AVIAN PREDATION IN THE COLUMBIA RIVER BASIN

2020 Final Annual Report

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EXECUTIVE SUMMARY

To address concerns about the impact of avian predation on the survival of Columbia River basin (CRB) juvenile salmonids (smolts), the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps of Engineers (USACE), and their management partners developed and implemented three separate management plans to reduce predation rates on smolts by piscivorous waterbirds nesting at four breeding colonies in the CRB: The Caspian tern (*Hydroprogne caspia*; hereafter referred to as "tern" or "terns") and double-crested cormorant (*Phalacrocorax auratus*; hereafter referred to as "cormorant" or "cormorants") breeding colonies on East Sand Island in the Columbia River estuary (CRE); and the tern colonies on Crescent Island (McNary Reservoir) and on Goose Island (Potholes Reservoir) in the Columbia Plateau region (CPR). The primary goal of these management initiatives was to reduce predation rates (proportion of available smolts consumed) on Endangered Species Act (ESA)-listed anadromous salmonid (*Oncorhynchus* spp.) populations (referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]) by reducing the size of or eliminating the breeding colonies at each of these four colony sites. Management initiatives implemented at these four colony sites have been primarily non-lethal strategies for terns (i.e. passive and active nest dissuasion) and primarily lethal (i.e. culling and egg-oiling) and non-lethal (i.e. nesting habitat management) strategies for cormorants. As part of the management plans for terns, the USACE created alternative nesting habitat for terns at various locations outside the CRB (i.e. northeastern California, southern Oregon, and San Francisco Bay) to compensate for reductions in tern nesting habitat on East Sand Island in the CRE and for elimination of tern nesting habitat on Crescent and Goose islands in the CPR.

Implementation of these management plans in 2020 was carried out by the USACE for the tern and cormorant colonies on East Sand Island and by the Bureau of Reclamation at the tern colony on Goose Island; active nest dissuasion was not conducted at the Crescent Island tern colony in 2020. The primary objectives of this study were to evaluate the efficacy of management to reduce avian predation on ESA-listed juvenile salmonids in the CRB, and to assess the magnitude of predation on smolts by piscivorous waterbirds nesting at unmanaged colonies, namely those of California gulls (*Larus californicus*) and ring-billed gulls (*L. delawarensis*; hereafter referred to as "gull" or "gulls") and American white pelicans (*Pelecanus erythrorhynchos*; hereafter referred to as "pelican" or "pelicans"). Specifically, we sought to (1) locate and estimate peak colony size for all piscivorous waterbird colonies within foraging range of juvenile salmonids in the middle Columbia River, lower Snake River, lower Columbia River, and CRE, including information on any new or incipient colonies; (2) estimate colony-specific predation rates on smolts by piscivorous waterbirds (with cost sharing from the USACE); and (3) measure the cumulative effects of predation by birds from multiple breeding colonies on salmonid survival, including an investigation of the additive effects of avian predation. The continued monitoring of colony sizes, locations, and predation rates on salmonids by piscivorous waterbirds nesting at both managed and un-managed colonies in the basin will help ensure that the intended benefits of management efforts are achieved and sustained, and that the accrued benefits from colony management are not offset by the emergence of new and growing colonies at other locations within the basin. In short, this research will guide managers in developing and monitoring long-term management initiatives for avian predators that are science-based, defensible, cost-effective, and have a high probability of success.

Colony Size

Columbia Plateau Region: The primary objective of the seventh year of implementation of the *Inland Avian Predation Management Plan* (*IAPMP*) was to limit the numbers of terns breeding on Goose Island and on other islands in Potholes Reservoir, and on Crescent Island in McNary Reservoir, to less than 40 breeding pairs each to reduce impacts from tern predation on ESA-listed juvenile salmonids in the CPR. To accomplish this task, the suitable tern nesting habitat at these sites was nearly eliminated by installing a variety of passive nest dissuasion materials on Goose Island and other islands in Potholes Reservoir prior to the 2020 breeding season, and by planting vegetation (planted in 2016) on Crescent Island. On both Goose and Crescent islands, passive nest dissuasion materials and/or vegetation covered all areas where terns had previously nested, as well as all areas of open, sparsely vegetated habitat that might be used by ground-nesting terns or gulls. Once terns arrived to initiate nesting in 2020, active nest dissuasion (i.e. human hazing) was used to dissuade terns from nesting on Goose Island and on other islands in Potholes Reservoir, activities described in detail in a separate report (see *above*). No hazing has been required to prevent terns from nesting on Crescent Island since the first year of management in 2015.

Despite the use of passive and active nest dissuasion techniques on Goose Island in each of the previous six years, some terns continued to display high fidelity to the island as a nesting site in 2020. As many as 6 breeding pairs of terns were successful in raising young (total of four fledglings) on Goose Island in 2020, the first year that terns were successful in establishing a breeding colony on Goose Island since 2015. Nesting success on Goose Island in 2020, although limited, was likely due to a decline in the frequency (i.e. number of days/week) and duration (i.e. number of hours/day), as well as changes in timing, of nest dissuasion activities (i.e. hazing) in 2020 relative to previous years. The continued fidelity of terns to Potholes Reservoir despite ongoing nest dissuasion activities is unexpected but is likely due to terns having a long nesting history at the site and the persistence of a large gull colony on the island, both before and during management, which continues to attract prospecting terns to the site. Another factor that might explain the strong fidelity of terns to the Potholes Reservoir is the paucity of alternative tern colony sites in the vicinity. As was the case in previous years during the management period, tern use of Goose Island for roosting and nesting in 2020 was largely limited to areas near the shoreline of the island that became exposed during the nesting season as reservoir levels receded. However, in 2020 terns were successful in establishing a small nesting colony on top of the island near the historic colony area, where we believe some, if not all, the fledglings were produced. Prior to management (2005–2013), an average of 367 breeding pairs of terns nested at the colony on Goose Island.

Willow and other native vegetation plantings on Crescent Island in 2016 have rendered the island unsuitable for tern nesting by eliminating all the open, bare-ground habitat that terns prefer for nesting. That, along with other passive nest dissuasion techniques used on Crescent Island (i.e. fencing, stakes, rope, flagging, and placement of woody debris), resulted in both terns and gulls abandoning the island as a nesting site during the 2015–2019 breeding seasons. In 2020, California gulls returned to Crescent Island to nest and established a breeding colony of approximately 400 nesting pairs at a site on and near the old tern colony site. Although terns were observed flying over Crescent Island during the 2020 breeding season, they did not attempt to nest at the site. The reason(s) for the return of a gull breeding colony on Crescent Island in 2020 are unknown but likely due to the removal of fencing and other passive nest dissuasion materials from the island and mortality of willows and other native vegetation on the island. Signs of intensive beaver herbivory have been observed on the island in previous years and was one of perhaps several factors affecting vegetation cover on the Crescent Island. If the gull colony on Crescent Island becomes more established in future years, we expect that the amount of open, unvegetated habitat on the island will also increase, which may ultimately lead to the reestablishment of a Caspian tern colony.

The complete abandonment of Crescent Island by nesting terns beginning in the first year of management (2015) was somewhat unexpected because terns and gulls had nested consistently on Crescent Island for nearly three decades prior to management. One factor that likely contributed to the absence of nesting terns on Crescent Island was the successful dissuasion of gulls from nesting on Crescent Island during 2015–2019; terns are breeding associates of gulls and are attracted to nest at the periphery of gull colonies. At Goose Island, gull nesting could not be prevented using passive and active nest dissuasion techniques, whereas at Crescent Island gulls never habituated to the nest dissuasion techniques implemented there. Instead, gulls abandoned Crescent Island as a nesting site and some, if not most, of these gulls established a new colony on Badger Island, located on the Columbia River just one kilometer upriver from Crescent Island, during 2015–2019. Similarly, most terns displaced from Crescent Island relocated to an unmanaged colony site in the Blalock Islands, John Day Reservoir on the Columbia River, 70 river kilometers downriver from Crescent Island, during 2015–2019. Small numbers of terns had nested in the Blalock Island during the previous decade. Prior to management (2005–2013), an average of 403 breeding pairs of terns had nested on Crescent Island.

Aerial, ground, and boat-based surveys were conducted in the CPR to determine where terns displaced from the managed colonies in Potholes Reservoir and at Crescent Island might attempt to nest. In 2020, terns attempted to nest at three different colony sites in the CPR: at one managed site (Goose Island) and at two unmanaged sites (Blalock Islands and Lenore Lake). In 2020, the tern colony at the Blalock Islands in John Day Reservoir was the largest in the CPR at 150 breeding pairs; this is down from 379 pairs that nested in the Blalock Islands in 2019 and down from the average colony size during the management period (391 pairs). After the Blalock Islands tern colony, the next largest tern colony in the CPR during 2020 was located on two islands (North Rock and Shoal Island) in Lenore Lake, at 53 breeding pairs; this is up from 48 pairs in 2019 and similar to the average colony size during the management period (52 breeding pairs). The Goose Island tern colony, at just 6 breeding pairs in 2020, was up from 0 pairs during 2016–2019. Only the tern colonies at Lenore Lake and on Goose Island were successful in rearing young to fledging in 2020, with a minimum of four fledglings produced at each colony. A total of 209 breeding pairs of terns nested in the CPR during 2020, the lowest number of terns to nest in the region since annual monitoring of Caspian tern colonies in the basin began in the late 1990's. The cause(s) of the decline in the regional breeding population of terns in 2020 is unknown; perhaps management at formerly the two largest tern colonies in the CPR during each of the previous seven breeding seasons has resulted in most terns relocating to nesting colonies outside the region, as was expected.

As part of this work, we also estimated the colony size of other piscivorous waterbirds (see *above*) in the CPR during the 2020 breeding season. Because visits to these colonies were less frequent (1–2 times per month) as compared to our monitoring of tern colonies (2–4 times per month) in the region, our estimates of peak colony size at these other piscivorous waterbird colonies are less accurate (i.e. actual peak colony size could be somewhat higher or lower). There were a total of nine active gull colonies in the CPR region during the 2020 breeding season, ranging in size from about 400 breeding individuals (Crescent Island) to over 11,000 breeding individuals (Goose Island). Cormorants were confirmed nesting at five colony sites in the CPR during 2020, with colony size ranging from 1 pair (Miller Rocks) to 333 pairs (Harper Island in Sprague Lake). Finally, the size of the Badger Island pelican colony was estimated to be 3,165 breeding individuals in 2020. These data were used to identify which colony sites to scan for smolt PIT tags following the 2020 breeding season (see *below*).

Columbia River Estuary: In 2020, both the implementation of avian predation management initiatives and monitoring of the tern and cormorant colonies on East Sand Island and elsewhere in the estuary were conducted by the USACE and the Oregon Department of Fish and Wildlife (ODFW) and are only briefly summarized here. The USACE estimated that 2,387 breeding pairs of Caspian terns nested on the prepared 1-acre designated colony site on East Sand Island in 2020 (K. Tidwell, USACE-FFU, personal communication), which is below the target colony size for terns on East Sand Island specified in the management plan (i.e. 3,125–4,375 breeding pairs). No estimates of tern nesting success on the designated colony area at East Sand Island are available, but nesting success was apparently very low, and perhaps the colony failed to produce any young. Despite ongoing nest dissuasion efforts to prevent terns from nesting outside the designated 1-acre colony area, large numbers of terns (hundreds to thousands of adults per week) once again attempted to nest along the southeast and east beaches of East Sand Island. None of these tern nesting attempts outside the 1-acre designated colony site were successful in producing fledged young. In 2020, terns did not establish a colony on Rice Island in the upper Columbia River estuary (K. Tidwell, USACE-FFU, personal communication), a location where terns have attempted to nest since the area of suitable tern nesting habitat on East Sand Island was reduced. Cormorants did not nest on East Sand Island in 2020. Based on monitoring conducted by ODFW, the size of the double-crested cormorant colony on the Astoria-Megler Bridge was estimated to be about 5,080 breeding pairs (J. Lawonn, ODFW, personal communication), which is the largest cormorant colony size ever recorded on the bridge and the sixth consecutive year of growth in the size of that cormorant colony since management of the cormorant colony on East Sand Island was first implemented in 2015.

Predation Rates

To investigate the impacts of predation by piscivorous colonial waterbirds on survival of juvenile salmonids and to determine the efficacy of on-going management actions to reduce predation on smolts by terns in the CPR, we estimated ESU/DPS-specific predation rates based on recoveries of smolt PIT tags on bird colonies following the 2020 nesting season. Estimates were generated using previously published, standardized methods, providing a means to compare predation rates across avian predator species and colonies, salmonid species and ESUs/DPSs, and years. To help ensure that enough ESA-listed Upper Columbia River (UCR) steelhead – a population that is highly susceptible to avian predation and therefore a suitable population to evaluate the efficacy of management actions – were tagged and available for predation analyses in 2020, 6,294 steelhead smolts were captured, PIT-tagged, and released into the tailrace of Rock Island Dam (RIS) in the middle Columbia River as part of this study. Tagging at RIS commenced in 2008, resulting in a long-term dataset (2008–2020) with which to evaluate the impacts of terns and other piscivorous colonial waterbird species on the survival of ESA-listed steelhead and to evaluate relative changes in predation rates associated with management actions.

Efficacy of Avian Management Plans: A primary goal of the *IAPMP* was to reduce predation rates by terns on ESA-listed juvenile salmonids to less than 2% per salmonid ESU/DPS, per colony, and to less than 5% per salmonid ESU/DPS for all tern colonies in the CPR combined. Recoveries of smolt PIT tags on tern colonies in 2020 were used to compare predation rates prior to and during tern management actions associated with the *IAPMP*. Results indicated that tern predation rates on juvenile salmonids during 2020 were among the lowest ever recorded in the CPR, with estimates ranging from < 0.1% to 2.2% (95% credible interval = 0.7–5.4%) per colony, per ESU/DPS. For the first time since management actions were implemented in 2014, the goal of reducing predation rates to less 2% per colony, per ESU/DPS was achieved for nearly all ESUs/DPSs evaluated. Predation rates by terns nesting at the managed colony sites on Goose Island and Crescent Island were greatly reduced or eliminated in 2020. Predation rates by terns nesting at the unmanaged tern colony on Lenore Lake (North Rock) were all less than 2.0% per ESU/DPS, with the highest rate observed on UCR steelhead at 1.0% (0.6–1.5%). Predation rates by terns nesting at the unmanaged tern colony in the Blalock Islands were also less than 2% per salmonid ESU/DPS, apart from Snake River steelhead, where the estimated predation rate was slightly above the 2% threshold at 2.2% (0.7–5.4%).

Comparisons of tern predation rates on juvenile salmonids prior to and during implementation of the *IAPMP* indicate that there have been benefits to several salmonid ESUs/DPSs, especially UCR steelhead from management associated with Goose Island and other islands in Potholes Reservoir where average annual predation rates have been reduced from an estimated 15.7% (14.1–18.9%) prior to management (2007–2013) to 0.2% (0.1–0.5%) during the management period (2014–2020). There was also evidence that survival of UCR steelhead smolts has increased significantly in the river reach where most terns from the Goose Island and Crescent Island colonies previously foraged (RIS to McNary Dam) during implementation of the *IAPMP*; increases in steelhead survival rates were commensurate with reductions in tern predation rates. In 2020, tern predation rates were the lowest recorded to date, coincident with record high survival rates for UCR steelhead smolts. For example, during smolt passage from RIS to McNary Dam predation by all tern colonies combined was estimated to be 1.0% (0.6–1.5%) and smolt survival was estimated to be 82.9% (64.8–91.2%), the lowest estimated level of tern predation and the highest estimated level of smolt survival in this river reach since tagging studies at RIS commenced in 2008. Due to continued predation on juvenile salmonids by terns nesting in the Blalock Islands downstream of McNary Dam, however, impacts to some ESA-listed ESUs/DPSs in 2020, particularly Snake River steelhead, remained at or near the 2% threshold. As such, adaptive management actions at the Blalock Islands tern colony, actions that have been proposed for 2021, will likely be necessary to achieve the overall goal of the *IAPMP* in the future.

Estimates of predation rates on juvenile salmonids by terns nesting on East Sand Island in the Columbia River estuary in 2020 were the lowest recorded since terns began nesting on East Sand Island in 1999, with estimates ranging from 0.4% (0.2–0.9%) on Upper Columbia River spring Chinook salmon to 5.9% (4.5–8.1%) on Snake River steelhead (Evans et al. 2021). Results indicated that predation on steelhead smolts by terns nesting on East Sand Island has been reduced by 65% to 76%, depending on the DPS, reductions that meet or exceed those anticipated in the *Estuary Caspian Tern Management Plan* (Evans et al. 2021). Large numbers of terns (several hundred to several thousand) attempted to nest outside of the designated 1-acre nesting area on East Sand Island in 2020, as well as nesting attempts by terns on Rice Island in the upper Columbia River estuary. Thus, continued nest dissuasion efforts and continued monitoring at prospective tern colony sites throughout the estuary will be necessary to ensure predation rates by terns nesting in the estuary do not exceed levels stipulated in the *Plan*.

There was no established colony of cormorants on East Sand Island in 2020 and only a small number of smolt PIT tags (n = 38) were recovered on areas where cormorants briefly attempted to nest during the 2020 nesting season. No estimates of predation were available for the large cormorant colony on the Astoria-Megler Bridge or for the other smaller, cormorant colonies on navigational aids in the upper estuary during 2020. Based on higher per-capita (per bird) estimates of predation rates by cormorants nesting in the upper estuary in previous years, and the relatively large number of smolt PIT tags (n = 1,048) recovered from a small portion of the cormorant colony on the Astoria-Megler Bridge in 2020, cormorant predation rates on juvenile salmonids were likely substantial in 2020 but were not quantified and are thus unknown.

Other Piscivorous Colonial Waterbirds: An investigation of predation rates by piscivorous colonial waterbirds (gulls, pelicans, and cormorants) nesting at other colonies in the Columbia Basin indicated that consumption rates by gulls from colonies in the CPR were as high or higher than those measured at nearby tern colonies in 2020. Unlike terns, cormorants, and pelicans, gulls are known to consume dead or moribund fish and to kleptoparasitize (steal) fish from other piscivorous waterbirds (e.g., Caspian terns) so impacts from gull colonies may be more indicative of consumption rates, rather than predation rates. Consumption rates by gulls nesting on Badger Island in McNary Reservoir and on Miller Rocks in John Day Reservoir were among the highest of any bird colony evaluated in 2020. For example, estimates of consumption rates as high as 9.2% (2.0–18.4%) for Snake River steelhead and as high as 4.4%

(1.8–9.2%) on UCR steelhead were documented for gulls nesting on Badger Island and on Miller Rocks, respectively, in 2020. Estimates of smolt consumption rates by gulls from these and other colonies in 2020, however, were generally lower than those in years past. In 2020, the cormorant colony on Foundation Island in McNary Reservoir was scanned for smolt PIT tags for the first time since 2014. Cormorant predation rates on smolts originating from the UCR were low (< 1.0% per ESU/DPS) but were significantly higher on smolts from Snake River ESUs/DPSs, with predation on Snake River steelhead and Snake River spring/summer Chinook salmon estimated at 4.0% (1.2–10.7%) and 2.5% (1.0–5.4%), respectively. Despite the high rates of consumption/predation on smolts observed at some gull and cormorant colonies in 2020, predation impacts by waterbirds nesting at several other colonies, particularly those located away from the Columbia River, were low to undetectable. For example, estimates of predation rates by cormorants nesting on Lenore Lake and at a small island in Hanford Reach were < 0.2% per salmonid ESU/DPS, as were estimates of consumption rates by gulls nesting at the large colony on Goose Island in Potholes Reservoir at < 0.1% per ESUs/DPSs. These results indicate that not all colonies of piscivorous waterbirds in the CPR posed a potential risk to smolt survival in 2020. Finally, accurate estimates of predation rates, those corrected for both on-colony PIT tag detection probabilities and deposition probabilities, were calculated for the first time for American white pelicans nesting on Badger Island in McNary Reservoir in 2020. Results indicated that smolt predation rates were < 0.5% per salmonid ESU/DPS; the highest estimated predation rate was on Snake River steelhead at 0.4% (0.1–4.2%). Significantly higher predation rates on other, non-ESA-listed juvenile salmonid stocks (e.g., subyearling Chinook salmon from the Upper River Bright stock) by Badger Island pelicans have been documented in other studies, and pelicans also consume some adult salmonids. Nevertheless, results of this study suggest that pelicans nesting at the colony on Badger Island posed only a small risk to actively migrating, ESA-listed UCR and Snake River juvenile salmonids in 2020.

Cumulative Predation and Smolt Survival: To investigate the cumulative effects of avian predation (predation by all avian predator species and colonies combined) and to determine what proportion of all sources of smolt mortality (1-survival) were due to predation by piscivorous colonial waterbirds, we conducted a mark-recapture-recovery analysis on UCR steelhead smolts that were PIT-tagged and released at RIS in 2020. We used previously published methods to jointly estimate predation and survival probabilities during smolt passage through multiple river reaches and we compared results from 2020 to those from previous years (2008–2019). Results from 2020 indicated that avian predation was a substantial source of mortality for UCR steelhead during out-migration from RIS to Bonneville Dam, with bird predation accounting for 56.1% (51.7–60.4%) of all mortality. In most previous years, the cumulative effects of avian predation and the proportion of total mortality that was due to predation were similar to or greater than those observed in 2020, with avian predation accounting for more than 50% of smolt mortality from all sources in 10 of the previous 12 years (2008–2019). Even after passage through the impounded sections of the middle and lower Columbia River upstream of Bonneville Dam, the impact of predation by piscivorous colonial waterbirds on UCR steelhead smolts in the CRE were substantial, with terns and cormorants breeding on East Sand Island annually consuming an average of 11.5% (10.3–12.9%) and 7.1% (5.9–8.6%), respectively, of available steelhead during 2008–2019. In 2020, cumulative predation/consumption rates (predation by all colonies combined) on UCR steelhead during smolt passage from RIS to the Pacific Ocean were estimated at 18.9% (15.0–22.8%), with the highest levels of predation/consumption by gulls at 14.0% (10.3–17.9%), followed by terns at 4.6% (3.5–5.9%), pelicans at 0.3% (0.1–1.0%), and cormorants at 0.1% (< 0.1–0.2%). These estimates exclude several colonies in the CRE (e.g., the large cormorant colony on the Astoria-Megler Bridge), however, and therefore underestimate total smolt losses due to predation by piscivorous colonial waterbirds in the estuary during 2020.

Additive Effects of Predation: To investigate to what degree avian predation on UCR steelhead smolts limited fish survival, we used a mark-recapture-recovery model to jointly estimate weekly and annual predation and survival probabilities among time-stratified cohorts to explicitly measure the strength, magnitude, and direction of the relationship between predation and survival. Data from 2019 and 2020 were used to update a previously published long-term dataset (2008–2018) and to evaluate more recent trends in predation and survival. Results indicated that the record low levels of tern predation on UCR steelhead in 2020 were associated with record high levels of survival during smolt outmigration from RIS to Bonneville Dam. An investigation of weekly and annual estimates of predation and survival probabilities suggested that a greater proportion of UCR steelhead smolts would have survived outmigration to Bonneville Dam in the absence of tern predation upstream of Bonneville Dam, with the estimated average annual difference in observed survival versus baseline survival (i.e. survival in the absence of tern predation) of 0.170 (0.097–0.227) during 2008–2020. Due to low levels of predation by cormorants upstream of Bonneville Dam (0.01 or 1%), differences in observed survival versus baseline survival were less than 0.01 (-0.005–0.008), indicating that only small increases in survival of UCR steelhead smolts to Bonneville Dam would be possible in the absence of cormorant predation upstream of Bonneville Dam. Although there was some evidence of a relationship between consumption probabilities by gulls and survival probabilities of UCR steelhead smolts, results were not statistically significant when considered across all years. The statistical power to accurately determine to what degree gull consumption influenced smolt survival was limited by a truncated time series, coupled with high levels of uncertainty in consumption and survival, and a lack of weekly variation in estimates of consumption probabilities. Since gulls are known to consume dead fish and to kleptoparasitize (steal) fish from other piscivorous waterbirds, it is likely that consumption of smolts by gulls was a more compensatory source of mortality compared to predation on smolts by terns and cormorants.

There was evidence that higher levels of tern predation on UCR steelhead smolts in the CRE were associated with lower returns of adults to Bonneville Dam, with increases in tern predation probabilities associated with statistically significant decreases in adult survival probabilities. These results suggest that in the absence of tern predation on UCR steelhead smolts, SARs for UCR steelhead would have nearly doubled, despite the fact that most smolts depredated by terns would have died from other causes before returning to Bonneville Dam. Results provide evidence that tern predation on UCR steelhead smolts was a partially additive source of mortality to the adult life-stage. There was some evidence that higher levels of cormorant predation on UCR steelhead smolts in the estuary were associated with lower adult returns to Bonneville Dam; however, results were not statistically significant when considered across all years. Furthermore, more recent data were not available for inclusion in this analysis due to the abandonment of the East Sand Island cormorant colony and the relocation of many of the cormorants that formerly nested at the large colony on East Sand Island to the Astoria-Megler Bridge. Impacts on smolt survival from predation by cormorants nesting on the Astoria-Megler Bridge are currently unknown but are likely greater on a per capita (per bird) basis. Collectively, these results suggest that efforts to reduce tern predation on UCR steelhead are increasing steelhead smolt survival in the CRB, particularly in years when predation rates are dramatically reduced as a result of management actions at Goose and Crescent islands, such as in 2020. More importantly from a conservation perspective, results suggest that in the absence of tern predation on UCR steelhead, significantly more adult steelhead would return to Bonneville Dam. Additional research is needed, however, to evaluate to what degree cormorant predation on, and gull consumption of, juvenile salmonids limits smolt survival and SARs in the CRB.

Biotic and Abiotic Factors: Previous research indicates that certain biotic and abiotic factors influence the susceptibility of juvenile salmonids to predation by colonial waterbirds during outmigration, as well as the probability of survival through the hydrosystem. As part of this study, and as recently recommended by the Independent Scientific Advisory Board (ISAB), we are investigating the influence of various covariates on the predation and survival probabilities of ESA-listed steelhead during outmigration. Covariates being investigated include biotic factors like fish size, rearing-type (hatchery, wild), abundance (density), run-timing, and environmental factors like spill, water transit time, and smolt arrival time in the estuary. The goal of this analysis is to describe those factors that best explain variation in survival and susceptibility to avian predation, and to identify potential "management relevant" variables, variables that resource managers may be able to control to some degree. Analyses of covariates are on-going, and results will be provided in our 2021 Annual Report to the Bonneville Power Administration and the Grant County PUD/PRCC.

Smolt Survival to Bonneville Dam

Due to restrictions imposed on travel/field work associated with the COVID-19 outbreak, several survival studies involving PIT-tagged smolts in the CRB were not conducted in 2020. One of those studies involved the operation of the National Marine Fisheries Service's pair trawl net detection system in the lower Columbia River downstream of Bonneville Dam. To mitigate for the loss of pair trawl PIT tag detections in 2020, we scanned for smolt PIT tags at several piscivorous waterbird nesting, roosting, and loafing sites in the CRE with the goal of increasing sample sizes of PIT-tagged fish known to have survived outmigration to Bonneville Dam for use in survival models. This work was conducted in collaboration with the USACE. A total of 6,239 smolt PIT tags were detected at avian nesting, roosting, and loafing sites in the CRE, including 2,530 PIT tags that were recovered at sites that were beyond the original scope of predation studies funded by BPA, Grant PUD/PRCC, and USACE in 2020. Survival analyses indicated that without detections of tags on avian colonies in the estuary, accurate and precise estimates of reach-specific and a cumulative survival estimate for UCR steelhead released at RIS would not have been possible in 2020. For example, without PIT tag detections from bird colonies in the

CRE, 95% credible bounds associated with estimates of UCR steelhead survival to Bonneville Dam were uninformative, ranging from 0.28 to 1.0 (i.e. 28 to 100%). Once detections from bird colonies in the CRE were added, credible bounds were 0.44 to 0.74 (i.e. 44 to 74%). This greater precision provided convincing evidence that a majority of UCR steelhead that were alive at RIS survived outmigration through the hydrosystem in 2020. Results of this and other studies indicate that recoveries of PIT tags on bird colonies in the CRE can be used to augment mark-recapture survival datasets to generate more accurate and precise estimates of smolt survival to Bonneville Dam.

BACKGROUND

Avian predation on out-migrating juvenile salmonids (smolts) has been identified as a factor that can significantly limit the survival of some Endangered Species Act (ESA)-listed anadromous salmonid (*Oncorhynchus* spp.) populations (referred to as Evolutionarily Significant Units [ESUs] or Distinct Population Segments [DPSs]) in the Columbia River basin (CRB). Addressing the impact of avian predation on smolt survival is a component of Biological Opinions and Reasonable and Prudent Alternatives (RPAs) associated with management of the Federal Columbia River Power System (FCRPS). Over the last two decades, numerous research, monitoring, and evaluation (RM&E) studies of avian predation have been conducted to assess the impacts on smolt survival of consumption by Caspian terns (*Hydroprogne caspia*; hereafter referred to as “tern” or “terns”), double-crested cormorants (*Phalacrocorax auritus*; 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 CRB smolts, the U.S. Fish and Wildlife Service (USFWS), the U.S. Army Corps of Engineers (USACE), and their management partners developed and implemented three separate management plans to reduce predation rates on smolts by piscivorous waterbirds nesting at four breeding colonies in the CRB (USFWS 2005, USACE 2014, USACE 2015): the largest tern and cormorant breeding colonies in North America, those on East Sand Island in the Columbia River estuary (CRE); and the two largest tern colonies in the Columbia Plateau region (CPR), those on Crescent Island in McNary Reservoir and on Goose Island in Potholes Reservoir. The primary goal of these management initiatives was to reduce predation rates (percentage of available smolts consumed) on ESA-listed salmonid ESUs/DPSs by reducing the number of birds breeding at each of these four colony sites. Management initiatives implemented at these four colony sites have involved primarily non-lethal strategies for terns (i.e. passive and active nest dissuasion) and primarily lethal strategies for cormorants (i.e. culling and egg-oiling). As part of the management plans for terns, the USACE created alternative nesting habitat for terns at various locations outside the CRB (i.e. northeastern California, southern Oregon, and the San Francisco

Bay area) to compensate for reductions in tern nesting habitat on East Sand Island in the CRE and for elimination of tern nesting habitat on Crescent and Goose islands in the CPR.

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

PROJECT OBJECTIVES

The primary objectives for ongoing work on avian predation on juvenile salmonids in the CRB in 2020 were to (1) assess the distribution and size of piscivorous waterbird colonies and (2) determine the colony-specific and cumulative, system-wide impacts of colonial waterbirds on juvenile salmonid survival in the CRB. As part of these objectives we (a) located and estimated colony size at piscivorous waterbird colonies within foraging range of juvenile salmonids in CRB, including information on any new or incipient colonies; (b) estimated colony-specific predation rates of multiple salmonid ESUs/DPSs of piscivorous waterbirds; (c) measure reach-specific and cumulative predation impacts by multiple avian predator species and survival rates on steelhead, an indicator species in studies of avian predation; (d) estimate the additive effects of avian predation on smolt survival and smolt-to-adult survival; and (e) investigate the biotic and abiotic factors that influence steelhead smolt susceptibility to avian predation. In addition to these primary objectives and tasks, we increased efforts to recover smolt PIT tags at piscivorous colonial waterbird nesting, loafing, and roosting sites in the CRE to help generate more accurate and precise estimates of smolt survival to Bonneville Dam in 2020. Detections associated with this additional task were necessary to mitigate for the loss of the National Marine Fisheries Services pair trawl PIT tag net detection system in the lower Columbia River in 2020, a system that was not operation due to travel restrictions associated with the coronavirus outbreak.

METHODS & ANALYSES

This work is part of a comprehensive program to implement, monitor, and evaluate avian predation management plans (including adaptive management) to reduce the impacts of avian predators on the survival of ESA-listed juvenile salmonids in the CRB. Action effectiveness monitoring was carried out in the CPR at both managed (i.e. Goose Island, Crescent Island, and elsewhere in northern Potholes Reservoir) and at unmanaged sites where terns might disperse to renest as a result of management. The main focus of this work was to measure colony size and predation rates on juvenile salmonids by piscivorous colonial waterbirds in the basin, both to evaluate the efficacy of ongoing management actions and to help identify new and emerging avian predation impacts on smolts that may warrant future management consideration. Finally, as part of this work we recovered smolt PIT-tags at piscivorous colonial waterbird nesting sites, loafing sites, and roosting sites in the CRE to help generate more accurate and precise estimates of smolt survival to Bonneville Dam in 2020 (see *above*). The methods used (see *below* for a brief description) were those used in previous studies so that results from 2020 could be compared to results collected in previous years, both before and during the implementation of avian predation management actions in the CRB.

NEST DISSUASION ACTIVITES

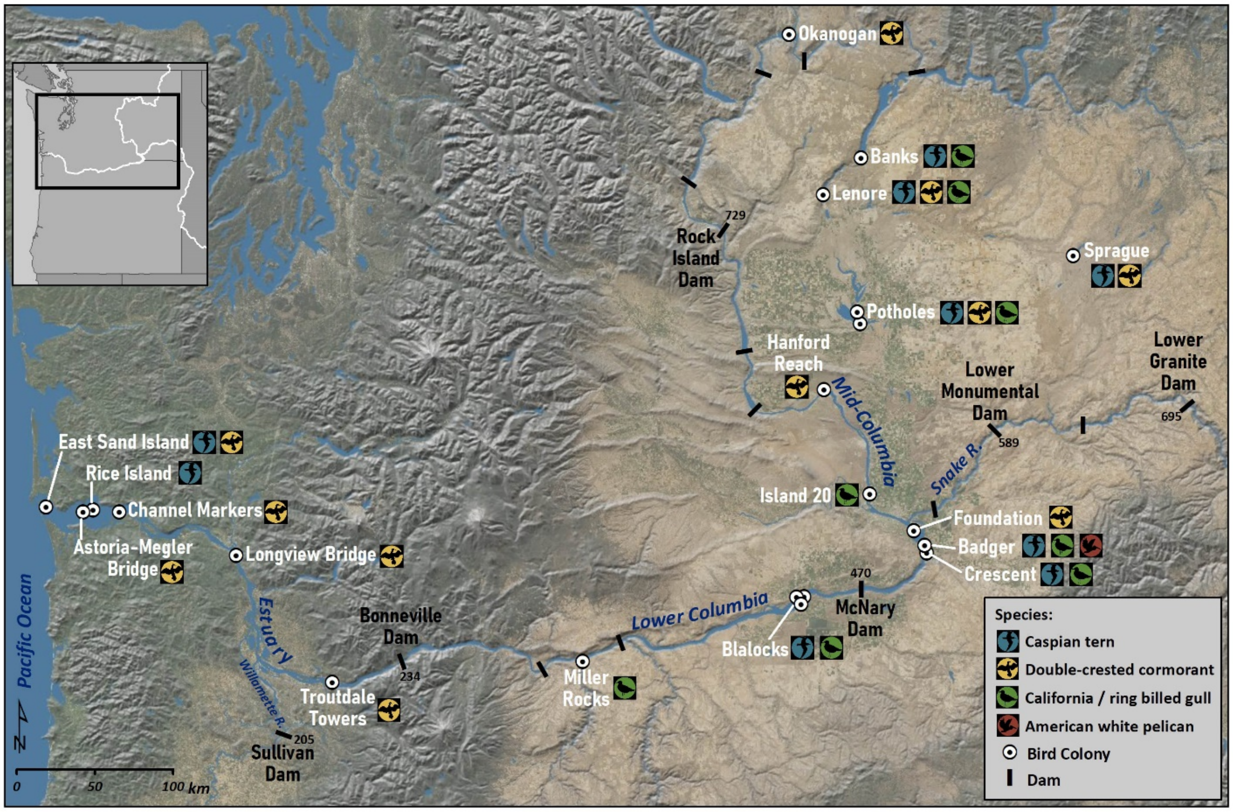
In 2020, nest dissuasion activities on Goose Island and elsewhere in Potholes Reservoir were carried out by the Bureau of Reclamation and its contractor, U.S. Department of Agriculture-Wildlife Services (USDA-WS). These activities were summarized in an annual report entitled, "Tern Deterrence at Goose Island: 2020 Annual Report" (USDA-WS 2021). The USACE Fisheries Field Unit (FFU) carried out nest dissuasion efforts at the East Sand Island tern colony in 2020 and were successful in limiting tern nesting to the prepared 1-acre colony area (USACE-FFU 2021).

NESTING DISTRIBUTION & COLONY SIZE

Action effectiveness monitoring was conducted both at the colony-level and the system-level (region-wide). Colony monitoring was designed to evaluate the efficacy of nest dissuasion efforts and the need for adaptive management at Goose Island, Crescent Island, and elsewhere in Potholes Reservoir in preventing terns from nesting at these sites (see USDA-WS 2021 for more details). Additionally, we monitored terns and other piscivorous colonial waterbirds (i.e. gulls, cormorants, and pelicans) nesting at unmanaged colonies to help identify new and emerging threats to juvenile salmonid survival in the CRB (see *Avian Predation Rates* section below).

First, we carried out a fixed-winged aerial survey in the CRB early in the breeding season (6-8 May) to help identify all active nesting colonies in 2020. Aerial surveys were followed by periodic (weekly to monthly) ground- and boat-based surveys throughout the breeding season (May-July) to identify all active nesting colonies in the region and to accurately assess nesting chronology and the number of birds attending nests. These surveys were used to identify the peak of nesting at each colony so that aerial-photo or ground-based surveys could be conducted to count the peak number of nesting birds or attended nests.

At Goose Island and other suitable nest sites in northern Potholes Reservoir, we periodically (weekly) monitored the activities of terns and other colonial waterbirds (i.e. gulls) throughout the breeding season using at least two field crew members stationed in the CPR. Crescent Island was monitored bimonthly to determine if active hazing and more frequent monitoring might be necessary (terns have not nested on Crescent Island since management commenced in 2015). Other piscivorous waterbird colonies (*Map 1*) were monitored 1-2 times per month throughout the breeding season. Monitoring was conducted from the air (fixed-wing or drone), a boat, and/or on foot, with precautions taken to minimize disturbance to actively nesting non-target species (e.g., gulls). Whenever possible, counts of piscivorous waterbirds 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 and in and around areas of passive nest dissuasion). Data collection methodologies used followed established protocols such that the data collected in 2020 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; Roby et al. 2021, Appendix A).



Map 1. Study area in the Columbia River basin in 2020.

Colony size (i.e. number of breeding pairs) for terns and cormorants nesting in the CRE in 2020 were provided by USACE and ODFW (see *above*). These estimates were used in our system-wide assessment of avian predation on juvenile salmonids in 2020.

AVIAN PREDATION RATES

We analyzed smolt PIT tags collected on bird colonies as part of this study to (1) estimate predation rates on ESA-listed salmonid ESUs/DPSs and to (2) assess relative differences in these predation rates prior to and during tern management actions associated with the *IAPMP*, with a focus on data collected in 2020. Comparisons between current and previous predation rates were made in the context of management initiatives for terns nesting on Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir and relative to the management goal of achieving predation rates of less than 2% per salmonid ESU/DPS, per colony, per year (USACE 2014). In 2020, we also estimated predation rates by terns at unmanaged colonies on the Blalock Islands in John Day Reservoir and on two islands in Lenore Lake. Although terns also nested on Harper Island in Sprague Lake, which is located 67 kilometers north of the lower Snake River, the island was not scanned for PIT tags following the breeding season due to a lack of permission from the landowner to access the site. PIT tag scanning during the pre-management and management periods were also conducted at other incipient tern colonies in years past (e.g., the Badger Island tern colony in McNary Reservoir and the Twinning Island tern colony in Banks Lake), but no terns nested at these sites in 2020.

PIT Tagging of Upper Columbia River Steelhead

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

A detailed description of the sampling methods used to PIT tag steelhead at RIS are provided in Evans et al. (2014). In brief, steelhead were captured at the RIS juvenile fish trap, PIT-tagged (*Biomark* model APT12), length- (mm; fork-length) and 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, origin, and size) and were tagged in concert with, and in proportion to, the run-at-large to ensure that the tagged sample was representative of the steelhead population at large (tagged and untagged fish at RIS). All juvenile steelhead captured at RIS were part of the ESA-listed UCR steelhead DPS, as all hatchery and wild steelhead originating from tributaries upstream of RIS are part of the ESA-listed population (NOAA 2014). The sampling approach used at RIS as part of this study ensures that (1) steelhead from all natural spawning stocks of the ESA-listed DPS are included in the sample (Entiat, Methow, Okanogan, and Wenatchee rivers), (2) that smolts of all sizes, conditions, and rear-types are included in the sample in proportion to their relative abundance, and (3) that smolt are tagged in-concert with the run at-large (tagging and untagged); criteria that allows us to make credible inference about the entire ESA-listed UCR population. The target sample size goal was to PIT-tag approximately 7,000 steelhead smolts for use in predation and survival analyses. 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) and because it was estimated to result in a predation rate precision of approximately ± 2% for those colonies that forage on smolts in the middle Columbia River immediately downstream of RIS, particularly managed tern colonies like those on Goose and Crescent islands.

For most other ESA-listed salmonid ESUs/DPSs, adequate numbers of PIT-tagged smolt were available for inclusion in predation rate analyses based regional tagging studies that occur upstream Lower Monumental Dam on the lower Snake River or upstream of McNary and

Bonneville dams on the lower Columbia River (see Predation Rate Estimates section *below* for details).

Predation Rate Estimates

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

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

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

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

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

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

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

Informative Beta priors were used to model deposition probability(ϕ). The shape parameters (α, β) are dependent on the predator species (cormorant, tern, gull, pelican) and are assumed to be mutually independent from colony to colony. For terns, we assumed α = 16.20 and β =

6.55, for cormorants, we assumed α = 15.98 and β = 15.29, for gulls we assumed α = 33.71 and β = 183.61, and for pelicans, we assumed α = 6.70 and β = 7.37.

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

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

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

Models were analyzed using the software STAN (SDT 2015), accessed through R version 3.6.3 (RDCT 2014), and using the rstan package (version 2.23.0; Stan Development Team 2020). Reported predation rate estimates represent simulated posterior medians along with 95% highest (posterior) density intervals (95% Credible Interval [CRI]) calculated using the HDInterval package (version 0.2.0; Meredith and Kruschke 2016). Annual predation rates were calculated for salmonid ESUs/DPSs where ≥ 500 PIT-tagged individuals were available to birds at each colony to avoid imprecise results that may occur from small sample sizes of available PIT-tagged smolts (Evans et al. 2012).

A more detailed description of the methods used to estimate colony- and ESU/DPS-specific predation rates, including methods to determine smolt availability, to recovery tags from bird colonies, and model assumptions are provided Monitoring Methods and in the Avian Predation Synthesis Report (Roby et al. 2021, Appendix A).

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

Inland Avian Predation Management Plan – A stated goal of the *IAPMP* is to reduce the impact of predation by terns on ESA-listed salmonid ESUs/DPSs to less than 2.0% per salmonid ESU/DPS, per colony and to less than 5.0% per salmonid ESU/DPS, for all tern colonies combined (i.e. cumulative predation; USACE 2014). To help evaluate the efficacy of the *IAPMP* to reduce predation impacts to these levels, predation rates were compared and contrasted between the pre-management period (2007–2013) and the management period (2014–2020) at both managed and unmanaged tern colonies in the CPR.

East Sand Island Caspian Tern and Double-crested Cormorant Management Plans – It is anticipated that reductions in the size of tern and cormorant colonies on East Sand Island in the

CRE will be commensurate with reductions in predation rates (USFWS 2005, USACE 2015). The methods to estimate colony and ESU/DPS-specific predation rates by terns and cormorant on East Sand Island are the same at those for colonies in CPR (see also Roby et al. 2021, Appendix A).

The USACE and its contractors were responsible for recovering smolt PIT tags and estimating predation rates by terns and cormorant nesting on East Sand Island in the CRE in 2020. As part of this study and as an independent analysis, we evaluated the cumulative, system-wide effects of predation on UCR steelhead tagged as part of this project (see *above*), which includes recoveries of smolt PIT tags from tern and cormorant colonies on East Sand Island. Estimates of cumulative predation by multiple predator species (terns, cormorants, pelicans, and gulls) and colonies, rely on a different analytical framework (see *below*) than those of colony- and ESU/DPS-specific estimates of predation (see *above*), so the two estimates, although often similar, are not directly comparably to one another.

Cumulative Predation and Survival: We evaluated the cumulative effects of avian predation (predation from all piscivorous colonial waterbird species and colonies combined) on ESA-listed UCR steelhead that were tagged at RIS in 2020. Salmonid smolts are also subject to numerous non-avian sources of mortality (e.g., hydroelectric dam passage, predation by piscivorous fish, disease, and other factors) and determining to what degree avian predation limits survival relative to these other sources of mortality may be critical for prioritizing recovery actions for ESA-listed salmonids (Evans et al. 2016). To investigate the cumulative effects of colonial waterbird predation and to estimate what proportion of all sources of smolt mortality (1-survival) were due to avian predation, we used the methods of Payton et al. (2019). Upper Columbia River steelhead smolts must migrate through the foraging areas of multiple avian predator species (terns, cormorants, gulls, and pelicans) from multiple breeding colonies. River reaches were defined by dams where PIT-tagged smolts were detected alive following tagging and release at RIS. Predation rates were based on the proportion of available smolts consumed by birds within each river reach, and survival rates were based on the proportion that survive out-migration through that reach. An estimate of the proportion of smolts that die from causes other than avian predation was also generated as part of these analyses, providing spatially explicit information on non-avian sources of smolt mortality.

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

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

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

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

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

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

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

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

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

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

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

We simplify the state-space model into succinct likelihood equations and employ Hamiltonian Monte Carlo (HMC) simulations to construct approximations of the joint posterior distributions of the parameters in question. The simulated posterior distributions are the basis from which we infer survival and recovery estimates (posterior medians), along with 95% credible intervals (2.5 and 97.5 posterior percentiles; 95% CRI). The HMC simulations are performed using the software STAN accessed through R version 3.1.2 (RDCT 2014), using the rstan package (version 2.8.0; SDT 2015). Four parallel HMC simulations will be run for 2,000 adaptation iterations, followed by 2,000 posterior iterations. Chain convergence will be visually evaluated and verified using the Gelman-Rubin statistic. Any evidence of a lack of convergence will be corrected by repeatedly doubling the prescribed number of adaptation and simulation intervals until convergence can be presumed. A more detailed description of these models and their use can be found in Evans et al. (2019) and Payton et al. (2019).

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

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

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

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

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

and

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

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

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

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

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

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

Independent of the variation described above, any further decrease/increase in baseline survival is assumed to be relative to the level of bird genus-specific predation associated with that reach. This relationship is generally assumed to be proportional, unless $\theta_w^{\{GENUS\}}$ encompasses all unaccounted-for mortality, at which point the relationship becomes directly inverse. The annual magnitude of this additive-mortality relationship will be measured with the parameter a. The measure of the resulting or "observed" weekly survival, s_w^*, can be expressed as

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

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

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

Results were used to estimate smolt survival and smolt-to-adult survival based on the degree to which bird predation was determined to be an additive or a compensatory source of smolt mortality for each salmonid ESU/DPS where there are adequate sample sizes of tagged, released, and consumed smolts, and where variation in predation rate is large enough to identify a statistically significant relationship. Addition details regarding the modelling framework used to investigate the relationship between predation and survival probabilities are provided in Payton et al. (2020, 2021).

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

To investigate factors that potential influence smolt susceptibility to avian predation we used a modified version of the avian predation rate model (see *above*). The modified model included all possible combinations of the biotic and abiotic factors identified as potential covariates. We define a $\boldsymbol{X}$ M by F-sized matrix of covariates with effects measured by F-sized parameter vector β.:

$$logit(\theta) = \mathbf{X}^{\mathrm{T}}\beta\,.$$

Covariates were limited to those factors with previous support based on the published literature and based on factors under a modicum of control by managers. Covariates included, colony size (number of breeding pairs), prey availability (based on weekly steelhead index as measured at RIS), individual fish characteristics (rear-type and size), river flows (as measured by estimates of water transit time), and operational strategies (spill). Model selection was performed using LOO-IC to reduce the litany of proposed parameters to a set of significant factors (Vehtari et al. 2017).

SMOLT SURVIVAL TO BONNEVILLE DAM

Due to restrictions imposed on travel/field work associated with the coronavirus outbreak, several on-going mark-recapture survival studies involving PIT-tagged smolts in the Columbia River did not occur as originally planned in 2020. One of those studies involved the operation of the National Marine Fisheries Services pair trawl net detection system in the lower Columbia River estuary downstream of Bonneville Dam (near Rkm 85). The pair trawl has become an important source of data regarding the number of PIT-tagged smolts that survive out-migration to Bonneville Dam (Rkm 234) each migration year. Two recently published studies, those of Hostetter et al. (2018) and Payton et al. (2019), indicate that detections (recoveries) of smolt PIT tags on bird colonies in the estuary can also be used to determine how many smolts survived outmigration to Bonneville Dam. As such, in lieu of detections of tagged smolts at the pair trawl in 2020, recoveries of smolt PIT tags on bird colonies in the estuary may providing an additional or alternative source of data for use in mark-recapture survival models. To address this possibility, we recovered smolt PIT tags from several piscivorous colonial waterbirds nesting, loafing, and roosting sites in the estuary, sites that were not included in the original scope of work for this study, but sites where we suspected PIT tags were being deposited by birds in the estuary. Scanning sites were based on fixed-wing aerial imagery of avian nesting, loafing, and roosting sites that were taken by this project and USACE Civil Air Patrol flights during the peak smolt out-migration period of April to June 2020. PIT tag recovery sites included (1) the flat, concrete crib section of the Astoria-Megler Bridge where large numbers of cormorants nested, (2) the western portion of East Sand Island where cormorants attempted but failed to nest, (3) Rice Island (Rkm 34) where both terns and pelicans attempted but failed to nest, (4) Miller Sands Island (Rkm 38) where cormorants were observed loafing and roosting and where pelicans attempted but failed to nest, and (5) two channel markers located near Miller Sands Spilt where cormorants nested.

Previous research has suggested that the information provided by PIT tags recovered from known mortality sources can be used to decrease bias in estimates of survival and increase the precision and coverage probabilities for the respective uncertainty intervals (Hostetter et al. 2018, Payton et al. 2019). To assess the additional information provided by tag recoveries from bird colonies is 2020, we developed reach-specific and cumulative estimates of survival using the JMS methods of Payton et al. (2019) both with and without the avian PIT tag recoveries using a Bayesian implementation of the Cormack-Jolly-Seber (see Hostetter et al. 2018).

RESULTS & DISCUSSION

NEST DISSUASION ACTIVITIES

In 2020, USDA-WS (contractor) installed flagging and repaired the existing passive nest dissuasion array (i.e. stakes and rope) on Goose Island. The contractor conducted 72 site visits to Goose Island from April to August to install additional dissuasion materials and haze terns that were attempting to nest on the island. Periodic visits to the northern portion of Potholes Reservoir were also conducted by the contractor to identify terns that might be prospecting to nest. During these surveys, terns were observed on 16 different islands and were dissuaded from nesting using active hazing alone (i.e. no passive dissuasion was used at these sites in 2020). For details on the nest dissuasion activities conducted by the contractor in 2020 see USDA-WS (2021).

The growth of willows planted on Crescent Island in 2016 has eliminated all upland habitat suitable for tern nesting on Crescent Island. As a consequence, the other passive nest dissuasion materials (i.e. fencing, stakes, and rope) that were installed on Crescent Island were removed prior to the 2020 breeding season. Up until 2020, passive nest dissuasion alone was sufficient to deter both terns and gulls from nesting on the island. In 2020, gulls reestablished a breeding colony in semi-vegetated habitat on Crescent Island (see *below* for details). Gulls were observed to be nesting in open areas where willows had died back and/or where beaver herbivory on the willows had created patches of ground with minimal ground cover. Despite the presence of an active gull colony on Crescent Island, terns did not attempt to nest on the island in 2020 (see *below*). Active hazing of terns or gulls was not conducted at Crescent Island in 2020.

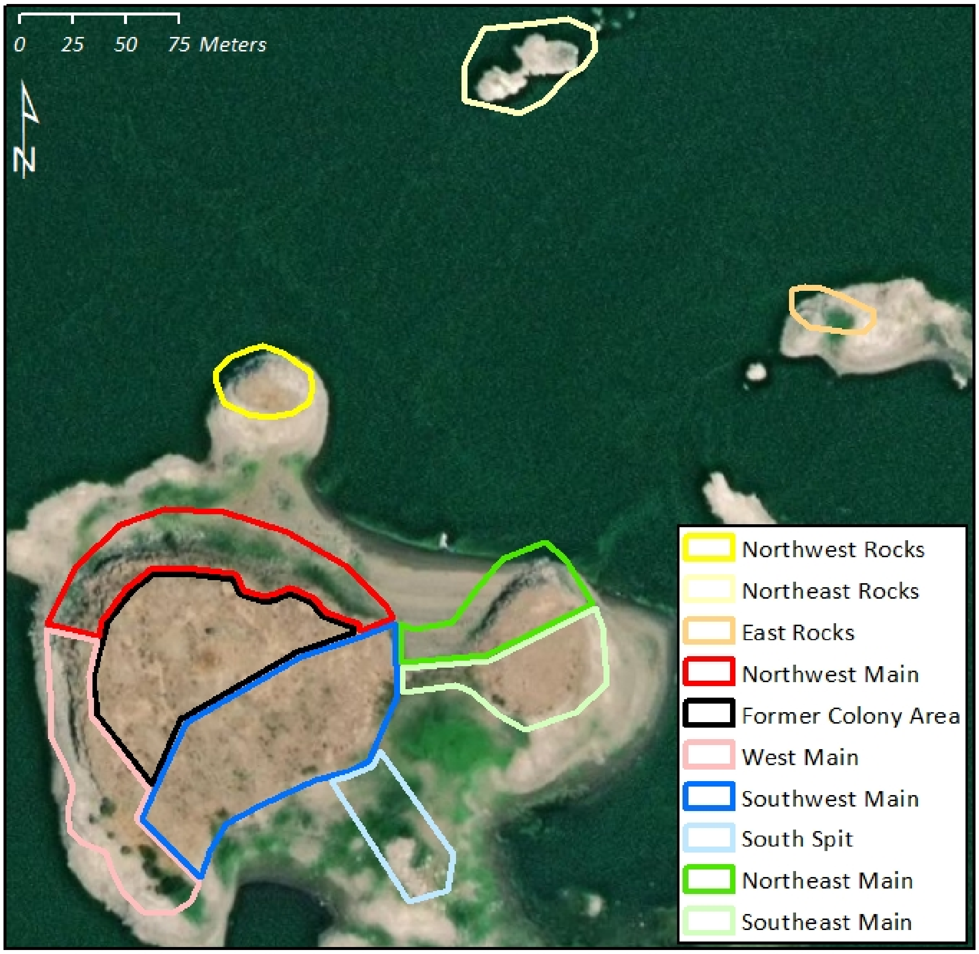
From 7 May to 6 August, the USACE-FFU conducted nest dissuasion efforts to prevent terns from nesting outside the prepared 1-acre colony area on East Sand Island in 2020 (USACE-FFU 2021). These efforts were mostly successful, however, large numbers of terns once again attempted (unsuccessfully) to nest in satellite colonies located at the wrack line located south and east of the designated 1-acre colony area. Although tern eggs were laid in 2020, no tern chicks were hatched at the satellite colonies and no fledglings were produced at either the satellite colonies or the designated 1-acre colony area (USACE-FFU 2021). No active nest dissuasion activities were performed at the former cormorant colony on East Sand Island in 2020 (K. Tidwell, USACE-FFU, personal communication).

NESTING DISTRIBUTION & COLONY SIZE

Columbia Plateau Region

Nest dissuasion efforts in 2020 were once again successful in preventing colony formation by terns on Crescent Island and at islands in northern Potholes Reservoir. Despite continued efforts to dissuade terns from nesting on Goose Island, a small colony became established near the historic colony site on top of the island in 2020 (see *below*). In total, there were three active tern colonies, 9 active gull colonies, 5 active cormorant colonies, and one active pelican colony in the CPR in 2020 (see *below* for more details).

Caspian Terns on Goose Island: As was the case in previous years, tern use of Goose Island for roosting and nesting attempts was largely limited to areas near the island's shoreline, which gradually was exposed during the nesting season as reservoir levels receded (*Map 2* and *Table 1*). As was the case in previous years, weekly attendance by terns on the island was greatly reduced from what was seen prior to management (*Figure 1* and *Table 1*). Unlike in recent years (2014-2019) when relatively few terns were present on Goose Island during the peak in the steelhead outmigration (early May), the peak number of terns observed on Goose Island in 2020 was on 4 May (180 adults; USDA-WS 2021). Active nest dissuasion (hazing) and collection of tern eggs (115 tern eggs collected under permit in 2020; USDA-WS 2021) were effective in preventing all but 6 breeding pairs of terns from successfully nesting at the colony in 2020. This small colony was established on the top of Goose Island near the historic colony area used by terns prior to management. A minimum of four fledgling terns were produced at the colony in 2020, the first year of colony establishment and successful nesting by terns on Goose Island since 2015.

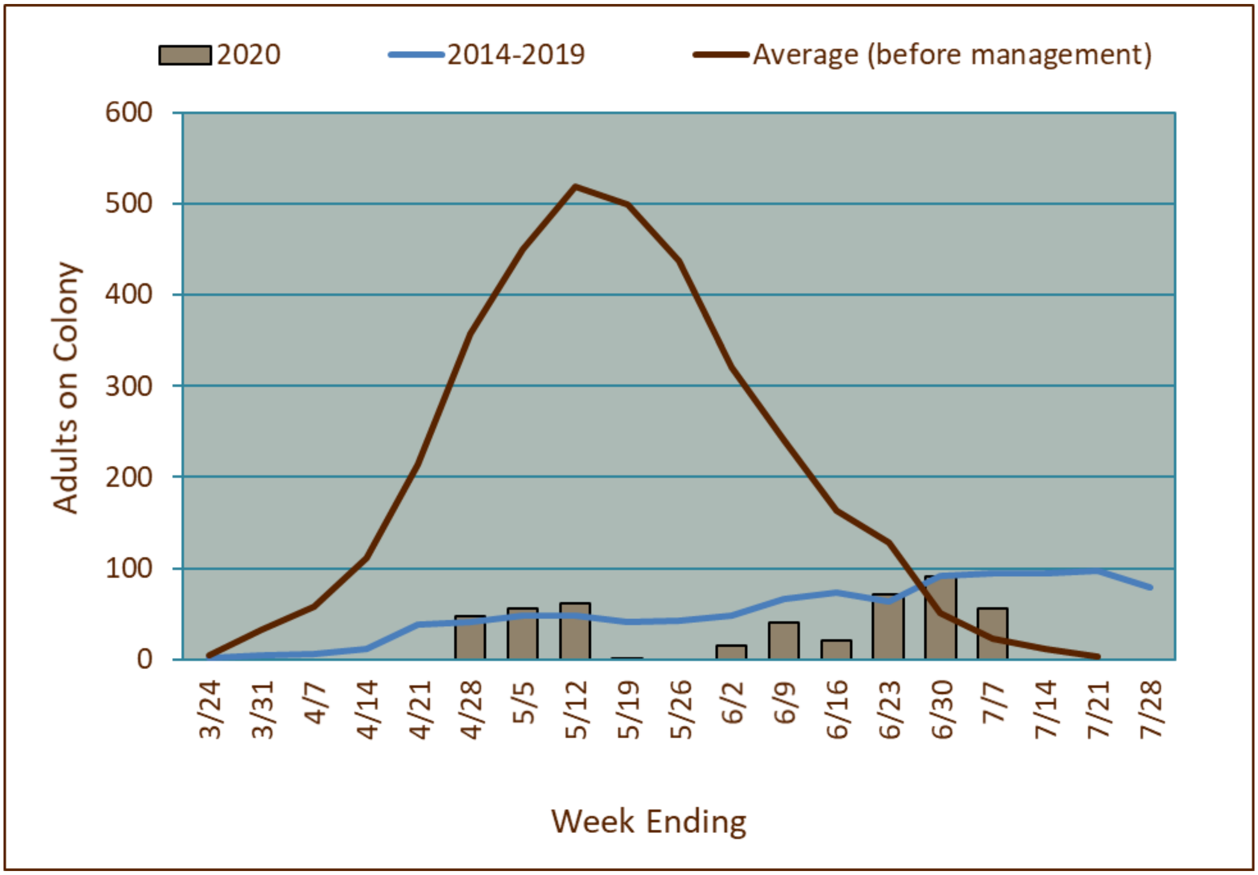


Map 2. Survey locations on Goose Island and nearby rocky islets, Potholes Reservoir in 2020.

Table 1. Counts of adult Caspian terns by location on Goose Island in 2020. Map 2 identifies the locations where daily counts of Caspian terns were conducted.

	Location										
Week Ending	Northwest Rocks	Northeast Rocks	East Rocks	Northwest Main	Former Colony	West Main	Southwest Main	South Spit	Northeast Main	Southeast Main	**Total**
4/26	47	0	0	0	0	0	0	0	0	0	**47**
5/4	43	2	0	0	0	0	0	0	11	0	**56**
5/16	12	0	0	0	0	0	0	0	0	50	**62**
5/22	0	0	0	0	0	0	0	0	1	0	**1**
6/8	0	0	0	0	0	0	0	0	0	0	**0**
6/17	0	0	0	0	0	0	0	0	0	15	**15**
6/26	0	0	0	0	0	0	0	0	0	40	**40**
7/8	3	0	0	0	0	0	0	0	0	21	**21**
7/17	54	0	0	0	0	0	0	0	17	0	**71**
7/24	47	4	0	0	0	0	0	0	40	0	**91**
7/30	0	20	0	26	0	0	0	0	0	0	**56**

Figure 1. Estimates from the ground of the average number of adult Caspian terns on Goose Island and the surrounding islets in Potholes Reservoir, by week, before (2010-2013) and during (2014-2019, 2020) tern management at Goose Island. No count data for the first 5 weeks and the last 3 weeks in 2020. Counts of terns conducted by USDA-WS as part of their nest dissuasion work on Goose Island indicated a peak count of 180 terns on 4 May in 2020, more than 2 months earlier than had been seen during the previous management years (2014-2019; see Results & Discussion section).



As was the case in previous years, gulls were present and nesting on the Goose Island in 2020. The index of gull colony size on Goose Island in 2020 was ca. 11,100 individuals, near the range (ca. 11,500–13,000) of gulls counted on the Goose Island during the four years prior to management (*Table 2*; Adkins et al. 2014; BRNW 2014). These index counts indicate that the colony size for gulls on Goose Island has not changed appreciably because of tern management activities on the island and support the conclusion that the combined effects of passive and active nest dissuasion efforts during the 2014-2020 nesting seasons had little impact on the establishment and size of the Goose Island gull colony.

Table 2. Sizes of mixed California/ring-billed gull breeding colonies (peak numbers of individuals counted) at managed sites in the Columbia Plateau region prior to (2008-2013) and during (2014-2020) management.

	Year												
Colony	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Goose Is. (Potholes Res.)	NA	13,021	NA	11,392	12,005	12,790	14,334	14,808	13,273	11,225	11,994	11,090	11,240
Crescent Is. (Columbia River)	8,567	8,575	8,108	7,108	7,187	5,707	6,404	0	0	0	0	0	400

Nesting success by terns on Goose Island in 2020, although limited, was likely due to a decline in the frequency (i.e. number of days/week) and duration (i.e. number of hours/day), as well as changes in timing, of active nest dissuasion activities (i.e. hazing) in 2020 relative to previous years. The continued fidelity of terns to Potholes Reservoir despite ongoing nest dissuasion activities is somewhat surprising and is likely due to terns having a long nesting history at the site and the persistence of a large gull colony on the island, both before and during management, which continues to attract prospecting terns to the site. Another factor that might explain the strong fidelity of terns to the Potholes Reservoir area is the paucity of alternative tern colony sites in the vicinity. Prior to management (2005-2013), an average of 367 breeding pairs of terns nested on Goose Island (*Figure 2*).

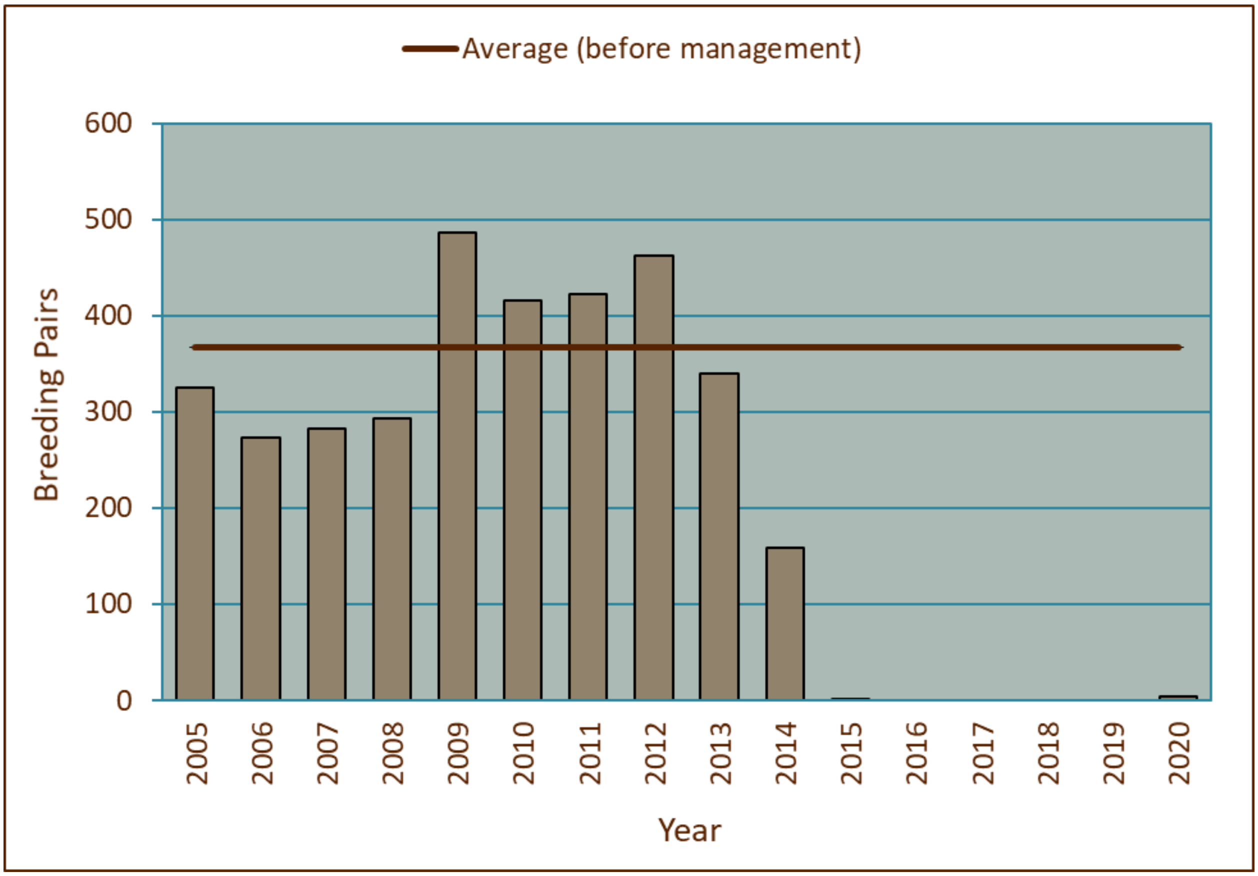


Figure 2. Size of the Caspian tern breeding colony (number of breeding pairs) on Goose Island and the surrounding islets in Potholes Reservoir before (2005-2013) and during (2014-2020) tern management in the region. Caspian terns did not nest on Goose Island and the surrounding islets in 2016-2019, while 6 breeding pairs nested on Goose Island in 2020.

Caspian Terns in Northern Potholes Reservoir: In 2020, loafing terns were documented on 16 different islands in northern Potholes Reservoir (USDA-WS 2021). Targeted hazing alone was successful in preventing the formation of an incipient tern colony on islands in northern Potholes Reservoir. Egg laying by terns at these sites was not documented in 2020. While these results are encouraging, some terns continue to show strong fidelity to Goose Island and other locations in Potholes Reservoir.

Caspian terns on Crescent Island: In 2020, the passive nest dissuasion that remained on Crescent Island (i.e. willows and other native vegetation) was not sufficient to prevent gulls from nesting on Crescent Island. Gulls, both California and ring-billed, were successful in established a breeding colony of approximately 400 pairs at a site on and near the old tern colony site. Although terns were observed flying over Crescent Island during the 2020 breeding season, they did not attempt to nest at the site during that year. The reason(s) for the return of a gull breeding colony on Crescent Island in 2020 are unknown but is likely due to removal of fencing and other passive nest dissuasion materials and mortality of willows and other native vegetation on the island. Beaver herbivory has been observed on the island in previous years and was one of perhaps several factors affecting vegetation cover on the island. As the gull colony becomes established (i.e. year after year) on the island, we expect it could lead to increases in the amount of open unvegetated habitat on Crescent Island, which may ultimately lead to the reestablishment of a tern colony there.

This was the sixth consecutive year when no nesting by terns occurred on Crescent Island, while prior to tern management in the CPR the average colony size for terns on Crescent Island was 461 breeding pairs (*Figure 3* and *Table 3*).

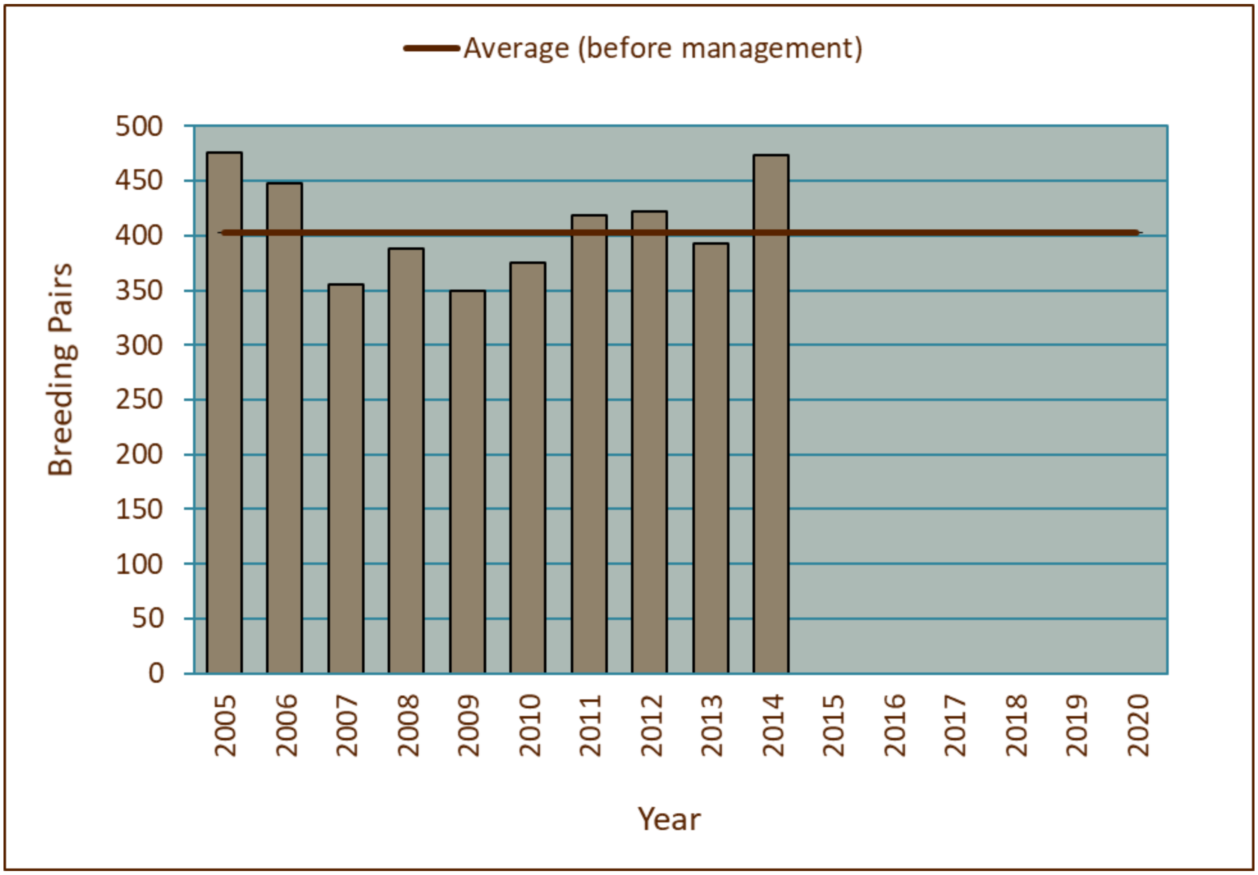


Figure 3. Size of the Caspian tern breeding colony (number of breeding pairs) on Crescent Island in the mid-Columbia River before (2005-2013) and during (2014-2020) tern management in the region. Caspian terns did not nest on Crescent Island in 2015-2020.

Table 3. Sizes of Caspian tern breeding colonies (number of breeding pairs) at both managed and unmanaged colonies in the Columbia Plateau region prior to (2005-2013) and during (2014-2020) management.

	Year															
Colony	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020
Goose Is. (Potholes Res)	325	273	282	293	487	416	422	463	340	159	2	0	0	0	0	6
Crescent Is. (Columbia River)	476	448	355	388	349	375	419	422	393	474	0	0	0	0	0	0
Blalock Is. (Columbia River)	6	110	43	104	79	136	20	6	26	45	677	483	449	313	379	150
Badger Is. (Columbia River)	0	0	0	0	0	0	33	60	0	0	0	0	41	8	0	0
Twinning Is. (Banks Lake)	13	23	31	27	61	34	19	22	13	67	64	6	0	0	0	0
Harper Is. (Sprague Lake)	7	7	0	11	4	4	4	30	1	8	10	3	92	79	18	0
North Rocks and Shoal Is. (Lenore Lake)	0	0	0	0	0	0	0	0	0	0	0	0	123	91	48	53
Total	**827**	**861**	**711**	**823**	**980**	**965**	**917**	**1003**	**773**	**755**	**769**	**675**	**705**	**491**	**445**	**209**

Other Caspian Tern Colonies: Nesting by terns was confirmed at three different colony sites in the CPR in 2020; on Goose Island in Potholes Reservoir (see *above*), on the Blalock Islands in John Day Reservoir, and on islands in Lenore Lake (see *Table 3* above). The historic tern colony sites on Crescent Island in McNary Reservoir (see *above*), Badger Island in McNary Reservoir, Twinning Island in Banks Lake, and Harper Island in Sprague Lake were not active in 2020 (see *Table 3* above). As was the case in 2015-2019, the largest tern colony in the CPR in 2020 was on the Blalock Islands, representing 72% of the total number of breeding pairs in the region in 2020 (see *Table 3* above).

The Blalock Islands are located on the Columbia River above John Day Dam near the town of Irrigon, OR, and are managed by the USFWS as part of Umatilla National Wildlife Refuge. The island group consists of several sizable, permanently vegetated islands, as well as numerous low-lying gravel islands and mudflats that were created by the John Day Dam impoundment. In 2020, terns were seen on the Blalock Islands during the crew's first visit to the site on 24 April, when 351 adult terns were observed nesting and loafing on multiple islands. As many as 132 and 18 attended tern nests were counted on Middle Island and Long Island, respectively. As was the case in previous years, periods of high wind and elevated reservoir elevations resulted in significant colony failure (i.e. egg loss) at the Blalock Island Complex in 2020, with complete nest failure (i.e. no young fledged from the colony) occurring in 2020. The colony size estimate of 150 breeding pairs in 2020 represents a decrease in colony size at the Blalock Island complex relative to 2019 (379 breeding pairs) and the average during the management period (2014-2020; 356 breeding pairs), but an increase compared to the pre-management average (2005-2013; 59 breeding pairs; *Figure 4* and see *Table 3* above. Inundation of tern nests on the low-lying islands at the Blalock Island Complex (see *above*) will likely continue to be a limiting factor on colony size and nesting success for terns at this site.

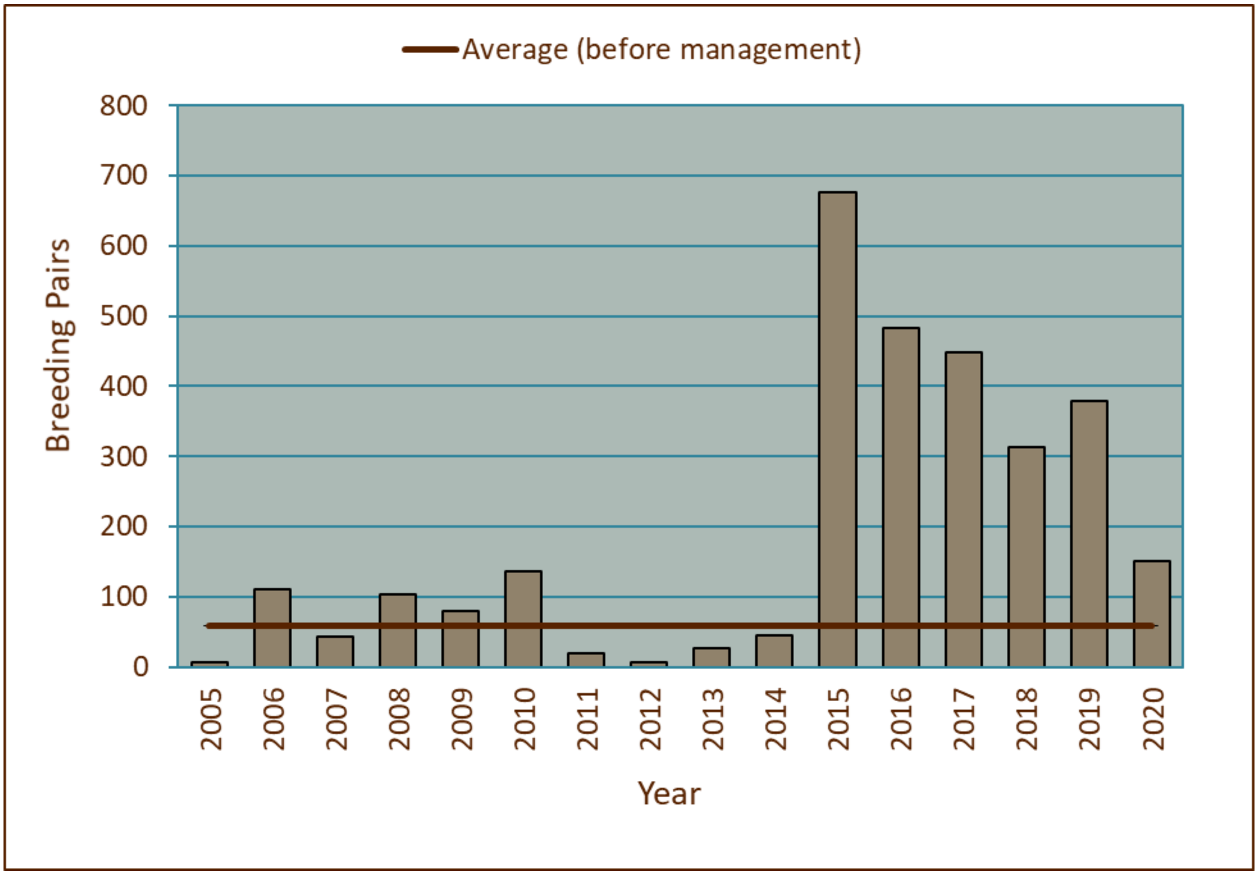


Figure 4. Size of the Caspian tern breeding colony (number of breeding pairs) at the Blalock Islands in the mid-Columbia River during 2005-2020. Also, provided is the average number of breeding pairs of Caspian terns on the Blalock Islands prior to tern management in the Columbia Plateau region (2005-2013).

Lenore Lake was formed by the Missoula Floods in the lower Coulee just north of Soap Lake, Washington and is managed by the Washington Department of Fish and Wildlife. Lenore Lake is known for its very alkaline waters that only Lahontan cutthroat trout can survive in. Terns first nested in Lenore Lake on Shoal Island (2014-2016) and then moved to North Rock, approximately 0.4 km NNE from the former colony site, in 2017. North Rock is a steep-sided, rocky island approximately one acre in area and is located about 48 km from the nearest section of the Columbia River. In 2020, terns were seen on islands in Lenore Lake during the crew's first visit to the site on 11 May, when 36 and 28 adult terns were counted on North Rock and Shoal Island, respectively. In total, we estimated that 53 breeding pairs of terns attempted to nest on these two islands in Lenore Lake in 2020, slightly higher than the colony size estimate in 2019 (48 breeding pairs) and identical to the average during the management period (2014-2020; 53 breeding pairs; *Figure 5* and see *Table 3* above). Terns did not nest at islands in Lenore Lake prior to the initiation of management in 2014. As has been the case in recent years, terns were successful in fledging some young (number unknown due to obstructed views of the colony) from sites in Lenore Lake in 2020.

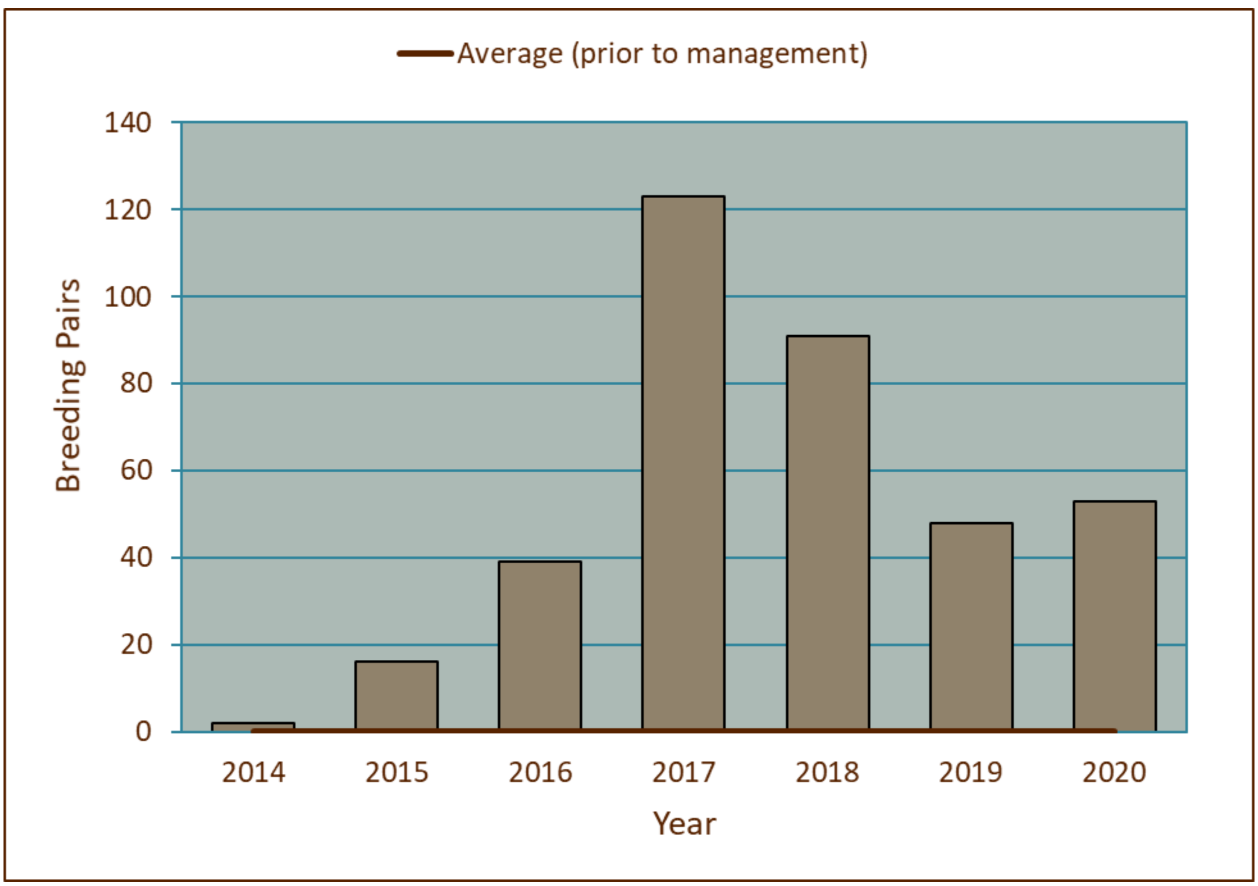


Figure 5. Size of the Caspian tern breeding colony (number of breeding pairs) at small islands in Lenore Lake during 2014-2020 (Shoal Island) and 2017-2020 (North Rock). Caspian terns did not nest on islands in Lenore Lake prior to tern management in the Columbia Plateau region (before 2014).

In total, an estimated 209 breeding pairs of terns nested at three different breeding colonies in the CPR during 2020. This represents a 76% decline in the regional breeding population size for terns compared pre-management average (873 breeding pairs; *Figure 6* and see *Table 3* above) and was the lowest total counted in the region since avian predation monitoring began in the late 1990's. The cause of the large decline in the regional breeding population in 2020 is unknown, however, it is possible that management of terns in the CPR in each of the last seven years has resulted in some terns relocating to nest outside the region which was one of the goals of the *IAPMP*.

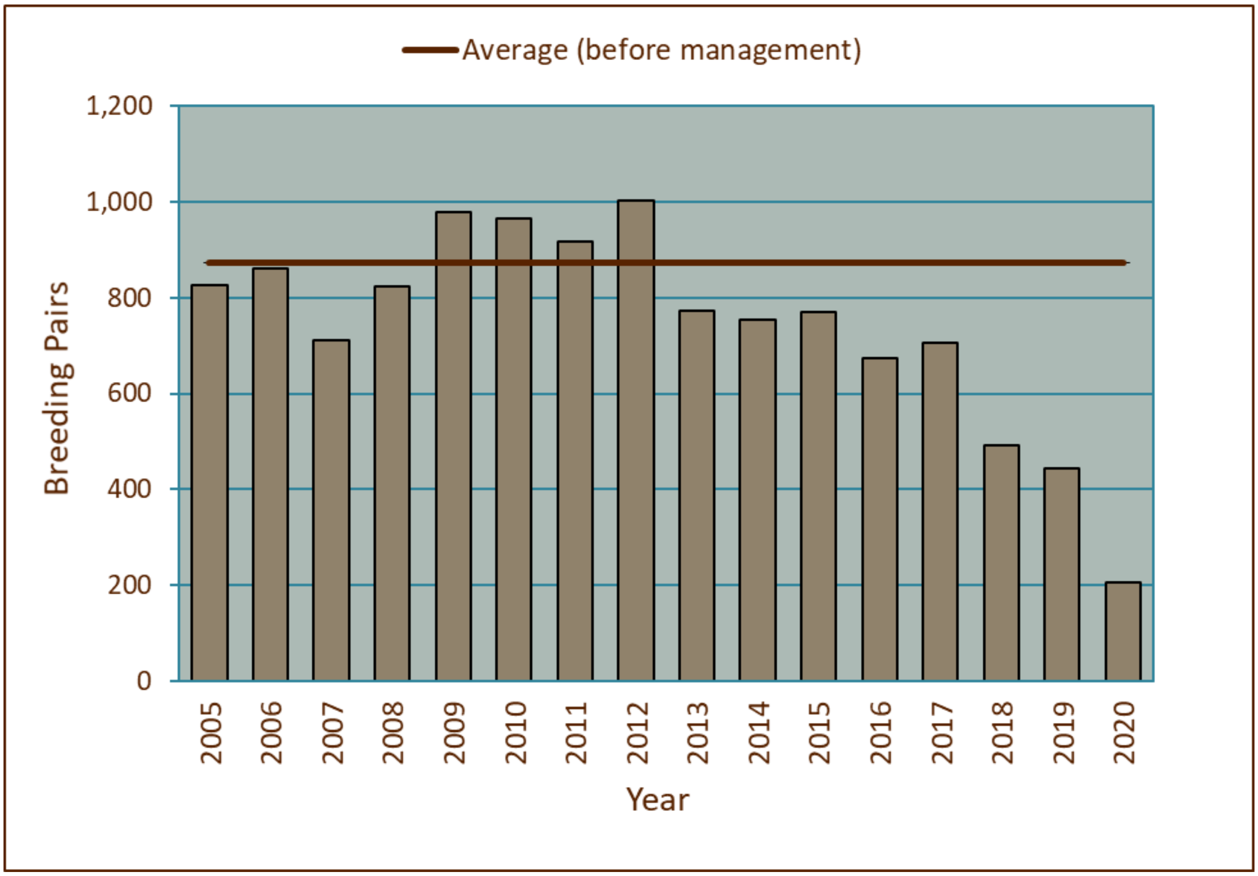


Figure 6. Total numbers of Caspian tern breeding pairs at all known colonies in the Columbia Plateau region during 2005-2020. Also, provided is the average number of breeding pairs of Caspian terns prior to tern management in the Columbia Plateau region (2005-2013).

Other Piscivorous Waterbird Colonies: As part of this work, we also estimated the colony size of other piscivorous waterbirds (i.e. gulls, cormorants, and pelicans) in the CPR in 2020. Because visits to these colonies were less frequent (1-2 times per month) as compared to our monitoring of tern colonies (2-4 times a month) in the region, our estimates of colony size at these other piscivorous waterbird colonies should be viewed with some caution (i.e. actual colony size could be higher or lower). In total, there were nine active gull colonies in the CPR region in 2020, ranging in size from 400 (Crescent Island) to over 11,000 (Goose Island) breeding individuals (*Figure 7* and *Table 4*). Cormorants were confirmed to be breeding at five colonies in the CPR in 2020, with colony size ranging from 1 (Miller Rocks) to 333 (Harper Island in Sprague Like) breeding pairs (*Figure 7* and *Table 4*). Finally, the size of the Badger Island pelican colony was estimated to be 3,165 breeding individuals in 2020 (*Figure 7* and *Table 4*). These data were used to identify which colonies to scan for smolt PIT tags following the 2020 breeding season (see *below*).

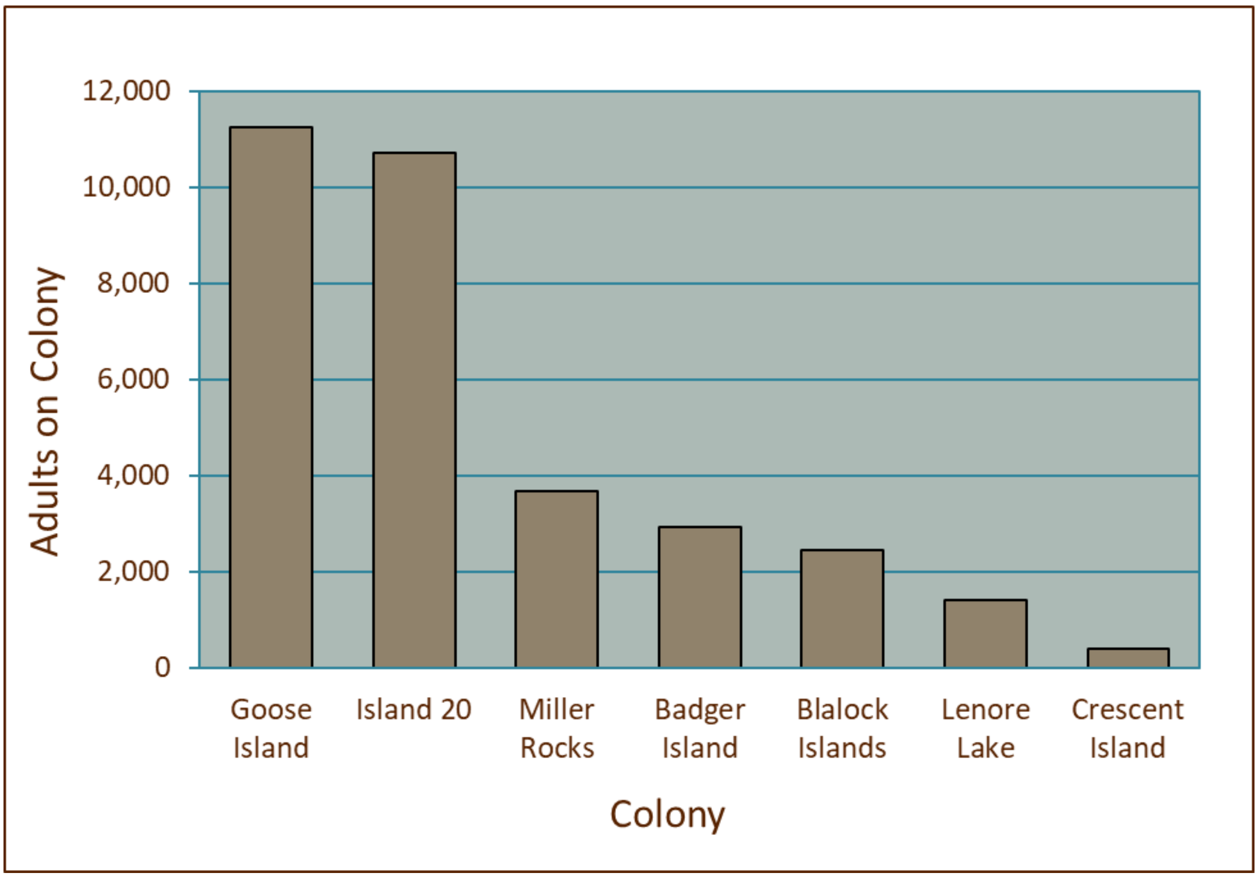


Figure 7. Sizes of gull breeding colonies (adults on colony) in the Columbia Plateau region during the 2020 breeding season. Gulls were also confirmed to be breeding at Twinning and Goose islands in Banks Lake and Harper Island in Sprague Lake, however no counts of breeding individuals are available for these sites.

Table 4. Size of California/ring-billed gull (LAXX), double-crested cormorant (DCCO), and American white pelican (AWPE) colonies in the Columbia Plateau region in 2020. "Active" denotes a colony that was active but not counted in 2020.

Colony	LAXX [1]	DCCO [2]	AWPE [1]
Miller Rocks (Columbia River)	3,690	1	
Blalock Islands (Columbia River)	2,465		
Crescent Is. (Columbia River)	400		
Badger Is. (Columbia River)	2,938		3,165
Foundation Is. (Columbia River)		215	
Island 20 (Columbia River)	10,703		
Hanford Reach (Columbia River)		active	
Goose Is. (Potholes Reservoir)	11,240		
North Rocks and Shoal Is. (Lenore Lake)	1,400	110	
Harper Is. (Sprague Lake)	active	333	
Twinning Is. and Goose Is. (Banks Lake)	active		

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

Columbia River Estuary

The USACE estimated that 2,387 breeding pairs of Caspian terns nested on the prepared 1-acre main colony on East Sand Island in 2020 (K. Tidwell, USACE-FFU, personal communication), which is below the target colony size for terns on East Sand Island specified in the management plan (i.e. 3,125–4,375 breeding pairs). However, large numbers of terns (hundreds to thousands of adults per week) once again attempted (unsuccessfully) to nest along the southeast and east beaches of East Sand Island, despite ongoing nest dissuasion efforts to prevent terns from nesting outside of the prepared colony area. In 2020, cormorants did not attempt to nest on East Sand Island, nor did terns establish a colony on Rice Island (K. Tidwell, USACE-FFU, personal communication). Based on monitoring conducted by ODFW, the size of the double-crested cormorant colony on the Astoria-Megler Bridge was estimated to be 5,081 breeding pairs (J. Lawonn, ODFW, personal communication), which represents the largest colony size for cormorants on the bridge ever recorded and was the sixth consecutive year of growth of that colony since cormorant management was first implemented on East Sand Island in 2015.

AVIAN PREDATION RATES

PIT-tagging of Upper Columbia River Steelhead

A total of 6,294 UCR steelhead smolts (4,388 hatchery, 1,906 wild) were captured, PIT-tagged, measured (fork-length), condition-scored, and released into the tailrace of RIS as part of this study in 2020. An additional 551 previously PIT-tagged (i.e. recaptured) steelhead smolts were also detected at the RIS trap, resulting in a total of 6,845 UCR steelhead available for predation rate analyses in 2020. Steelhead were tagged and released at RIS from 9 April to 16 June 2020, a period which accounted for 98% of all steelhead encountered in the trap. Fish were randomly selected for tagging and were tagged in-concert with, and in proportion to, the run at-large (see also Evans et al. 2014). Mean steelhead fork length was 190 mm (standard deviation [SD] = 26 mm; range = 109 to 304 mm). An evaluation of external fish condition indicated that most steelhead were in good over-all external condition in 2020, with only 4.5% of steelhead observed with severe body injuries (subcutaneous wounds/scars), disease (fungal or viral infections), severe descaling (>20% of scales missing), and/or major fin damage (>50% of fin tissue missing). The most common type of damage was body injuries, followed by severe descaling. By comparison, between 3.6% and 23.1% of steelhead tagged in years past were in compromised condition (see also Evans et al. 2014 and Collis et al. 2020).

PIT Tag Recovery

As part of this study, a total of twelve avian colonies in the CPR were scanned for smolt PIT tags following the 2020 breeding season, including two tern colonies, six gull colonies, three cormorant colonies, and one pelican colony (*Table 5*). Scanning was also conducted at four known avian loafing/roosting sites in the CPR, areas where large numbers of piscivorous waterbirds were observed during the smolt out-migration period. From these locations (breeding and loafing combined), a total of 9,432 PIT tags from 2020 migration-year smolts (Chinook, coho, sockeye, and steelhead) were detected (*Table 5*). The vast majority (>98%) of tags were recovered from breeding colonies, with the largest number of smolt PIT tags found on the Badger Island gull and pelican colonies (n = 3,892), followed by the Miller Rocks gull colony (n = 2,374), the Foundation Island cormorant colony (n = 1,002), and Blalock Islands tern colony (n = 596; *Table 5*). In addition to smolt tags, 29 tags from other salmonid age-classes (i.e. adults) and fish species were recovered on Badger Island, which included 19 adult sockeye, 7 northern pikeminnow (*Ptychocheilus oregonensis*), 2 white sturgeon (*Acipenser transmontanus*), and 1 bull trout (*Salvelinus confluentus*). We did not attempt to recover PIT tags from the two managed tern colony sites on Crescent Island and Goose Island because, in the case of Crescent Island, no terns were observed nesting at the site in 2020, or, in the case of Goose Island, hazing associated with the *IAPMP* prevented colony formation so there was no known nesting area from which to scan for PIT tags (see also *below*).

Table 5. Numbers of 2020 migration year PIT-tagged salmonid smolts (Chinook, sockeye, coho, and steelhead combined) and other fish species (adult salmonids {Badger Island only}, sea-run cutthroat trout {estuary colonies only}, and northern pikeminnow combined) recovered on avian breeding colonies and loafing sites in the Columbia River Basin. Piscivorous colonial waterbird species include Caspian terns (CATE), California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), and American white pelican (AWPE). Mixed colonies represent an unknown combination of these and possibly other avian species. Numbers of recovered PIT tags were not adjusted to account for tag loss due to on-colony PIT detection and deposition probabilities (see Table 6) and thus represent minimum numbers of consumed tagged fish by birds at each site (see Methods & Analysis section).

Location	Rkm	Bird Species	Area Use	Smolt PIT Tags Recovered	Other Recovered
Potholes Reservoir	Off-river	LAXX	Breeding	3	
Lenore Lake	Off-river	CATE	Breeding	179	
		DCCO	Breeding	1	
		Mixed	Loafing	64	
Hanford Island	592	DCCO	Breeding	51	
			Loafing	20	
Island 20	549	LAXX	Breeding	568	
Foundation Island	518	DCCO	Breeding	1,002	
		Mixed	Loafing	18	
Badger Island	512	LAXX	Breeding	277	
		AWPE/LAXX	Breeding	3,892	29
Blalock Islands	441-439	CATE	Breeding	596	
		LAXX	Breeding	322	
		Mixed	Loafing	65	
Miller Rocks	331	LAXX	Breeding	2,374	
Miler Sand Island	38	AWPE	Breeding	7	
Channel Markers	35-39	DCCO	Breeding	90	
Rice Island	34	CATE	Breeding	293	2
		AWPE	Breeding	1	
		Mixed	Loafing	92	
Astoria-Megler Bridge		DCCO	Breeding	1,048	
East Sand Island	8	CATE [1]	Breeding	3,705	15
			Breeding (Satellite)	890	3
		DCCO	Breeding	38	1
		Mixed [1]	Breeding	62	
			Loafing	13	1
Total				**15,671**	**51**

[1] Tags recovered by the USACE.

In association with survival models (see *below*), PIT tags were also recovered at avian breeding and loafing sites in the CRE downstream of Bonneville Dam, including the Astoria-Megler Bridge (cormorants, n = 1,048), Rice Island (terns, n = 386), East Sand Island (cormorants, n = 113; mixed species, n = 62), two channel markers (cormorants, n = 90), and Miller Sands Spit (pelicans, n = 7; see *Table 5* above). As part of USACE funded efforts, scanning for smolt PIT tags also occurred at the East Sand Island tern colony in the CRE, where 3,705 and 890 PIT tags from 2020 migration year smolts were recovered from the prepared 1-acre colony area and at satellite breeding sites along the southeast and east beaches, respectively. From all estuary locations combined, a total of 6,234 PIT tag from 2020 migration year smolts were recovered following the breeding season. In addition to salmon and steelhead smolt tags, 22 PIT tags from sea-run cutthroat trout (*O. clarki*) were also recovered at breeding and loafing sites on East Sand and Rice islands in 2020 (see *Table 5* above).

PIT Tag Detection & Deposition Probabilities

Table 6 provides results on the probability that a consumed PIT tag was deposited on a bird colony and subsequently detected by researchers following the breeding season, 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 colonies (see also Hostetter et al. 2015 and Payton et al. *In prep.*). Detection probabilities were directly measured in 2020 based on the proportion of tags that were intentionally sown by researchers at each colony during the breeding season that were subsequently detected after the breeding season (see also Hostetter et al. 2015). Detection probabilities were highly variable (range = 0.14 – 0.99), depending on the colony and when during the nesting season tags were sown (at the beginning or end of the breeding season; *Table 6*). The lowest detection probabilities were on the Blalock Islands tern colony and the cormorant colonies on Foundation and Hanford islands, locations where detection efficiency has steadily decreased over time due to high rates of PIT tag collision. PIT tag collision is caused by high densities of tags, a phenomenon that renders tag codes in close proximity to one another unreadable (see also Evans et al. 2019).

Table 6. Detection efficiency (range during nesting season) and deposition (95% credible interval) estimates (depicted as a proportion) for smolt PIT tags on bird colonies during 2020. Results were used to estimate predation rates (see Tables 7-13) based on the number of smolt PIT tags recovered following the breeding season (see Table 5 above). Piscivorous colonial waterbird species include Caspian terns (CATE), California and ring-billed gulls (LAXX), double-crested cormorants (DCCO), and American white pelican (AWPE).

Location	Rkm	Bird Species	Detection (Range)	Deposition (95% CRI) [1]
Lenore Lake	Off-river	CATE	0.50 – 0.82	0.71 (0.51-0.89)
Lenore Lake		DCCO	0.50 – 0.98	0.51 (0.34-0.70)
Goose Island	Off-river	LAXX	0.32 – 0.96	0.15 (0.11-0.21)
Hanford Island	592	DCCO	0.20 – 0.52	0.51 (0.34-0.70)
Island 20	549	LAXX	0.60 – 0.90	0.15 (0.11-0.21)
Foundation Island	549	DCCO	0.15 – 0.34	0.51 (0.34-0.70)
Badger Island	512	AWPE	0.55 – 0.75	0.47 (0.36-0.60)
		LAXX	0.44 – 0.66	0.15 (0.11-0.21)
Blalock Islands	441-439	CATE	0.38 – 0.53	0.71 (0.51-0.89)
		LAXX	0.88 – 0.96	0.15 (0.11-0.21)
Miller Rocks	331	LAXX	0.83 – 0.84	0.15 (0.11-0.21)
East Sand Island - Main Colony	8	CATE	0.48 – 0.84 [2]	0.71 (0.51-0.89)
East Sand Island - Satellite Colonies	8	CATE	0.11 – 0.94 [2]	0.71 (0.51-0.89)

[1] Deposition estimates for CATE, DCCO, and LAXX are those of Hostetter et al. (2015); estimates for AWPE are those of Payton et al. (In prep).
[2] From Evans et al. (2021)

Efficacy of Avian Predation Management Plans

Currently there are three avian predation management plans being implemented to increase survival of ESA-listed juvenile salmonids in the CRB. To evaluate the efficacy of these management plans, we measured predation rates at both managed and unmanaged colonies in 2020 and compared those rates to those prior to implementation of management (see *below*).

Inland Avian Predation Management Plan: Tern predation rate estimates in 2020 were amongst the lowest ever recorded in the CPR, with estimates ranging from < 0.1% to 2.2% (95% CRI = 0.7–5.4) per colony, per ESA-listed salmonid ESU/DPS (*Appendix, Table A1-A2*). For the first time since management actions were implemented in the CPR starting in 2014, the goal of reducing predation to less 2% per colony, per ESU/DPS, were achieved for nearly all salmonid ESUs/DPSs evaluated in 2020.

Predation rates estimates at the unmanaged colony in Lenore Lake were below the 2% threshold for all salmonid ESUs/DPSs evaluated in 2020, with the highest rate being 1.0% (95% CRI = 0.6–1.5) on UCR steelhead (*Table 7*). Rates were 0.3% (0.1-0.8) for UCR spring Chinook and were <0.1% for all other ESUs/DPSs (*Table 7*). Predation rate estimates for terns nesting on islands in Lenore Lake were available starting in 2015, the first year a colony was observed on islands in the lake, and estimates have been consistently at or below 1.0% per ESU/DPS (*Appendix, Table A3*). These results suggest that at its current size the tern nesting colony on Lenore Lake poses a relatively minor risk to the survival of UCR steelhead and no to little risk to other salmonid populations. Based on average annual per-capita predation rates on UCR steelhead by terns nesting at Lenore Lake, the nesting colony would have to increase to more than 150 breeding pairs to potentially reach the 2% predation threshold identified in the *IAPMP* (see also Collis et al. 2021).

Table 7. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESU/DPS) by Caspian terns nesting on an unnamed island in Lenore Lake and on the Blalock Islands during 2020. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section and Appendix for details and estimates of predation by these colonies in years past).

ESU/DPS	Lenore Lake	Blalock Island
SR Sockeye	NA	NA
SR Sp/Su Chinook	< 0.1%	0.1% (<0.1-0.4)
UCR Sp Chinook	0.3% (0.1-0.8)	0.1% (<0.1-0.5)
SR Fall Chinook	< 0.1%	0.2% (0.1-0.8)
SR Steelhead	< 0.1%	2.2% (0.7-5.4)
UCR Steelhead	1.0% (0.6-1.5)	1.3% (0.3-4.2)

Based on aerial imagery taken during the peak of breeding season in May of 2020, there were no terns nesting on Twinning Island in Banks Lake or on Harper Island in Sprague Lake (see *Map 1* above), locations where small colonies (< 40 breeding pairs) occurred in most but not all years in the past. Given there were no terns nesting at these sites in 2020, predation rates were assumed to be zero or close to zero (i.e. <0.1%). In 2014 and 2015, following implementation of managed actions at the nearby tern colony on Goose Island in Potholes Reservoir (see *below*), 66 and 64 pairs nested, and predation rates on UCR steelhead were 1.2% (0.3–6.4) and 2.6% (1.8–3.9), respectively, demonstrating that terns were commuting to middle Columbia River to forage on smolts and that impacts exceed the 2% threshold (*Appendix, Table A1*). As such, continued monitoring of Twinning Island and other islands in Banks Lake for the reestablishment of a tern colony is warranted. Scanning for PIT tags at the Harper Island tern colony has occurred only once in the past (2012), the only year when island access was granted for PIT tag recovery by the private landowner (Collis et al. 2021). Predation rate estimates in

2012 indicated that terns consumed less than 0.1% of available Snake River and UCR ESUs/DPSs, with the highest rates observed on Snake River steelhead at 0.2% (see Roby et al. 2021, Appendix B). Here too, predation impacts were low, in part, due the size of the colony (30 breeding pairs; Collis et al. 2021). Future monitoring of predation rates at this colony may be warranted if a tern colony become established at this site and landowner permission is granted to access the island.

Passive dissuasion, coupled with human hazing of terns on Goose Island by the USDA-WS prevented the formation of a large, sustained breeding colony on Goose Island in 2020. For the first time since 2015, however, terns were successful in rearing young on Goose Island (see *above*). Furthermore, the attendance of terns at the island during the peak of the smolt outmigration (May) was the highest (180 adults; USDA-WS 2021) observed since 2015. Due to the aforementioned hazing of terns on Goose Island, there was no discrete area to scan for smolt PIT tags following the breeding season and, as such, no estimates of predation rates were unavailable. We assume that terns that attempted to nest on Goose Island during May were commuting to the middle Columbia River to consume juvenile salmonids but based on per capita (per breeding pair) predation rate estimates from years past and the fact that most terns were only briefly present on Goose Island in 2020, predation rate estimates were likely less than 2% per ESU/DPS. In 2014, the first year of management at Goose Island, a colony of 159 pairs were present throughout the smolt outmigration period (April to June) and consumed an estimated 2.9% (95% CI = 1.9–5.1) of UCR steelhead. Since 2015 and prior to 2020, only small numbers of terns (less than 50 adults) were observed on Goose Island during the peak breeding and smolt outmigration period (Collis et al. 2021). Predation rates on UCR steelhead by Goose Island terns prior to implementation of management actions in 2014 were the highest of any tern colony in the region averaging 15.7% (14.1–18.9) during 2007-2013 (*Table 8*). Estimated average annual predation rates on UCR Spring Chinook were 2.5% (1.7–3.6) during 2007-2013 (*Table 8*). Finally, in 2016, a colony of 144 pairs formed on an unnamed island in northern Potholes Reservoir. Recoveries of smolt PIT tags indicated that terns consumed an estimated 4.1% (2.9–6.3) of UCR steelhead in 2016 (*Appendix, Table A1*), impacts that prompted adaptive management actions at this and other surrounding islands in northern Potholes Reservoir during 2017–2020. Since then, active and passive dissuasion techniques have been successful at preventing terns from nesting on islands in northern Potholes Reservoir thereby eliminated predation impacts.

Table 8. 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 (2014-2020 for Goose Island and North Potholes; 2015-2020 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 (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 (2017–2020). See Appendix for annual estimates. NC denotes that no colony existed during that period.

	Goose Is.			North Potholes Is.			Crescent Is.		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2020 [2]	Last 3-years 2018–2020 [2]	Pre-Management 2007–2013	Management 2016 [1]	Last 3-years 2018–2020	Pre-Management 2007–2014	Management 2015–2020	Last 3-years 2018–2020
SR Sockeye	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	1.5% (1.2–2.0)	< 0.1%	< 0.1%
SR Sp/Su Chinook	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	0.8% (0.7–1.0)	< 0.1%	< 0.1%
UCR Sp Chinook	2.5% (1.7–3.6)	< 0.1%	< 0.1%	NC	0.1% (0.1–0.3)	< 0.1%	0.5% (0.3–0.9)	< 0.1%	< 0.1%
SR Fall Chinook	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	1.0% (0.9–1.2)	< 0.1%	< 0.1%
SR Steelhead	< 0.1%	< 0.1%	< 0.1%	NC	< 0.1%	< 0.1%	4.5% (4.1–5.1)	< 0.1%	< 0.1%
UCR Steelhead	15.7% (14.1–18.9)	0.2% (0.1–0.5)	< 0.1%	NC	4.1% (2.9–6.3)	< 0.1%	2.5% (2.2–2.9)	< 0.1%	< 0.1%

[1] Nesting colony formed in 2016 and was than actively managed during 2017–2019.
[2] Large numbers of Caspian terns were observed on Goose Island during the peak smolt out-migration period in May of 2020 (USDA-WS 2021) but since birds were dissuaded from nesting, predation rate estimates were not available (see Results & Discussion section).

For the sixth consecutive year, the tern colony at Crescent Island was eliminated in 2020 and predation rates were thus assumed to be zero or close to zero. Prior to management actions in 2015, predation rates by Crescent Island terns were highest on steelhead populations, with an average annual predation rate estimate of 2.5% (2.2–2.9) and 4.5% (4.2–5.1) on UCR steelhead and SR steelhead, respectively (see *Table 8* above). Average annual predation rates on salmon ESUs were less than 2% prior to management, with the highest impacts observed on SR sockeye at 1.5% (1.2–2.0; see *Table 8* above). Management actions at Crescent Island have been successful and have essentially eliminated predation impacts by terns nesting at this site since 2015 (see *Table 8* above). The greatest benefits of management have been to Snake River salmonids due to the greater susceptibility of Snake River salmonid ESUs/DPSs compared with UCR ESUs/DPSs (see also Collis et al. 2021).

Predation rates on ESA-listed salmonid populations by terns nesting on the Blalock Islands, the largest tern colony in CPR in 2020, ranged from 0.1% (< 0.1–0.4) in SR spring/summer Chinook to 2.2% (0.7–5.4) in SR Steelhead (*Table 9*). In the case of SR steelhead, rates were the highest for any individual tern colony in the CPR in 2020, but rates were just slightly above the 2% threshold identified in the *IAPMP*. Predation rates by Blalock Islands terns have been, on average, significantly higher since management actions on the Crescent Island tern colony were implemented in 2015 (*Appendix, Table A2*). Increases in predation rates were commensurate with the over-all increase in the size of the Blalock Island tern colony, with the colony increasing from an average of 57 breeding pairs (range = 6 to 136) during 2007–2014 to average of 460 breeding pairs (range = 313 to 677) during 2015–2019 (see *Figure 4* above; see also Collis et al. 2021). In 2020, however, colony size was greatly reduced (150 pairs; see *Table 3* above) as were predation rates, with the lowest estimates observed since 2015, the first year following management at Crescent Island. Despite significantly reduced predation rates in 2020, average annual predation rate estimates by terns nesting on the Blalock Islands during the management period were comparable to or higher than those of terns nesting on Crescent Island during the pre-management period for most of the ESUs/DSPs evaluated, particularly for ESUs/DPSs originating from the Snake River (*Table 9*). For instance, predation rates on Snake River steelhead by Blalock Island terns increased from an average of 0.5% (95% CRI = 0.4-0.9) prior to management to an average of 4.0% (95% CRI = 3.3-4.9) during implementation of management at Goose and Crescent islands (*Table 9*). Consequently, as described by Collis et al. (2021) increases in predation rates on salmonid smolts by terns nesting on the Blalock Islands has offset the benefits achieved by the elimination of the tern colonies on Crescent and Goose islands associated with management in most years. Although average annual predation rates have been significant higher during the management period, rates in 2020 were the lowest observed since 2015, particularly impacts on UCR and SR steelhead which were as high as 8% in some years (*Appendix, Table A2*).

Table 9. Average annual predation rates (95% credible intervals) by Caspian terns nesting at unmanaged colonies during the pre-management period (2007-2013) and during the management period (2014-2020). 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 (Fall) fish were evaluated. See Appendix for annual estimates. Time periods represent the average of all years or data from the last three years of the management (2018–2020). NA denotes that predation estimates were not available. NC denotes that no colony existed during that period.

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

	Lenore Lake Islands			Harper Island		
ESU/DPS	Pre-Management 2007–2013	Management 2014–2020	Last 3-years 2018–2020	Pre-Management 2007–2013	Management 2014–2020	Last 3-years 2018–2020
SR Sockeye	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
SR Sp/Su Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
UCR Spr Chinook	NC	0.1% (0.0–0.3)	0.1% (0.0–0.3)	< 0.1%	NA	NA
SR Fall Chinook	NC	< 0.1%	< 0.1%	< 0.1%	NA	NA
SR Steelhead	NC	< 0.1%	< 0.1%	0.2% (0.1–1.3)	NA	NA
UCR Steelhead	NC	0.7% (0.5–1.0)	0.8% (0.6–1.1)	< 0.1%	NA	NA

[1] *No established nesting colony existed during 2018–2020.*

In summary, predation rates by tern colonies in the CPR were amongst lowest recorded since management actions associated with the *IAPMP* were first implement in 2014. Reductions in tern colony sizes at the managed colony site on Goose Island in Potholes Reservoir have, on average, greatly reduced predation rates on salmonids smolts, particularly on UCR steelhead. Similarly, the complete elimination of the Crescent Island tern colony during 2015-2020 reduced predation by terns breeding at this site to zero or nearly so. An unintended consequent of management action at Goose Island and Crescent Island, however, was the rapid expansion of the tern colony on the Blalock Islands, where predation rates have exceeded the 2% threshold for multiple ESA-listed salmonid ESUs/DPSs and years, indicating that adaptive management at this colony site will likely be needed to achieve the over-all goals of the *IAPMP* (see also Collis et al. 2021). Adaptive management at the Blalock Island nesting sites could benefit ESA-listed populations originating from both UCR and Snake River ESUs/DPSs, but the greatest net benefit would be to Snake River populations, populations that have yet to receive the full benefits of the tern management actions in the CPR due the dispersal of terns from Crescent Island in McNary Reservoir to the Blalock Islands in John Day Reservoir.

East Sand Island Caspian Tern & Double-crested Cormorant Management Plans: Estimates of predation rates on juvenile salmonids by terns nesting on East Sand Island in 2020 were the lowest recorded since terns began nesting on East Sand Island in 1999, with estimates ranging from 0.4% (0.2–0.9) on Upper Columbia River spring Chinook to 5.9% (4.5–8.1) on Snake River steelhead (*Appendix A, Table A.3*; see also Evans et al. 2021). Results indicated that predation by terns on East Sand Island have been reduced by 65% to 76% on steelhead DPSs; reductions that meet or exceed those anticipated in the *Estuary Caspian Tern Management Plan* (Evans et al. 2021). Large numbers of terns (several hundred to several thousand) attempted to nest outside of the designated 1-acre nesting areas on East Sand Island in 2020, as well as nesting attempts by terns on Rice Island in the upper Columbia River estuary. Thus, continued nest dissuasion techniques and continue monitoring at sites throughout the estuary will be necessary to ensure predation impacts do not exceed levels stipulated in the *Plan* in the future.

There was no established colony of cormorants on East Sand Island in 2020 and only a small number of smolt PIT-tags (n = 38) were recovered on areas where cormorants briefly attempted to nest (see *Table 5* above). No estimates of predation were available for the large cormorant colony on the Astoria-Megler Bridge or for the other smaller, cormorant colonies in the upper estuary in 2020. Based on higher per-capita (per bird) predation rates of cormorants nesting in the upper estuary in years past (see Cramer et al. 2021) and the relatively large number of PIT tags recovered (n = 1,048) from a small nesting area (i.e. north crib) on the Astoria-bridge in 2020 (*Table 5* above), cormorant predation rates on juvenile salmonids in the Columbia River estuary were likely substantial in 2020 but were not quantified and are thus unknown.

Historic colony- and ESU/DPS-specific predation rates by East Sand Island terns and cormorants dating back 2000 for terns and 2003 for cormorants are provided in the *Appendix (Table A3* and *Table A4*, respectively; see also Roby et al. 2021, Appendix B).

Predation Rates by Other Piscivorous Colonial Waterbirds

Predation rates by gulls and cormorants nesting at several colonies in CPR were equal to or often greater than those of tern colonies nesting in 2020, with predation rates being more than 2% and as high as 9% for several ESA-listed ESUs/DPSs (see *below*). Despite high levels of predation at some colonies, predation impacts by other colonies were low to non-existent (i.e. < 0.1%), indicating that not all of the colonies evaluated posed a potential threat to smolt survival in 2020. Also, gulls are known to consume dead or moribund fish and to kleptoparasitize (steal) fish from other piscivorous waterbirds, such as terns. Consequently, smolt PIT tag recoveries on gull colonies may be more indicative of consumption rates, rather than predation rates (Cramer et al. 2021a). Like terns, however, cormorants and pelicans are strictly piscivorous and known to consumed live fish (Collis et al. 2021). Below is summary of colony- and salmonid ESU/DPS-specific predation/consumption rates by non-managed gull, cormorant, and pelican colonies in the CPR in 2020, along with comparisons to predation/consumption estimates from years past.

Goose Island Gulls: Consumption rates of salmonids by gulls nesting on Goose Island in Potholes Reservoir were amongst the lowest of any colony evaluated in 2020, with consumption estimates of <0.1% of available smolts, per salmonid ESU/DPS (*Table 10*). Despite the large size of the gull colony on Goose Island (11,240 adults counted; see *Table 2* above), only three smolt PIT tags were recovered from the colony following the breeding season (see *Table 5* above). Results indicate that gulls were not commuting to the middle Columbia River to forage on salmonids. In 2012, the last year the Goose Island gull colony was scanned for smolt PIT tags, BRNW (2013) estimated consumption rates by gulls as high as 2.8% of available UCR steelhead smolts (*Appendix, Table A6*). A large colony of terns (463 breeding pairs), however, also nesting on Goose Island in 2012 and BRNW (2013) hypothesized that most, if not all, of the tags deposited on the gull colony in 2012 were from smolts that were stolen (kleptoparasitized) from terns. The extremely low consumption estimates of gulls in 2020 support this hypothesis and indicate that gulls breeding on Goose Island pose little to no threat to the survival of anadromous juvenile salmonids. With a large tern colony nearby, however, gulls on Goose Island may still pose a threat to steelhead survival because the energy demands of terns are presumably higher in the presence of gulls that kleptoparasitize smolts from terns.

Table 10. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESU/DPS) by California and ring-billed gulls nesting on Goose Island, Island 20, Badger Island, Blalock Island, and Miller Rocks Island during 2020. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section and Appendix for details and estimates of predation by these colonies in years past).

ESU/DPS	Goose Is.	Island 20	Badger Is.	Blalock Is.	Miller Rocks Is.
SR Sockeye	NA	NA	NA	NA	NA
SR Sp/Su Chinook	<0.1%	0.3% (<0.1-1.3)	1.0% (0.1-3.1)	0.1% (<0.1-0.4)	1.1% (0.5-2.0)
UCR Sp Chinook	<0.1%	0.1% (<0.1-1.4)	0.6% (0.1-2.0)	0.1% (<0.1-0.9)	3.1% (1.4-6.2)
SR Fall Chinook	<0.1%	0.1% (<0.1-1.4)	1.0% (0.3-6.1)	0.1% (<0.1-0.7)	1.9% (0.8-3.9)
SR Steelhead	<0.1%	1.6% (0.3-4.9)	9.2% (2.0-18.4)	2.6% (0.7-6.8)	4.4% (1.8-9.2)
UCR Steelhead	<0.1%	2.1% (1.1-3.6)	4.9% (1.7-9.3)	3.2% (0.7-9.6)	2.5% (0.5-8.0)

Island 20 Gulls: Consumption rates on salmonid smolts by gulls nesting on Island 20 varied by salmonid ESU/DPS, with rates ranging from 0.1% (<0.1-1.4) in SR Fall Chinook to 2.1% (1.1-3.6) in UCR steelhead (see *Table 10* above). As observed in years past, estimates of consumption were significantly higher on steelhead DPSs compared with salmon ESUs. Previous suggest that higher consumption rates of steelhead smolts by gulls compared with salmon smolts is associated with the larger average size of steelhead smolts and the surface orientation of steelhead smolts, factors the increase the susceptibility of steelhead to plunge-diving predator like gulls and terns (Evans et al. 2016, Evans et al. 2019; Cramer et al. 2021a). Estimates of Island 20 gulls consumption from 2020 were generally similar to those observed in years past, with the exception of 2015. In 2015 the highest estimates recorded to-date were observed, with an estimated 3.6% and 7.9% of SR steelhead and UCR steelhead, respectively, consumed (Roby et al. 2016; see *Appendix, Table A6*). In most other years, however, predation rates on steelhead DPSs were less than 3.0% of available smolts and were less than 1.0% of available salmon ESUs (*Appendix, Table A5*). Increased rates of consumption by Island 20 gulls in some years, like 2015, may be associated with increases in colony size and/or due to environmental conditions (e.g., low flows) that increase smolt exposure times to gull predation during out-migration (Roby et al. 2016, Payton et al. 2016, Hostetter et al. 2021).

Badger Island Gulls: Similar to gulls on Island 20, consumption rates by gulls nesting on Badger Island on juvenile salmonids were highly variable, ranging from 0.6% (0.1–2.0) of UCR spring Chinook to 9.2% (2.0–18.4) of SR steelhead, the highest ESU/DPS-specific predation rate observed by any colony in the CPR in 2020. Smaller sample sizes of PIT tagged smolts in 2020, coupled with uncertainty in consumption probabilities between gulls and co-nesting pelicans on Badger Island (see *below)* resulted in imprecise estimates of consumption, as denoted by the width of 95% credible intervals (see *Table 10* above; see *Appendix, Table A6* for sample sizes of available tagged smolts). Predation rates by Badger Island gulls were also high on UCR steelhead in 2020 with an 4.9% (1.7–9.3) of available fish consumed (see *Table 10* above). Consumption rates by Badger Island gulls in 2020 were similar to those observed in years past, with the exception of SR steelhead, where estimates in 2020 were the highest observed since the colony formed in 2015 (*Appendix, Table A6*). Again, smaller samples sizes of tagged smolts, particularly SR steelhead, where sample size were an order of magnitude smaller in 2020 compared to most other years (*Appendix, Table A6*), resulted in imprecise estimates of consumption, so relative comparisons of estimates between years should be interpreted cautiously.

Blalock Island Gulls: Since at least 2012, there have been two separate gull colonies present within the Blalock Islands complex in John Day Reservoir, one on Anvil Island and one on Straight Six Island. Of the two gull colonies in the Blalock Islands complex, smolt consumption rates have been significantly higher for gulls nesting on Anvil Island as compared to gulls nesting on Straight Six Island (see also Roby et al. 2016). Differences in relative predation rates are attributed to difference in the size of the colonies (with substantially more gulls on Anvil Island) and difference in species composition (Anvil Island was dominated by nesting California gulls and Straight Six Island was dominated by nesting ring-billed gulls, the former are known to consume a higher proportion of juvenile salmonids; Collis et al. 2002). Also, data from Hostetter et al. (2015) and Cramer et al. (2021a) indicated that per capita (per bird) consumption of juvenile salmonids was consistently greater for gull colonies dominated by California gulls as compared to those dominated by ring-billed gulls. This difference in smolt consumption rates between the gull species is likely due to differences in body size and energy requirements (Winkler 1996), as well as the proportion of the diet that consists of fish (Collis et al. 2002), both of which are greater for California gulls compared with ring-billed gulls (see also Cramer et al. 2021a).

For the purpose of this and other predation rate studies (Evans et al. 2019, Cramer et al. 2021a), we have combined estimates of gulls nesting on Anvil and Straight Six islands (collectively referred to as "Blalock Islands gulls"; see also Evans et al. 2019). Results of both colonies combined indicate that consumption rates were less than 1% for all ESA-listed salmon ESU but were higher for steelhead DPSs with estimates of 2.6% (0.7–6.8) and 3.2% (0.7–9.6) in UCR and SR steelhead, respectively (see *Table 10* above). Consumption rate estimates by Blalock Island gulls in 2020 were like, but slightly higher than, those observed in years past (*Appendix, Table A6*). Similar to estimates at the Badger Island gull colony and at other gull colonies, small sample sizes of PIT-tagged smolts resulted in imprecise estimates of consumption in 2020.

Miller Rocks Island Gulls: Consumption rates by gulls nesting at Miller Rocks Island were amongst the highest of any gull colony evaluated in 2020, with estimates consistently higher than those of other nearby gull colonies, particularly predation on salmon ESUs. For instance, consumption rates by other gull colonies were consistently < 0.5% on salmon ESUs but ranged from 1.1% (0.4–2.0) on SR sockeye to 3.1% (1.4–6.2) on UCR spring Chinook (see *Table 10* above). Similar to other gull colonies, the consumption estimates of gulls nesting on Miller Rocks were higher on steelhead DPSs compared with salmon ESUs but differences between steelhead and salmon were not as pronounced at Miller Rocks, with Miller Rock gulls consuming an estimated 2.5% (0.5–8.0) and 4.4% (1.8–0.2) of UCR and SR steelhead, respectively (see *Table 10* above). Consumption rate estimates by Miller Rocks gulls in 2020 were similar to those observed in years past for salmon ESUs but lower than those of years past on steelhead DPSs (*Appendix, Table A7*).

Miller Rocks Island is in close proximity to John Day and The Dalles dams (18 Rkm and 23 Rkm, respectively). Evans et al. (2016) observed that gulls disproportionately consumed smolts near dams and hypothesized that smolts may be more vulnerable near dams as a result of (1) increased smolt travel times or delayed migration in the forebay of dams, (2) smolt morbidity or mortality associated with dam passage, or (3) smolts being temporarily stunned or disoriented by hydraulic conditions in the tailrace of dams. Gull consumption of smolts, however, is not limited to foraging near dams, with gulls consuming substantial numbers of good-condition smolts from open reservoirs and free-flowing sections of the river as well (see Evans et al. 2016 for a detailed discussion). Results indicate that gulls consumed smolts that were tagged and released in apparently good condition and preferentially consumed large smolts, attributes that suggest that gull consumption was a partially additive source of mortality for juvenile salmonids. Nevertheless, the proportion of smolts consumed by gulls that were dead, moribund, or otherwise compromised when consumed, rather than depredated in an alive and healthy state, is currently not known. Addressing this question is paramount to understanding what proportion of salmonid smolts consumed by gulls were depredated, and to what degree the mortality of smolts that were depredated by gulls was additive mortality, and therefore limits smolt survival in the CRB (see also *below* and Cramer et al. 2021a).

Lenore Lake Double-crested Cormorants: Despite an estimated colony size of 110 breeding pairs, only one smolt PIT tag was recovered from the Lenore Lake cormorant colony following the breeding season in 2020 (*Table 11*). In 2018 and 2019, when an estimated 111 and 112 breeding pairs, respectively, were present, zero (no) and one PIT tags were recovered, respectively. Results provide strong evidence the Lenore Lake cormorants pose little or no threat to out-migrating juvenile salmonids and unlike terns breeding on islands in Lenore Lake (see *above*), cormorants apparently do not commute to the middle Columbia River to forage on juvenile salmonids during the breeding season (see also Cramer et al. 2021a).

Table 11. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESU/DPS) by double-crested cormorants nesting on unnamed island in Lenore Lake, Hanford Island, and Foundation Island during 2020. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section and Appendix for details and estimates of predation by these colonies in years past).

ESU/DPS	Lenore Lake	Hanford Island	Foundation Island
SR Sockeye	NA	NA	NA
SR Sp/Su Chinook	<0.1%	<0.1%	2.5% (1.0-5.8)
UCR Sp Chinook	<0.1%	0.1% (<0.1-0.9)	0.1% (<0.1-2.4)
SR Fall Chinook	<0.1%	<0.1%	0.8% (0.1-3.6)
SR Steelhead	<0.1%	<0.1%	4.0% (1.2-10.7)
UCR Steelhead	<0.1%	<0.1%	0.1% (<0.1-0.3)

Hanford Island Double-crested Cormorants: The colony on Hanford Island or Locke Island in Hanford Reach on the Columbia River has periodically been scanned for smolt PIT tags since nesting birds were first confirmed in 2007 (Cramer et al. 2021a). Predation rate estimates in 2020 were low, with less than 0.2% of available fish predated, per ESU/DPS. The highest estimate to-date was just 0.2% (0.1–0.7) in UCR spring Chinook in 2017, when 68 pairs were observed nesting trees. The small size of the colony is presumably associated with the low rate of predation on juvenile salmonids, especially given the colonies location on the middle Columbia River. It should be noted that the island regularly floods during the nesting season, however, flooding which may have washed away or buried some unknown proportion of deposited tags.

Foundation Island Double-crested Cormorants: The cormorant colony on Foundation Island, the largest cormorant colony located on the Columbia River upstream of Bonneville Dam, was scanned for PIT tags in 2020 for the first time in the last five years (*Appendix, Table A7*). Predation rate estimates on some salmonid ESUs/DPSs by cormorants nesting at Foundation Island were substantial and were the highest of three cormorant colonies evaluated in 2020 (see *Table 11* above). Predation rate estimates were 2.5% (1.0–5.8) and 4.0% (1.2–10.7) on SR spring/summer Chinook and SR steelhead, respectively (see *Table 11* above). Predation rates on UCR steelhead and spring Chinook smolts, however, were low at less than 0.2% per ESU/DPS (see *Table 11* above). Higher predation rates on SR smolts compared with UCR smolts is consistent with data from years and has been attributed to Foundation Island cormorants disproportionately foraging in the lower Snake River compared with the middle Columbia River (Evans et al. 2016). Increased river turbidity and the greater abundance of salmonids in lower Snake River compared with the middle Columbia River are factors that may explain the increased susceptibility of SR salmonids to Foundation Island cormorant predation (Hostetter et al. 2012). Predation rates by Foundation Island cormorant in 2020 were very similar to those of years past (*Appendix, Table A7*), as were estimates of colony size (215 breeding pairs in 2020; see *Table 4* above).

Predation impacts by Foundation Island cormorants have been similar to or greater than those of nearby tern colonies (e.g., Crescent Island and the Blalock Island). For instance, predation rates by Foundation Island cormorants on SR steelhead and SR sockeye have average 4.0% (3.4–4.7) and 3.4% (2.4–4.5), respectively, compared with 4.5% (4.1–5.1) and 1.5% (1.2–2.0), respectively, by Crescent Island terns during the same time period (2007-2014; Cramer et al. 2021a; see also *Appendix, Tables A1* and *A7*). Despite similar levels of predation, however, Foundation Island cormorants were not included in management plans associated with the *IAPMP* because at the time the management plan was written, only minimum estimates of predation were available due to a lack of information on PIT tag deposition probabilities by cormorants (see also Cramer et al. 2021b).

Badger Island American White Pelicans: Estimates of on-colony PIT tag deposition probabilities for pelicans breeding on Badger Island were available for the first time in 2020 (Payton et al. *In prep*), providing the necessary input data to generate accurate estimates of predation rates for the first time since 2007, the first year the colony was scanned for fish tags. Results indicated that predation rates on SR and UCR smolts in 2020 were consistently lower than those of nearby gull and cormorant colonies, ranging from 0.1% (<0.1–0.9) in SR spring/summer Chinook to 0.3% (0.1–1.0) in UCR steelhead (*Table 12*). Like predation rate estimates at gull and cormorant colonies, small samples sizes of PIT-tagged smolts in 2020 resulted in imprecise estimate of predation for several salmonid ESUs/DPSs as denoted by wide 95% credible intervals (see *Tables 6-7* above; see *Appendix, Table A9* for sample sizes). Despite this caveat, results from 2020, where the first accurate estimates of predation are reported, indicate that pelicans pose little threat to ESA-listed UCR and SR smolts. Pelicans generally forage in shallow water less than three meters deep by dipping their bill into the water and scooping prey items (Knopf and Evans 2004). Pelicans have also been observed congregating and foraging near diversion structures such as Horn Rapids Dam on the Yakima River and in the tailrace of hydroelectric dams such as John Day Dam and The Dalles Dam, particularly during June and July when large numbers of American shad and subyearling Fall Chinook salmon are migrating (Stinson 2016; R. Cordie, USACE, personal communication). In a study of non-ESA-listed subyearling Chinook salmon, Payton et al. (2020) estimated that predation rates by pelicans nesting on Badger Island were substantial in some but not all years, with estimates as high as 10% reported in 2016. Because estimates of predation by Badger Island pelicans reported prior to 2020 were minimum estimates of predation, comparisons of pelican predation rates from 2020 to years past were not available and given evidence that fish from particular stocks may be more susceptible to pelican predation than others (e.g., sub-yearling versus yearling), additional research regarding predation by pelicans nesting at Badger Island and at other colonies in the CRB (e.g., Miller Sands Spit in the CRE) is warranted (Payton et al. 2020).

Table 12. Estimated predation rates (95% credible interval) on Snake River (SR) and Upper Columbia River (UCR) salmonid populations (ESU/DPS) by American white pelicans nesting on Badger Island during 2020. NA indicates that sample sizes of PIT-tagged smolts were too small (< 500) to generate reliable estimates (see Methods & Analysis section). Estimates of predation by pelicans on Badger Island were reported as minimum estimates in years past because estimates of PIT tag deposition probabilities were unavailable (see Collis et al. 2021b and Appendix).

ESU/DPS	Badger Island
SR Sockeye	NA
SR Sp/Su Chinook	0.1% (<0.1-0.9)
UCR Sp Chinook	0.2% (<0.1-1.3)
SR Fall Chinook	0.4% (0.1-1.6)
SR Steelhead	0.4% (0.1-4.2)
UCR Steelhead	0.3% (0.1-1.0)

Unlike terns, cormorants, and gulls, pelicans are capable of consuming adult salmonids, with PIT tags implanted in adult sockeye salmon, adult steelhead (both pre- and post-spawned), and jack Chinook salmon recovered on the Badger Island pelican colony. Adult salmonids ranging in size from 425 mm fork length to 675 mm fork length were consumed by Badger Island pelicans (Roby et al. 2017). In 2020, 19 tags from adult sockeye salmon that were tagging at the Bonneville Dam fishway were recovered on Badger Island pelican colony in 2020 (see *Table 5* above). In addition to adult sockeye, tags from other species were also recovered on Badger Island, including white sturgeon, bull trout, and pikeminnow (see also Cramer et al. 2021a for a more detailed discussion of Badger Island pelican predation impacts). The diverse diet and presence adult-sized fishes highlight the differences in diet composition between pelicans and other piscivorous waterbirds in the CRB.

Cumulative Predation & Survival

An investigation of predator-specific (tern, cormorant, pelican, and gull) and cumulative (all predator species and colonies combined) effects of avian predation on UCR steelhead indicated that predation impacts varied considerably by predator species and river reach in 2020. Of the predator species capable of foraging in the section of river between RIS and McNary Dam, the highest predation/consumption rates were those of gulls at 8.3% (95% CRI = 5.6–12.2), followed by tern colonies at 1.0% (0.6–1.5), pelicans on Badger Island (the lone pelican colony evaluated) at 0.3% (0.1–1.0), and cormorants on Foundation Island (the lone cormorant colony in this reach) at 0.1% (<0.1–0.3) (*Figure 8;* see *Tables 7-12* above for colony-specific estimates). As noted *above*, previous research indicates that cormorants nesting on Foundation Island in McNary Reservoir disproportionally forage on smolts on the lower Snake River (Evans e al. 2016), which presumably explains why predation rates on UCR steelhead were lower than that of the other salmonid ESUs/DPSs consumed by cormorants in this river reach. Cumulative predation probabilities (predation by all avian species and colonies combined) on UCR steelhead smolt during passage from RIS to McNary Dam were estimated at 9.2% (6.4–13.1) of available smolts (*Figure 8*).

Of those colonies capable of foraging on steelhead between McNary Dam and Bonneville Dam, consumption by gull colonies were again the highest of those predator species evaluated, with an estimated 5.9% (3.7–9.6) of available UCR steelhead smolts consumed (*Figure 8*). Predation by terns, cormorants, and pelicans in this river reach were significantly lower at 1.1% (0.4–1.9) for terns and < 0.1% for both cormorants and pelicans (*Figure 8*).

The measurable effects of predation by birds foraging on UCR steelhead between Bonneville Dam and the Pacific Ocean were limited to terns breeding on East Sand Island in 2020, with an estimated 3.9% (2.4–5.7) of available UCR steelhead consumed (*Figure 8*). Terns also attempted but failed to nest at two temporary, satellite colony sites along the shoreline of East Sand Island and on Rice Island in the upper CRE in 2020. Only small numbers of cormorants were observed on East Sand Island and no established colony site was identified in 2020 (K. Tidwell, USACE-FFU, personal communication). Although PIT tags were recovered from these failed nesting sites and from avian loafing/roosting sites in the estuary (see *Table 5* above; see *below* for details), estimates of predation rates by birds that briefly attempted but failed to nest in 2020 were unavailable because detection and deposition probabilities were unknown, information that is necessary to accurately estimate predation rates (see *above*). A large colony of cormorants (greater than 5,000 breeding pairs; J. Lawonn, ODFW, personal communication) successfully nested on the Astoria-Megler bridge in 2020 and smolt PIT tags were recovered at some but not all areas occupied by nesting cormorants on the bridge (see *Table 5* above). Estimates of predation by cormorants that nested on the Astoria-Megler bridge were unavailable because there were no estimates of PIT tag detection and deposition probabilities, nor were tags recovered from all areas where cormorants nested. As such, the effects of colonial waterbird predation on UCR steelhead in the CRE in 2020 were presumably much greater than the 3.9% of available UCR steelhead consumed by terns on East Sand Island, but the full impact of predation was unknown.

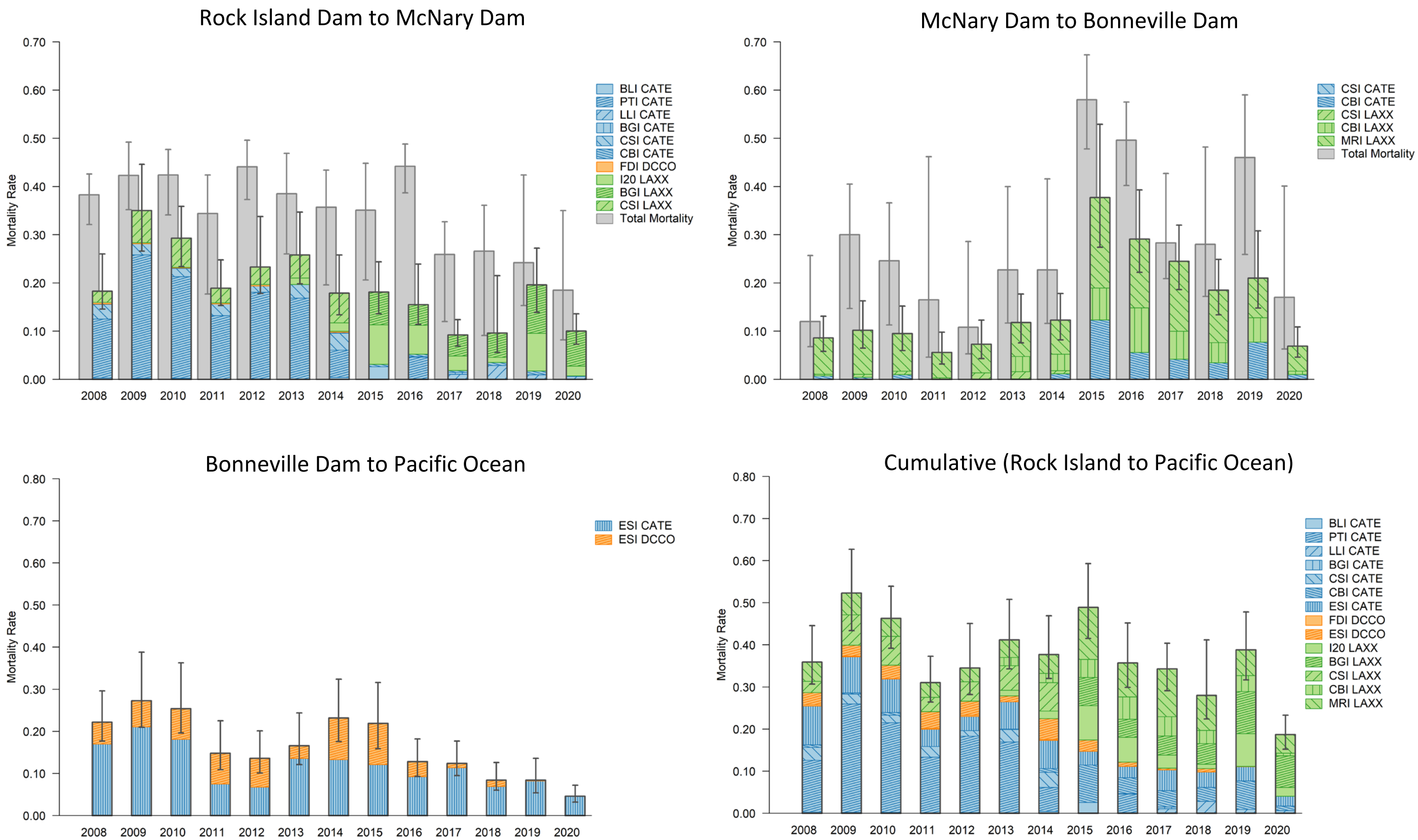


Figure 8. Estimated total mortality (grey bars) and mortality due to colonial waterbird predation (colored bars) on steelhead smolts during passage from Rock Island Dam to McNary Dam, McNary Dam to Bonneville Dam, Bonneville Dam to the Pacific Ocean, and cumulative predation from Rock Island Dam to Pacific Ocean. Colony locations include Banks Lake Island (BLI), Potholes Reservoir (PTI), Lenore Lake Island (LLI), Island 20 (I20), Foundation Island (FDI), Badger Island (BGI), Crescent Island (CSI), central Blalock Islands (CBI), Miller Rocks Island (MRI), and East Sand Island (ESI). Avian species include Caspian terns (CATE), double-crested cormorants (DCCO), and California and ring-billed gulls (LAXX). Error bars represent 95% credible intervals for total mortality and avian predation. Result from 2008-2018 are those of Evans et al. (2021).

Cumulative predation/consumption rate estimates (predation by all colonies included in the study) on UCR steelhead during smolt passage from RIS to the Pacific Ocean were substantial at 18.9% (15.0–22.8; see *Figure 8* above). Of the predator species evaluation, consumption by gulls was the greatest at an estimated 14.0% (10.3–17.9), followed by tern predation at 4.6% (3.5–5.9), pelican predation at 0.3% (0.1–1.0), and measurable cormorant predation at 0.1% (<0.1–0.3). Again, these estimates exclude multiple tern and cormorant breeding sites in the CRE in 2020. It is also important to note that the cumulative effects of predation differ significantly by salmonid species and ESA-listed salmonid ESUs/DPSs, particularly predation on salmon ESUs, where predation impacts by colonial waterbirds are often significantly lower than those of steelhead DPSs (Evans et al. 2021; see also *Tables 7-12* above). An analysis of the cumulative effects of avian predation on the survival of several; other ESA-listed salmonid ESU/DPS, including UCR spring Chinook, Snake River steelhead, Snake River spring and fall Chinook, and Snake River sockeye during 2008-2018 is presented in Evans et al. (2021).

In 2020, estimates of UCR steelhead smolt survival from RIS to McNary Dam were the highest observed since smolt tagging efforts at RIS commenced in 2008, with an estimated annual survival probability of 82.6% (66.6–92.3; see *Figure 8* above). Survival from McNary Dam to Bonneville Dam in 2020 was also amongst the highest recorded since 2008 at an estimated 73.9% (51.5–91.7) of UCR steelhead smolts (see *Figure 8* above). Cumulative survival from RIS to Bonneville Dam was also the highest recorded since 2008, with an estimated 60.0% (44.4–74.0) of UCR steelhead smolts surviving out-migration through the hydrosystem. Annual estimates of UCR steelhead survival from RIS to Bonneville Dam ranged annually from 27.2% (23.3–32.3) to 54.0% (47.9-59.5) during 2008-2019 (Payton et al. 2020). Due to lower-than-average detection probabilities of live PIT-tagged smolts passing PIT tag arrays (see also *below*) and lower recovery probabilities of dead PIT tagged fish on bird colonies in 2020 (due to reduction predation), estimates of UCR steelhead survival in 2020 were less precise than in years past, as denoted by the width of 95% credible intervals (see *Figure 8* above). Despite this uncertainty, results were still precise enough to be informative and suggest that a record high proportion of UCR steelhead survived outmigration to Bonneville Dam in 2020.

Comparisons of total steelhead smolt mortality (1-survival) to predation by colonial waterbirds indicated that predation by colonial waterbirds in 2020 accounted for 59.3% (51.6–67.1) and 60.0% (49.1-71.6) of all sources of mortality during smolt passage from RIS to McNary Dam and McNary Dam to Bonneville Dam, respectively. Comparisons from RIS to Bonneville Dam indicated that predation by colonial waterbirds in 2020 accounted for 56.1% (51.7–60.4) of all sources of morality. Due to lack of PIT tag detection sites downstream of the bird colonies on East Sand Island in the estuary, estimates of total smolt mortality from Bonneville Dam to the Pacific Ocean were not available (see *Figure 8* above). Similar to results from 2020, comparisons of total UCR steelhead smolt mortality to predation by colonial waterbirds in years past (2008-2019), indicated that predation was the one of the greatest, and in many years the single greatest, direct sources of UCR steelhead mortality during out-migration to Bonneville Dam, with predation accounting for an estimated 42% (95% CRI = 30–56%) to 70% (95% CRI = 53–87%) of all UCR steelhead smolt mortality, on average, during 2008-2019 (see *Figure 8* above;

see also Evans al 2019). Results from this and several other published studies (e.g., Evans et al. 2016, Evans et al. 2019, Payton et al. 2019) indicate that the direct impact of colonial waterbird predation on UCR steelhead smolts was greater than the direct impact of all other mortality sources combined in 11 of the last 13 years (2008-2020). For example, UCR steelhead smolt losses to piscivorous colonial waterbirds upstream of Bonneville Dam were greater than direct losses associated with passage through five hydroelectric dams (Wanapum, Priest Rapids, McNary, John Day, and Bonneville dams), predation from piscivorous fish, predation by piscivorous waterbirds species and colonies that were not included in the study, mortality from disease, and all remaining mortality factors. Results suggest that avian predation, although not the original cause of salmonid declines in the CRB, in now a factor limiting the survival of some salmonid populations that are listed under the U.S. Endangered Species Act (see also Evans et al. 2021).

Record low (since 2008) estimates of cumulative avian predation rates on UCR steelhead were coincident with record high estimates of smolt survival in all river-reaches evaluated in 2020. The greatest reduction in predation rates in 2020 were associated with the elimination of tern colonies on Goose Island in Potholes Reservoir and Crescent Island in McNary Reservoir, the two colonies managed as part of the *IAPMP* (USACE 2014, Collis et al. 2021). Terns breeding on Goose and Crescent Island forage on smolts between RIS and McNary Dam and an investigation tern predation rate estimates indicates that predation rates on UCR steelhead smolts significantly decreased during the management period at these two colonies (*Figure 9*). Correspondingly, estimated annual survival rates of UCR steelhead smolts during passaging from RIS to McNary Dam were, on average, significantly higher following reductions in predation rates by terns (*Figure 9*). Results suggest that efforts to reduce tern predation rates increases UCR steelhead smolt survival rates within the area where management actions have occurred (*Figure 9*; see also Collis et al. 2021). The reverse trends for estimated predation rates and survival rates were observed the river reach between McNary Dam and Bonneville Dam, however, where tern predation rates on steelhead smolts increased significantly and steelhead survival rates decreased significantly; these changes were attributable to the elimination of the tern colony on Goose and Crescent island and the subsequent dispersal of terns to the Blalock Islands, where terns predominately forage downstream of McNary Dam (*Figure 9*). In 2020, however, predation rates by terns breeding on the Blalock Island on UCR steelhead were the lowest recorded since management commenced in 2014 and estimates in 2020 were comparable to those observed prior to implementation of *IAPMP* during 2008–2013 at 1.1% of available steelhead (*Figure 9*). Comparisons of tern predation rates on steelhead smolts and smolt survival rates downstream of Bonneville Dam were not available due to the lack of smolt survival estimates from Bonneville Dam to the Pacific Ocean (see also Collis et al. 2021).

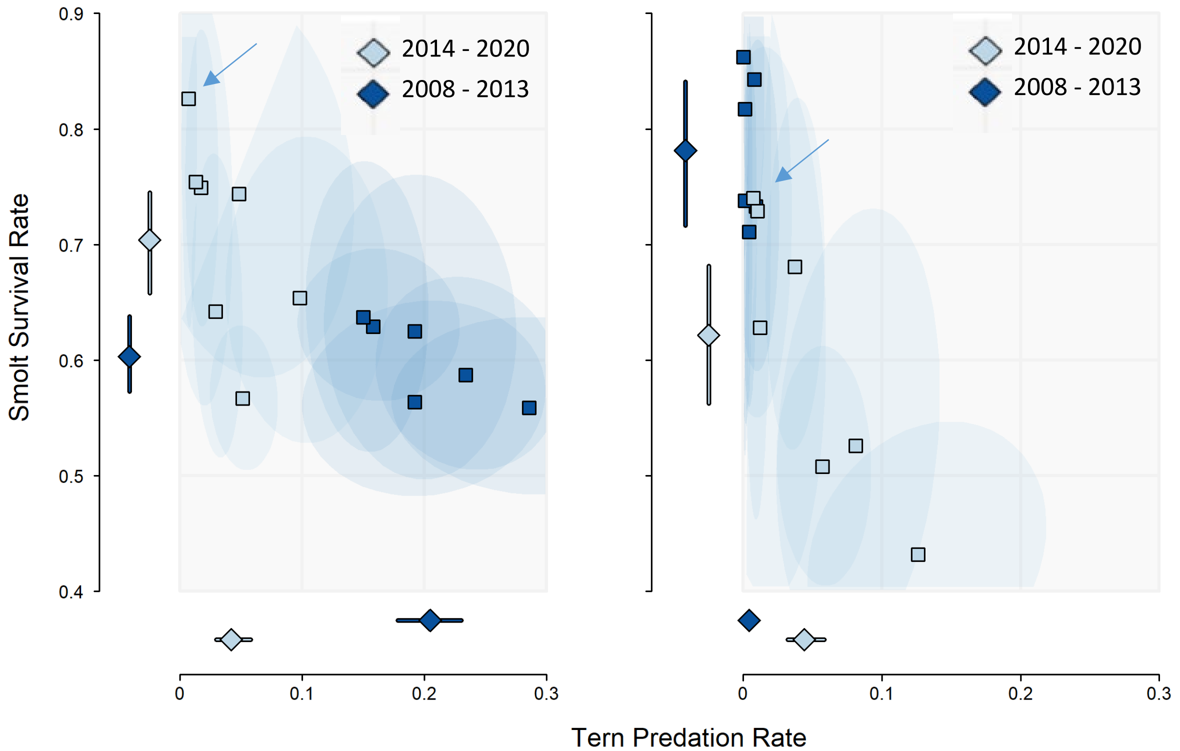


Figure 9. Caspian tern predation rates and survival rates of Upper Columbia River steelhead smolts during passage from Rock Island Dam to McNary Dam (left panel) and McNary Dam to Bonneville Dam (right panel) prior to (dark blue) and following (light blue) management actions that reduced the size of tern colonies at Goose Island and Crescent Island, colonies which forage within Reach 1. 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). Arrow indicates estimates from 2020. Results are those of Collis et al. (2021) updated with estimates from 2020.

In summary, although the cumulative effects of bird predation on UCR steelhead in 2020 remain substantial and continue to be one of greatest direct sources of smolt mortality during outmigration, predation rates in 2020 were amongst the lowest recorded since system-wide studies of predation began in 2008, particularly predation by tern colonies. Consumption rates by gull colonies were the greatest of the four predator species evaluated. Because gulls are known to consume dead, moribund, and injured smolts and are known to kleptoparasitize (steal) smolts that have been depredated by other piscivorous waterbirds, like terns, the full impact of gull consumption on smolt survival is unknown (see Additive Effects of Predation section *below*). Reductions in cumulative predation rates and, especially tern predation rates, were coincident with increases in UCR steelhead smolt survival upstream of Bonneville Dam, where estimate of both predation and survival were available (see Additive Effect of Predation section *below*).

Additive Effects of Predation

A multiple year (2008-2020) investigation of the additive effects of predation on UCR steelhead smolts indicates at increases in tern predation on UCR steelhead smolts above Bonneville Dam (BON) were associated with statistically significant decreases in smolt survival to BON (*Table 13* and *Figure 10*). Results suggest that significantly more steelhead smolts would have survived out-migration to BON in the absence of tern predation, with an estimated difference between observed survival and baseline survival (i.e. survival in the absence of tern predation) of 0.170 (0.097–0.227) across all years (*Table 13*). Results also suggest that tern predation was a super-additive source (**a** > 1.0) of steelhead smolt mortality upstream of BON (*Table 13*). Results indicated that predation by cormorants nesting on Foundation Island, the only cormorant colony upstream of BON that was included in the study, depredated a negligible proportion of available UCR steelhead smolts, with predation probabilities averaging less than 0.01 (1%) annually in those years were the cormorant colony was scanned for smolt PIT tags (*Table 13*). As such and not surprisingly due to extremely low levels of predation, there was no evidence of a statistically significant relationship between cormorant predation probabilities on UCR steelhead smolts and UCR steelhead survival probabilities to BON (*Table 13*). Similar to terns, gulls consumed a substantial proportion of UCR steelhead smolts, with an average annual consumption probability of 0.21 (0.19–0.23) of available smolts during 2013–2020 (*Table 13*). Despite such high levels of consumption by gulls, however, there was no evidence of a statistically significant relationship between gull consumption of UCR steelhead smolts and smolt survival to BON. Although the average annually point estimate of additivity due gull consumption was 0.19, indicating partially additivity, estimates of gull consumption ranged from super-additive in some years to over-compensatory (**a** < 0) in other years, with no consistent trend identified across all years. In should also be noted that weekly probabilities of smolt consumption by gulls were less variable and less precise (based on the width of 95% credible intervals) compared with those by terns, factors that make it difficult to determine to what degree predation limits survival (see also Payton et al. 2021).

Table 13. Average annual (2008-2020) predation probabilities by Caspian terns, double-crested cormorants, and California and ring-billed gulls (gulls) and survival probabilities for Upper Columbia River steelhead smolts during out-migration from Rock Island Dam to Bonneville Dam and smolt-to-adult survival probabilities from Rock Island Dam to Bonneville Dam. Estimates of the magnitude of the association between predation probabilities on survival ($\boldsymbol{a}$, additivity) and the difference in survival probabilities from estimated baseline survival probabilities ($\boldsymbol{\Phi}^{\Delta}$) are also provided. Values are reported as medians with 95% credible intervals. Statistically significant ($prob[a > 0] > 0.95$) relationships between predation and survival are in bold and denoted in red. A dashed line denotes estimates of predation were too small to investigate the relationship between predation and survival (see Methods & Analysis section). Estimates from 2008-2018 are those of Payton et al. 2021. Estimates of additivity in other salmonid species (e.g., Chinook and sockeye) and ESA-listed populations (e.g., those from the Snake River) are available in Payton et al. 2021.

Predator	Reach (life-stage)	Year(s)	Survival	Predation	a	Φ^{Δ}
Caspian terns	RIS to BON (smolt)	2008-2020	0.464 (0.440–0.484)	0.14 (0.12–0.15)	**1.35 (0.93–1.78)**	**0.170 (0.097–0.227)**
	BON to BON (SAR)	2008-2017	0.026 (0.025–0.028)	0.11 (0.10–0.13)	**0.12 (0.04–0.19)**	**0.015 (0.005–0.025)**
Double-crested cormorants	RIS to BON (smolt)	2008-2012, 2014,2020	0.452 (0.402–0.496)	< 0.01	0.39 (-1.47–2.46)	0.114 (-0.019–0.232)
	BON to BON (SAR)	2008-2015 [2]	0.030 (0.028–0.032)	0.07 (0.06–0.08)	0.05 (-0.03–0.13)	0.004 (-0.002–0.010)
California and ring-billed gulls [1]	RIS to BON (smolt)	2013-2020	0.513 (0.468–0.567)	0.21 (0.19–0.23)	0.18 (-0.73–1.00)	0.156 (-0.014–0.329)

[1] No information on gull impacts from colonies located downstream of Bonneville Dam were available as these colonies were not included in the study (see also Payton et al. 2020, 2021).

[2] No information on cormorant impacts in CRE during 2016-2020 due to colony abandonment on East Sand Island and dispersal to other colony sites in the estuary where impacts were unknown (see also Payton et al. 2020, 2021).

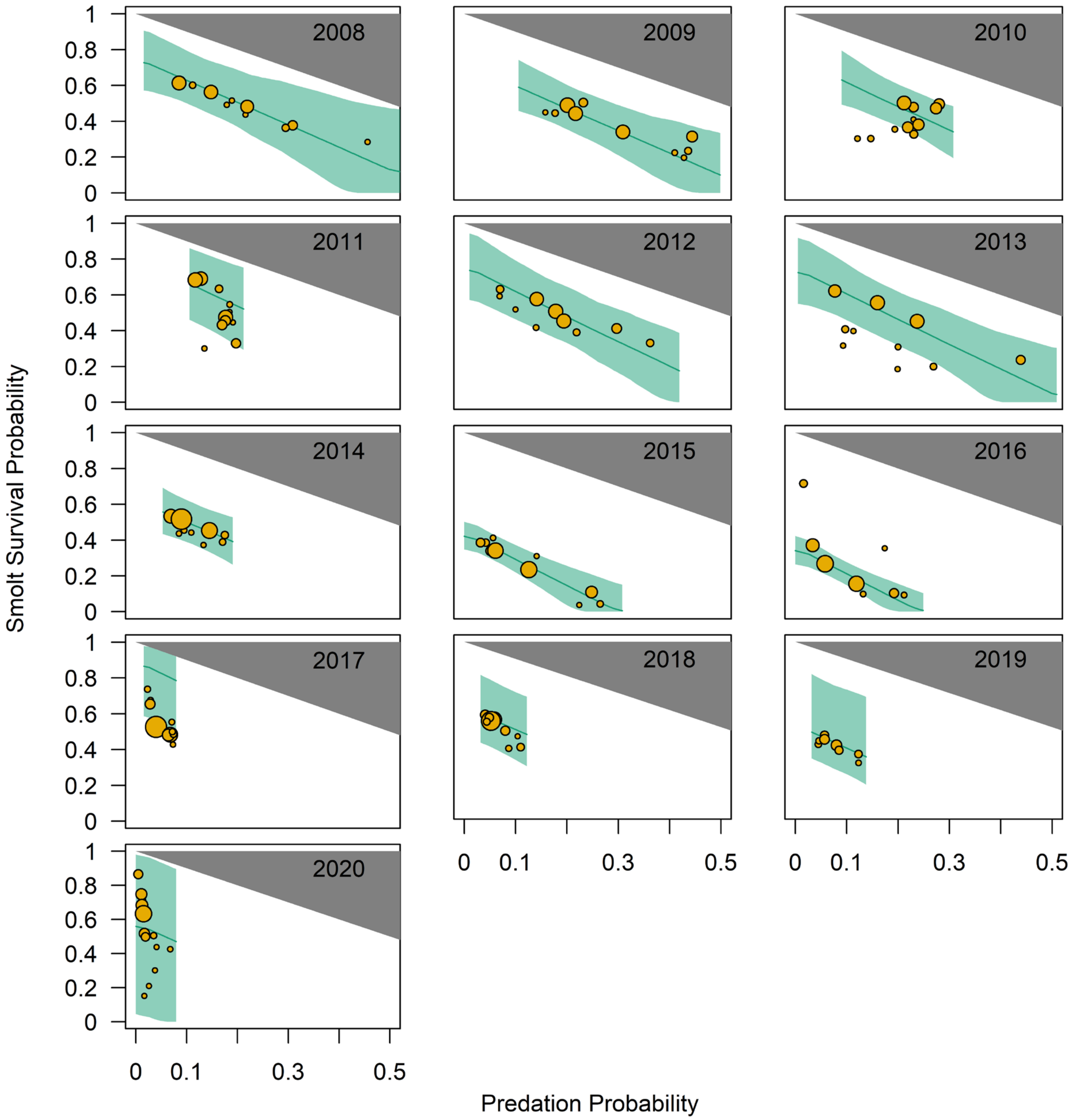


Figure 10. Weekly probability estimates of steelhead smolt survival and Caspian tern predation along with the estimated annual relationships between survival and predation during out-migration from Rock Island Dam to Bonneville Dam. The size of circles depicts relative numbers of steelhead smolts tagged and released each week at Rock Island Dam. Dashed lines represent the best fit estimate of the linear relationship and shading denotes 95% credible intervals around the best fit. Results are those Payton et al. (2021) updated with estimates from 2019 and 2020.

There was strong evidence that tern predation on UCR steelhead smolts was a partially additive source of mortality to the adult steelhead life-stage, with increases in tern predation probabilities on smolts associated with statistically significant decreases in adult returns to BON (*Figure 11* and see *Table 13* above). For instance, results suggest that in the absence of tern predation in the estuary, average annual smolt-to-adult return (SAR) probabilities could have been 0.041 (0.031–0.052) instead of 0.026 (0.025–0.026) with tern predation (see *Table 13* above). As such, although results suggest that the majority of smolts would have died before returning as adults to BON, SARs would have, on average, nearly doubled to BON in the absence of tern predation in the CRE. Results indicate that the best available science and most prudent conclusion from this and other studies (Payton et al. 2020, 2021) is that tern predation on steelhead smolts is a partially additive source of mortality to the adult steelhead life-stage (ISAB 2021). Managers, regional stakeholders, and the public at-large, however, must decide whether these potential increases SARs warrant efforts to manage terns in the estuary. Although resulted presented herein apply specifically to UCR steelhead, the same additive relationship between tern predation on smolts in the estuary and adult steelhead returns to BON was observed in Snake River steelhead (see Payton et al. 2021), suggesting the tern predation limits the survival of multiple steelhead DPSs during both the smolt and SAR life stages.

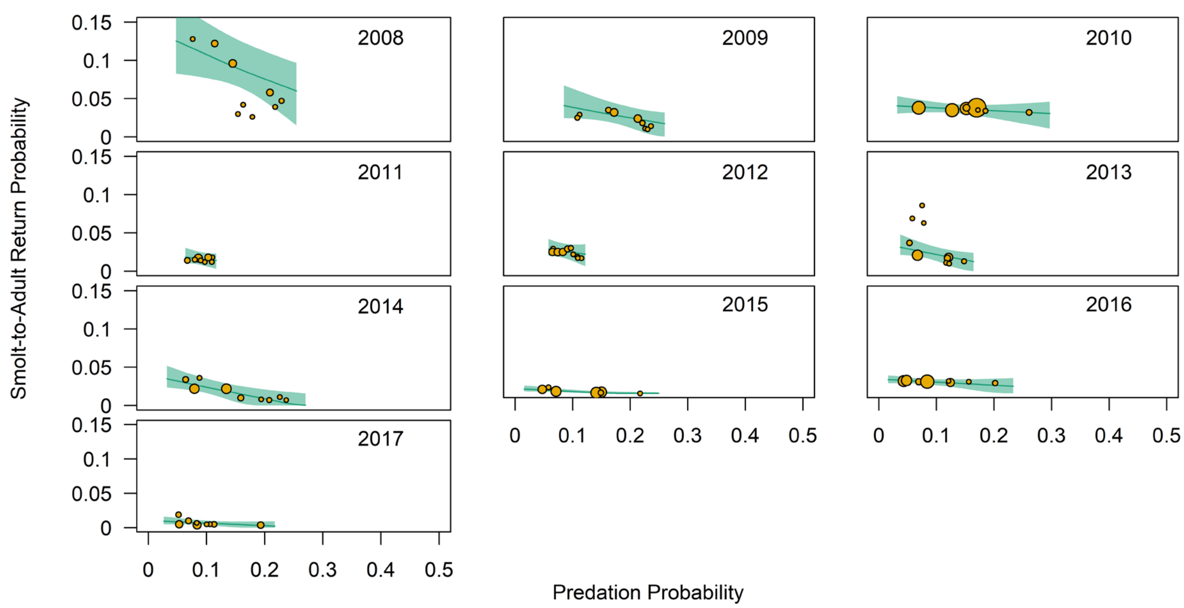


Figure 11. Weekly smolt-to-adult survival probabilities for Upper Columbia River steelhead as a function of Caspian tern predation probabilities during out-migration from Bonneville Dam as smolts to return to Bonneville Dam as adults in each year from 2008 to 2017. The size of circles depicts the relative number of PIT-tagged smolts detected at Bonneville Dam each week. Lines represent the estimate of the best linear fit to the data and shading denotes 95% credible intervals around the best fit. Results are those of Payton et al. (2021) updated with estimates from 2017, the most recent year with complete adult returns.

The average annual predation probability on UCR steelhead smolts by cormorants nesting on East Sand Island was 0.07 (0.06–0.08) of available fish during 2008-2015 (years with accurate estimates of cormorant predation and complete adult returns; see *Table 13* above) and ranged annually from 0.04 (0.02–0.06) to 0.12 (0.07–0.18). There was some evidence that cormorant predation on UCR steelhead smolts was a partially additive source of mortality to the adult life stage, with increasing weekly cormorant predation probabilities on UCR steelhead smolts associated with decreasing adult returns to BON in some, but not all, of the years evaluated. When data from all years were considered, there was not a statistically significant relationship at > 0.95 criteria level (see *Table 13* above); results would, however, be significant at the > 0.90 level. Lower levels of cormorant predation, less variable levels of predation, and high levels of total morality (1-survival) to the adult life stage, contributed to uncertainty in estimates of additivity in cormorant predation in CRE. Additional research, including the inclusion of covariates that account for density dependent relationships (Hostetter et al. 2021) and the effects of smolt arrival times in the estuary on adult returns (Payton et al. 2021), are warranted to better understand to what degree cormorant predation in the estuary limits adult salmonid returns.

As discussed in Payton et al. (2020), tern predation upstream of BON was estimated to be a super-additive source of smolt mortality (**a** > 1.0), whereby greater levels of tern predation were associated with additional indirect mortality, mortality that further reduced the survival of smolts that were not directly consumed by the tern colonies included in the study. Mechanisms of super-additivity may be related to the foraging behavior of terns, where unsuccessful depredation attempts resulted in some proportion of the prey being injured (lethally- or sub-lethally), losses that are akin to latent mortality or crippling losses in harvest management (Williams et al. 2002, Schaub and Lebreton 2004, Payton et al. 2020). In additional to crippling losses, some proportion of smolts captured by terns are stolen by gulls (i.e. kleptoparasitism) prior to the fish being consumed by the foraging tern, its mate, or its young. For example, an unknown, but possibly substantial, proportion of tagged smolts captured by terns were kleptoparasitized by gulls (Garcia et al. 2010, Patterson 2012, Adkins et al. 2014). These smolt losses, however, were not fully incorporated into tern predation probabilities (Payton et al. 2020, 2021). As such, estimates of tern predation probabilities may underestimate the full impact of all terns on smolts during out-migration, and thus exacerbating estimates of super-additivity during the smolt life stage (Payton et al. 2020). The difficulties in calculating unbiased estimates of additivity and the ambiguity associated with the interpretation of this parameter are the primary motivations to instead focus on differences in survival probabilities with and without predation ($i.e.\ \boldsymbol{\Phi}^{\Delta}$). Unlike estimate of additively, this metric provided management-relevant information with which to quantify the degree to which predation limits both smolt and SARs.

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

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

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

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

Biotic & Abiotic Factors

Results of covariate analyses are on-going and will be provided in our 2021 Annual Report to Bonneville Power Administration and Grant County PUD/PRCC. Thus far we have begun the investigation as to what covariates are most strongly associated with intra- and inter-annual variation in predation for those colonies that pose the greatest potential risk to fish survival in CRB. For many colonies, however, variation in predation is influenced by levels of previous predation upstream. For example, each year, the demographic makeup of steelhead passing RIS was altered considerably by the depredation Caspian terns nesting in Potholes Reservoir and, therefore, there is added uncertainty in what fish were available to avian predators nesting closer to McNary Reservoir. A more comprehensive model simultaneously estimating impacts is needed (i.e. a JMS-type model). Implementing the appropriate suite of covariates to such a model involve several deliberate choices and is an iterative process that are we steadily progressing through.

SMOLT SURVIVAL TO BONNEVILLE DAM

In 2020, in effort to increase samples size of PIT tagged smolts know to have survival passage to Bonneville Dam in lieu of the NMFS pair trawl, a total of 6,239 smolt PIT tags (from 2020 migration year steelhead, Chinook, coho, and sockeye; *Table 14*) were detected at avian nesting, roosting, and loafing sites in the estuary. Of these, 2,530 tags were recovered at sites that were beyond the original scope of predation studies funded by BPA in 2020. A total of 947 smolt PIT tag were passively detected at PIT tag antennas located on a floating barge (near Rkm 230) and at pile dikes (near Rkm 70), the only live detection sites available on the lower mainstem Columbia River downstream of Bonneville Dam in 2020 (*Table 14*).

Table 14. Numbers of 2020 migration year PIT-tagged juvenile salmonids detected alive passing PIT tag antennas located at a floating barge or pile dikes (recaptured) or detected dead (recovered) on avian nesting, loafing, or roosting sites located downstream of Bonneville Dam.

	Steelhead	Chinook	Coho	Sockeye
Recaptured	231	596	100	20
Recovered	3,649	2,241	245	104

The ability to generate accurate and precise estimate of UCR steelhead smolt survival depends on the number of tagged fish releases at RIS and the number subsequently recaptured alive or recovered dead on bird colonies following release at RIS. In general, larger sample sizes of released, recaptured, and recovered tags result in more accurate and precise estimates of survival (Hostetter et al. 2018). In 2020, in lieu of the NMFS pair trawl, the majority (ca. 91%) of UCR steelhead smolt tags detected at locations downstream of Bonneville Dam were on those on avian nesting, roosting, and loafing sites (n = 64) compared with live detections associated with the PIT barge and pile dike antennas (n = 6; *Table 15*). Survival analyses indicate that without detections of tags on avian colonies in the estuary, accurate and precise estimates of reach-specific and cumulative survival probabilities of UCR steelhead released at RIS would not have been possible in 2020. For instance, without avian detections, 95% credible bounds associated with estimates of steelhead survival to Bonneville Dam were uninformative and ranged from 0.28 to 1.0 (i.e. 28 to 100%), but with avian detections, credible bounds were informative, ranging from 0.44 to 0.74 (i.e. 44 to 74%; *Figure 12*). The inclusion of avian detections at colonies upstream of Bonneville Dam also increased the precision and accuracy of smolt survival estimates to Bonneville Dam in 2020 (*Figure 12*).

Table 15 Weekly numbers (N) of PIT-tagged juvenile steelhead released at Rock Island Dam (RIS) that were subsequently detected alive (recaptured) passing McNary Dam (MCN), John Day Dam (JDA), Bonneville Dam (BON), and PIT detections sites at barge and pile dike located down-stream (post) of BON and the number detected dead (recovery) on waterbirds colonies. Weeks are those of the Julian calendar.

		Recapture				Recovery			
Week	N	@ MCN	@ JDA	@ BON	Post-BON	RIS-MCN	MCN-JDA	JDA-BON	Post-BON
15	10	0	0	2	0	0	0	0	0
16	206	1	7	35	0	3	0	0	2
17	678	9	40	117	0	6	0	0	1
18	1,019	16	39	118	1	11	2	4	6
19	1,146	24	85	104	1	19	2	4	7
20	1,967	15	174	182	3	18	8	12	27
21	915	14	44	58	1	19	3	5	12
22	598	10	39	43	0	4	2	8	8
23	234	2	8	18	0	2	3	1	0
24	62	1	0	4	0	0	3	0	1
25	5	0	0	0	0	0	0	0	0
26	1	0	0	0	0	0	0	0	0
29	2	0	0	0	0	0	0	0	0
Total	**6,843**	**92**	**436**	**681**	**6**	**82**	**23**	**34**	**64**

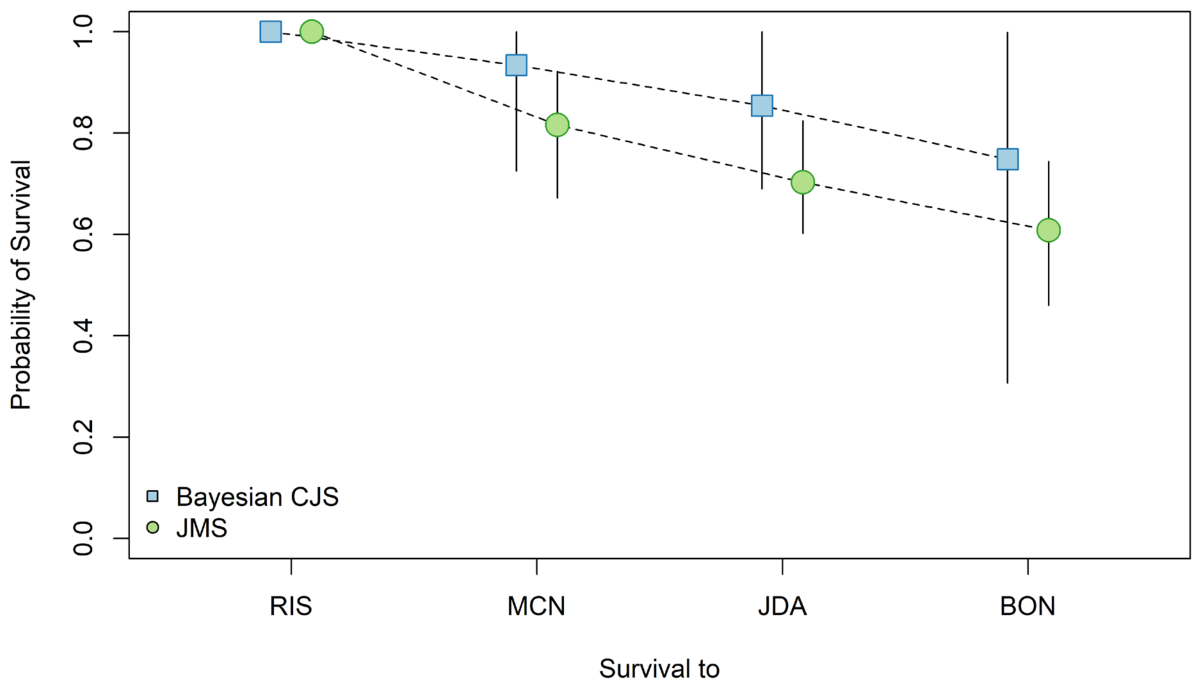


Figure 12. Cumulative migration survival probabilities (95% credible interval) for juvenile steelhead using mark–recapture (CJS; square) or mark–recapture–recovery (circle) models. Cumulative survival estimates are from PIT-tagging and release at Rock Island Dam to McNary Dam (MCN), John Day Dam (JDA), and Bonneville Dam (BON).

MANAGEMENT RECOMMENDATIONS

Based on results from 2020 and those from previous years (Roby et al. 2021), we provide the following management recommendations to maximize the benefits to ESA-listed juvenile salmonids associated with management of piscivorous colonial waterbirds in the CRB:

1. Terns continued to show high fidelity to managed colony sites in the CRB in 2020 (i.e. East Sand Island and Rice Island in the CRE and Goose Island in the CPR). Continued efforts to dissuade terns from nesting at satellite colony sites on East Sand Island or at the former colony site on Rice Island will be necessary to achieve management objectives outlined in the *Estuary Tern Management Plan*. Continued efforts to dissuade terns from nesting on Goose Island in Potholes will also likely be required to prevent tern colony formation there. Nest dissuasion strategies should include multiple techniques, both passive and active, and should focus on implementing these strategies early in the breeding season, particularly during the peak of the smolt outmigration period (i.e. April–June).

2. After a series of nesting seasons when the tern colony on East Sand Island either produced very few young or completely failed, colony size dropped below the target colony size (3,125–4,375 breeding pairs) for the first time in 2020. Nesting conditions for terns on East Sand Island have apparently declined (e.g., high rates of disturbance and predation by eagles and gulls, poor quality of nesting substrate that does not drain well after a heavy rainstorm) to such an extent that the colony is at risk of being abandoned. If the tern colony on the designated 1-acre colony site on East Sand Island were abandoned, it would place much greater pressure on Rice Island and satellite colony sites on East Sand Island as alternative colony sites. We recommend that lethal control of predatory gulls and nesting habitat improvements be carried out at the designated colony site on East Sand Island so that tern colony size is within the target size range specified in the *Estuary Tern Management Plan* and nesting success is sufficient to ensure the colony is not abandoned.

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

sizes do not exceed those stipulated in the *Plan*; and (3) recovery of smolt PIT tags to estimate predation rates on juvenile salmonids by terns using previously established methods at all active and incipient colony sites in the estuary to ensure impacts do not exceed acceptable levels.

4. Although terns did not attempt to nest on Crescent Island in 2020, a moderate-sized gull colony became established on Crescent Island near the site of the former tern colony. This is the first year since management was initiated on Crescent Island that gulls have successfully nested there. The presence of a gull colony on Crescent Island will likely enhance the prospects that terns will again nest on the island in future years, both because nesting gulls will likely suppress vegetation growth at their colony site, creating more open nesting habitat for terns, and because nesting gulls will attract terns to nest at the site because they are nesting associates. Additional nest dissuasion, both passive and active, are needed on Crescent Island to prevent formation of a new tern colony.

5. Although predation on juvenile salmonids by terns at managed colonies in the CPR has been nearly eliminated, predation rates on Snake River steelhead by terns nesting at the unmanaged colony in the Blalock Islands continues to exceed the 2% threshold specified in the *IAPMP*. Adaptive management by way of colony size reductions at the Blalock Islands tern colony is needed to reduce Caspian tern predation rates to levels specified in the *Plan*.

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

7. An investigation of the cumulative effects of avian predation indicates that smolt consumption by gulls, primarily California gulls, nesting at several unmanaged colonies is a source of substantial mortality of UCR steelhead during outmigration to Bonneville Dam. Research to better understand factors that influence smolt susceptibility to gull consumption and to quantify to what degree gull consumption limits smolt survival (i.e. is an additive source of mortality) are needed. Management actions to reduce the size of specific gull colonies in the CPR (e.g., Miller Rocks and Badger Island) may warrant consideration.

8. In 2020, based on monitoring conducted by USACE, cormorants did not attempt to nest on East Sand Island. Based on monitoring conducted by ODFW, the size of the double-crested cormorant colony on the Astoria-Megler Bridge was estimated to be 5,080 breeding pairs, which represents the largest colony size for cormorants on the bridge

ever recorded and was the sixth consecutive year of growth of that colony since cormorant management in the estuary began in 2015. It is likely that the relocation of nesting cormorants from the colony on East Sand Island to the Astoria-Megler Bridge has increased the predation impacts of cormorants on ESA-listed juvenile salmonids in the CRE. Additional work is needed to estimate predation rates on smolts by cormorants nesting on the bridge, and to devise a plan to reduce those impacts by relocating the bridge colony back to East Sand Island and to other colony sites outside the CRB.

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

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APPENDIX

Table A1. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on Crescent Island and Badger Island in McNary Reservoir, Goose Island in Potholes Reservoir, an unnamed island in Potholes Reservoir, an unnamed island 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 2007-2019. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimated from 2000-2018 are those previously reported by Roby et al. (2021).

			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
			Badger Island, McNary Reservoir			
Year	SR	SR	UCR	SR	SR	UCR

	Sp/Su Chinook	Fall Chinook	Sp Chinook	Sockeye	Steelhead	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
			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
			Unnamed Island, Northern Potholes Reservoir			
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2016	<0.1%	<0.1%	0.1% (<0.1-0.3)	<0.1%	<0.1%	4.1% (2.9-6.3)
N	38,633	5,461	1,956	522	20,729	7,003
			Lenore Lake Island, 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

			Banks Lake Island, Banks Lake			
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
2008	<0.1%	<0.1%	-	<0.1%	<0.1%	<0.1%
N	30,142	22,968		767	28,653	8,403
2009	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)
N	20,679	26,567	738	2,651	52,102	8,025
2010	<0.1%	<0.1%	<0.1%	0.2% (<0.1-1.2)	<0.1%	0.1% (<0.1-0.3)
N	5,790	28,067	929	566	7,913	8,382
2012	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.3)
N	41,258	24,772	1,812	2,884	25,841	6,845
2014	<0.1%	<0.1%	0.5% (<0.1-7.9)	0.1% (<0.1-0.6)	<0.1%	1.2% (0.3-6.4)
N	22,195	6,043	641	1,414	16,599	7,757
2015	<0.1%	<0.1%	0.2% (<0.1-0.9)	0.1% (0-0.5)	<0.1%	2.6% (1.8-3.9)
N	7,706	3,449	766	1,262	3,601	7,222
2016	<0.1%	<0.1%	<0.1%	<0.1%	<0.1%	0.1% (<0.1-0.2)
N	38,633	5,461	1,956	522	20,729	7,003

Table A2. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on the Blalock Islands on ESA-listed salmonid populations originating from the Snake River (SR) and Upper Columbia River (UCR) based on detections at McNary Dam during 2007-2019. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimated from 2000-2018 are those previously reported by Roby et al. (2021).

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

Table A3. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by Caspian terns nesting on East Sand Island on ESA-listed salmonid populations originating from the Snake River (SR; based on detection at Bonneville Dam), Upper Columbia River (UCR; based on detections at Bonneville Dam), middle Columbia River (MCR; based on detection at Bonneville Dam), and Upper Willamette River (WR; based on detections at Sullivan Dam) during 1999-2020. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimated from 2000-2018 are those previously reported by Roby et al. (2021). Estimates from 2020 are those reported by Evans et al. (2021). Accurate and comparable estimates of predation were not available (NA) in 2019.

	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

Table A4. Number of available PIT-tagged smolts (N) and annual predation rates (95% credibility intervals) by double-crested cormorants nesting at locations in the Columbia River estuary 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 1999-2018. 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. There was no established double-crested cormorant colony (NC) on East Sand Island in 2020. Estimated from 2000-2018 are those previously reported by Roby et al. (2021).

	East Sand Island, Columbia River Estuary							
Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
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*	NC	NC	NC	NC	NC	NC	NC	NC

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 2007-2020. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimated from 2000-2019 are those reported by Roby et al. (2021).

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

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
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)
sN	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
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

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 Central 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 2007-2019. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. Estimated from 2000-2018 are those previously reported by Roby et al. (2021).

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

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
Central 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

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, an unnamed island in Hanford Reach of the middle Columbia River an unnamed island in Lenore Lake, and in Northern Potholes Reservoir on ESA-listed salmonid populations originating from the Snake River (SR; based on detections at Lower Monumental Dam) and Upper Columbia River (UCR; based on detections at Rock Island Dam) during 2007-2019. A dashed line indicates that sample sizes of PIT-tagged smolts interrogated passing dams were too small (< 500) to generate reliable predation rates. The Foundation Island colony was active in 2013 and during 2015-2019 but was not scanned for smolt PIT tags so estimate in those years are unavailable. Estimated from 2000-2018 are those previously reported by Roby et al. (2021).

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
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

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	SR Sockeye	SR Steelhead	UCR Steelhead
			Lenore Lake Island, Lenore Lake			
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