities (proportion of available fish) of Upper Columbia River steelhead by rearing-type (hatchery, wild) to predation by colonies of Caspian terns (CATE), California and ring-billed gulls (LAXX), and double-crested cormorants (DCCO) during smolt passing from Bonneville Dam (BON) to the Pacific Ocean during 2008–2024 (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 overlapping 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). Due to these low predation rates, adequate data to investigate relative differences in predation by rear-type due to DCCO and AWPE predation in this river reach were not available. Furthermore, birds from these colonies do not 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 19.6% (18.0–21.2%) and 14.6% (12.9–16.6%) 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 28.1% (25.7–30.1%) and 27.1% (24.0–31.0%) for hatchery and wild smolts, respectively. Annual trends in CATE and DCCO predation also followed very similar patterns across the 17-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).

These 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 differences 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 upstream CATE and LAXX colonies 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: COMPARISONS OF PREDATION AND SURVIVAL OF UPPER COLUMBIA RIVER STEELHEAD SMOLTS FROM SELECT HATCHERIES

Introduction: Steelhead smolt hatchery programs in the middle Columbia River differ in their methods of fish rearing and release. Understanding if and how rearing and release strategies influence smolt susceptibility to avian predation/consumption may help managers select management strategies that aid in steelhead smolt survival during outmigration. The goal of this study was to create a relative comparison of predation and survival of steelhead released into the middle Columbia River to determine if specific hatchery programs produced fish that were less susceptible to predation and more likely to survive outmigration. Predation and survival rates were estimated for large groups of PIT-tagged steelhead smolts (e.g., >5,000) that were released in the Methow and Okanogan Rivers and were compared based on the hatchery program, release site, and year.

Data Compilation: Large groups of hatchery reared steelhead from the Methow and Okanogan River programs were selected for comparison from 2021 to 2024. Key rearing and release characteristics varied by each program, including fish age (1 or 2 years), brood type (Hatchery {H}xH, HxWild {W}, and WxW), acclimation period/site, and release type (volitional, direct; *Table C1*). Hatchery programs were identified using uploads from the Columbia River Data Access in Realtime (DART) database by querying hatchery, release site, and brood year (Columbia Basin Research 2024). Methow Basin Conservation 1-Year programs were reared at the Winthrop National Fish Hatchery (WINT) or Wells Hatchery (WELH), were released from the Twisp River (TWISPR) or WINT, and had a brood year one less than release year. Methow Basin Conservation 2-Year programs were fish reared in the WINT or WELH hatchery, were released from TWISPR or WINT, and had a brood year two less than release year. Methow Conservation Program Early Winters Natural Acclimation Pond were identified by their hatchery as WINT and release site EARLWP. Methow Safety Net Program fish were identified as originating from the Wells Hatchery and being released into the Methow River (METHR). Columbia River Safety Net Program fish were identified as originating from and being released from the Wells Hatchery. Okanogan Safety Net fish originated from the Wells Hatchery and were released at Salmon Creek (SALMOC), Antoine Creek (ANTOIC), or Similkameen Acclimation Pond (SIMILP). Okanogan Steelhead Conservation were fish that originated at Wells Hatchery and were released at St Mary's Acclimation Pond (STMARP). Finally, previously tagged wild fish that were recaptured (passively detected) at the Rocky Reach Dam juvenile bypass facility (RRJ) were included in the analysis for comparison to hatchery fish detected at RRJ from each program.

Table C1. Key characteristics of each hatchery program, including brood type, age at release, acclimation, and release type. For Brood Type, W represents wild, and H represents hatchery.

Program/Production Group	Brood Type	Age at Release	Acclimation	Release Type
Methow Basin Conservation 1-Year Smolt	Mix of WxW and HxW	1 year	Limited spring acclimation	Short-term Volitional Release
Methow Basin Conservation 2-Year Smolt	Mix of WxW and HxW	2 years	Small subset, spring acclimation	Short-term Volitional Release
Columbia River Safety Net Program	HxH	1 year	No acclimation	Direct Plant
Methow River Safety Net Program	HxH	1 year	No acclimation	Direct Plant
Okanogan Conservation 1-year smolts	Mix of WxW and HxW	1 year	Spring Acclimation	Short-term Volitional Release
Okanogan Safety Net 1-year smolts	HxH	1 year	Reared at Wells Fish Hatchery	Direct Plant
Wenatchee River Steelhead Conservation	Mix of WxW and HxH	1 year	Overwinter Acclimation	Limited Volitional
Early Winters	Mix of WxW and HxH	2 years	Spring Acclimated in Natural Pond	True Volitional

Methods: Steelhead smolts were tagged and released into the middle Columbia with some proportion subsequently detected (recaptured) alive at a downstream location (Rocky Reach Dam, Rock Island Dam {RIS}, McNary Dam {MCN}, John Day Dam, Bonneville Dam {BON}, or pair trawl and/or pile dike antennas in the estuary) or recovered dead on an avian colony (see *Map 1* for list of colonies and locations). Rocky Reach dam (RRJ) was used as the 'starting point' for estimates of predation/consumption and survival, thus only individuals that were last detected alive at RRJ were included in relative comparisons of predation and survival. Annual survival and predation/consumption probabilities (proportion of available fish) were generated using the joint mortality and survival (JMS) model of Payton et al. (2019) for each conservation program by year. Full details of the JMS model are provided in Payton et al. (2019; see also Methods & Analysis, Cumulative Predation & Survival section). Comparisons of survival and predation/consumption probabilities by hatchery program were investigated based on the avian predator species (terns, gulls, cormorants, pelicans, or all birds combined) and the river reach (RRJ to RIS, RIS to MCN, MCN to BON}, and BON to the estuary) where predation occurred (see also Methods & Analysis, Cumulative Predation and Survival section).

Results & Discussion: Predation/consumption probabilities by hatchery program varied by avian predator species, river reach, and year (2021–2024; *Table C2*, *Figure C1*). Results indicated that on average across years, there was little to no difference in predation rates between hatchery programs, but there were differences in predation between the hatchery and wild fish in some years and river reaches. Total mortality (1 – survival) among hatchery groups and wild fish were similar within years, though there was some variation across years. In most years, credible intervals for each program overlapped, indicating that there is little to no evidence that certain programs had significantly higher or lower predation and survival probabilities (*Figure C1*). On average, predation rates from all bird colonies during smolt passage from RRJ to BON across all hatchery programs evaluated, averaged 39.7% (33.4–47.1%) in 2021, 25.5% (20.2–32.8%) in 2022, 25.6% (20.5–31.5%) in 2023, and 30.8% (24.0–39.1%) in 2024. Estimates of predation on wild steelhead during passage from RRJ to BON were lower than that of the hatchery program fish in all years at 19.2% (13.1–27.6%) in 2021, 10.3% (6.1–17.1%) in 2022, 10.0% (5.3–19.5%) in 2023, and 13.0% (8.7–18.5%) in 2024. However, the overlap of credible intervals between wild and hatchery program steelhead suggests little difference in some years, particularly when considering predation by colonies downstream of BON in the estuary, where an appreciable proportion of both hatchery and wild steelhead were consumed (*Figure C2*).

The hatchery program with the greatest average predation estimates across years was the Okanogan Steelhead Conservation program, with an average of 35.1% (27.6–44.3%) from all bird colonies during passage from RRJ to BON (*Figure C1*). This was followed by the Methow Basin Year-2 and Year-1 programs, with average estimates of predation of 33.6% (28.5–39.8%) and 33.5% (27.0–41.3%) respectively. Next, the average estimate of predation for the Methow Safety Net steelhead smolts was 30.3% (24.2–37.6%). The Okanogan Safety Net program had an average annual predation estimate of 26.9% (21.9–32.9%). The hatchery program with the lowest average predation estimates during 2021–2024 was the Columbia River Safety Net at 21.8% (16.8–27.8%). There is little evidence, however, that the differences in these estimates were significant enough to conclude that any one of these programs produces smolts that were less susceptible to predation by predator species (terns, gulls, cormorants, and pelicans) and colonies included in the analysis.

From RRJ to BON, gull species overall represented the greatest consumption rates across hatchery programs, averaging 31.3% (24.9–38.0%) in 2021, 21.5% (16.1–28.0%) in 2022, 17.9% (13.6–23.0%) in 2023, and 22.6% (16.3–30.4%) in 2024. Conversely, gull consumption rates for wild fish averaged 7.8% (4.0–13.8), and ranged from 6.3% (3.2–12.4%) in 2022 to 10.4% (5.8–16.1%) in 2021. Gulls are plunge-diving, surface-oriented predators and are known to select their prey based on size. Hatchery steelhead are, on average, larger than wild steelhead. As such, it is possible that size-selective consumption is contributing to greater predation of hatchery smolts compared with wild smolts by gulls (Hostetter et al. 2012, Roby et al. 2016, Hostetter et al. 2023).

Terns were the second greatest predation threat to hatchery and wild steelhead from RRJ to BON but estimates were considerably lower than that of gulls. Estimates of predation from terns on hatchery program steelhead ranged from an average of 2.9% (2.0–4.7%) in 2022 to 6.4% (4.4–9.3%) in 2024. Results found that wild steelhead experienced similar estimates of predation by terns, ranging from 1.1% (0.2–2.9%) in 2023 to 7.6% (4.4–12.2%) in 2021.

Results indicated that cormorants posed a relatively small predation risk to smolts between RRJ and BON but was a significant source of predation downstream of BON in the Columbia River estuary (*Figures C1, C2*). Predation estimates from RRJ to the Pacific Ocean demonstrated the growing impact of predation by cormorants that exist downstream of BON (*Figure C2*). While groups from different hatchery programs experienced similar predation rates within the same year, estimated predation rates from cormorants for all groups increased from 2022 to 2024 (8.0% [4.6–15.6%] in 2022, 14.3% [7.6–26.2%] in 2023, and 13.9% [18.2–24.0%] in 2024). Increases in predation were largely attributed to increases in the size of cormorant colonies on the Astoria-Megler Bridge and the Troutdale Towers (see also the Cumulative Predation & Survival section). Unlike gulls and terns, cormorants are pursuit-predators that do not select prey based on size, which may explain, in part, why wild and hatchery are more equally susceptible to cormorant predation (see also Hostetter et al. 2023 and the Predation Rate section). Colonies of cormorants and pelicans that foraged on UCR steelhead between RIS and MCN maintained low predation probabilities across hatchery groups and wild fish: Hanford Island cormorants averaged 0.6% (0.2–2.2%), Foundation Island cormorants averaged 0.4% (>0.1–2.1%), and Badger Island pelicans averaged 0.3% (>0.1–1.3%). Collectively, results of this study are congruent with past studies, which concluded that there is not consistent evidence that cormorants disproportionately consumed steelhead or other salmonid species based on size or rear-type (Hostetter et al. 2012, Payton et al. 2016, Hostetter et al. 2023).

Total mortality across hatchery programs was often similar within years but varied across years (*Figure C1*). While some programs within a year had significant differences in total mortality (e.g., Methow Basin Year-1 Smolts had greater total mortality the Methow Basin Year-2 Smolts in 2021), no clear patterns emerged. Averaged across all years, mortality from avian predation constituted 59.7% (55.6–66.6%) of all mortality sources (i.e., total mortality) for the Okanogan

Conservation Program, 54.2% (50.1–59.5%) for the Methow Basin Year-2 programs, 52.9% (49.3–59.0%) for the Methow River Safety Net program, 47.1% (42.9–54.3%) for the Early Winter program (2024 only), 46.0% (45.2–52.3%) for the Methow Basin Year-1 programs, 42.8% (38.8–48.6%) for the Okanogan Safety Net program, and 37.2% (35.8–42.1%) for the Columbia River Safety Net program. On average, mortality due to avian predation constituted 20.4% (15.0–29.0%) of all wild steelhead mortality during the study period. Differences in total mortality between hatchery and wild steelhead highlights the need to explore other, non-avian factors associated wild smolt mortality during outmigration, especially during outmigration through the hydro system (*Table C2, Figure C1*). For hatchery steelhead, however, avian predation was the dominate sources of smolt mortality during smolt passage from RRJ to BON.

Conclusion: The main goal of this study was to evaluate if certain hatchery programs produced fish that were less likely to succumb to predation and more likely to survive outmigration. Across four years (2021–2024), however, there was little evidence that certain hatchery programs yielded consistently higher or lower estimates of predation or total mortality during smolt passage from RRJ to BON. When compared to wild steelhead, results of this study indicated that hatchery-reared steelhead have increased susceptibility to colonial waterbird predation for some, but not all, predator species and colonies in the Columbia River basin (see also Predation Rate section and Hostetter et al. 2023). Further research to better understand if different rearing practices benefit steelhead smolt survival and/or reduce predation may include investigating specific components of each program such as release timing, release location, brood origin, fish length, and fish condition or health.

Table C2. Estimates of annual avian predation/consumption rates (95% credibility intervals), annual total mortality (1-survival) (95% credible intervals), and number of available PIT-tagged smolts (N) from Rocky Reach Dam (RRJ) to Bonneville Dam (BON) of tagged and released smolts from select steelhead release programs for the Methow and Okanogan Rivers between 2018–2024. The first releases for the Early Winters program began in 2024, thus only has one year of data for this analysis. Wild tagged fish detected at RRJ were included for comparison.

Year	Metric	Methow Basin 1-year Smolts*	Methow Basin 2-year Smolts**	Columbia River Safety Net	Methow River Safety Net	Okanogan Conservation	Okanogan Safety Net	Early Winters	Wild
2021	Predation	45.0% (38.7-52.9)	40.0% (34.7-46.4)	28.9% (23.5-35.6)	37.7% (31.5-44.5)	46.8% (37.7-57.2)	40.0% (34.4-46.0)	-	19.2% (13.1-27.6)
	Total Mortality	81.7% (78.3-84.2)	66.4% (63.4-69.4)	68.8% (62.9-73.4)	63.0% (57.9-67.4)	68.4% (62.3-74.1)	68.8% (64.3-72.5)	-	75.1% (70.3-78.8)
	N Released	10,674	24,589	4,990	4,976	4,998	14,983	-	-
	N at RRJ	3,578	8,686	2,574	2,714	1,846	4,822	-	1,293
2022	Predation	31.0% (25.4-38.7)	29.3% (24.6-34.9)	15.2% (11.0-20.1)	24.6% (19.4-31.6)	32.7% (25.1-41.6)	20.4% (15.8-26.3)	-	10.3% (6.1-17.1)
	Total Mortality	70.8% (62.7-76.0)	65.0% (60.3-69.3)	66.0% (58.0-72.5)	50.1% (40.8-57.8)	59.9% (54.1-65.5)	61.9% (55.1-67.2)	-	68.0% (59.2-74.9)
	N Released	16,244	24,430	4,996	4,999	4,993	14,991	-	-
	N at RRJ	3,907	8,326	2,448	2,559	1,911	4,251	-	972
2023	Predation	25.6% (20.3-31.2)	26.1% (21.9-30.7)	20.4% (15.8-25.7)	28.4% (22.1-35.3)	30.4% (24.4-37.6)	22.9% (18.4-28.8)	-	10.0% (5.3-19.5)
	Total Mortality	58.7% (45.8-65.4)	44.4% (39.6-49.7)	44.9% (36.4-52.3)	48.6% (40.5-55.2)	42.0% (34.7-49.7)	48.4% (42.1-54.2)	-	42.8% (30.4-42.8)
	N Released	10,483	25,390	4,992	4,986	4,989	14,976	-	-
	N at RRJ	2,912	7,467	2,763	1,773	1,592	3,942	-	825
2024	Predation	32.4% (23.7-42.5)	39.4% (32.8-47.2)	22.9% (17.0-29.7)	30.5% (23.9-39.0)	30.5% (23.3-40.7)	24.3% (19.1-30.4)	35.9% (28.6-44.3)	13.0% (8.7-18.5)
	Total Mortality	77.7% (51.1-88.8)	74.5% (65.8-79.7)	59.5% (39.0-69.1)	68.5% (58.3-75.5)	70.1% (54.3-81.1)	73.8% (65.2-79.0)	76.2% (81.4-66.7)	74.3% (64.1-80.4)
	N Released	4,983	18,325	5,021	4,964	4,994	14,995	8,869	-
	N at RRJ	859	4,445	2,629	2,048	1,524	3,791	2,394	2,016

** 1-year smolts are Methow Conservation Programs (WINT/WELH) 1-year programs (released from TWISPR and WINT)*

*** 2- year smolts are Methow Conservation Programs (WINT/WELH) 2-year programs (released from TWISPR and WINT)*

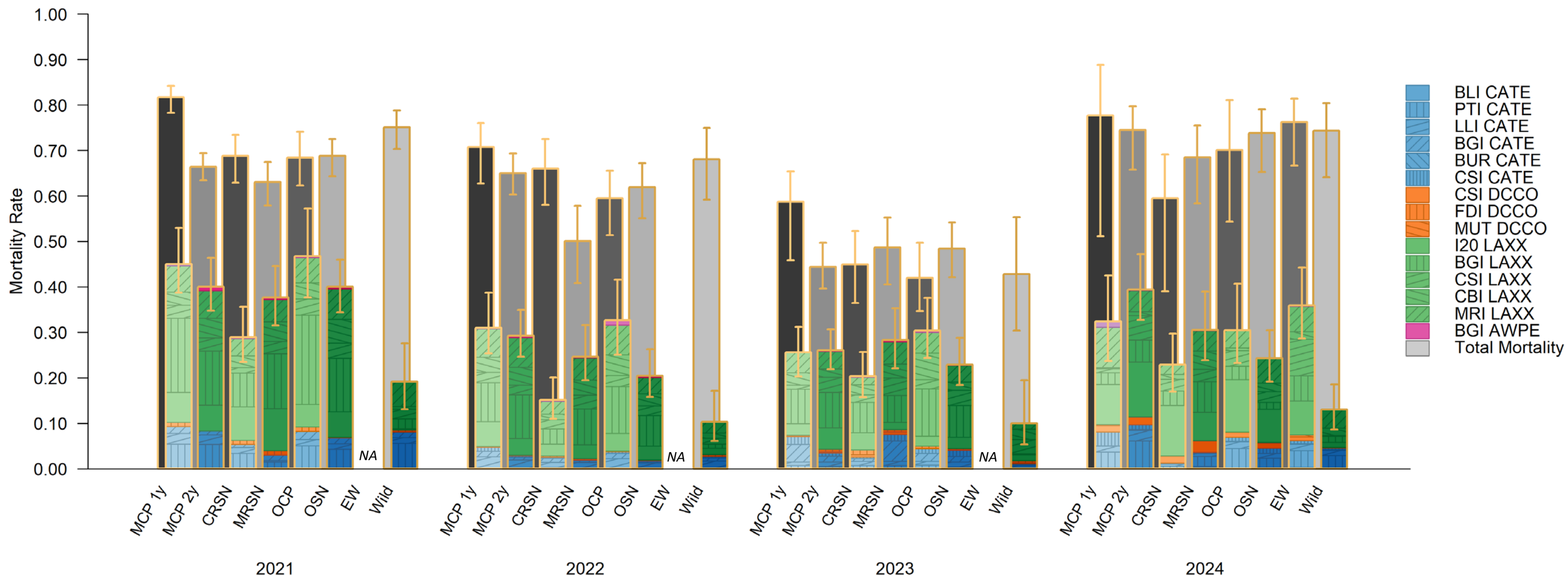

Figure C1. Estimated total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for select hatchery program and wild steelhead during passage from Rocky Reach Dam to Bonneville Dam. Hatchery programs include the Methow Basin Conservation 1-Year smolts (MCP 1y), Methow Basin Conservation 2-Year smolts (MCP 2y), Columbia River Safety Net (CRSN), Methow River Safety Net (MRSN), Okanogan Conservation Program (OCP), Okanogan Safety Net (OSN), and Early Winter (EW). No data exists for the Early Winter program before 2024. Wild steelhead detected at RRJ are labeled as 'Wild.' 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.

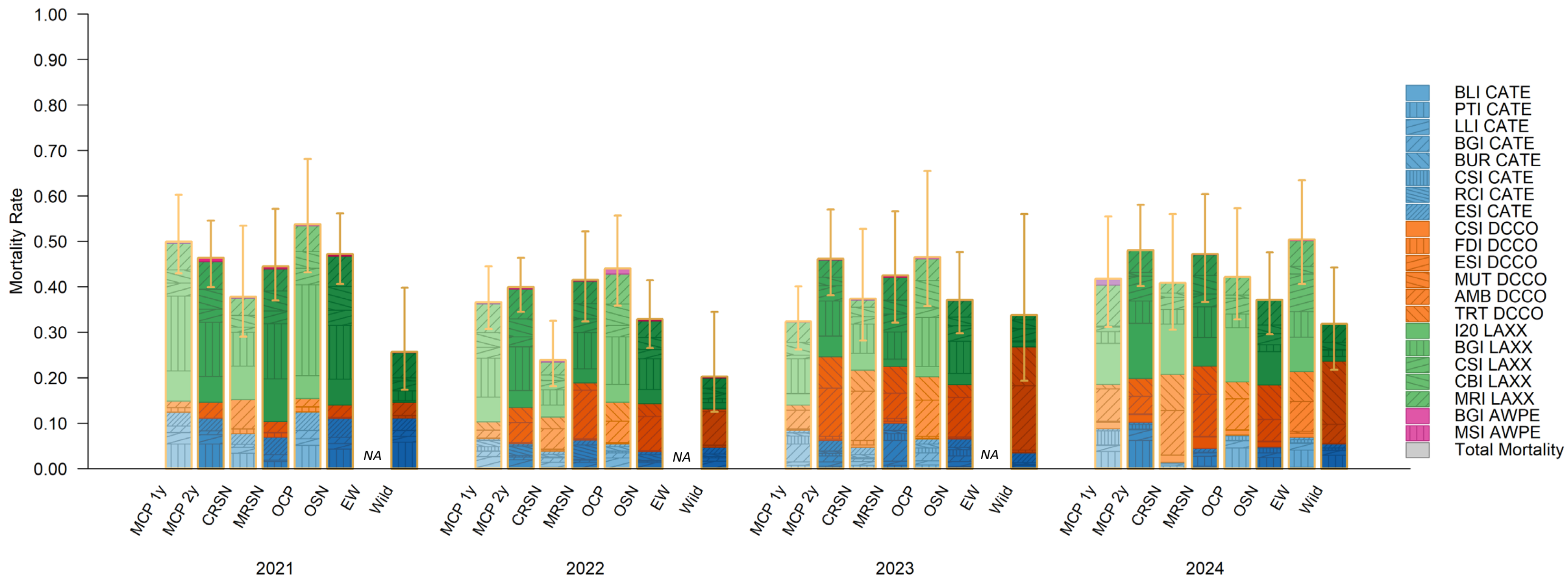

Figure C2. Estimated mortality associated with avian predation/consumption (colored bars) for select hatchery program and wild steelhead during passage from Rocky Reach Dam to the Pacific Ocean. Hatchery programs include the Methow Basin Conservation 1-Year smolts (MCP 1y), Methow Basin Conservation 2-Year smolts (MCP 2y), Columbia River Safety Net (CRSN), Methow River Safety Net (MRSN), Okanogan Conservation Program (OCP), Okanogan Safety Net (OSN), and Early Winter (EW). No data exists for the Early Winter program before 2024. Wild steelhead detected at RRJ are labeled as 'Wild.' 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.

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

Upper Columbia River Steelhead
Weekly Estimates of Predation and Total Mortality

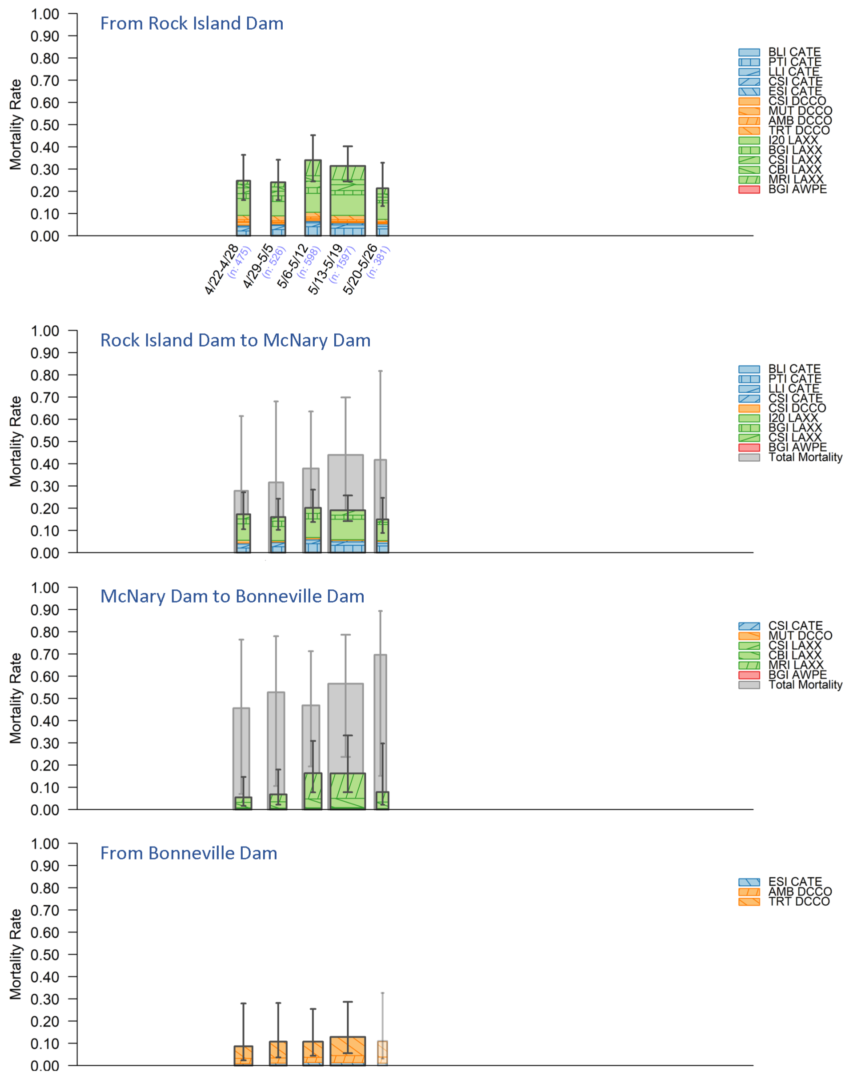

Figure D1. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Upper Columbia River steelhead during smolt passage from Rock Island Dam to the Pacific Ocean (top), Rock Island Dam to McNary Dam, McNary Dam to Bonneville Dam, and following 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).

Snake River Steelhead
Weekly Estimates of Predation and Total Mortality

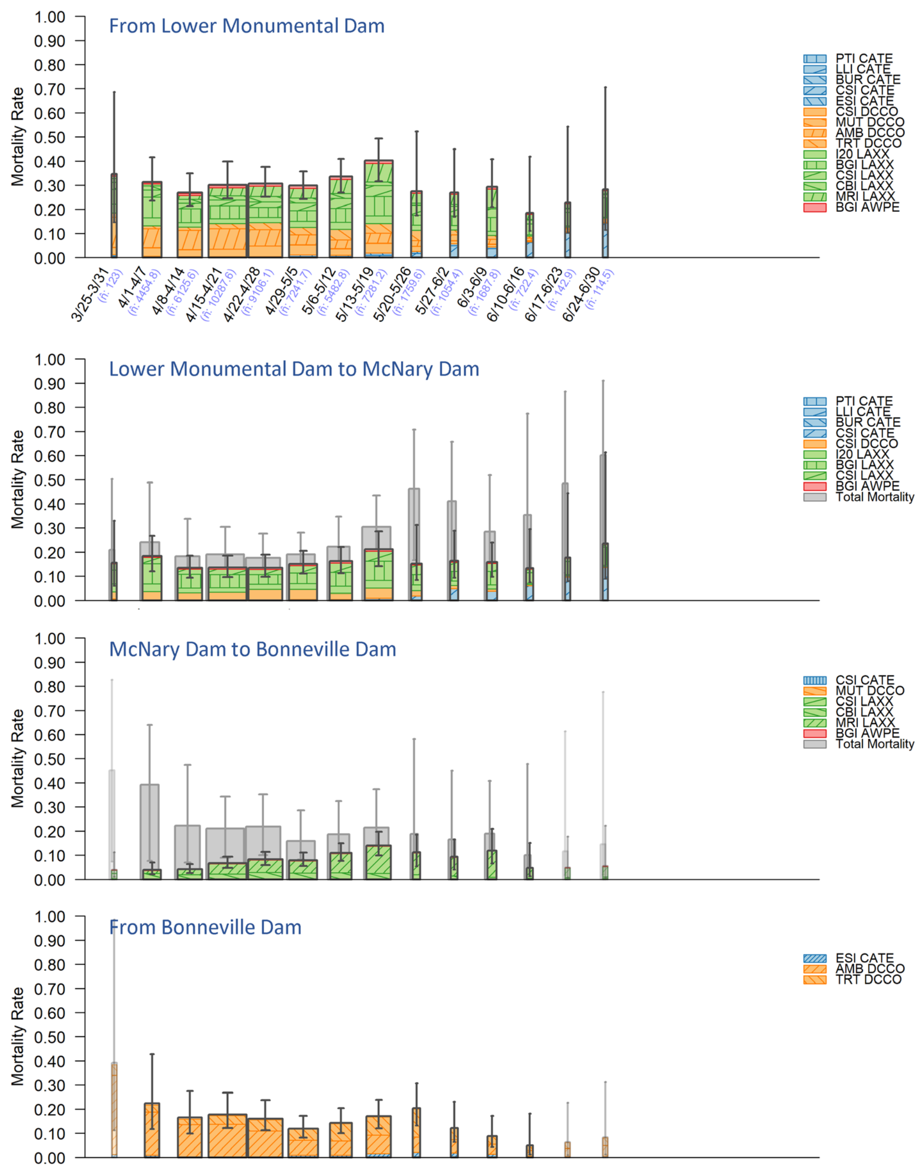

Figure D2. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River steelhead during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and following 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).

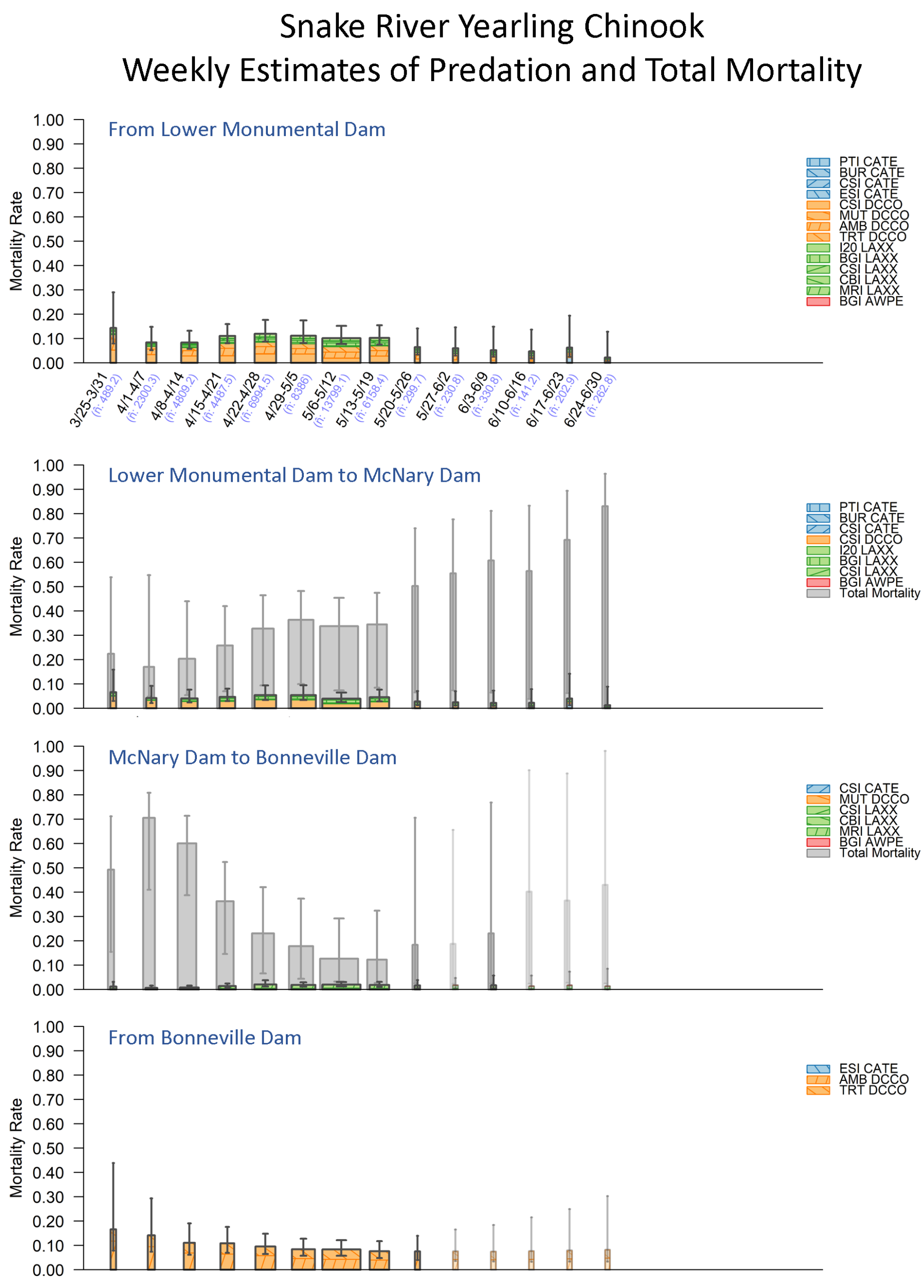

Figure D3. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River yearling Chinook during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and 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).

Snake River Subyearling Chinook Weekly Estimates of Predation and Total Mortality

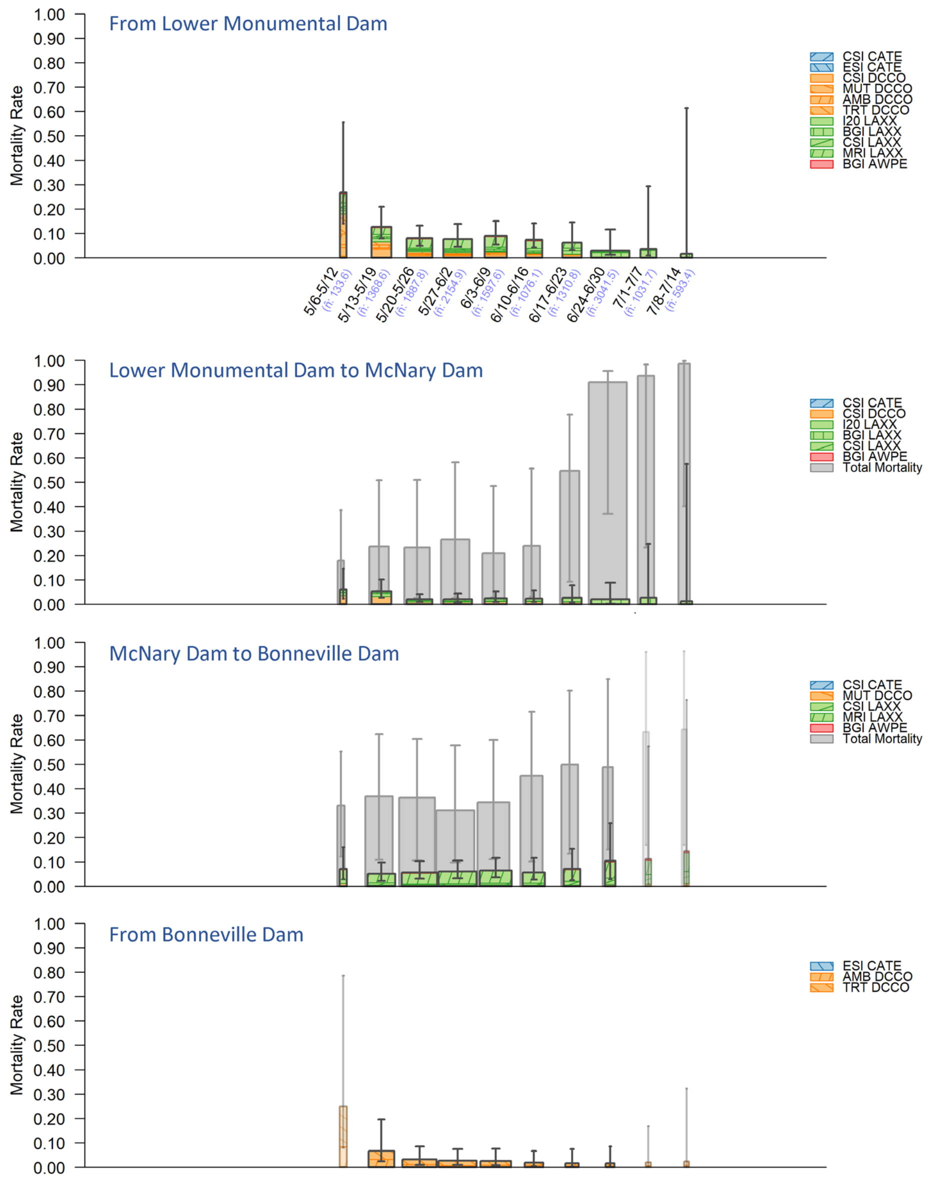

Figure D4. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River sub-yearling Chinook during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, 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).

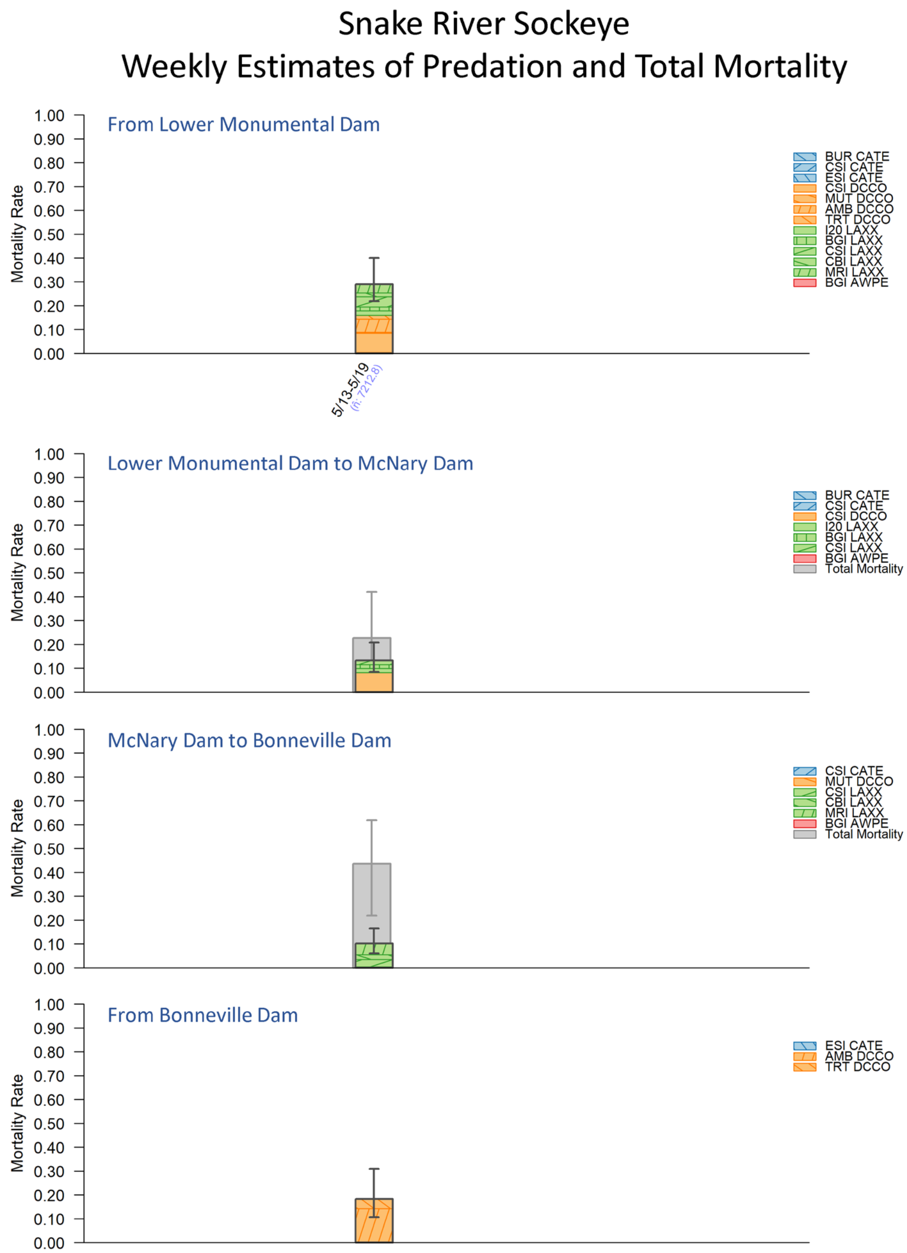

Figure D5. Estimated weekly total mortality (grey bars) and mortality associated with avian predation/consumption (colored bars) for Snake River sockeye during passage from Lower Monumental Dam to the Pacific Ocean (top), Lower Monumental Dam to McNary Dam, McNary Dam to Bonneville Dam, and following 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).

APPENDIX E: DOUBLE-CRESTED CORMORANT COUNTS ON NAVIGATION AIDS DURING 2022–2024

Table E1. Location of navigation aids for double-crested cormorant colonies in the Columbia River estuary. Location names are from Roby et al. 2021, Lawonn 2023, and the United States Coast Guard (U.S.C.G.) number (No.) 2024.

Colony Complex	U.S.C.G No.	Name in Lawonn 2023	Name in Roby et al. 2021	Latitude	Longitude
Estuary Navigation Aids RKM 0-22	-	Jetty A Tower	Jetty A Channel Marker	46.265954	-124.037809
	9945	Sand Island Range Front Light	Sand Island Channel	46.26582	-123.9929
	9950	Sand Island Range Rear Light	Sand Island Channel	46.2673	-123.98103
Estuary Navigation Aids RKM 22-51	10185	Harrington Point Channel 52	Estuary Channel Marker	46.2342	-123.71417
	10186	Tongue Point Channel Range	Estuary Channel Marker	46.23294	-123.71352
	10187	Tongue Point Channel Range	Estuary Channel Marker	46.23515	-123.70581
	10175	Harrington Point Range Front	Estuary Channel Marker	46.25591	-123.677
	10180	Harrington Point Range Rear	Estuary Channel Marker	46.26176	-123.6656
	10215	Miller Sands Dike Light 5	Estuary Channel Marker	46.25652	-123.66856
	10230	Miller Sands Dike Light 11	Estuary Channel Marker	46.26118	-123.64197
	10195	Miller Sands Range Front Light	Estuary Channel Marker	46.26241	-123.63666
	10235	Pillar Rock Lower Range Front	Estuary Channel Marker	46.25275	-123.54342
	10240	Pillar Rock Lower Range Rear	Estuary Channel Marker	46.25173	-123.52938
	10275	Pillar Rock Upper Range Front	Estuary Channel Marker	46.26073	-123.51556
	10280	Piller Rock Upper Range Rear	Estuary Channel Marker	46.2617	-123.50288
Estuary Navigation Aids RKM 51-234	10950	Martin Island Bybee Ledge	Not Reported	45.9571	-122.80797
	10955	Martin Island Bybee Ledge	Not Reported	45.95585	-122.80626
	11570	Vancouver to Bonneville Gary	Not Reported	45.5518	-122.33982
	11630	Fashion Reef Lower Range	Not Reported	45.58511	-122.12714
	11635	Fashion Reef Lower Range Rear	Not Reported	45.58629	-122.1193
	11720	Warrendale Lower Range Rear	Not Reported	45.61359	-122.03763

Table E2. Number of double-crested cormorant breeding pairs at known colonies on navigation aids downstream of Bonneville Dam during 2022–2024. NA denotes that counts were not available.

Colony	2022	2023	2024
Jetty A Tower	21	NA	37
Sand Island Range Front Light	23	NA	20
Sand Island Range Rear Light	22	NA	15
Harrington Point Channel 52 Light	0	0	0
Tongue Point Channel Range Front Light	7	8	5
Tongue Point Channel Range Rear Light	10	0	0
Harrington Point Range Front Light	41	47	53
Harrington Point Range Rear Light	7	1	0
Miller Sands Dike Light 5	46	65	47
Miller Sands Dike Light 11	0	0	0
Miller Sands Range Front Light	10	12	11
Pillar Rock Lower Range Front Light	0	30	45
Pillar Rock Lower Range Rear Light	19	21	0
Pillar Rock Upper Range Front Light	44	50	34
Piller Rock Upper Range Rear Light	36	78	70
Martin Island Bybee Ledge Channel Range Front Light	NA	15	18
Martin Island Bybee Ledge Channel Range Rear Light	NA	50	49
Vancouver to Bonneville Gary Island Light A	NA	10	NA
Fashion Reef Lower Range Front Light	NA	15	9
Fashion Reef Lower Range Rear Light	NA	32	28
Warrendale Lower Range Rear Light	NA	10	20
Total	286	444	461

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