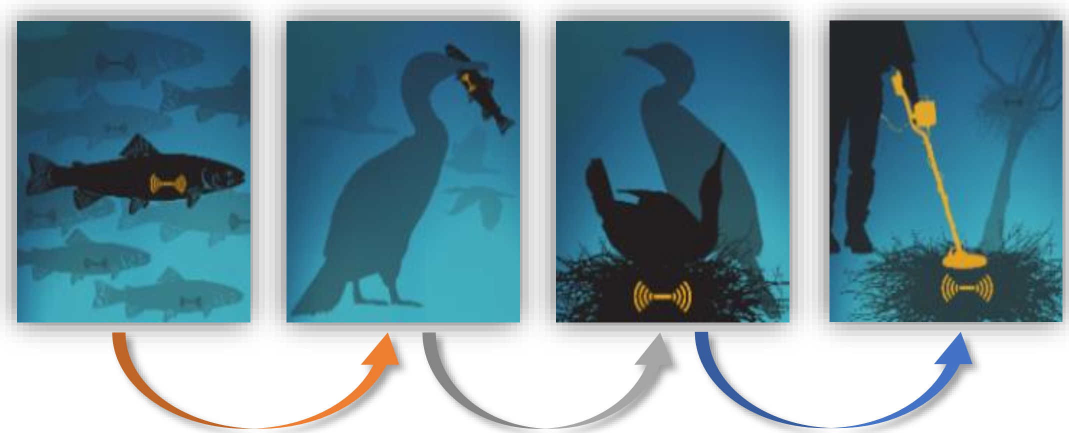
FINAL TECHNICAL REPORT: East Sand Island Passive Integrated Transponder Tag Recovery and Avian Predation Rate Analysis, 2018



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East Sand Island Passive Integrated Transponder Tag Recovery and Avian Predation Rate Analysis, 2018

Final Technical Report

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EXECUTIVE SUMMARY

To address concerns of avian predation on Endangered Species Act (ESA) listed juvenile salmonids (*Oncorhynchus spp.*) in the Columbia River Estuary, management plans have been developed to reduce the number of Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus*) nesting on East Sand Island. The primary goal of this study was to provide the U.S. Army Corps of Engineers (USACE) and other regional stakeholders with information to evaluate the effectiveness of management plans in reducing predation rates (percentage of available fish consumed) on ESA-listed juvenile salmonids by terns and cormorants nesting on East Sand Island in 2018. The primary tasks were to (1) recover juvenile salmonid passive integrated transponder (PIT) tags from the tern and cormorant colonies on East Sand Island and (2) use those data to model predation rates. More specifically, we generated population-specific (salmonid evolutionary significant units [ESU] or distinct population segments [DPS]) predation rates on juvenile salmonids that integrated multiple factors of uncertainty in the tag recovery process, including imperfect detection of tags on bird colonies, on-colony tag deposition probabilities that varied by bird species (tern, cormorant), and temporal changes in fish availability to avian predators nesting on East Sand Island. To ensure relative comparability of predation rate results collected in 2018 to years past, we use the tag recovery and analytical methods of Evans et al. (2012) and Hostetter et al. (2015), previously peer-reviewed methods that allow for direct comparisons of predation rates among predator species, salmonid ESUs/DPSs, and years. To facilitate the review of data collected in 2018 relative to previous years, the organization and general content of this report is similar to that of past reports (see Evans et al. 2016a, Evans et al. 2018).

PIT Tag Recovery

Following the 2018 nesting season, a total of 10,886 and 2,680 PIT tags from 2018 migration year smolts (Chinook salmon [*O. tshawytscha*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*], and steelhead trout [*O. mykiss*] tags combined) were recovered on the East Sand Island Caspian tern and double-crested cormorant colonies, respectively. PIT tags were detected by systematically scanning the entire area (referred to as a “pass”) occupied by nesting birds during the breeding season, with a total of five passes conducted on the tern colony and four passes on the cormorant colony. Average annual detection efficiency (proportion of deposited tags detected by researchers after the breeding season) was estimated at 77% (seasonal range = 57–97%) on the tern colony and 88% (seasonal range = 76–100%) on the cormorant colony, amongst the highest rates of detection ever recorded on East Sand Island. All newly detected PIT tags recovered on the tern and cormorant colonies were uploaded to the PIT Tag Information System (PTAGIS) on 2 December 2018, so that the data was readily available to other researchers, managers, and the general public.

Predation Rates

Caspian terns – Predation rates on juvenile salmonids by Caspian terns nesting on East Sand Island in 2018 were amongst the lowest ever recorded, with rates on most ESA-listed ESUs/DPSs significantly lower than in years past. Predation rates on salmon ESUs in 2018 ranged from 1.3% (95% credible interval [CRI] = 0.7–2.1%) on Snake River Fall Chinook to 4.2% (95% CRI = 2.9–6.4%) on Snake River sockeye. Predation rates on steelhead DPSs in 2018 ranged from 5.3% (95% CRI = 3.8–8.0%) on Middle Columbia River steelhead to 6.9% (95% CRI = 5.3–10.2%) on Snake River steelhead. By comparison, predation rates averaged 14.9% (95% CRI = 13.1–17.6%) and 22.2% (95% CRI = 20.3–24.8%) for Middle and Snake River steelhead DPSs, respectively, prior to reductions in the size of the tern colony on East Sand Island associated with management.

Reductions in Caspian tern predation rates in 2018 were commensurate with reductions in tern colony size, indicating that tern management actions to reduce the number of terns on East Sand Island are resulting in lower rates of predation on juvenile salmonids by this colony. The East Sand Island colony has been reduced from an average of 9,221 breeding pairs (range = 8,283–10,668 pairs) prior to management (2000-2010) to 5,957 breeding pairs (annual range = 3,500–7,387 pairs) following management (2011-2018). The 2018 colony size of 4,959 breeding pairs is approaching, but has not reached, the target colony size goal of 3,125 breeding pairs (USFWS 2005). More importantly from a fish conservation perspective, predation rates on salmonid ESUs/DPSs by East Sand Island Caspian terns are now significantly lower than those observed prior to management, with an approximately one-half reduction in steelhead predation rates observed since management was implemented and approaching the two-thirds reduction stated in the tern management plan (USFWS 2005). 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 on Rice Island in the upper Columbia River Estuary, birds presumably displaced from East Sand Island as a result of management. The impact on smolts from terns that have attempted but were prevented from nesting on Rice Island partially off-set, to an unknown degree, reductions in predation rates by terns nesting on East Sand Island. To fully evaluate the efficacy of the Caspian Tern Management Plan for reducing predation rates on ESA-listed smolts throughout the Columbia River Estuary, an investigation of cumulative predation rates by all Caspian terns – those on East Sand Island and Rice Island – in the estuary would be necessary but was beyond the scope of this study.

An investigation of East Sand Island Caspian tern predation rates based on a fish's ESU/DPS, rear-type (hatchery, wild), outmigration history (in-river, transported), and abundance (density) indicated that multiple factors influence smolt susceptibility to tern predation. A relative comparison of predation impacts from 2018 and from years past (2006-2017) indicated that predation rates on steelhead DPSs were significantly higher than those of salmon ESUs. There was also evidence that hatchery spring Chinook salmon from both the Upper Columbia River and Snake River ESUs were more susceptible to tern predation than their wild counterparts, while there were no statistically credible differences in susceptibility by rear-type (wild, hatchery) for steelhead DPSs. There was some evidence that fish with an in-river migration history were more susceptible to Caspian tern predation on East Sand Island than those that were transported, although differences in susceptibility between in-river and transported fish were generally small and inconsistent across the study period. Weekly differences in smolt susceptibility based on the relative abundance of PIT-tagged smolts in the estuary were also observed, with tern predation rates generally decreasing as the number of steelhead smolts in the estuary increased.

Double-crested cormorants – ESU/DPS-specific predation rates by double-crested cormorants in 2018 were amongst the lowest ever recorded on East Sand Island, ranging from just 0.4% (95% CRI = 0.1–1.0%) of Middle Columbia River steelhead to 0.9% (95% CRI = 0.5–1.9%) of Snake River sockeye. The relatively small colony (3,672 nesting pairs), coupled with a late or right-shifted nesting chronology for East Sand Island cormorants in 2018, explained the record low rates of predation observed in 2018. Predation rates in 2018 were significantly lower than those observed prior to management actions for all ESUs/DPSs evaluated. For instance, prior to management, predation rates on Snake River spring/summer Chinook salmon and Snake River steelhead averaged 5.2% (95% CRI = 4.4–1.9%) and 9.3% (95% CRI = 8.0–11.0%), respectively, during 2003-2015. The East Sand Island cormorant colony has been reduced from an average of 12,744 breeding pairs (range = 10,646–14,916 pairs) prior to management actions that reduced peak colony size (2003-2015) to 4,663 breeding pairs (annual range = 544–9,771 pairs) following management actions that reduce peak colony size (2016-2018). The goal of the Double-crested Cormorant Management Plan was to reduce the peak size of the East Sand Island colony to no more than approximately 5,600 nesting pairs, and to limit steelhead predation rates to approximately 3% or less per DPS (USACE 2015), goals that were achieved in 2018.

Prior to 2016, when the management plan was first implemented, the vast majority of double-crested cormorants in the Columbia River Estuary nested on East Sand Island, allowing for a holistic evaluation of predation rates based on recoveries of smolt PIT tags at just that single colony. Starting in 2016, however, cormorants have not successfully established a nesting colony on East Sand Island throughout the entire peak of the smolt outmigration period (April to June). Instead, large numbers of birds have dispersed from East Sand Island to other locations, some of which are within the Columbia River Estuary (e.g., Astoria-Megler Bridge). As such, analogous to the presence of Caspian terns on Rice Island, future predation rate monitoring and evaluation studies may need to consider the cumulative impact of all cormorants in the estuary – those on East Sand Island and the Astoria-Megler Bridge – on smolts to fully evaluate the efficacy of management plans to reduce predation rates on ESA-listed salmonid ESUs/DPSs.

A multiyear (2006-2018) investigation of East Sand Island double-crested cormorant predation impacts indicated significant annual variation in predation rates, with differences in predation rates often greater between years within the same ESU/DPS than differences between ESUs/DPSs in the same year. An investigation of smolt abundance and predation rates indicated that, unlike Caspian terns, predation rates increased as the number of prey available in the Columbia River Estuary increased. Also, unlike Caspian terns, there was no consistent evidence that hatchery fish or in-river migrants were more or less susceptible to cormorant predation, suggesting that cormorants indiscriminately consumed juvenile salmonids in the estuary. Collectively, result suggest that a different suite of mechanisms may regulate juvenile salmonid susceptibly to double-crested cormorant predation compared with Caspian tern predation. Taken together, results indicate that predator-prey interactions in the Columbia River Estuary are dynamic and that multiple factors were associated with variation in avian predation rates on juvenile salmonids.

BACKGROUND

Avian predation has been identified as a factor that limits the survival of ESA-listed juvenile salmonids in the Columbia River Basin (NOAA 2008, 2010). Previous research has demonstrated that Caspian terns and double-crested cormorants nesting on East Sand Island in the Columbia River Estuary consume millions of juvenile salmonids annually (Roby et al. 2003; Lyons 2010). An evaluation of avian predation rates (proportion of available fish consumed) revealed that cormorants and terns nesting on East Sand Island consumed upwards of 10% and 20% of some ESA-listed Chinook and steelhead populations, respectively (Evans et al. 2016a). Predation losses in the Columbia River Estuary also affect juvenile salmonids belonging to every Evolutionary Significant Unit (ESU) and Distinct Population Segment (DPS) of salmonid from the Columbia River Basin, fish that have survived freshwater migration through the Federal Columbia River Power System (FCRPS) and have a higher probability of survival to adulthood compared to those fish that have yet to complete outmigration (Roby et al. 2003).

While levels of tern and cormorant predation on some populations of juvenile salmonids have been high on average, there has also been substantial intra- and inter-annual variability in predation rates. For instance, predation rates on the same salmonid ESU/DPS can vary significantly by year (Evans et al. 2012; Sebring et al. 2013) and by week within the same year (Evans et al. 2016a). Past studies indicate that differences in colony size (number of breeding pairs) explain variation in predation rates, with decreases in colony size associated with a decrease in predation rates. In addition to colony size, other factors have also been linked to variability in predation rates, including a fish's rear-type (hatchery, wild), outmigration history (e.g., transported from the Snake River), and abundance (Ryan et al. 2003; Hostetter et al. 2012; Lyons et al. 2014a; Evans et al. 2016a). Collectively, results from these studies indicate that predation by Caspian terns and double-crested cormorants is not only a substantial source of smolt mortality, but also that predator-prey interactions are dynamic and may vary based on different biotic and abiotic conditions in the Columbia River Estuary (see Lyon et al. 2014a and Evans et al. 2016a for a more detailed review of biotic and abiotic factors known to influence fish susceptibility to avian predation in the estuary).

To address concerns about the impact of avian predation on ESA-listed juvenile salmonids in the Columbia River Estuary, two management plans are currently underway, entitled "*Caspian Tern Management to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary"* (USFWS 2005, 2006) and "*Double-crested Cormorant Management to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary"* (USACE 2015). These management plans aim to reduce the number of Caspian terns and double-crested cormorants nesting on East Sand Island to reduce predation rates and, ultimately, increase the survival of juvenile salmonids migrating through the estuary. Efforts to reduce colony sizes have been primarily through lethal (i.e., culling and egg oiling) and non-lethal (i.e., passive and active nest dissuasion) strategies for double-crested cormorants and non-lethal strategies for Caspian terns. Management plans were developed in response to Reasonable and Prudent Alternatives (RPAs) specified in Biological Opinions on the operation of the FCRPS issued by NOAA Fisheries (NOAA 2008, 2010, 2014a). RPAs 66 and 67 specify annual monitoring of juvenile salmonid predation impacts by Caspian terns and double-crested cormorants in the Columbia River Estuary. More specifically, management plans require that salmonid PIT tags be recovered on the East Sand Island tern and cormorant colonies after the breeding season to document annual trends in predation rates (USACE 2015). Management plans also specify that annual changes in predation rates be used to evaluate the efficacy of the management actions to reduce predation by terns and cormorant in the Columbia River Estuary (USFWS 2005; USACE 2015).

To address the monitoring requirements of these two management plans, two tasks were conducted as part of this study in 2018: (1) recovery of smolt PIT tags on the Caspian tern and double-crested cormorant colonies on East Sand Island and use of those data to (2) estimate predation rates on ESA-listed salmonid populations. Predation rates from 2018 were then compared with predation rate estimates from years past to evaluate the effectiveness of management actions in reducing predation rates through reductions in the colony size of Caspian terns and double-crested cormorants on East Sand Island. To ensure the relative comparability of predation rate results collected in 2018 to years past, we used the PIT tag recovery and analytical methods of Evans et al. (2012) and Hostetter et al. (2015), previously peer-reviewed methods that allow for direct comparisons of predation rates among predator species (terns, cormorants), salmonid ESUs/DPSs, and years. Finally, given that the reporting requirements associated with this study were the same as those of years past, and to facilitate review of results from 2018 to those of years past, the organization and general content of this report is similar to past reports (see Evans et al. 2016a and Evans et al. 2018).

METHODS

PIT Tag Recovery

We used the previously established methods of Evans et al. (2012, 2016a) to recover (detect) PIT tags on the East Sand Island Caspian tern and double-crested cormorant colonies in 2018. Below is a summary of those methods by colony site:

East Sand Island Caspian tern colony – Data from aerial and ground-surveys conducted during the breeding season were used to determine where Caspian terns nested on East Sand Island in 2018 (*Figure 2*). A custom built eight-coil flat-plate PIT tag detection system attached to an all-terrain vehicle (ATV) was then used to detect PIT tags *in situ* on the East Sand Island Caspian tern colony after nesting birds dispersed from the colony in September of 2018. PIT tags were detected by systematically scanning the entire area (referred to as a "pass") occupied by nesting terns during the breeding season (*Figure 1*). Additional passes were conducted until the number of newly identified, previously undetected tags were less than 5% of the total number found during all previous passes, which resulted in five passes of the tern colony in 2018. Passes were conducted in varying directions, a technique that results in higher detection efficiency (Ryan et al. 2003), and at consistent speed and antenna height to optimize antenna performance (see Evans et al. 2016a for additional details). Hand-held PIT tag detection systems (*Biomark*, model HPR) were also used to detect PIT tags in areas inaccessible to the ATV (e.g., areas adjacent to dissuasion fencing and vegetated habitat; see *Figure 2*). PIT tag transceivers were optimized to detect ISO FDXB tags, the most common type of PIT tag implanted in juvenile salmonids from the Columbia River Basin in 2018 (PSFMC 2018).

In addition to electronic detection of PIT tags using the flat-plate and hand-held antenna systems, PIT tags were also physically removed from the Caspian tern colony using a tow behind sweeper magnet (*Bluestreak,* model Hog series) attached to the ATV (*Figure 1*). The physical removal of PIT tags reduces tag collision, a phenomenon that renders PIT tags in close proximity to each other undetectable using electronics. The physical removal of PIT tags, and subsequent hand scanning of each tag to acquire its unique code, increases tag detections at sites like the East Sand Island tern colony where tag densities are very high (Evans et al. 2016a). Both physical and electronic PIT tag recovery were conducted concurrently in 2018, when conditions permitted (i.e., use of magnet requires dry substrate).

PIT tag codes stored locally on the flat-plate system's transceiver were uploaded to a central storage drive at the completion of each scanning session, along with metadata regarding the date and pass number. After each scanning day, tag data were uploaded to a cloud-based server for redundancy. Following validation and removal of duplicate records, newly detected tag codes, including codes from tags physically removed with the sweeper magnet, were uploaded to PTAGIS on 2 December 2018, using guidelines and protocols established by the PIT-tag Steering Committee (PSFMC 2018). Tag codes can be downloaded directly from PTAGIS as Raw Tagging Files, under the APD (Avian Predation Detection) Directory (PSMFC 2018).

<u>*East Sand Island double-crested cormorant colony*</u>: Data from aerial and ground-surveys conducted during the breeding season were used to determine where double-crested cormorants nested on East Sand Island in 2018 (*Figure 2*). Hand-held PIT tag detection systems (*Biomark*, model HPR) were then used to detect PIT tags *in situ* on the East Sand Island cormorant colony after nesting birds dispersed from the island following the breeding season in October of 2018. Analogous to scanning the tern colony, PIT tags were recovered by systematically scanning the entire area occupied by nesting

cormorants during the 2018 breeding season with additional passes conducted until the number of newly identified previously undetected tags were less than 5% of the total number found during all previous passes, which resulted in a total of four passes of the cormorant colony in 2018. In addition to scanning areas occupied by double-crested cormorants, we also independently scanned nesting areas exclusively used by Brandt's cormorants (*P. penicillatus*) during the breeding season, an unmanaged predator species on East Sand Island. This was necessary because Brandt's cormorants nested adjacent to double-crested cormorants and efforts to delineate tags deposited by the two species were needed to minimize potential bias in predation rate estimates from double-crested cormorants (i.e., erroneously attributing tags consumed by Brandt's cormorants to those of double-crested cormorants; see Evans et al. 2016a for additional details).

PIT tag codes stored locally on each transceiver were uploaded to a central storage drive at the completion of each scanning session, along with metadata regarding the scan date, species (double-crested, Brandt's), and pass number. After each scanning day, tag data were uploaded to a cloud-based server for redundancy. Following validation and removal of duplicate records, newly detected tag codes were uploaded to PTAGIS on 2 December 2018. Tag codes can be downloaded directly from PTAGIS as Raw Tagging Files, under the APD (Avian Predation Detection) Directory (PSMFC 2018).

Predation Rate Calculations

Following previously established methods (Hostetter et al. 2015), a Bayesian hierarchical model was used to estimate predation rates based on recoveries of smolt PIT tags on the East Sand Island Caspian tern and double-crested cormorant colonies in 2018. Predation rate estimates were derived using the proportion of juvenile salmonid PIT tags found on each bird colony from the available population of PIT-tagged fish (i.e., smolt availability), and then adjusting by the probability that a consumed PIT tag was subsequently deposited on that colony (i.e., deposition probability) and later detected by researchers following the nesting season (i.e., detection probability; *Figure 3*).

The predation rate model used in 2018 was the same model used to estimate predation rates on smolts by East Sand Island Caspian terns and double-crested cormorants in years past and the same model used in the Affected Environment Analysis of the *Double-crested Cormorant Management Plan in the Columbia River Estuary* (USACE 2015). The one exception to the use of a standardized method to estimate predation rates was for the East Sand Island double-crested cormorant colony in 2016, where a different analytical approach and group of PIT-tagged fish were used to estimate hypothetical rates of double-crested cormorant predation rates that year (see Skalski and Townsend 2017). To ensure a standardized method was used in all study years, we re-analyzed PIT tag data recovered from the East Sand Island cormorant colony in 2016 using the Bayesian methods of Hostetter et al. (2015). Revised, standardized predation rate estimates from all study years are presented in the *Results* (see *Appendix A, Historical Predation Rates).*

<u>*Smolt Availability*</u> – Smolt availability to birds nesting in the Columbia River Estuary was based on detections of live PIT-tagged fish last interrogated passing Bonneville Dam (Rkm 234 on the lower Columbia River) and Sullivan Dam (Rkm 203 on the lower Willamette River), referred to as "in-river fish". Bonneville and Sullivan dams are considered the upper most reaches of the Columbia River Estuary as defined by the USACE for the purposes of evaluating avian predation rates (USACE 2015; *Map 1*). In addition to in-river migrants, PIT-tagged smolts that were loaded into barges at dams on the lower Snake River and transported and released below Bonneville Dam near Skamania Landing (Rkm 225; *Map 1*) were also included in predation rate analyses, referred to as "transported fish". Availability of transported fish was based on fish interrogated (detected alive) at the Lower Granite Dam (Rkm 695),

Little Goose Dam (Rkm 635), or Lower Monumental Dam (Rkm 589) Juvenile Bypass Systems (JBS) and subsequently loaded into a fish barge. Fish were classified as being collected for transportation based on a unique combination of the interrogation site antennas (e.g., detected entering a raceway) and date at each JBS. Downstream interrogation histories, JBS facility collection reports, and other sources (e.g., NOAA, USACE, and FPC Technical Reports) were used to validate and otherwise proof classifications to ensure accurate assignment of each fish's outmigration history (in-river, transported). Due to small numbers of PIT-tagged fish (generally < 500), smolts collected at JBS facilities and transported using trucks during the study period were not included in study results (see also Evans et al. 2016a).

For both in-river and transported groups of fish, smolt availability was defined as those fish last detected or released (for transported fish) between 1 March and 31 August each year, which reflects the annual periods of overlap in active PIT-tagged smolt out-migration and Caspian tern and double-crested cormorant nesting activity on East Sand Island (Evans et al. 2012; Adkins et al. 2014). PIT-tagged fish were then grouped by salmonid ESU/DPS, representing a unique combination of species (steelhead trout, Chinook salmon, or sockeye salmon), run-type (spring, summer, fall), and river-of-origin (Columbia, Snake, or Willamette). The designation of ESU/DPS followed that of NOAA (2014b) and was largely based on the tagging and release location of each PIT-tagged fish relative to the geographic boundary of each ESU/DPS. Fish within each ESU/DPS were further grouped by rear-type (hatchery, wild), outmigration history (in-river, transport for Snake River ESUs/DPSs), and week.

Not all ESA-listed salmonid ESUs/DPSs in the Columbia River Basin were included in predation rate analyses, as populations that originate wholly or partially below Bonneville and Sullivan dams were excluded. These populations were excluded because spatially- and temporally-explicit detections of live fish during outmigration were not available for these ESUs/DPSs and because sufficient sample sizes of tagged fish from these ESUs/DPSs were lacking for some populations (see also Lyons et al. 2014b). These ESUs/DPSs were: (1) Lower Columbia River steelhead, (2) Lower Columbia River Chinook, (3) Lower Columbia River coho, and (4) Columbia River chum [*O. keta*]. In addition to ESA-listed salmonids, non-listed juvenile salmonids and other fishes (e.g., Pacific lamprey [*Lampetra tridentate*], Eulachon [*Thaleichthys pacificus*]*,* White sturgeon [*Acipenser transmontanus*], coastal cutthroat trout [*O. clarki clarki*]) were also available as prey to Caspian terns and double-crested cormorants nesting on East Sand Island; fish that are of cultural, economic, and/or conservation concern (Lyons et al. 2014b). Including these other fishes was beyond the scope of this study, but efforts to reduce the number of terns and cormorants nesting on East Sand Island will presumably benefit these other fishes as well (USACE 2015).

<u>Deposition and Detection Probabilities</u> – Not all smolt PIT tags that are ingested by colonial waterbirds are subsequently deposited on their nesting colony (Hostetter et al. 2015). A portion of PIT tags implanted in depredated fish are damaged and rendered unreadable following digestion, or are regurgitated off-colony at loafing, staging, or other off-colony areas used by birds during the nesting season. Deposition probability (i.e., probability that a tag consumed by a nesting bird will be deposited on its breeding colony) was previously estimated by feeding PIT-tagged fish to Caspian terns and double-crested cormorants and subsequently recovering those tags on East Sand Island (see Hostetter et al. 2015 for details). The distribution of the median deposition probability derived from these studies was 0.71 (95% CRI = 0.51–0.89) for Caspian terns and 0.51 (95% CRI = 0.34–0.70) for double-crested cormorants (see also *Results*). Deposition probabilities measured during these previous experiments were used to infer deposition probabilities for data collected in 2018. Use of deposition probabilities from data collected in years past was deemed appropriate because results of deposition experiments indicated that deposition probabilities did not vary significantly within or between years for each predator species evaluated (see Hostetter et al. 2015 for additional details).

Not all PIT tags deposited by birds on their nesting colony are subsequently found by researchers after the nesting season (Evans et al. 2012; Sebring et al. 2013; Hostetter et al. 2015). For example, tags can be blown off the colony during wind storms, washed away during flooding events, or otherwise damaged or lost during the nesting season. Furthermore, the detection methods used to find PIT tags on bird colonies are not 100% efficient, with some proportion of detectable tags missed by researchers during the scanning process. Unlike deposition probabilities, detection probabilities (i.e., probability that a tag deposited by a bird on the colony is detected by researchers after the nesting season) often vary significantly within and between breeding seasons, variation that necessitates a direct measure of detection probabilities in each study year, for each colony (Hostetter et al. 2015). To address this in 2018, PIT tags with known tag codes were intentionally sown on the East Sand Island tern and cormorant colonies (hereafter referred to as "control tags") immediately prior to and immediately following the nesting season to quantity PIT tag detection probability during the nesting season. Control tags were the same size and type as those used to mark most juvenile salmonids from the Columbia River Basin (12 mm, FDXB, *model HPT*). During each discrete sowing period, control tags were haphazardly sown throughout the area occupied by nesting birds during the breeding season (see *Figure 2*). Detections (i.e., recoveries) of control tags during scanning efforts after the nesting season were then used to model the probability of detecting tags that are deposited at different times during the nesting season via logistic regression (see *Predation Rates* below for details). Equal number of control tags were sown during each discrete period and sample sizes (n = 300 on the tern colony and n = 300 on the cormorant colony) were selected by considering historic releases (see Evans et al. 2016a). This allows direct comparisons of independent detection probabilities, with similar precision between years.

In 2018, a newer model PIT tag, the Advance Performance Tag (APT), was used to tag some juvenile salmonids from the Columbia River Basin. Although APT tags operate on the same frequency and are the same size and shape as the more commonly used High Performance Tag (HPT), the read range of the APT tags is expected to exceed that of the HPT tag, a difference that could influence rates of detection on bird colonies. To investigate potential differences in detection rates by tag model, we sowed both HPT and APT tags on the East Sand Island Caspian tern and double-crested cormorant colonies prior to and following the 2018 nesting season. A description of the methods and results associated with this ancillary, detection efficiency experiment is provided in *Appendix B.*

<u>*Predation Rates*</u> – Following the methodology of Hostetter et al. (2015), predation rates were modeled independently for each salmonid ESU/DPS, bird colony (Caspian tern, double-crested cormorant) and year. The probability of recovering a PIT tag from a smolt on each colony was modelled as the product of the three probabilities described above, the probability that (1) the fish was consumed (θ), (2) the PIT tag was deposited on-colony (ϕ), and (3) the PIT tag was detected on-colony after the breeding season (ψ_i; see also *Figure 3*):

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

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:

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

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. 2016a). 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:

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

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

Informative Beta priors were used to model deposition probability(ϕ). The shape parameters (α, β) are dependent on the predator species (cormorants, terns) and are assumed to be mutually independent from colony to colony. For terns, we assumed α = 16.20 and β = 6.55 and for cormorants, we assumed α = 15.98 and β = 15.29 (see Hostetter et al. 2015 for details).

Weekly predation estimates were defined as the estimated number of PIT-tagged smolts consumed divided by the total number last detected passing Bonneville Dam, Sullivan Dam, or released from barges in the tailrace of Bonneville Dam 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 last detected at Bonneville Dam, Sullivan Dam, or released from barges:

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

Summation of weekly consumption estimates is necessary to accurately reflect weekly variation and autocorrelation of predation rates and thus to create unbiased annual rates with accurate assessments of precision (Butler and Stephens 1993; Hamilton 1994).

Rear-type and outmigration history comparisons: We updated our previous methods (see Evans et al. 2016) for the calculation of predation rates specific to each rear-type (hatchery, wild) and each migration history (in-river, transported) among different ESUs/DPSs. We implemented a Bayesian approach that acknowledged the autocorrelative nature of predation rates (as outlined above) while additionally allowing for the common estimate of deposition and detection across cohorts. Predation rates specific to each cohort were thus developed by partitioning the release and recovery of tags such that:

$$k_{iv} \sim Binomial(n_{iv}, \theta_{iv} * \phi * \psi_i)$$

where k_{iv} is the number of smolt PIT tags in category *v* recovered from the number available (n_{iv}) in week *i*. Annual and weekly predation probabilities for each cohort were then calculated simultaneously, with shared estimates of deposition and detection, using the autoregressive methods described above.

To more thoroughly evaluate differences in predation rates among cohorts we accessed the prevalence of differences by category (rear-type, migration history) across multiple years of data. We defined v_0 to be the cohorts of wild fish and transported fish for the rearing and migration history analysis, respectively. Thus v_1 respectively represented hatchery and in-river fish. We defined ρ to be the average proportional difference in the log-odds of predation among cohorts over the entire study period. A value less than or greater than 1.0 indicating a preference for a group or category of fish and a value of 1.0 showing no preference. A random error term was included to account for extraneous variation of the proportional difference in log-odds among weeks. Therefore, we assumed:

$$k_{iv_0} \sim Binomial(n_{iv_0}, \theta_i * \phi * \psi_i)$$

and

$$k_{iv_1} \sim Binomial\left(n_{iv_1}, logit^{-1}\left((\rho + \eta_i) * \ logit(\theta_i)\right) * \phi * \psi_i\right)$$

where $\eta_i \sim normal(0, \sigma_\rho)$. Credible intervals for ρ which overlapped 1 were defined as not statistically significant. This test was applied to all appropriate ESUs/DPSs for each comparison.

<u>Predation impacts prior to and following management actions</u>: If given enough time and a significant decrease in the number of nesting birds, it is expected that the management of Caspian terns and double-crested cormorants on East Sand Island will have a measurable effect on the level of predation. For the Caspian tern colony on East Sand Island, comparisons of predation rates by management period were defined as those during 2000-2010 (pre-management) and those during 2011-2018 (management). The management time period was considered to have started in 2011 (as opposed to 2008 when efforts to reduce nesting habitat were first initiated) because this was the first year that reductions in nesting habitat at East Sand Island resulted in a significant reduction in the number of terns below the pre-management average (Evans et al. 2016a; BRNW 2017).

For the double-crested cormorant colony on East Sand Island, management time periods were defined as 2003-2015 (pre-management) and 2016-2018 (management). Although cormorants have nested on East Sand Island since the late 1980's, the steady increase in the number of nesting cormorants and predation rates was first considered to be a significant threat to salmonid recovery by the federal government in 2003 (referred to as the "current period"; NOAA 2014a; USACE 2015). The first year of management actions on the East Sand Island cormorant colony were initiated in 2015. Management activities in 2015, however, were limited in scope and started after most PIT-tagged fish passed Bonneville and Sullivan dams that year. For instance, only 158 adult cormorants (< 1% of the estimated 12,150 breeding pairson East Sand Island in 2015) were culled between 22 May and 31 August 2015 (USACE 2015), a number too small to influence predation rates in a meaningful way that year (Evans et al. 2016a). As such, impacts from the 2015 East Sand Island cormorant colony on PIT-tagged juvenile salmonids should be considered more comparable to the pre-management impacts, than impacts during the management period.

To monitor the effectiveness of avian predation management plans to reduce predation rates by reducing colony size, we compared estimates of ESU/DPS-specific predation rates and peak colony sizes (number of nesting pairs) across study years, to the extent possible (e.g., where sufficient data existed for the analysis). Per capita (per nesting pair) predation rates were calculated by dividing the annual ESU/DPS-specific predation rate by the peak measure of colony size each year:

$$Annual\ Per\ Capita\ Predation\ Rate_y = {\sum_w (\theta_{wy} * n_{wy}) / \sum_w (n_{wy})} \Big/ {C_y}$$

where C_y is the peak colony size in year, *y.* The relationship between colony size and predation rates across years was further evaluated by generating a posterior distribution of least squares regression lines, calculated from random samples of per capita predation rates taken from the posterior distributions associated with the above defined estimates. The strength and direction of the relationship can be inferred from the median value of the resulting posterior distribution associated with the slope of the resulting posterior distribution with the 95% CRI defined as the 2.5th and 97.5th quantiles. Statistical credibility was identified by the credibility interval of the slope parameter not overlapping with zero.

Modelling software and assumptions: All predation rate models were implemented using the software JAGS accessed through R version 3.1.3 (R Core Team 2015) using the R2jags (Su and Yajima 2012) and dclone (Sólymos 2010) R packages. Three parallel chains were run for 80,000 iterations each, after an initial 10,000 iteration burn-in, to diagnose and confirm convergence. Chain convergence was tested using the Gelman-Rubin statistic ($\hat{R}$; Gelman et al. 2004). A single "long-run" of 150,000 Markov Chain Monte Carlo (MCMC) iterations were run to produce the final posterior distribution from which estimates were derived (Raferty 1992). Chains were thinned by 20 to reduce autocorrelation inherent to successive MCMC samples. Results were reported as posterior medians along with the 2.5 and 97.5 percentiles, which are referred to as 95% credible intervals. Annual predation rates were calculated for salmonid ESUs/DPSs where ≥ 500 PIT-tagged individuals were available to birds to avoid imprecise results that may occur from small sample sizes of available PIT-tagged smolts (Evans et al. 2012). In addition, predation rates had to be at least 1% per group for comparisons of impacts, based on rear-type and outmigration history; rates potentially large enough to be biologically relevant to salmon survival.

A detailed list of predation rate model assumptions (A) and procedures used to evaluate the validity of those assumptions is provided in Hostetter et al. (2015) and Evans et al. (2016a). Briefly, the predation model assumed that (A1) PIT tag interrogation data obtained at dams from PTAGIS were accurate, (A2) PIT-tagged fish passing dams were available to birds nesting downstream, (A3) predation, detection, and deposition were independent variables, and in the case of detection and deposition, were accurately measured in field studies, (A4) PIT-tagged fish were consumed in a relatively short (one week) period following interrogation at upstream dams, and (A5) PIT-tagged fish were representative of non-tagged fish belonging to the same ESU/DPS and passing the same detection (dam) or release (barge) sites. All assumptions were validated to the extent possible, or possible violation of the assumption (e.g., predation within a week of detection/release) had little influence on predation rates (see Hostetter et al. 2015 and Evans et al. 2016a for additional details).

RESULTS AND CONCLUSIONS

PIT Tag Recovery

East Sand Island Caspian tern colony – Following the nesting season, 10,886 PIT tags from 2018 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead trout combined) were recovered on the East Sand Island Caspian tern colony (*Table 1*). The number of smolt tags recovered from the tern colony in 2018 was slightly higher than the number in 2016 (9,930 tagged smolts) and 2017 (8,407 tagged smolts), but substantially lower than in other years (annual range during 2000-2015 = 13,059 to 44,947 tagged smolts recovered).

In total, five passes of the tern colony with the flat-plat antenna were conducted following the nesting season, with 3.4% unique tag codes detected during the fifth and final pass. A total of six passes were also conducted with magnet sweeper, with 14,796 tags detected and physically removed from the colony site following the 2018 nesting season. Physical removal of tags from the colony site helps increase detection efficiency on the East Sand Island tern colony in future years by reducing tag collision (Evans et al. 2016a).

Recoveries of control PIT tags sown on the East Sand Island tern colony (n = 300) indicated that estimated detection efficiency averaged 77% (seasonal range = 57–97%) during the 2018 nesting season (*Table 2*). Estimated average detection efficiency in 2018 was similar to the range during 2011-2017 (range of annual averages = 64–87%; Evans et al. 2018). Based on previous studies that empirically measured deposition rates for Caspian terns nesting on East Sand Island, deposition rates were estimated to be 71% (95% CRI = 51–89%; *Table 2* and Hostetter et al. 2015).

East Sand Island double-crested cormorant colony – Following the nesting season, 2,680 PIT tags from 2018 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead trout combined) were recovered on the East Sand Island double-crested cormorant colony (*Table 1*). The number of smolt PIT tags recovered on the double-crested cormorant colony in 2018 was higher than that recovered in 2017 (1,340 smolt tags) but substantially lower than that recovered in years past (annual range = 9,047 to 31,984 tagged smolts during 2005-2016). In 2017, nesting cormorants were not consistently present on East Sand Island during the peak smolt outmigration period (April to May) and as a result, a record low number of smolt PIT tags were deposited by double-crested cormorants on East Sand Island (1,340 smolt tags; Evans et al. 2018). A similar but less dramatic temporal gap between peak smolt run-timing and colony attendance was observed in 2018, where a large portion of available PIT-tagged smolts in the estuary migrated past East Sand Island prior to the arrival of most cormorants on East Sand Island in May of 2018 (*Figure 4*). Unlike 2017, where multiple mass dispersal events occurred during the peak outmigration period (Turecek et al. 2018a), once birds arrived on East Sand Island in 2018 they were consistently on-colony throughout the breeding season (Turecek et al. 2018b). A delay in the nesting chronology of double-crested cormorants in 2018 was largely attributed to predation and disturbance by bald eagles (*Haliaeetus leucocephalus*) that were present on the East Sand Island cormorant colony during April and May (Turecek et al. 2018b).

In total, four passes of the double-crested cormorant colony with hand-held PIT tags antennas were conducted following the 2018 nesting season, with 3.7% unique tag codes detected during the fourth and final pass. Control PIT tags sown to measure detection efficiency on the cormorant colony (n = 300) indicated that detection efficiency averaged 88% (range = 76–100%) during the nesting season in 2018 (*Table 2*). Detection efficiency estimates in 2018 were the highest reported since comparable estimates were first generated in 2004 (annual range = 36–81% during 2004-2017; Evans et al. 2016a). Based on previous studies that empirically measured deposition rates for double-crested cormorants nesting on East Sand Island, deposition rates were estimated to be 51% (95% CRI = 34–70%; *Table 2* and Hostetter et al. 2015).

East Sand Island Brandt's cormorant colony – A total of just 240 PIT tags from 2018 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead trout combined) were recovered on the East Sand Island Brandt's cormorant colony following the nesting season in 2018 (*Table 1*). Analogous to double-crested cormorants, Brandt's cormorants attempted to nest on East Sand Island after the peak smolt outmigration period. Tags recovered from the Brandt's cormorant colony were not used in predation rate analyses, but tag codes were reported to PTAGIS as avian mortalities (see

Methods). It is worth noting that an estimated 2,120 pairs of Brandt's cormorants nested on East Sand Island in 2018, the highest number observed since the colony formed in 2006 (Evans et al. 2016a; Turecek et al. 2018b). Previous research indicated that Brandt's cormorants consumed a small proportion of available juvenile salmonids in the estuary, with annual estimated predation rates consistently less than 0.5% of available fish, per ESU/DPS (Evans et al. 2016a). The paucity of PIT tags recovered from the Brandt's cormorant colony in 2018 suggests that predation impacts remain low on East Sand Island, despite increases in the size of Brandt's cormorant colony from 44 pairs in 2006 to 2,120 pairs in 2018 (Evans et al. 2016a and Turecek et al. 2018b).

Predation Rates

*East Sand Island Caspian terns – P*redation rate estimates varied by salmonid ESU/DPS in 2018 (*Table 3*). Similar to years past (see *Appendix A*), results indicated that steelhead DPSs were the most susceptible to predation by Caspian terns nesting on East Sand Island, with predation rate estimates ranging from 5.3% (95% CRI = 3.8–8.0%) on Middle Columbia River steelhead to 6.9% (95% CRI = 5.3–10.2%) on Snake River steelhead (*Table 3*). By comparison, predation rates on most salmon ESUs were significantly lower than those on steelhead DPSs, ranging from just 1.3% (95% CRI = 0.7–2.1%) on Snake River Fall Chinook salmon to 1.4% (95% CRI = 0.9–2.3%) on Upper Columbia River spring Chinook salmon (*Table 3*). The one exception was for Snake River sockeye in 2018, where terns consumed an estimated 4.2% (95% CRI = 2.9–6.5%) of available fish.

Differences in steelhead susceptibility to Caspian tern predation relative to salmon susceptibility are well documented in past studies (Collis et al. 2001; Ryan et al. 2003; Evans et al. 2012; Evans et al. 2016b), with predation rate estimates on steelhead populations often 5 to 10 times greater than those on salmon populations (*Appendix A, Table A1*). Differences in the relative size (length) and behavior of steelhead compared with salmon species are two possible explanations. Beeman and Maule (2006) observed that steelhead smolts were more surface-oriented compared with salmon smolts and surface orientation is believed to render fish more vulnerable to predation by Caspian terns, a plunge diving species that forages in the top meter of the water column (Cuthbert and Wires 1999). Furthermore, Hostetter et al. (2012) and Evans et al. (2016a) noted size-selectivity amongst avian predators, with larger fish typically predated at higher rates than smaller fish, which may explain differences between salmon and steelhead in their susceptibility to tern predation (see *Impacts by rear-type and outmigration history* below for additional discussion and Evans et al. 2016a).

An investigation of smolt run-timing, based on the passage date of PIT-tagged fish at Bonneville Dam, indicated that most fish were available as prey to Caspian terns nesting on East Sand Island in 2018 (*Figure 4*). Unlike past years, however, the peak colony attendance in 2018 was in late June instead of late May (Evans et al. 2016a; Roby et al. 2017). A late or right-shifted breeding chronology for terns may have reduced smolt susceptibility to Caspian tern predation in 2018 relative to past years. Similar to the double-crested cormorant colony on East Sand Island, predation and disturbance by bald eagles during April and May were believed to be the primary reason why the tern breeding season was delayed in 2018 relative to years past.

An investigation of weekly predation rates by East Sand Island Caspian terns indicated that estimated predation rates were generally lower when the largest number or greatest density of PIT-tagged smolts were available as prey in the estuary (*Figure 5*). Estimated impacts on steelhead DPSs were the lowest during the peak of the run in May and slightly higher before (April) and especially after (June) the peak passage period. Hostetter et al. (2012) theorized that the inverse relationship between prey density and Caspian tern predation rates was due to prey swamping, with the probability of an individual fish being consumed decreasing as the number of available prey increases (see also Ims 1990). A multi-year analysis of weekly East Sand Island Caspian tern predation rates and smolt abundance estimates in the estuary conducted by Roby et al. (2017) indicated that the trend of decreasing tern predation rates with increasing smolt availability was statistically significant across all weeks and years (2006-2016), with the odds of predation estimated to decline by a factor 0.82 (95% CRI = 0.75-0.89) for every 10% increase in the relative availability of PIT-tagged smolts. A functional relationship between decreasing levels of predation with increases in prey abundance is often referred to as a *Type II* mortality curve (see Roby et al. 2017 for additional discussion).

<u>*Impacts by rear-type and outmigration history*</u>: There was evidence that hatchery Snake River spring/summer Chinook were more susceptible to predation by East Sand Island Caspian terns compared with their wild counterparts in 2018 (*Table 4*). No difference in relative susceptibility, however, was observed between hatchery and wild Snake River steelhead in 2018 (*Table 4*). Comparisons in other ESUs/DPSs were constrained by small sample sizes of PIT-tagged wild fish in 2018. An investigation of weekly and annual trends over the course of the last 13 years (2006-2018) indicated that hatchery Snake River spring/summer Chinook and hatchery Upper Columbia River spring Chinook were also, on average, more susceptible to East Sand Island tern predation as compared to their wild counterparts (*Appendix C;* see also Evans et al. 2016a). Differences in predation rates on steelhead DPSs by rear-type in years past, however, were often minor (based on the proximity of the effect-factor estimate to 1.0), with no consistent long-term trend was identified over the course of the last 13 years (*Table 4 and Appendix C).*

Data from other studies indicates that both behavior and physical traits associated with hatchery-raised juvenile salmonids may enhance susceptibility to predation (Olla and Davis 1989, Fritts et al. 2007, Hostetter et al. 2012). Evans et al. (2016a) attributed difference in the vulnerability of hatchery spring/summer Chinook salmon to Caspian tern predation to differences in the size (mm; fork length) of hatchery (mean = 144 mm) and wild (mean = 111 mm) Chinook salmon smolts last detected passing Bonneville Dam during 2006-2015. An analysis of associations between predation rates and length data (based on lengths collected within the same month fish were interrogated passing Bonneville Dam) indicated that the odds of Caspian tern predation on spring/summer Chinook salmon increased by an estimated 12% (95% CRI = 11.9–12.6%) for every 10-mm increase in fork-length (Evans et al. 2016a). Hostetter et al. (2012) also found evidence of size-selectivity in Caspian terns nesting at Crescent Island in McNary Reservoir (Columbia River), with larger fish more likely to be preyed upon than smaller fish up to about 175 mm, at which point susceptibility to tern predation was estimated to be similar for fish up to about 225 mm. Tern predation rates on fish > 225 mm was estimated to rapidly decrease as fish reached or exceeded the maximum prey size for Caspian terns of about 275 mm (Cuthbert and Wires 1999; Lyons 2010). The majority (> 80%) of hatchery and wild PIT-tagged steelhead last detected passing Bonneville Dam were between 175-225 mm, fish with similar length-dependent selectivity profiles (Roby et al. 2017), a finding that explains why differences in relative susceptibility of steelhead by rear-type were less pronounced than those observed in salmon ESUs.

There was some evidence that Caspian terns disproportionately consumed in-river migrating fish from the Snake River relative to transported migrants from the Snake River, although no statistically significant difference by migration history was observed in 2018 (*Table 4*). An investigation of the transported versus in-river migrant data over the course of the last thirteen years indicated that in-river fish were, on average, more likely to be consumed then their transported counter-parts. Odds-ratios were close to 1.0 (no preference), however, for most ESUs/DPSs and no consistent trend across all weeks and years was observed (*Appendix D*; see also Evans et al. 2016a). Collectively, results indicate that in-river fish were more likely to be consumed than transported fish in most weeks and years but that differences in susceptibly were relatively small and inconsistent. Roby et al. (2017) theorized that differences in the relative susceptibility of in-river versus transported fish were due to differences in run-timing (arrival times in estuary) and how run-timing coincided with the nesting chronology of Caspian terns on East Sand Island.

<u>Impacts prior to and following management actions</u>: An investigation of predation impacts prior to and following management actions indicates that predation rates were, on average, significantly lower following management actions to reduce colony size on East Sand Island during 2011-2018, compared with predation rates prior to management during 2000-2010 (*Table 5*). For instance, average annual predation rates on Snake River steelhead during 2000-2010 were estimated to be 22.2% (95% CRI = 20.3–24.8%), compared with 9.2% (95% CRI = 8.2–10.4%) following management actions that reduced the number of terns on East Sand Island. Significant reductions in predation rates were also observed in all other ESA-listed ESUs/DPSs evaluated (*Table 5*), with the exception of Snake River sockeye, where no statistically significant difference by management period was observed. Insufficient samples sizes of Snake River sockeye and some other ESUs/DPSs (e.g., Upper Willamette River spring Chinook; see *Appendix A*), limited comparisons in all study years, however.

Addressing high rates of steelhead predation by Caspian terns was the primary impetus of the Caspian Tern Management Plan in the Columbia River Estuary (USFWS 2006). Average estimated predation rates following East Sand Island Caspian tern management on steelhead DPSs were roughly one-third to one-half (depending on the DPS) lower than those observed during the pre-management period (*Table 5*). In all three steelhead DPSs evaluated (Snake River steelhead, Upper Columbia River steelhead, Middle Columbia River steelhead), reductions in Caspian tern predation rates were proportional to reductions in colony size (*Figure 6*). Trends were particularly evident when large differences in colony sizes were observed and less evident when smaller or no difference in colony sizes were observed due variability in predation rate estimates (*Figure 6*). Using data from all years, comparisons indicate a linear relationship between annual predation rates and colony sizes ($p < 0.01$ in all steelhead DPS evaluated; *Figure 6*). Results indicate that Caspian tern management initiatives aimed at reducing the number of Caspian terns nesting on East Sand Island have resulted, on average, in reduced estimated annual predation rates on steelhead DPSs. This was particularly evident in 2017 and 2018, when the lowest numbers of Caspian terns ever recorded on East Sand Island (3,500 and 4,959 nesting pairs, respectively) coincided with the lowest estimated predation rates (*Figure 6*). Prior to management actions that reduced the size of the colony, the East Sand Island Caspian tern colony averaged 9,221 breeding pairs (range = 8,283–10,668 pairs during 2000-2010; *Figure 6* and BRNW 2017). Following management actions that reduced the size of colony, the East Sand Island Caspian tern colony averaged 5,957 breeding pairs (range = 3,500–7,387 pairs during 2011-2018; *Figure 6*).

The Caspian Tern Management Plan target colony size goal for East Sand Island is no more than 3,125 breeding pairs (USFWS 2005). Based on an analysis of average (2000-2018) per capita (per nesting pair) predation rate estimates (*Table 6*), an East Sand Island tern colony of 3,125 breeding pairs would consume an estimated 6.9% (95% CRI =6.3–8.1%), 5.9% (95% CRI = 5.3–6.3%), and 4.7% (95% CRI = 4.1–5.6%) of Snake River, Upper Columbia River, and Middle Columbia River steelhead, respectively. If achieved, rates would represent an approximately two-thirds reduction in steelhead predation rates relative to average pre-management predation rates; another stated goal of the Caspian Tern Management Plan (USFWS 2005). In the context of over-all smolt mortality due to Caspian tern predation in the estuary, it is important to note that predation rate results presented herein are specific to terns that nested on East Sand Island, but large numbers (hundreds to thousands of pairs annually) of

Caspian terns have attempted to nest on Rice Island (Rkm 34; *Map 1*) in the upper Columbia River estuary during the East Sand Island management period (USACE, unpublished data; Evans et al. 2018). The impact of Caspian terns that have attempted to nest on Rice Island on smolt survival in the Columbia River Estuary during the management period, however, is currently unknown. Roby et al. (2002) reported that juvenile salmonids were more prevalent in the diet of Caspian terns that successfully nesting on Rice Island compared with terns nesting on East Sand Island in years past, with salmonids comprising 77% and 90% of the diet of terns on Rice Island in 1999 and 2000 compared with 46% and 47% juvenile salmonids in the diet of terns on East Sand Island in 1999 and 2000. It is unknown, however, if similar differences in smolt susceptibility to Rice Island and East Sand Island tern predation existed during the management period. Regardless, because Caspian terns were observed on both East Sand Island and Rice Island in recent years, an investigation of cumulative predation rates by all terns would be necessary to characterize the total or net impact of tern predation on ESA-listed smolts throughout the estuary.

East Sand Island double-crested cormorants – Estimated predation rates by double-crested cormorants in 2018 were amongst the lowest ever recorded on East Sand Island, ranged from 0.4% (95% CRI = 0.1–1.0%) of Middle Columbia River steelhead to 0.9% (95% CRI = 0.5–1.9%) of Snake River sockeye (*Table 4*). Predation rate estimates were similar between steelhead and salmon ESUs/DPSs in 2018. An analysis of historic data indicates that cormorant predation rates were often, but not always, higher on steelhead DPSs compared with salmon ESUs (*Appendix A*, *Table A2*) and the lack of difference between steelhead and salmon in 2018 may be due in part to the over-all low estimates of predation in 2018. Significant annual variation in ESU/DPS-specific predation rates by cormorants on juvenile salmonid populations were also evident in years past, with differences in predation rates often greater between years within the same ESU/DPS than differences between ESUs/DPSs in the same year. For instance, estimated predation rates by cormorants on Snake River spring/summer Chinook ranged from just 0.5% (95% CRI =0.3–0.0%) in 2018 to 14.5% (10.5–22.4%) in 2015 (*Appendix A, Table A2*).

The small size (3,672 nesting pairs) of the double-crested cormorant colony on East Sand Island in 2018 compared with years past (annual range = 10,050–13,771 pairs during 2003 to 2015) was one presumed reason why predation rate estimates in 2018 were amongst the lowest ever recorded. An investigation of weekly predation rates in 2018 indicated that predation rates were at or close to zero (0) in many weeks during 2018, particularly during the peak smolt outmigration period of April and May (*Figure 7*). Measured predation was virtually non-existent during much of April because cormorants did not establish a nesting colony on East Sand Island until May, after the peak of the 2018 smolt migration period had ended (*Figure 4*). Analogous to Caspian terns on East Sand Island, the double-crested cormorant nesting season was delayed or right-shifted in 2018 relative to years past (*Figure 4*). Collectively, a combination of a smaller colony size and late colony formation resulted in some of the lowest East Sand Island cormorant predation rates observed since 2000, the first-year predation rate estimates are available (Evans et al. 2016a).

Roby et al. (2015) and Evans et al. (2016a) observed that estimated predation rates by double-crested cormorants increased in concert with the number of smolts available in the estuary, with the highest estimates of predation occurring during the peak outmigration period for each ESU/DPS evaluated. Results indicated that as more fish became available, double-crested cormorants consumed a larger proportion; a finding that suggests that a swamping affect might not exist for double-crested cormorants, at least at the smolt densities observed during these studies (Ims 1990; Hostetter et al. 2012). The trend observed in cormorants – analogous to a *Type III* mortality curve – was the opposite of that observed in East Sand Island Caspian terns, where estimated predation rates decreased as more

PIT-tagged fish became available (*Figure 5;* see also Roby et al. 2017). Unlike Caspian terns, double-crested cormorants are pursuit divers that can consume multiple fish in a single foraging bout (Hatch and Weseloh 1999) and as such, highly concentrated prey may be especially vulnerable to predation by double-crested cormorants (Lyons 2010; Evans et al. 2016a).

Impacts by rear-type and outmigration history: Insufficient sample sizes of recovered PIT tags prevented a meaningful comparison of predation impacts based on a fish's rear-type (hatchery, wild) to East Sand Island double-crested cormorants in 2018. A summary of cormorant predation rate estimates by rear-type and outmigration history from data collected during 2006-2015 (years when the vast majority of cormorants in the estuary nested on East Sand Island during the entire smolt outmigration period) are provided in *Table 4* and *Appendix C* (see also Evans et al. 2016a). Results indicated that there were no consistent trends in the relative susceptibility of fish by rear-type to East Sand Island double-crested cormorant predation in years past. There was limited evidence that wild Snake River steelhead were more susceptible to cormorant predation than their hatchery counterparts, but differences were not consistent across weeks and years (*Appendix C*). There was some evidence that hatchery Upper Columbia River spring Chinook were more susceptible, on average, than wild fish, but again, no consist trend across all weeks and years was observed (*Appendix C*). Collectively, results indicated that hatchery and wild smolts last detected passing Bonneville Dam had no appreciable difference in susceptibility to double-crested cormorant predation on East Sand Island for most ESU/DPS, weeks, and years evaluated (*Appendix C*). Other studies have also observed small and inconsistent differences in predation rates between hatchery and wild juvenile salmonids to cormorant predation in the Columbia River Estuary (Collis et al. 2001; Ryan et al. 2003; Hostetter et al. 2012). A more detailed discussion of mechanisms that potentially explain differences (or lack thereof) in the relative susceptibility of smolts based on their rear-type to cormorant predation is provided in Evans et al. (2016a).

Insufficient samples of recovered PIT tags also prevented a meaningful comparison of cormorant impacts based on fish's outmigration history (in-river, transport) in 2018. A multiple year summary of comparisons in years past showed no consistent trend between the relative susceptibility of in-river versus transported fish to East Sand Island cormorant predation (*Appendix D*; see also Evans et al. 2016a). There was some evidence that transported Snake River sockeye and fall Chinook salmon were more likely to be consumed by cormorants than in-river migrants and some evidence that in-river Snake River spring/summer Chinook salmon and in-river Snake River steelhead were slightly more likely to be predated than transported fish (*Table 4*), but again, no consistent trend across all weeks and years were identified (Appendix D).

Impacts prior to and following management actions: An investigation of predation impacts prior to and following management actions indicates that estimated predation rates were significantly lower following management actions on East Sand Island (*Table 7*). In 2018, predation rates were less than 1.0% for all ESU/DPS evaluated (Table 4 and 7). Conversely, prior to management, average annual steelhead predation rates were estimated at 5.1% (95% CRI = 4.1–6.1%), 8.3% (95% CRI = 6.8–10.1%), and 9.3% (95% CRI = 8.0–11.0%) of Upper Columbia River, Middle Columbia River, and Snake River steelhead, respectively (*Table 7*). Average annual predation rate estimates prior to management were also appreciable on some salmon ESUs, particularly Snake River spring/summer Chinook, with an estimated 5.2% (95% CRI = 4.4–6.1%) of available fish consumed by East Sand Island cormorants during 2003-2015.

Reductions in estimated predation rates in 2018 were largely attributed to a smaller colony and the late (right-shifting) arrival of cormorants on East Sand Island (*Figure 4*; see also Predation Rate estimates above for additional details). As part of efforts to reduce colony size in 2018, cormorants were confined to nesting habitat on the far west end of the island (*Figure 2*), with passive and active dissuasion techniques used to prevent nesting east of the western portion of the island. Actions were successful, with an estimated 3,672 breeding pairs in 2018, all of which nested on the designated colony area on the far west end of the island. The average size of the East Sand Island double-crested cormorant colony prior to management was estimated to be 12,744 breeding pairs (range = 10,646 –14,916 pairs during 2003-2015; BRNW 2017). The goal of Double-crested Cormorant Management Plan was to reduce the East Sand Island double-crested cormorant colony to no more than approximately 5,600 breeding pairs (USACE 2015), a target goal that was achieved in 2018. Based on per capita predation rates (*Table 6*), an East Sand Island double-crested cormorant colony would consume an estimated 2.8% (95% CRI = 2.2–3.9%), 3.2% (95% CRI = 2.8–5.0%), and 3.4% (95% CRI = 2.8–4.5%) of Upper Columbia, Middle Columbia, and Snake River steelhead, respectively. With rates in 2018 less than 1.0% for all ESU/DPS evaluated, target predation rates goals were also achieved in 2018. It should be noted, however, this outcome was a product of both a historically small colony (due to management) and a historically late nest initiation period that occurred after the peak 2018 smolt migration period (*Figure 4*). Given how highly variable annual rates of predation by cormorants have been in years past (*Appendix A, Table A2*), the benefits of reducing the number of double-crested cormorants on East Sand Island in any given year will likely vary from year to year and as such future monitoring maybe warranted, particularly if more normal nesting behavior occurs in 2019 and/or if a larger colony again becomes established on East Sand Island.

Prior to management actions in 2016, the vast majority (> 95%) of double-crested cormorants present in the Columbia River Estuary nested on East Sand Island during the smolt outmigration period (Lyons et al. 2014a; Evans et al. 2016a). In 2016 and 2017, cormorants repeatedly attempted but did not successfully nest on East Sand Island during the peak smolt outmigration period of April and May, with several *en masse* dispersal events occurring during the breeding season. Rather than completely dispersing to colony sites outside of the Columbia River Estuary, large numbers of cormorants (at least 7,000 adults in 2017) remained in the estuary and continued to forage on juvenile salmonids to an unknown degree (Evans et al. 2018). Since 2016 there has also been a substantial increase in the number of double-crested cormorants nesting further upriver on the Astoria-Megler Bridge (Rkm 23), with the colony increasing from an average 111 breeding pairs prior to 2016 to an average of 1,040 breeding pairs since 2016, peaking at 1,736 pairs in 2018 (Turecek et al. 2018b). As such, analogous to impacts from Rice Island Caspian terns foraging in the upper estuary, estimates of East Sand Island cormorant predation rates presented herein do not represent the total or net impact of all cormorants in the estuary on juvenile salmonids in 2018. Additional research would be needed to characterize the total or cumulative impact of all cormorants in the estuary on ESA-listed smolts but was beyond the scope of this study. Finally, predation rate estimates from 2016 and 2017, where cormorants temporarily dispersed *en masse* from East Sand Island, are not directly comparable to estimates in other years, where more normal nesting behavior on East Sand Island occurred.

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MAPS

Map 1. Columbia and Snake rivers depicting Lower Granite, Little Goose, and Lower Monumental dams (sites where PIT-tagged smolts were loaded into transportation barges) and Bonneville and Sullivan dams (interrogation sites for in-river fish) and Skamania landing (release site for transported fish). Interrogation and release sites were used to determine the availability of PIT-tagged fish to terns and cormorants nesting on East Sand Island in the Columbia River estuary.

FIGURES

Figure 1. PIT tag detection equipment used on East Sand Island in 2018 which included an eight-coil flat-plate antenna with tow-behind sweep magnet attached to an ATV for use on the Caspian tern colony (left) and hand-held portable antenna systems (right) for use the double-crested cormorant colony and areas not accessible to the ATV on the Caspian tern colony.

Figure 2. Areas scanned for PIT tags deposited by nesting double-crested cormorants (DCCO; top), Brandt's cormorants (BRAC: top), and Caspian terns (CATE: bottom) on East Sand Island in 2018.

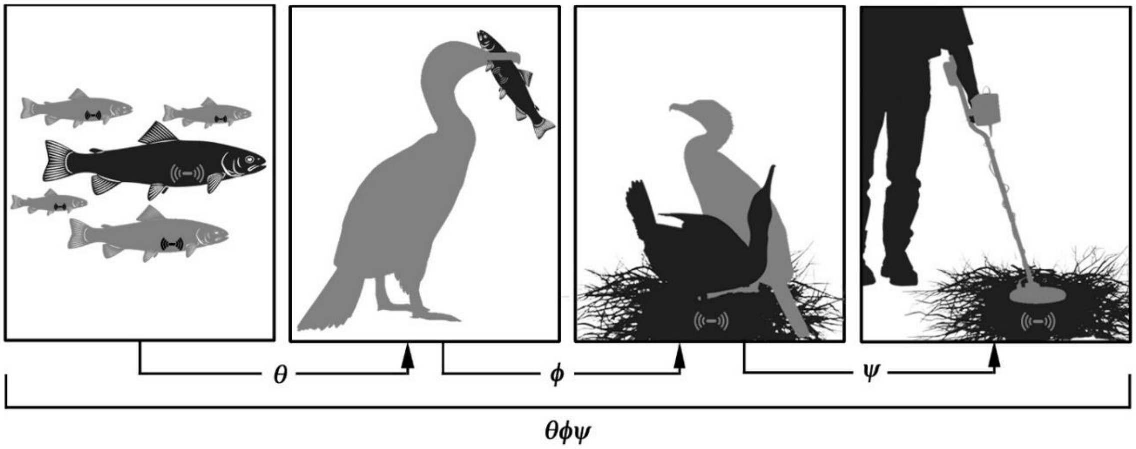

Figure 3. Conceptual model of the tag-recovery process in studies of avian predation. The probability of recovering a fish tag on a bird colony is the product of three probabilities: a fish was consumed (predation probability, ϑ), deposited on the nesting colony (deposition probability, ϕ), and detected by researchers (detection probability, ψ). Figure from Hostetter et al. (2015).

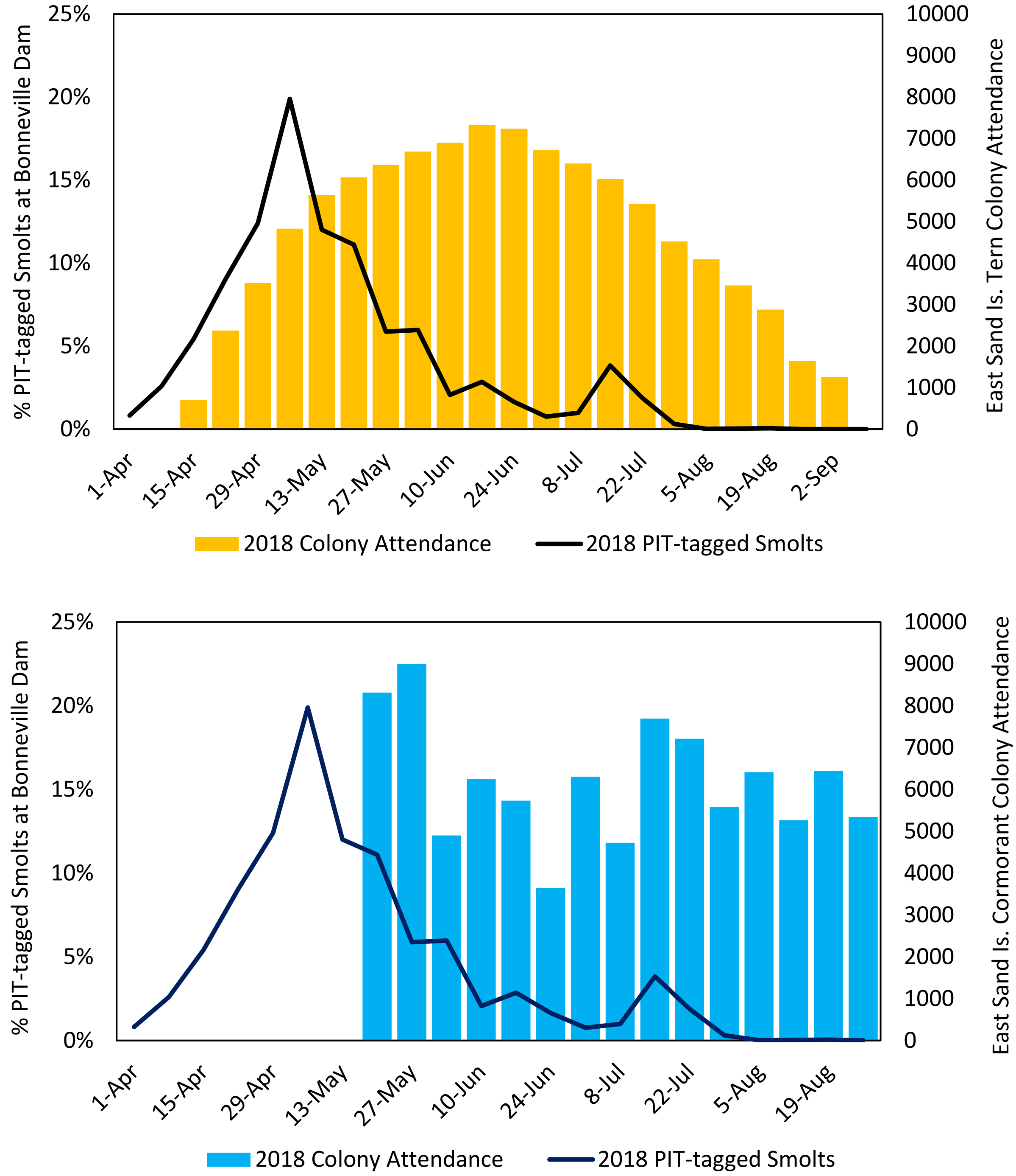

Figure 4. Attendance of Caspian terns (top; orange bars) and double-crested cormorants (bottom; blue bars) on East Sand Island and the run-timing of PIT-tagged juvenile salmonids last detected passing Bonneville Dam in 2018 (top and bottom; black line).

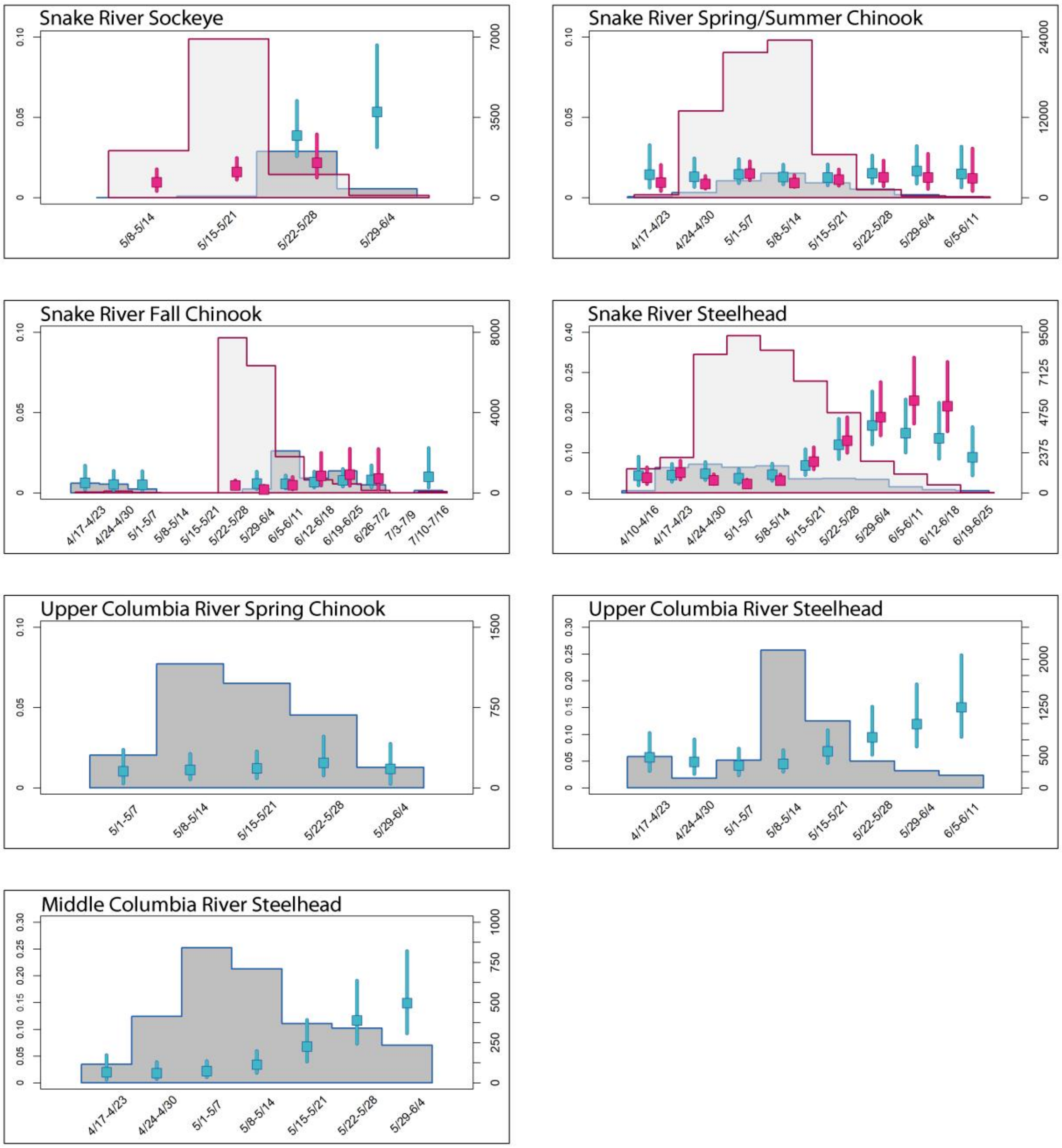

Figure 5. Estimated weekly predation rates (y_1; proportion of fish consumed) on in-river (blue squares) and transported (red squares) PIT-tagged juvenile salmonids last detected passing Bonneville or Sullivan dams (y_2; number available, dark gray bars) or transported from the lower Snake River (y^2; number available; light gray bars) by Caspian terns on East Sand Island during 2018. Error bars represent 95% credible intervals for predation rates.

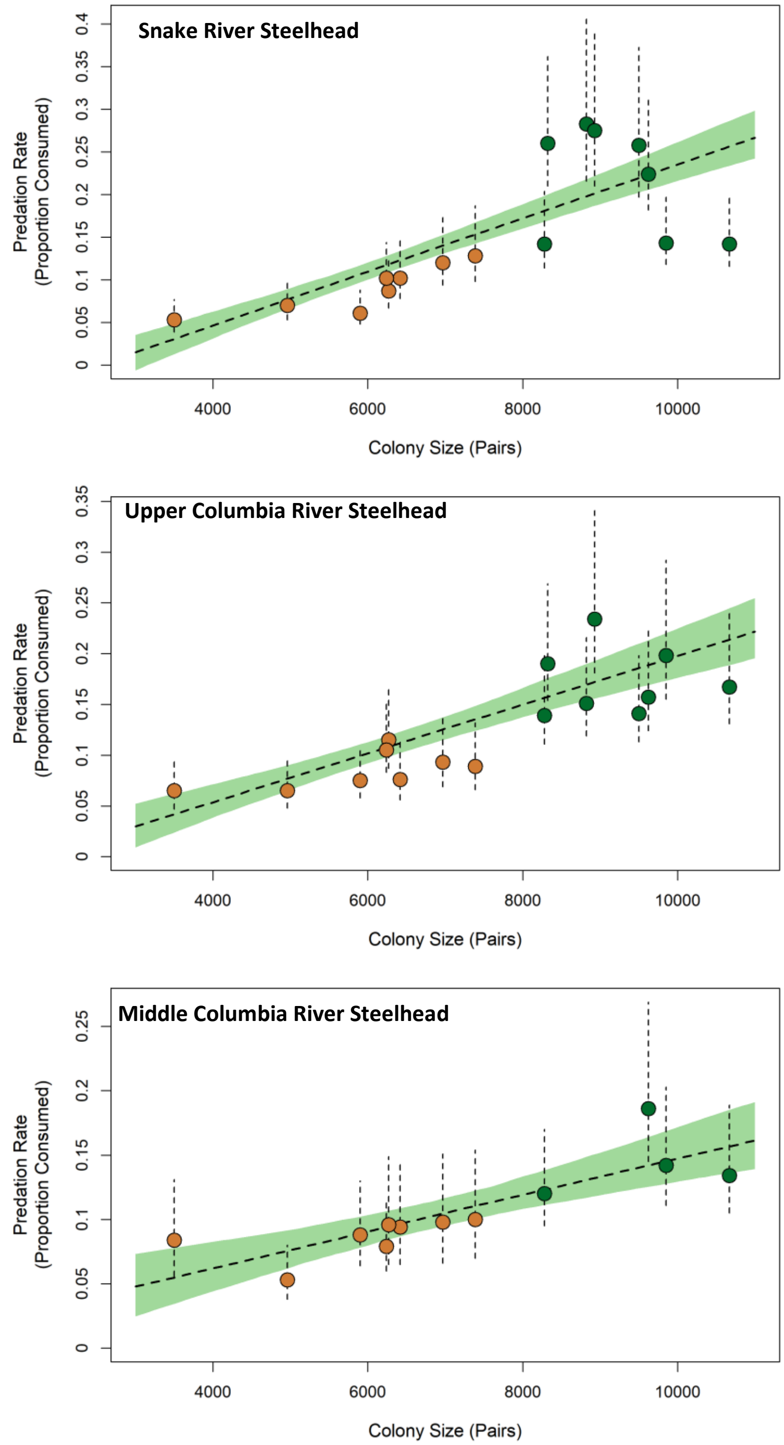

Figure 6. Estimated annual predation rate (95% credible intervals) and Caspian tern colony size (nesting pairs) prior to (green dots) and following (orange dots) management actions on East Sand Island. Predation rates are on PIT-tagged Upper Columbia River, Snake River, and Middle Columbia River steelhead last detected passing Bonneville Dam during 2003-2018 (for years with adequate sample sizes of tagged fish; see Methods).

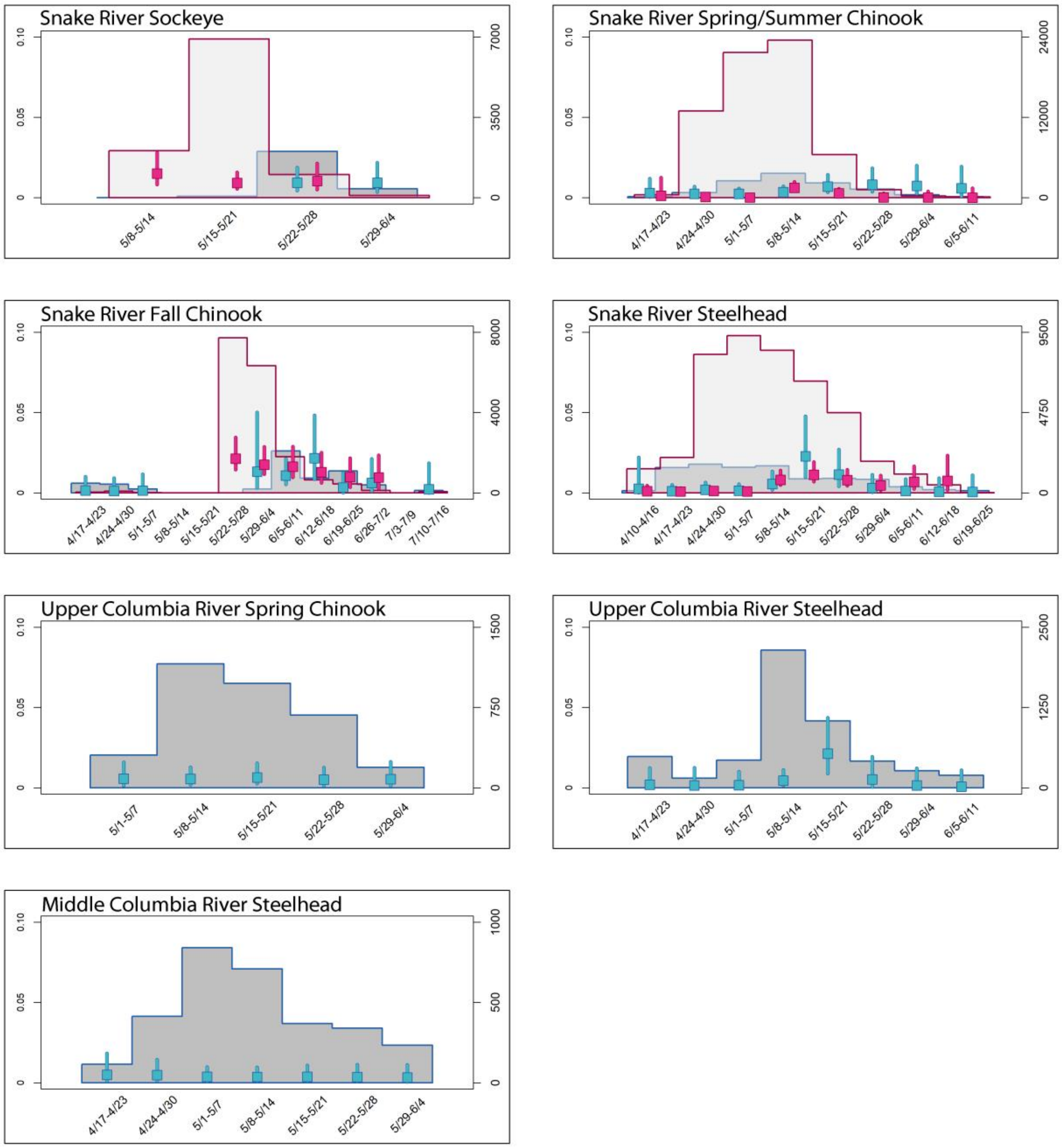

Figure 7. Estimated weekly predation rates (y_1; proportion of fish consumed) on in-river (blue squares) and transported (red squares) PIT-tagged juvenile salmonids last detected passing Bonneville or Sullivan dams (y_2; number available, dark gray bars) or transported from the lower Snake River (y^2; number available; light gray bars) by double-crested cormorants on East Sand Island during 2018. Error bars represent 95% credible intervals for predation rates.

TABLES

Table 1. Number of 2018 migration year PIT-tagged juvenile salmonids (Chinook salmon, coho salmon, sockeye salmon, and steelhead trout combined) recovered (electronic detections and physical removal; see Methods) on bird colonies on East Sand Island following the 2018 breeding season.

Location	Colony	Tags Recovered
East Sand Island	Caspian tern	10,886
	Double-crested cormorant	2,680
	Brandt’s cormorant	240

Table 2. Average deposition (95% credible interval) and detection (range; first-to-last week of nesting season) probability estimates for PIT tags on the East Sand Island Caspian tern and double-crested cormorant colonies in 2018. Results were used to estimate the proportion of PIT-tagged smolt consumed by birds that were deposited on their nesting site and the proportion of deposited tags subsequently detected by researchers after the nesting season (see Methods). Deposition estimates are those reported by Hostetter et al. (2015).

Location	Colony	Deposition	Detection
East Sand Island	Caspian terns	0.71 (0.51-0.89)	0.77 (0.57-0.97)
	Double-crested cormorants	0.51 (0.34-0.70)	0.88 (0.76-1.00)

Table 3. Estimated predation rates (95% credible interval) of PIT-tagged salmonid smolts last detected at Bonneville Dam on the Columbia River or Sullivan Dam on the Willamette River (In-river) or released from transportation barges (Transported) below Bonneville Dam by Caspian terns or double-crested nesting on East Sand Island in 2018. Predation rates were adjusted to account for tag loss due to on-colony PIT tag detection efficiency and deposition rates (see Table 2). The number (N) of in-river and transported PIT-tagged smolts and current U.S. Endangered Species Act (ESA) status of each evolutionarily significant unit (ESU) or distinct population segment (DPS) of PIT-tagged fish are provided. Only fish originating from and collected for transport on the Snake River (SR) were used in this analysis. Due to the limited use of East Sand Island by double-crested cormorants in 2018 (see Figure 4) predation rates on available smolts are biased low to unknown degree.

		N		Caspian terns		Double-crested cormorants	
ESU/DPS[1]	ESA[2]	In-river	Transported	In-river	Transported	In-river	Transported
SR Sockeye	E	2,546	10,087	4.2% (2.9-6.4)	1.5% (1.1-2.4)	0.9% (0.5-1.9)	1.1% (0.7-1.8)
SR Spr/Sum Chinook	T	11,174	66,723	1.4% (1.0-2.1)	1.1% (0.9-1.7)	0.5% (0.3-0.8)	0.3% (0.2-0.4)
UCR Spr Chinook	E	3,370	-	1.4% (0.9-2.3)	-	0.6% (0.3-1.2)	-
SR Fall Chinook	T	5,981	17,402	1.3% (0.7-2.1)	0.4% (0.3-0.7)	0.9% (0.5-1.6)	1.9% (1.3-3.0)
UWR Spr Chinook	T	76	-	-	-	-	-
SR Steelhead	T	9,572	44,241	6.9% (5.3-10.2)	6.1% (4.8-8.8)	0.5% (0.3-0.9)	0.5% (0.3-0.8)
UCR Steelhead	T	5,322	-	6.5% (4.8-9.7)	-	0.7% (0.4-1.4)	-
MCR Steelhead	T	3,209	-	5.3% (3.8-8.0)	-	0.4% (0.1-1.0)	-

[1] MCR = Middle Columbia River, SR = Snake River, UCR = Upper Columbia River, UWR = Upper Willamette River
[2] E = Endangered, T = Threatened

Table 4. Relative susceptibility of PIT-tagged smolts by rear-type and outmigration history to predation by Caspian terns and double-crested cormorants nesting on East Sand Island during 2006-2017 (average across all years) and in 2018. Values represent the odds-ratio of predation, with values < 1 indicating greater predation odds for hatchery fish and in-river fish and values > 1 indicating greater predation odds for wild fish and transported fish (see Methods). Dashed lines denote that insufficient sample sizes (< 500 PIT-tagged fish of each category) or extremely low rates of predation rates (< 1.0% of each category) prevented comparisons in 2018. Asterisk denotes a statistical significance difference. See Appendix C and D for weekly and year-specific results during 2006-2018. Salmonid populations included fish from the Snake River (SR) and Upper Columbia River (UCR), with runs of spring (Sp) and summer (Su) fish.

	Caspian Terns		Double-crested Cormorants	
	2006-2017	2018	2006-2015 [1]	2018
Hatchery versus Wild				
SR Sp/Su Chinook	0.37 (0.31-0.45)*	0.24 (0-0.82)*	0.91 (0.78-1.04)	-
UCR Sp Chinook	0.25 (0.1-0.4)*	-	0.64 (0.37-0.88)*	-
SR Steelhead	1.02 (0.94-1.12)	1.01 (0.67-1.41)	1.25 (1.11-1.39)*	-
UCR Steelhead	0.86 (0.7-1.02)	-	1.02 (0.73-1.25)	-
Transport versus In-river				
SR Sp/Su Chinook	0.85 (0.78-0.92)*	0.81 (0.48-1.12)	0.74 (0.66-0.82)*	-
SR Fall Chinook	0.92 (0.81-1.05)	0.81 (0.17-1.78)	1.39 (1.19-1.61)*	-
SR Sockeye	0.67 (0.45-0.94)*	0.47 (0.01-1.02)	1.23 (0.74-1.70)	-
SR Steelhead	0.87 (0.81-0.93)*	0.98 (0.68-1.33)	0.77 (0.67-0.89)*	-

[1] Cormorant predation rates in 2016 and 2017 were excluded from the time series due to mass dispersal events during the breeding season (see Methods)

Table 5. Average annual predation rates (95% credible intervals) by Caspian terns nesting on East Sand Island prior to and following periods of management. Salmonid populations (ESU/DPS) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated, where applicable. Asterisks denotes statistically credible differences between management periods (see Methods).

	Pre-management Period	Management Period
Salmonid ESU/DPS	2000-2010	2011-2018
Snake River Sockeye [1]	1.5% (0.9-2.2)	1.9% (1.4-2.4)
Snake River Spr/Sum Chinook	4.8% (4.3-5.4)	1.5% (1.3-1.8)*
Upper Columbia River Spr Chinook	3.9% (3.4-4.6)	1.6% (1.2-1.9)*
Snake River Fall Chinook	2.5% (2.2-3.0)	0.7% (0.6-0.9)*
Upper Willamette River Spr Chinook [2]	2.5% (1.9-3.3)	1.0% (0.6-1.4)*
Snake River Steelhead	22.2% (20.3-24.8)	9.2% (8.2-10.4)*
Upper Columbia River Steelhead [3]	17.2% (15.7-19.3)	8.7% (7.7-9.9)*
Middle Columbia River Steelhead [4]	14.9% (13.1-17.6)	8.9% (7.6-10.2)*

[1] Predation rate estimates were not available in 2000-2008 and in 2016-2017 due to insufficient sample sizes

[2] Predation rate estimates were not available in 2000-2006 and in 2017 due to insufficient sample sizes

[3] Predation rate estimates were not available in 2000-2002 due to insufficient sample sizes

[4] Predation rate estimates were not available in 2000-2006 due to insufficient sample sizes

Table 6. Average annual per capita (nesting pair) predation rates (95% credible intervals) by Caspian terns and double-crested cormorants nesting on East Sand Island. Salmonid populations (ESU/DPS) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated, where applicable.

Salmonid ESU/DPS	Caspian Terns	Double-crested Cormorants
	2000-2018	2003-2015,2018 [5]
Snake River Sockeye [1]	0.0002% (0.0002-0.0003)	0.0003% (0.0002-0.0005)
Snake River Sp/Su Chinook	0.0005% (0.0004-0.0005)	0.0005% (0.0003-0.0006)
Upper Columbia River Sp Chinook	0.0004% (0.0003-0.0005)	0.0003% (0.0003-0.0005)
Snake River Fall Chinook	0.0002% (0.0002-0.0003)	0.0003% (0.0002-0.0004)
Upper Willamette River Sp Chinook [2]	0.0002% (0.0002-0.0003)	0.0001% (0.0001-0.0002)
Snake River Steelhead	0.0022% (0.0020-0.0026)	0.0006% (0.0005-0.0008)
Upper Columbia River Steelhead [3]	0.0019% (0.0017-0.0020)	0.0005% (0.0004-0.0007)
Middle Columbia River Steelhead [4]	0.0015% (0.0013-0.0018)	0.0006% (0.0005-0.0009)

[1] Predation rate estimates were not available in 2000-2008 and 2016-2017 due to insufficient sample sizes

[2] Predation rate estimates were not available in 2000-2006 and 2017 due to insufficient sample sizes

[3] Predation rate estimates were not available in 2000-2002 due to insufficient sample sizes

[4] Predation rate estimates were not available in 2000-2006 due to insufficient sample sizes

[5] Predation rate estimates from 2016 and 2017 were excluded because cormorants dispersed from East Sand Island during the peak smolt outmigration period (see Results)

Table 7. Average annual predation rates (95% credible intervals) by double-crested cormorants nesting on East Sand Island prior to and following management. Salmonid populations (ESU/DPS) with runs of spring (Sp), summer (Su), and fall (Fall) fish were evaluated, where applicable. NA denotes insufficient sample sizes of available fish. Asterisks denotes statistically credible differences between management periods (see Methods).

	Pre-management Period	Management Period [1]
Salmonid ESU/DPS	2003-2015	2018
Snake River Sockeye [2]	3.6% (2.7-4.5)	0.9% (0.4-1.7)*
Snake River Spr/Sum Chinook	5.2% (4.4-6.1)	0.5% (0.2-0.8)*
Upper Columbia River Spr Chinook	3.1% (2.4-3.9)	0.6% (0.2-1.1)*
Snake River Fall Chinook	3.0% (2.6-3.6)	0.9% (0.5-1.5)*
Upper Willamette River Spr Chinook [3]	1.3% (0.5-1.8)	NA
Snake River Steelhead	9.3% (8.0-11.0)	0.5% (0.3-0.9)*
Upper Columbia River Steelhead [4]	5.1% (4.1-6.1)	0.7% (0.4-1.3)*
Middle Columbia River Steelhead [5]	8.3% (6.8-10.1)	0.4% (0.1-0.9)*

[1] Predation rate estimates in 2016 and 2017 were not included in averages because cormorants dispersed from East Sand Island en masse during the peak smolt outmigration period and consumed an unknown percentage of tagged fish in those years (see also Evans et al. 2018).

[2] Predation rate estimates were not available in 2000-2008 and in 2016-2017 due to insufficient sample sizes

[3] Predation rate estimates were not available in 2000-2006 and in 2017 due to insufficient sample sizes

[4] Predation rate estimates were not available in 2000-2002 due to insufficient sample sizes

[5] Predation rate estimates were not available in 2000-2006 due to insufficient sample sizes

APPENDIX A: HISTORICAL PREDATION RATES

This appendix provides annual PIT tag predation rate estimates for Caspian terns and double-crested cormorants nesting on East Sand Island during 2006-2018. Predation rate estimates were based on the number (N) of PIT-tagged fish interrogated passing Bonneville Dam or Sullivan Dam (in-river migrants; *Table A1* and *A2*) or the number released from barges downstream of Bonneville Dam (transported migrants; *Table A3*). Predation rates were corrected for PIT detection and deposition probabilities unique to each colony and year. Salmonid populations originating from the Snake River (SR), Upper Columbia River (UCR), Middle Columbia River (MCR) and Upper Willamette River (UWR) were evaluated, with runs of spring (Sp), summer (Su), and fall (Fall) fish included, where applicable.

Predation rate estimates dating back to 2000 are also available for some ESUs/DPSs and years, depending on sample sizes, and can be found in Evans et al. (2016a).

Table A1. Annual predation rates (95% credible interval) of PIT-tagged juvenile salmonid last detected (N) passing Bonneville or Sullivan dams by Caspian terns nesting on East Sand Island during 2006-2018. Dashes denote insufficient sample sizes (< 500 PIT-tagged fish were available) for generating predation rates.

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
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% (1-2.6)	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

Table A2. Annual predation rates (95% credible interval) of PIT-tagged juvenile salmonid last detected (N) passing Bonneville or Sullivan dams by double-crested cormorants nesting on East Sand Island during 2006-2018. Dashes denote insufficient sample sizes (< 500 PIT-tagged fish) for generating predation rates.

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
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 [1]	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 [1]	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	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

[1] *Due to colony dispersal events during the peak smolt outmigration period in 2016 and 2017, estimates are minimums (see Results).*

Table A3. Annual predation rates (95% credible interval) of PIT-tagged juvenile salmonid collected at Lower Granite Dam, Little Goose Dam, and Lower Monumental Dam on the Snake River and released from barges downstream of Bonneville Dam by double-crested cormorants and Caspian terns nesting on East Sand Island during 2006-2018. Dashes denote insufficient sample sizes (< 500 PIT-tagged fish) for generating predation rates. NA denotes that comparable rates of predation were not available that year (see Methods).

	Predation by Caspian terns				Predation by Double-crested cormorants			
Year	SR Sp/Su Chinook	SR Fall Chinook	SR Sockeye	SR Steelhead	SR Sp/Su Chinook	SR Fall Chinook	SR Sockeye	SR Steelhead
2006	4.0% (3.2-5.6)	1.8% (1.4-2.6)	-	22.7% (18.2-31.1)	4.9% (3.5-7.7)	1.7% (1.2-2.6)	-	8.1% (5.9-12.8)
N	78,532	48,661		70,988	78,532	48,661		70,988
2007	2.3% (1.8-3.4)	3.0% (1.6-5.5)	-	16.7% (13.4-24.5)	2.1% (1.4-3.3)	0.9% (0.1-3.4)	-	3.9% (2.7-6.1)
N	*32,184*	*607*		*45,276*	*32,184*	*607*		*45,276*
2008	4.2% (3.4-5.9)	1.6% (1.2-2.2)	-	18.7% (15.2-26.1)	3.9% (2.8-6.1)	5.3% (3.9-8.2)	-	6.0% (4.3-9.1)
N	*95,267*	*48,039*		*65,097*	*95,267*	*48,039*		*65,097*
2009	4.3% (3.5-6.3)	1.8% (1.4-2.6)	1.1% (0.8-1.6)	16.1% (13.1-23.1)	6.8% (4.9-10.3)	5.8% (4.2-8.9)	8.9% (6.4-13.8)	10.7% (7.8-16.8)
N	*51,805*	*34,407*	*10,167*	*22,627*	*51,805*	*34,407*	*10,167*	*22,627*
2010	3.6% (2.9-5.1)	0.9% (0.7-1.3)	-	14.9% (12.0-21.2)	4.7% (3.4-7.2)	5.3% (3.8-8.1)	-	9.4% (6.8-14.3)
N	*40,996*	*46,843*		*32,904*	*40,996*	*46,843*		*32,904*
2011	1.9% (1.5-2.7)	0.5% (0.4-0.8)	0.4% (0.2-0.7)	9.2% (7.3-13.0)	3.6% (2.6-5.6)	4.0% (2.9-6.2)	8.6% (6.2-13.5)	6.5% (4.8-10.1)
N	*64,858*	*53,093*	*7,038*	*26,862*	*64,858*	*53,093*	*7,038*	*26,862*
2012	2.4% (1.8-3.4)	1.0% (0.8-1.5)	1.0% (0.7-1.5)	8.2% (6.5-12.0)	2.7% (1.9-4.2)	6.6% (4.8-10.3)	6.2% (4.4-9.7)	4.4% (3.1-6.9)
N	*38,963*	*41,537*	*14,013*	*30,542*	*38,963*	*41,537*	*14,013*	*30,542*
2013	1.1% (0.8-1.6)	1.3% (0.6-2.5)	0.5% (0.3-0.9)	8.9% (6.8-13.3)	4.0% (2.9-6.3)	9.7% (6.6-15.5)	1.3% (0.8-2.1)	4.4% (3.2-6.8)
N	*49,592*	*2,106*	*9,280*	*32,490*	*49,592*	*2,106*	*9,280*	*32,490*
2014	1.1% (0.8-1.6)	0.9% (0.4-2.0)	0.8% (0.4-1.3)	9.5% (7.4-13.4)	8.4% (6.2-13.2)	4.4% (2.6-7.6)	7.6% (5.4-12.0)	8.5% (6.2-13.1)
N	*66,759*	*1,539*	*5,839*	*33,327*	*66,759*	*1,539*	*5,839*	*33,327*
2015	1.3% (1.0-2.0)	2.1% (1.6-3.1)	2.4% (1.7-3.6)	8.9% (7.0-12.8)	16.1% (11.7-24.8)	5.3% (3.8-8.3)	7.8% (5.4-12.3)	9.3% (6.7-14.5)
N	*20,575*	*8,347*	*4,357*	*10,461*	*20,575*	*8,347*	*4,357*	*10,461*
2016 [1]	0.8% (0.6-1.1)	1.1% (0.8-1.6)	5.9% (4.2-8.7)	11.3% (8.9-16.2)	6.3% (4.5-9.7)	0.1% (0-0.2)	0.2% (0-0.7)	2.9% (2.0-4.5)
N	43,068	10,948	2,829	13,608	43,068	10,948	2,829	13,608
2017 [1]	0.8% (0.6-1.3)	0.3% (0.2-0.5)	2.3% (1.3-4.0)	6.4% (5.0-9.2)	0.4% (0.3-0.7)	0.2% (0.1-0.3)	1.4% (0.6-3.0)	0.9% (0.6-1.4)
N	32,395	13,205	1,589	28,964	32,395	13,205	1,589	28,964
2018	1.1% (0.9-1.7)	0.4% (0.3-0.7)	1.5% (1.1-2.4)	6.1% (4.8-8.8)	0.3% (0.2-0.4)	1.1% (1.3-3.0)	1.1% (0.7-1.8)	0.5% (0.3-0.8)
N	66,723	17,402	10,087	44,241	66,723	17,402	10,087	44,241

[1] *Due to colony dispersal events during the peak smolt outmigration period in 2016 and 2017, estimates are minimums (see Results).*

APPENDIX B: DETECTION PROBABILITIES BY PIT TAG MODEL

Passive integrated transponder (PIT) tag types and models have changed over the years as the technology has improved. For instance, the number of frequencies available, sizes, transmission modes, and modulation rates of PIT tags used to mark juvenile salmonids in the Columbia River has changed since PIT tags were first developed to mark fish in the region in the 1990's (Prentice et al. 1990). Previous research indicates that the type of PIT tag used in avian predation studies, where tags are recovered on land as opposed to underwater, can influence detection probabilities (Ryan et al. 2003). Recently, *Biomark Inc.* has introduced an updated tag model, the Advanced Performance Tag (APT). The frequency (134 kHz), transmission mode (full duplex), and size (12 x 2 mm) of the APT is identical to the more commonly used High Performance Tag (HPT), but the APT tags have a slightly different rate of modulation, which is controlled by the microchip within the tag. According to the manufacturer, the modulation rate in APT could result in a greater read range (distance of detection between the tag and antenna), a difference that could improve detection probabilities on bird colonies where tag densities and collision rates can be high (Steve Angela, Biomark, personnel communication).

To investigate if the two tag models (HPT, APT) had similar detection probabilities on bird colonies we intentionally sowed equal numbers of both types on tags on East Sand Island in 2018 and compared rates of detection by colony site (Caspian tern, double-crested cormorant) and release date (pre-season, post-season). A brief summary of the methods and results of this ancillary, detection efficiency experiment is provided below.

Methods

The same methods used to sow tags as part of detection efficiency trails on East Sand Island in 2018 were used to sow APT tags (see *Methods, Detection and Deposition Probabilities* in the main body of this report) on East Sand Island. In brief, both HPT and APT PIT tags were intentionally sown on the Caspian tern and double-crested cormorant colonies immediately prior to and following the 2018 nesting season and the proportion of tags recovered after the nesting season used to determine rates of detection. Pre-season and post-season sowing dates were 15 April and 13 September on the tern colony and 30 March and 28 October on cormorant colony. Equal numbers of both tag models (n= 50 APT and 50 HPT) were sown on each bird colony (tern, cormorant) during each release period (pre-season, post-season). A proportions test was used to determine if differences in detection rates by tag model, colony, and release period were statistically significant.

Results

There was some evidence that the APT tags were more likely to be detected following the nesting season by researchers than HPT tags (*Table B1*). Rates of APT tag detection were slightly higher on both the Caspian tern and double-crested cormorant colonies compared with HPT tags, especially for tags sown prior to the nesting season in late March and April. No statistically significant difference in detection rates by tag model and release period, however, were detected on either the Caspian tern or double-crested cormorant colonies in 2018 (*Table B1*). Collectively, results suggest that APT tags will be detected at a similar, if not slightly higher, rates compared with HPT tags on bird colonies in the future, as APT tags start to replace HPT tags in the coming years.

Table B1. The number and proportion of PIT tags sown and subsequently recovered on the East Sand Island Caspian tern (tern) and double-crested cormorant (cormorant) colonies in 2018 by tag model (HPT, APT) and release period (pre-season, post-season). Statistical differences in the proportion of tags recovered by model and release period are also provided based on results of Proportions Test.

		Proportion (number) Recovered		Proportions Test
Colony	Release Period	HPT	APT	P-value
Tern	Pre-season	0.54 (27)	0.64 (32)	0.4161
	Post-season	0.96 (48)	1.00 (50)	0.4751
	Combined	*0.75 (75)*	*0.82 (82)*	*0.1116*
Cormorant	Pre-season	0.62 (31)	0.74 (37)	0.2838
	Post-season	1.00 (50)	1.00 (50)	1
	Combined	*0.81 (81)*	*0.87 (87)*	*0.3348*
All		0.78 (156)	0.85 (169)	0.1242

References

Prentice, E. F., T. A. Flagg, and C. S. McCutcheon. 1990. Feasibility of using implantable passive integrated transponder (PIT) tags in salmonids. Pages 317–322 *in* N. C. Parker, A. E. Giorgi, R. C. Heidinger, D. Jester Jr., E. D. Prince, and G. A. Winans, editors. Fish-marking techniques. American Fisheries Society, Symposium 7, Bethesda, Maryland.

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

APPENDIX C: HATCHERY VERSUS WILD COMPARISONS

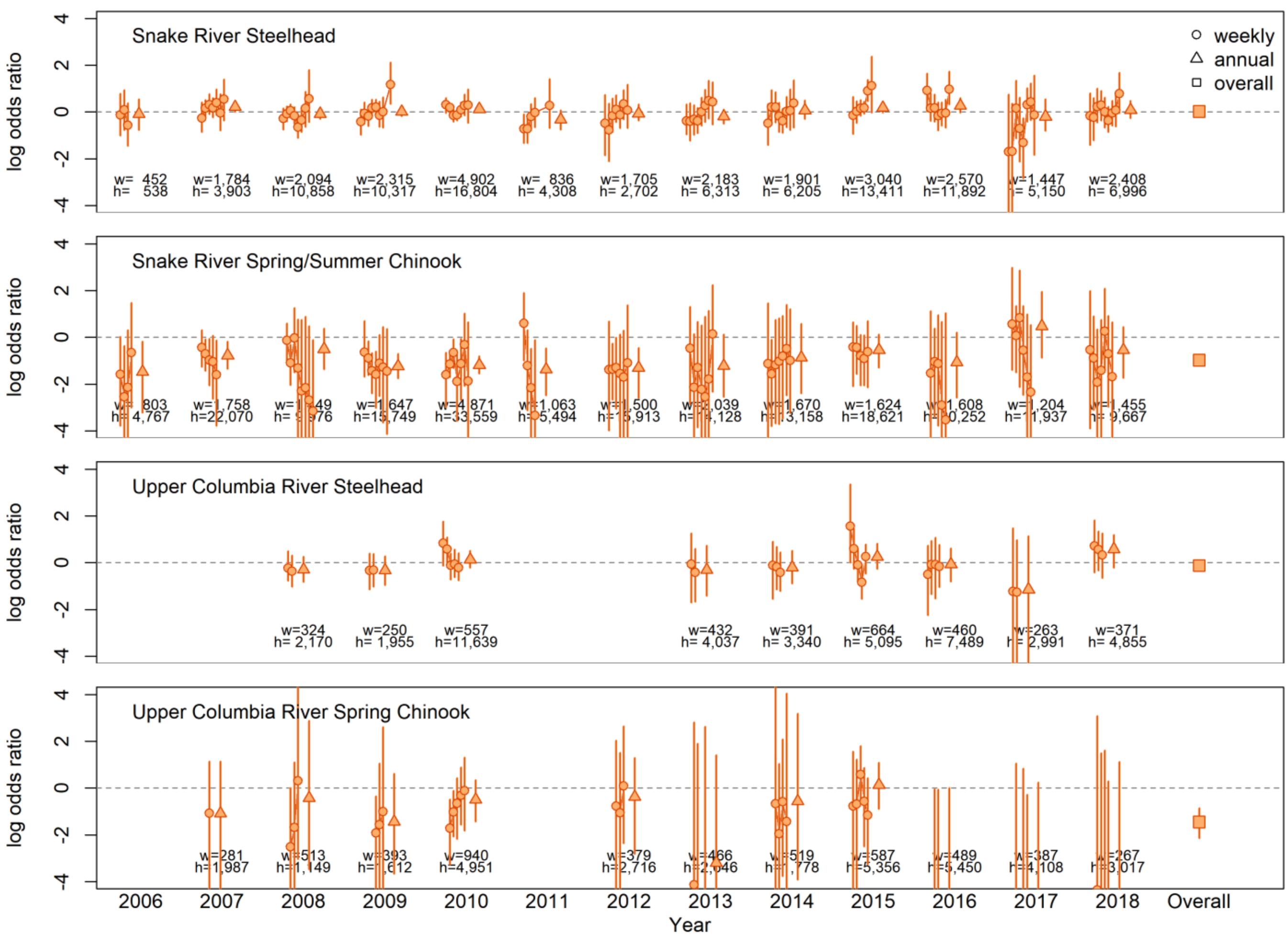

Figure C1. Relative susceptibility of salmonids among rearing types to predation by Caspian terns nesting on East Sand Island during 2006-2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for hatchery fish and values > 0 indicating greater predation odds for wild fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Only years were > 500 PIT-tagged fish of each rearing type available. Only weeks in which > 50 PIT-tagged fish of each rearing type available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right.

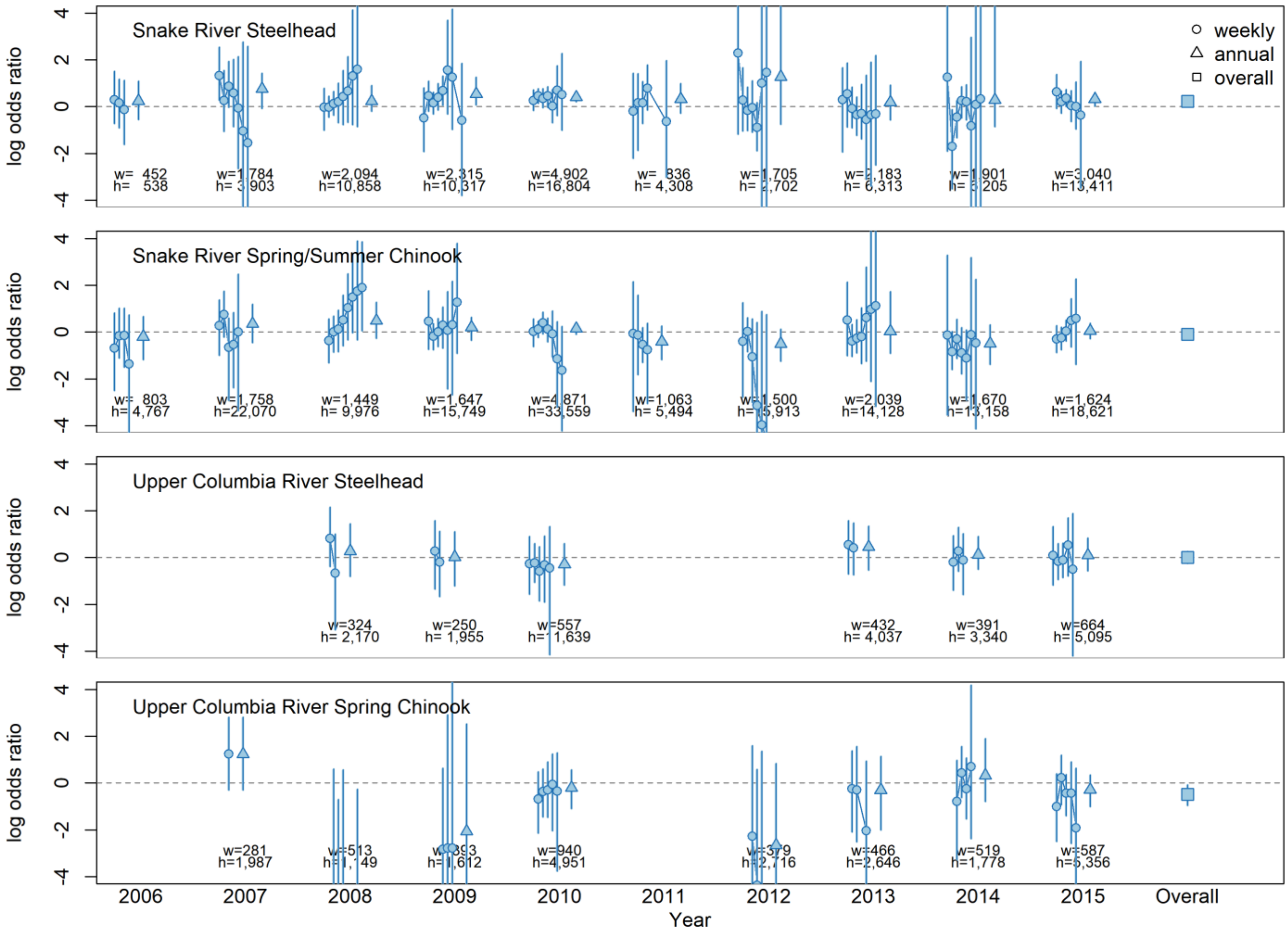

Figure C2. Relative susceptibility of salmonids among rearing types to predation by double-crested cormorants nesting on East Sand Island during 2006-2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for hatchery fish and values > 0 indicating greater predation odds for wild fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Only years were > 500 PIT-tagged fish of each rearing type available. Only weeks in which > 50 PIT-tagged fish of each rearing type available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right.

APPENDIX D: INRIVER VERSUS TRANSPORTED COMPARISONS

Figure D1. Relative susceptibility of salmonids among migration types to predation by Caspian terns nesting on East Sand Island during 2006-2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for in-river fish and values > 0 indicating greater predation odds for transported fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Only years were > 500 PIT-tagged fish of each migration history available. Only weeks in which > 50 PIT-tagged fish of each migration history available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right.

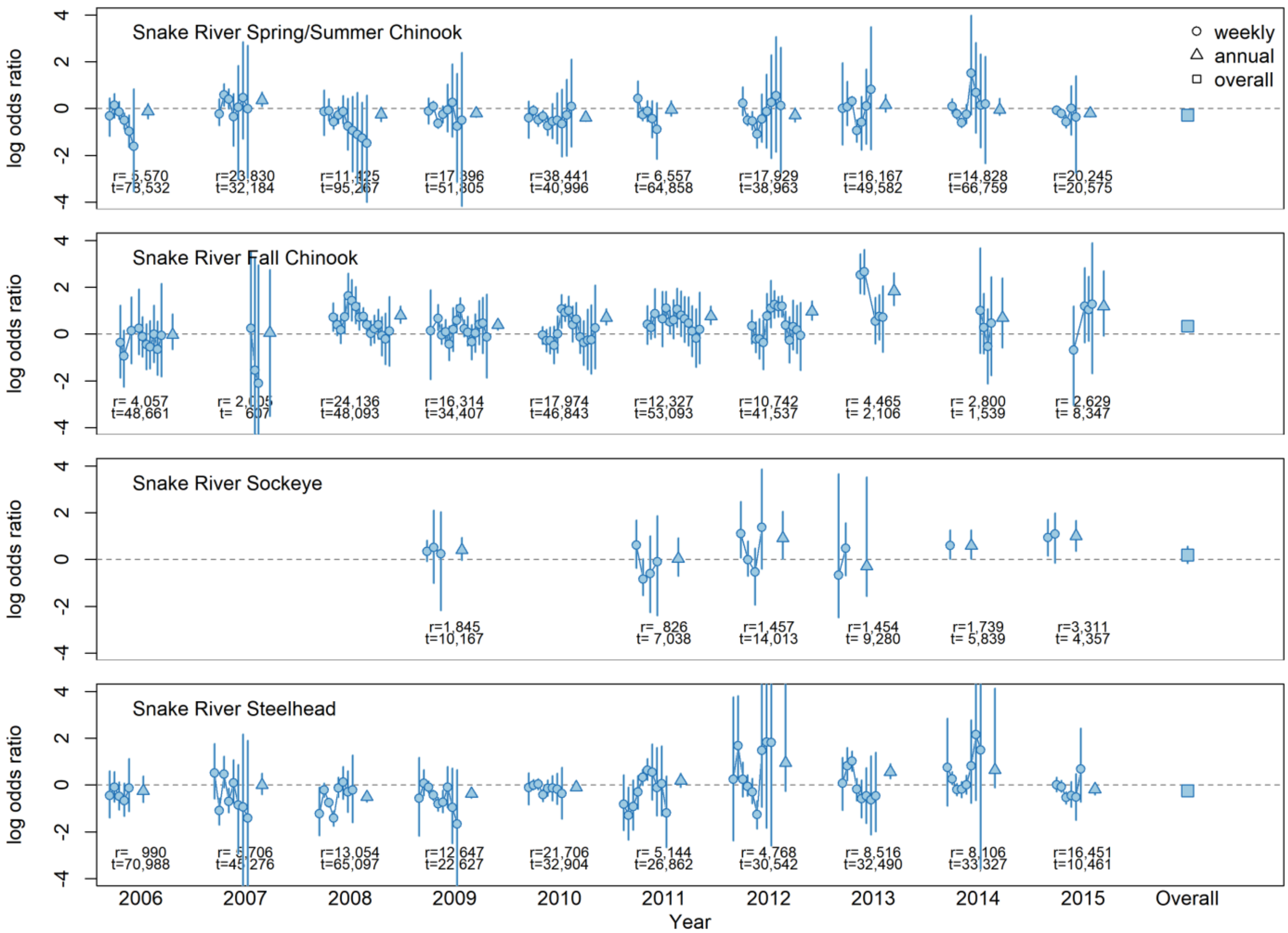

Figure D2. Relative susceptibility of salmonids among migration types to predation by double-crested cormorants nesting on East Sand Island during 2006-2018. Values represent the log odds ratio of predation (y_1), with values < 0 indicating greater predation odds for in-river fish and values > 0 indicating greater predation odds for transported fish. Error bars represent 95% credible intervals, with uncertainty ranges over-lapping 0 associated with relative differences that were not considered statistically significant. Only years were > 500 PIT-tagged fish of each migration history available. Only weeks in which > 50 PIT-tagged fish of each migration history available are included in the plot. Within each year weekly estimates (circles) are followed by an annual estimate (triangles) with the overall log-odds ratio estimate (square) presented on the far right.