

FINAL TECHNICAL REPORT: Predation Impacts on Juvenile Salmonids by Double-crested Cormorants and Caspian Terns Nesting on East Sand Island in the Columbia River Estuary

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Predation Impacts on Juvenile Salmonids by Double-crested Cormorants
and Caspian Terns Nesting on East Sand Island in the
Columbia River Estuary

2015 Final Technical Report

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EXECUTIVE SUMMARY

As a component of a comprehensive strategy for salmonid (*Oncorhynchus* spp.) recovery in the Columbia River Basin, management plans have been developed to reduce the impacts of double-crested cormorants (*Phalacrocorax auritus*) and Caspian terns (*Hydroprogne caspia*) nesting on East Sand Island on the survival of juvenile salmonids listed under the Endangered Species Act (ESA). Management initiatives are currently being implemented to reduce the size of the East Sand Island colonies through primarily lethal strategies (i.e., culling and egg oiling) for double-crested cormorants and dispersal of nesting birds to areas outside the Columbia River Basin for Caspian terns. The primary goal of this study was to estimate predation rates (percentage of available tagged fish consumed by birds) based on recoveries of passive integrated transponder (PIT) tags implanted in juvenile salmonids on the double-crested cormorant and Caspian tern colonies on East Sand Island. More specifically, the objective was to generate population-specific (salmonid evolutionary significant units [ESU] or distinct population segments [DPS]) predation rates on ESA-listed juvenile salmonids from the Columbia River Basin.

Predation rate estimates generated as part of this study integrated multiple factors of uncertainty in the tag recovery process, including imperfect detection of PIT tags on bird colonies, on-colony PIT-tag deposition probabilities, and temporal changes in fish availability to predators nesting on East Sand Island. Predation rates were used to compare and contrast impacts within a given fish population based on the fish's rear-type (hatchery, wild) and migration history (in-river, transported). Predation rates were also used to compare and contrast smolt losses prior to and following bird management actions on East Sand Island; data critical in evaluating the effectiveness of management plans aimed at reducing predation rates. We also evaluated the relationship between predation rates and various biotic and abiotic conditions in the estuary that have potentially influenced these rates during 2000-2015. The goal of this analysis was to describe the influence of "management relevant" variables on predation rates, variables that resource managers may be able to control to some degree.

Predation rates indicated that impacts by double-crested cormorants on ESA-listed juvenile salmonids in 2015 were some of the highest ever recorded, with estimates of 14.5% (95% creditable interval [CI] = 10.5–22.4%) and 12.8% (95% CI = 9.3–19.6%) for Snake River spring/summer Chinook salmon (*O. tshawytscha*) and Snake River steelhead (*O. mykiss*), respectively. Impacts on salmon ESUs were comparable to those of steelhead DPSs in 2015, although the lowest rates were observed on two salmon populations (Snake River sockeye salmon *O. nerka* and upper Willamette River spring Chinook salmon, with < 2.5% of available fish consumed by cormorants in 2015). An investigation of temporal trends in predation rates indicated that double-crested cormorants consumed smolts in proportion to their relative availability, with the highest predation rates observed in May, when the largest numbers of PIT-tagged fish were available in the estuary. An investigation of predation rates dating back to 2000 indicated that smolt losses to double-crested cormorants were substantial in most years but highly variable over time. For example, annual predation rates by East Sand Island double-

crested cormorants on Snake River steelhead have ranged from 1.9–16.6% for that particular DPS during 2000-2015. In comparison to smolt losses by double-crested cormorants, results indicated that Brandt's cormorants (*Phalacrocorax penicillatus*) nesting on East Sand Island posed little risk to smolt survival in the estuary, with predation rates $\leq 0.6\%$ for each salmonid ESU/DPS evaluated in 2015. Despite a steady increase in the size of the Brandt's colony since it was first established on East Sand Island in 2006, predation rate estimates have remained below 1.0% for each salmonid ESU/DPS in all years.

Predation rates indicated that impacts by Caspian terns nesting on East Sand Island on ESA-listed juvenile salmonids were generally lower in 2015 compared with years past, with estimates ranging from 0.4% (95% CI = 0.1–1.5%) to 10.5% (95% CI = 8.2–15.0%) in Upper Willamette River spring Chinook and Upper Columbia River steelhead, respectively. Of those ESUs/DPSs evaluated, steelhead DPSs were predated at significantly higher rates than salmon ESUs. For instance, predation rates were 10.2% (95% CI = 8.2–14.6%) for Snake River steelhead but just 0.8% (95% CI = 0.2–1.5%) for Snake River fall Chinook in 2015. The finding that Caspian terns disproportionately consumed steelhead DPSs compared with salmon ESUs has been consistent since research was initiated on East Sand Island Caspian tern colony in 2000. Contrary to temporal results from double-crested cormorants, Caspian tern predation rates were the lowest when PIT-tagged smolt availability in the estuary was the highest; a finding consistent with predator-swamping, whereby an individual fish's susceptibility to tern predation decreases when larger numbers of fish were available in the estuary.

If management is successful in significantly reducing the number of double-crested cormorants and Caspian terns that reside in the Columbia River Estuary during the spring/summer smolt outmigration, it is expected that there will be a commensurate reduction in smolt mortality. In 2015, the first year of double-crested cormorant management in the Columbia River Estuary, actions to reduce the number of double-crested cormorants nesting on East Sand Island were implemented after the peak of the smolt outmigration period. The percent of adult cormorants culled (ca. < 1% of adults) was also not large enough to have an effect in reducing cormorant predation. There was some evidence that management efforts to reduce the number of Caspian terns nesting on East Sand Island are associated with lower predation rates on ESA-listed salmonid populations. For instance, predation rates by East Sand Island terns on Upper Columbia River steelhead averaged 17.2% (95% CI = 15.7–19.3%) during 2000-2010, but were 9.9% (95% CI = 8.5–12.0%) during 2011-2015, associated in part with reductions in colony size due to tern nesting habitat restrictions on East Sand Island during the latter period. Despite the apparent reductions in Caspian tern predation associated with management, the target colony size for Caspian terns in the Columbia River Estuary has not been met, so the gains in survival of ESA-listed juvenile salmonids associated Caspian tern management have not been fully realized.

A relative comparison of predation impacts based on the fish's rearing-type (hatchery, wild) indicated that hatchery and wild fish were equally susceptible to double-crested cormorant predation in the Columbia River estuary, with no consistent predation trend or preference by rear-type identified during 2006-2015. Unlike cormorants, there was evidence that Caspian

terns disproportionately consumed hatchery spring/summer Chinook salmon compared with their wild counterparts during 2006-2015. Comparisons of predation impacts by a fish's migration history (in-river, transported) also indicated differences in the relative susceptibility of smolts to bird predation, with transported Snake River fall Chinook and transported Snake River sockeye salmon disproportionately consumed by double-crested cormorants compared with in-river migrants during 2006-2015. There was also some evidence that in-river steelhead and in-river spring/summer Chinook were disproportionately consumed by Caspian terns and double-crested cormorants compared with transported fish but results were not consistent across all weeks and years. Data from this and other studies suggest the difference in fish susceptibility to bird predation is related to several behavioral and physical traits, including the size and condition of fish, the run-timing of fish, the abundance of fish, and predator-specific foraging techniques and behaviors. Additional research is needed to better understand these factors and how (or if) these factors can be managed to decrease predation rates on juvenile salmonids by double-crested cormorants and Caspian terns in the Columbia River estuary.

An evaluation of predation impacts from both double-crested cormorants and Caspian terns (i.e., cumulative predation rates) indicated that birds nesting on East Sand Island have annually consumed between 1.5–16.5% of available salmon per ESUs and between 12.0–40.6% of available steelhead per DPS during 2000-2015. Losses were comparable to or greater than those of other documented sources of smolt mortality (e.g., dam passage, predation by piscine predators) in the Columbia River Basin. Furthermore, impacts from bird predation in the estuary are on juvenile salmonids that have survived freshwater outmigration, including passage through the Federal Columbia River Power System, fish that have a higher probability of surviving than those that have yet to complete outmigration. Additionally, juvenile salmonids belonging to every ESA-listed ESU/DPS from the Columbia River Basin must pass through the Columbia River Estuary and are therefore susceptible to predation by birds nesting on East Sand Island. Despite recent reductions in the numbers of Caspian terns nesting on East Sand Island and predation rates by these birds on juvenile salmonids, the total number of birds (terns and cormorants combined) nesting on East Sand Island has remained fairly constant since 2000, as have average annual predation rates at the level of steelhead DPS and salmon ESU.

A multivariate analysis of factors influencing predation rates indicated that colony size was just one of several factors that explained variation in predation rates by Caspian terns and double-crested cormorants nesting on East Sand Island during 2000-2015. Fluctuations in large-scale climate indices (Multivariate El-Nino Index, North Pacific Gyre Oscillation), river operational strategies (spill, discharge), smolt abundance, and other factors were also important variables; demonstrating that a fish's susceptibility to bird predation in the estuary is determined by a complex set of interacting factors. Results suggest that reductions in colony size will have to be both large and sustained before management goals to reduce avian predation can be fully realized in the Columbia River estuary.

BACKGROUND

Predation on juvenile salmonids during outmigration to the Pacific Ocean is considered a limiting factor in the recovery of salmonid populations from the Columbia River Basin that are listed under the U.S. Endangered Species Act (ESA; NOAA 2008, 2010). Studies of avian predation in the Columbia River Basin have mostly focused on colonial waterbirds nesting in the estuary (Collis et al. 2001; Roby et al. 2003; Ryan et al. 2003; Lyons 2010), where the largest known colonies of Caspian terns and double-crested cormorants in western North America currently reside (Lyons 2010). Previous research has demonstrated that cormorants and terns nesting on East Sand Island in the Columbia River Estuary consume millions of juvenile salmonids annually (Lyons 2010), including salmonids listed under the ESA.

Since 1987, PIT tags have been placed in juvenile salmonids from the Columbia River Basin to study their behavior and survival following release. PIT tags can provide specific information on individual fish, including species (steelhead, Chinook, coho, sockeye), run-type (spring, summer, fall, winter), rear-type (hatchery, wild), and release information (date, location). Post-release interrogations of individual PIT-tagged fish can provide information on migration timing, migration route (in-river, transported), and survival based on detections of live fish passing hydroelectric dams or other in-river PIT tag interrogation sites. PIT tag interrogations can also be used to assess mortality of individual fish based on detections of tags from fish that succumbed to harvest or predation. Comparisons of post-release interrogations among different groups of PIT-tagged fish (e.g., species, runs, rear-types, etc.) can allow for a thorough evaluation of fish survival and mortality, thereby providing for in depth action effectiveness monitoring of fish management initiatives and strategies (Hostetter et al. 2015; Evans et al. *in press*).

Beginning in 1996, recoveries of PIT tags on bird colonies in the Columbia River Basin have been used to calculate predation rates (percentage of available tagged fish consumed by birds) and to compare and contrast the relative susceptibility of different fish populations to avian predators (Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005, Maranto et al. 2010, Evans et al. 2012, Hostetter et al. 2012, Sebring et al. 2013, Hostetter et al. 2015, Evans et al. *in press*). These studies have been conducted at piscivorous waterbird colonies located throughout the Columbia River Basin, both in the Columbia River estuary and in the Columbia Plateau region. A recent evaluation of avian predation on juvenile salmonids revealed that Caspian terns and double-crested cormorants nesting on East Sand Island consumed large proportions of available PIT-tagged ESA-listed salmonids, with upwards of 10% and 20% of ESA-listed Chinook and steelhead populations, respectively, consumed by double-crested cormorants and Caspian terns in some years (Lyons et al. 2014b; Hostetter et al. 2015; BRNW 2015). Although certain salmonid species and populations may also be susceptible to avian predation in the Columbia Plateau region (e.g., Upper Columbia River steelhead to Caspian terns nesting on Goose Island in Potholes Reservoir; Evans et al. 2012), avian predation in the Columbia River estuary affects juvenile salmonids belonging to every ESU/DPS from the Columbia River Basin, fish that have

survived freshwater migration and have a higher probability of survival to adulthood compared to those fish that have yet to complete outmigration (Roby et al. 2003).

Resource management agencies working in the Columbia River Basin recognize the importance of addressing avian predation in efforts to restore ESA-listed salmonids (NOAA 2008, 2010). Plans to recover ESA-listed salmonids have been developed by the United States government, and specifically call for developing strategies to manage avian predation as a means to bolster juvenile salmonid survival (USACE 2014; USACE 2015a). Many of these strategies were developed in response to Reasonable and Prudent Alternatives (RPA) specified in Biological Opinions issued by NOAA Fisheries, specifically RPAs 66 and 67 in the Supplement Federal Columbia River Power System Biological Opinion (NOAA 2014). Two avian predation management plans are currently underway in the Columbia River Estuary, entitled “*Caspian Tern Management to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary*” (USFWS 2005, 2006) and “*Double-crested Cormorant Management Plan to Reduce Predation on Juvenile Salmonids in the Columbia River Estuary*” (USACE 2015a). These management plans aim to reduce the number of Caspian terns and double-crested cormorants nesting in the Columbia River estuary, thereby reducing predation impacts and increasing the survival of juvenile salmonids. Colony reductions will be primarily through lethal (i.e., culling and egg oiling) strategies for double-crested cormorants and the non-lethal dispersal of Caspian terns to alternative colony sites outside the Columbia River Basin.

It is well documented that biotic and abiotic conditions can play an important role in the survival of juvenile salmonids during outmigration to the ocean and after ocean entry (e.g., Petrosky and Schaller 2010, Burke et al. 2013, Peterson and Burke 2013; Evans et al. 2014). While levels of tern and cormorant predation on some populations of juvenile salmonids have been high on average, there has been substantial intra- and inter-annual variability in predation impacts (Lyons et al. 2014b). For instance, predation rates on the same salmonid population can vary significantly by week (BRNW 2015) and year (Evans et al. 2012). Furthermore, even within the same salmonid population, differences in predation probabilities based on a fish’s rear-type (hatchery, wild), migration history (in-river, transported), and run-timing have been observed (Ryan et al. 2003; Zamon et al. 2013; Lyons et al. 2014b; BRNW 2015). An exploratory analysis of the effects of biotic and abiotic conditions has previously been conducted for double-crested cormorants at East Sand Island (Lyons et al. 2014b), but is lacking for Caspian terns. Additionally, this previous cormorant analysis focused on the effects of strictly environmental variables after considering just one management relevant variable, colony size. An analysis investigating the potential effects of a more inclusive set of management relevant variables might identify additional factors that influence smolt susceptibility to avian predation.

The primary objective of this study was to generate population-specific (ESU/DPS) predation rates on ESA-listed juvenile salmonids by double-crested cormorants and Caspian terns nesting on East Sand Island in 2015 and to compare and contrast those estimates with results from years past. To ensure predation rate estimates generated from data collected in 2015 were comparable to estimates from years past, we used the same PIT tag predation rate modeling

techniques used in years past, those of Evans et al. (2012) and Hostetter et al. (2015). These methods integrated multiple factors of uncertainty in the tag recovery process, including imperfect detection of tags on bird colonies, on-colony tag deposition probabilities that vary by bird species, and temporal changes in fish availability to predators nesting on East Sand Island. A secondary objective was to evaluate relative differences in predation impacts by terns and cormorants nesting on East Sand Island based on a fish's rear-type (hatchery, wild) and migration history (in-river, transported). Predation rates were used to compare and contrast smolt losses prior to and following bird management actions on East Sand Island; data critical in evaluating the effectiveness of management plans aimed at reducing predation rates. Finally, as part of an exploratory analysis, we investigated the relationship between predation rates and various biotic and abiotic conditions (covariates) in the estuary using data collected since 2000. These results may be important in describing natural variation in predation impacts and identifying factors that influence a fish's susceptibility to bird predation in the estuary.

METHODS

Recovery of PIT tags on East Sand Island

Cormorant colonies: Following previously established field methods (Evans et al. 2012; BRNW 2015), portable PIT tag detection systems (*Biomark*, model HPR; [Figure 1](#)) were used to recover PIT tags *in situ* on the East Sand Island double-crested cormorant and Brandt's cormorant (*P. penicillatus*) colonies after birds dispersed from their colonies from 11-23 October 2015. Recovery of tags on the Brandt's cormorant colony, a non-managed species on East Sand Island, was necessary because Brandt's cormorants nested amongst double-crested cormorants and efforts to delineate tags deposited by the two species (Brandt's, double-crested) 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). Portable PIT tag antennas were optimized to detect ISO FDXB tags implanted in juvenile salmonids from the Columbia River Basin. PIT tags were recovered by systematically scanning the entire area (referred to as a "pass") occupied by nesting double-crested cormorants and Brandt's cormorants during the breeding season, with a total of three passes conducted. The orientation or directionality of antennas relative to nesting substrate was changed following each pass, a technique that results in higher detection efficiency (Ryan et al. 2003).

Aerial imagery (Smith and Associates 2016) taken during the nesting season and ground-surveys (this study) conducted immediately after the nesting season were used to distinguish double-crested cormorant nesting areas from Brandt's cormorant nesting areas on East Sand Island in 2015 ([Figure 2](#)). This was possible because the two species used different nesting material, material identifiable in aerial imagery and by researchers surveying the colony area immediately after the nesting season. Due to temporal and spatial overlap between double-

crested and Brandt's cormorant nesting areas in 2015, only nesting areas exclusively used by Brandt's cormorants during the nesting season were scanned and assigned to Brandt's cormorants. Furthermore, small groups of Brandt's cormorant nests (aggregations of 20 or fewer nests) were not scanned in isolation due to their proximity to double-crested cormorant nests (BRNW 2015). For instance, the irregular distribution of small groups of Brandt's nests does not allow detected PIT tags to be assigned to an individual cormorant species (Brandt's, double-crested) given the detection range of antennas can over-lap adjacent nests. As a result of these criteria, PIT tags detected in the areas where both species overlapped or were small groups of Brandt's nests existed, were assigned to the larger double-crested cormorant colony.

Tag codes stored locally on each PIT tag detection system were transferred to a central storage drive at the completion of each scanning session, along with metadata regarding the date, species (double-crested cormorant, Brandt's cormorant), and pass number. At the conclusion of each scanning day, tag data was uploaded to a cloud-based server for redundancy. Following validation and removal of duplicate records, newly detected tag codes on the cormorant colonies were uploaded to the PIT Tag Information System (PTAGIS) using guidelines and protocols established by the PIT-tag Steering Committee (PSFMC 2015).

Caspian tern colony: Following previously established methods (Evans et al. 2012; Zamon et al. 2013; BRNW 2015), an eight coil flat-plate PIT tag detection system attached to an ATV (*Figure 1*) was used to detect PIT tags *in situ* on the East Sand Island Caspian tern colony after birds dispersed from the breeding colony from 15-28 September 2015. Analogous to the system used to detect tags on cormorant colonies, the eight coil flat-plate system was optimized to detect ISO FDXB tags. PIT tags were detected by systematically scanning the entire area occupied by nesting terns during the breeding season (*Figure 2*), with nine complete passes conducted. Passes were conducted in varying directions, but at a consistent speed and antenna height to optimize the antenna performance (< 2" above the underlying substrate with the ATV operated at a constant speed of < 5 mph). In total, the flat-plate antenna coils create a detection field that measured 48" x 12". To ensure the colony area was completely scanned each pass, ATV tread marks were used to guide overlapping transects. Additionally, portable PIT tag detection systems, the same units used on the cormorant colonies, were used to recover PIT tags in areas inaccessible to the ATV (e.g., areas adjacent to dissuasion fencing and vegetated habitat; see *Figure 2*).

In addition to electronic detection of PIT tags using the flat-plate antenna system, PIT tags were also physically removed from the Caspian tern colony using a tow behind sweeper magnet attached to the ATV (*Bluestreak*, Hog Series; *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 where tag densities are high, like on the East Sand Island Caspian tern colony (BRNW 2015). Both physical and electronic PIT tag recovery were conducted concurrently, when conditions permitted (i.e., use of magnet required dry substrate).

Tag codes stored locally on the flat-plate system's transceiver (*Biomark*, IS1001 MC) were transferred to a central storage drive at the completion of each scanning session, along with metadata regarding the date and pass number. At the conclusion of each scanning day, tag data was uploaded to a cloud-based server for redundancy. Following validation and removal of duplicate records, newly detected tag codes on the tern colony, including codes from tags physically removed with the sweeper magnet, were uploaded to PTAGIS using guidelines and protocols established by the PIT-tag Steering Committee (PSFMC 2015). Physically recovered PIT tags were transferred to the funding agency following completion of the contract.

Predation Rate Calculations

Following previously established methods (Hostetter et al. 2015; BRNW 2015), a Bayesian hierarchical model was used to estimate predation rates based on recoveries of smolt PIT tags on the East Sand Island double-crested cormorant, Brandt's cormorant, and Caspian tern colonies. The Bayesian model simultaneously incorporates uncertainty due to on-colony detection probabilities and on-colony deposition probabilities (see [Figure 3](#) for conceptual illustration), resulting in a more accurate and defensible estimate of predation rates (Hostetter et al. 2015). The Bayesian predation model used in 2015 was the same model used to quantify predation rates on PIT-tagged smolts in 2014 (BRNW 2015) and the same model used in the Affected Environment Analysis in the Double-crested Cormorant Management Plan in the Columbia River Estuary (USACE 2015a).

Smolt Availability: Smolt availability to birds nesting on East Sand Island were based on interrogations of live PIT-tagged fish detected 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 migrating fish". Bonneville and Sullivan dams are the nearest upstream dams to East Sand Island with adequate smolt PIT tag interrogation capabilities (Hostetter et al. 2015; BRNW 2015) and the location of each dam is considered the upper most reaches of the Columbia River Estuary as defined by the USACE for the purposes of evaluating avian predation rates (USACE 2015a; [Map 1](#)). In addition to in-river migrants, PIT-tagged smolts that were loaded into barges at dams on the lower Snake River, 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". Smolt availability of transported fish were based on fish interrogated or tagged at the Lower Granite Dam (Rkm 695), Little Goose Dam (Rkm 635), or Lower Monumental Dam (Rkm 589) Juvenile Bypass Systems (JBS). Fish were classified as being collected for transportation based on a unique combination of the interrogation site (e.g., detected entering a raceway) and date at each JBS. Downstream interrogation histories, weekly JBS facility reports, and other sources (e.g., NOAA, USACE and FPC Technical Reports) were used to validate and otherwise proof classifications to ensure accurate assignment (in-river, transported). Due to small numbers of PIT-tagged fish and the use of a different transportation method, fish loaded into trucks at each JBS during the study period were not included in the study.

Availability of in-river fish to bird predation on East Sand Island was considered annually during 2000-2015, the historic period of PIT tag recovery on East Sand Island. Availability of transported fish to bird predation was considered annually during the last decade (2006-2015), a time period that reflects contemporary transportation scenarios for smolts based on operational guidelines for fish passage in Federal Columbia River Power System (NOAA 2008). For both in-river and transported fish, smolt availability within a given year was defined as those fish last detected or released (transported fish) from 1 March to 31 August, which reflects the periods of overlap in active smolt out-migration and cormorant and tern nesting activity on East Sand Island (Evans et al. 2012; Adkins et al. 2014). PIT-tagged fish were grouped by ESA-listed salmonid population or ESU/DPS, representing a unique combination of the species (steelhead trout, Chinook salmon, sockeye salmon), run-type (spring, summer, fall, or winter), and river-of-origin (Upper Columbia River, Middle Columbia River, Snake River, or Willamette River). The designation of ESUs/DPSs follows that of NOAA (2014) and was largely based on the rearing (if hatchery) and release location of tagged fish relative to the geographic boundary of each ESU/DPS in the Columbia River Basin. Fish within each ESU/DPS were further grouped by rear-type (hatchery/wild), year, and week (see [Predation Rate Calculations](#) below). Interrogation data were retrieved from PTAGIS, a database maintained by the Pacific States Marine Fisheries Commission (www.ptagis.org).

Not all ESA-listed ESUs/DPSs in the Columbia River Basin were included in predation rate analyses, as four populations originate wholly or partially below Bonneville and Sullivan dams and were thus excluded because temporal and spatial interrogation records of live fish were not available, these ESUs/DPSs were: (1) Lower Columbia River steelhead trout, (2) Lower Columbia River Chinook salmon, (3) Lower Columbia River coho salmon *O. kisutch*, and (4) Columbia River chum salmon *O. keta*. In addition to ESA-listed salmonids, non-listed juvenile salmonids and other fishes (e.g., Pacific lamprey *Lampetra tridentate*, Eulachon *Thaleichthys pacificus*) were available as prey to double-crested cormorants and Caspian terns; fish that are of cultural, economic, and/or conservation concern (Lyons et al. 2013). Including these other fishes was beyond the scope of this study but efforts to reduce the number of double-crested cormorants and Caspian terns nesting on East Sand Island will presumably also benefit these other fishes (USACE 2015a).

Deposition and Detection Probabilities: 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 double-crested cormorants and Caspian terns and subsequently recovering those tags on East Sand Island (Hostetter et al. 2015). Deposition probabilities measured during these experiments (2004-2006 for Caspian terns and 2012-2013 for double-crested cormorants) were used to infer deposition probabilities for data collected in 2015 and other years where retrospective analyses were conducted as part of this study. The distribution

of the median deposition probability derived from these studies (cormorant = 0.51; 95% credible interval [CI] = 0.34–0.70; tern = 0.71; 95% CI = 0.51–0.89) was applied to data in all study years. Use of deposition probabilities from data collected in different years is deemed appropriate because results of deposition experiments indicate that deposition probabilities did not vary significantly within or between years for each predator species (Hostetter et al. 2015).

Not all PIT tags deposited by birds on their nesting colony are subsequently found by researchers after the nesting season (Ryan et al. 2003; Evans et al. 2012). For example, tags can be blown off the colony during wind storms, washed away during flooding events, or otherwise damaged or lost during the course of 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 often vary significantly within and between nesting season (Evans et al. 2012; Hostetter et al. 2015), variation that necessitates a direct measure of detection probabilities in each study year, for each colony. To address this, PIT tags with known tag codes were intentionally sown on the East Sand Island cormorant and tern colonies (hereafter referred to as “control tags”) prior to, during (terns only), and following the nesting season to quantify PIT tag detection probability (i.e., probability that a tag deposited by a bird is detected by researchers after the nesting season). Control tags were the same size and frequency as the majority of PIT tags used to mark juvenile salmonids from the Columbia River Basin (12 mm, ISO FDXB). During each discrete sowing time period, control tags were haphazardly sown throughout the area occupied by nesting birds during the breeding season (see [Figure 1](#) and [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 Calculations](#) below for details). Equal number of control tags were sown during each discrete time period and sample sizes were selected by considering historic releases (see [Results and Conclusions](#) for sample sizes in 2015). This allows direct comparisons of independent detection probabilities, with similar precision between years. Additionally, sample sizes were purposefully limited to not amplify tag collision, whereby additional control tags could suppress detection probabilities, especially in areas where high densities of PIT tags may occur, like on East Sand Island Caspian tern colony (BRNW 2015).

Predation Rates: Following the methodology of Hostetter et al. (2015), predation rates were modeled independently for each salmonid ESU/DPS, bird colony (double-crested cormorants, Brandt’s cormorants, and Caspian terns) and year. The probability of recovering a PIT tag from a smolt on a particular 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 (ψ_i) (see also [Figure 3](#)):

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

where k_i is the number of smolt PIT tags recovered from the number available (n_i) in week i . The probable values of these parameters were modeled using a Bayesian approach. The detection probability (ψ_i) and predation probability (θ_i) were each modeled as functions of time. The probability, ψ_i , that a tag, consumed in week i and then deposited on the colony and detected was assumed to be a logistic function of week. That is:

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

where β_0 and β_1 were both derived from non-informative priors (normal [0, 1000]). Weekly predation probability, θ_i , was modeled as a random walk process with mean μ_θ and variance σ_θ^2 , where:

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

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

Informative Beta (α, β) priors were used to model deposition probability (ϕ). The shape parameters are dependent on species type and are assumed to be mutually independent from colony to colony, and from year to year. For cormorant colonies we assumed $\alpha = 15.98$ and $\beta = 15.29$, for tern colonies we assumed $\alpha = 16.20$ and $\beta = 6.55$ (Hostetter et al. 2015).

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 near Skamania landing in a given week. Annual predation probabilities 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:

$$\frac{\sum_{i \in \text{breeding season}} (\theta_i * n_i)}{\sum_{i \in \text{breeding season}} (n_i)}$$

Cumulative predation rates from double-crested cormorants and Caspian terns were calculated as a sum of these two consumption estimates, divided by total smolt available per salmonid ESU/DPS.

All predation rate models were implemented using the software JAGS (Plummer 2003) accessed through R version 3.1.3 (R Core Team 2015) using the R2jags (Su 2015) and dclone (Solymos 2013) 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 iterations were run to produce the final posterior distribution from which final 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 (95% CI). Annual predation rates were calculated for salmonid ESUs/DPSs where ≥ 500 PIT-tagged individuals were considered available to birds in a given year to avoid imprecise results that may occur from small sample sizes of available PIT-tagged smolts (Evans et al. 2012).

Predation impacts by rear-type and migration history: To date, efforts to compare and contrast predation probabilities based on fish’s rear-type and migration history have been sporadic or antiquated (e.g., not corrected for both detection and deposition probabilities; Ryan et al. 2003; Sebring et al. 2013). These types of comparisons, however, can be readily incorporated into the modelling of predation rates for each salmonid ESU/DPS described above. Inclusion of these variables is achieved through a re-parameterization such that

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

where k_{iv} is the number of smolt PIT tags in category v recovered from the number available (n_{iv}) in week i . This approach allows a common estimate of deposition and detection across categories which facilitates increased precision (i.e., smaller bounds around the estimate). Annual and weekly predation probabilities for each classification or subset can then be calculated using the methods described above.

Building on this approach, we compared and contrast predation probabilities by double-crested cormorants and Caspian terns based on a fish’s (1) rear-type (hatchery, wild) and (2) migration history (in-river, transported). To evaluate if one subset or group of fish (e.g., hatchery fish) were more susceptible to bird predation than the other group (e.g., wild fish), comparison of rates were made on a weekly and annual bases and then analyzed across the entire study period (2006-2015). Analysis across the entire study period further included a random effect for year. Comparisons between groups were made letting ρ represent the average proportional difference in the odds of predation over the study period, with a value less than or greater than 1.0 indicating a preference for a particular group of fish and a value of 1.0 showing no preference. For instance, a value 1.55 indicates that the odds of a fish being consumed are 55% greater for that particular group of fish. We tested for statistically significant differences using logistic regression. The weekly estimates of predation were treated as mutually independent, allowing the focus to be limited to only the proportion of recovered tags (corrected for detection and deposition probabilities) from those last seen at the nearest upstream dam or release site. Therefore

$$k_{iv_0} \sim \text{Binomial}(n_{iv_0}, \theta_{iv} * \phi * \psi_i)$$

and

$$k_{iv_1} \sim \text{Binomial}(n_{iv_1}, \rho * \theta_{iv} * \phi * \psi_i)$$

and we are testing the hypothesis $H_0: \rho = 1.0$. This test was applied to all appropriate ESUs/DPSs for each comparison. To simplify visual comparisons of output, data were plotted as the log-odds ratio, with values greater than or less than 0 indicating a preference for a particular group of fish (95% confidence intervals that over-lap 0 were not statistically significant).

Predation impacts prior to and following management actions: If given enough time and a significant decrease in the number of nesting birds, it is expected that the management of double-crested cormorants and Caspian tern on East Sand Island will have a measurable effect on the level of predation in the Columbia River Estuary. This hypothesis can be tested comparing posterior distributions of average annual predation rates prior to and following management actions. Management actions on the double-crested cormorant colony were not initiated until after the majority of available PIT-tagged smolts used in the study had passed Bonneville and Sullivan dams (see *Results and Conclusions*). As such, there were no meaningful post-management measures of predation rates to analyze with PIT tag data for double-crested cormorants as part of this study. 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 2011-2015 (post-management). The post-management time period was considered to have started in 2011 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 (BRNW 2015).

Analysis assumptions: Results of predation analyses were based on the following assumptions:

- A1. Information from PTAGIS on release and interrogation of PIT-tagged salmonids were complete and accurate.
- A2. PIT-tagged smolts detected passing Bonneville Dam, Sullivan Dam, or released from barges were available to birds nesting downstream.
- A3. The predation, deposition, and subsequent detection probabilities for PIT-tagged smolts were all independent.
- A4. The detection probabilities for control PIT tags sown on-colony were equal to those of PIT tags deposited by birds on-colony.
- A5. The deposition probabilities for PIT tags measured in previous years were equal to those of smolt PIT tags consumed by birds during the current study year.
- A6. PIT tags from consumed fish were egested by avian predators within a relatively short time period (one week) of the PIT-tagged fish being detected passing an upstream dam or released from a barge.

A7. PIT-tagged fish are representative of non-tagged fish belonging to the same ESU/DPS and passing the same detection (dam) or release (barge) site.

To help meet the first assumption (A1), irregular entries in PTAGIS were either verified by the respective coordinator of the PIT-tagging effort or were censored from the analysis. Detections of PIT-tagged salmonids at dams or release sites upstream of bird colonies is deemed the most appropriate measure of fish availability given the downstream movement of smolts, the ability to standardize data across sites, and the ability to define unique groups of salmonids by a known location and passage date (Assumption A2). Assumption A2 assumes all PIT-tagged fish last detected passing a dam or released via barge were alive and available to predators downstream. If large numbers of fish halt their out-migration or died immediately following passage/release and prior to reaching the foraging range of the colony, predation rates would be biased low. The fate of each PIT tag implanted in a smolt is assumed to be independent (A3). Lack of independence among PIT-tagged fish could potentially bias predation probabilities and overinflate measures of precision (i.e., credibility intervals). Detection probability estimates (A4) were generally high (ca. 70%, depending on the colony and year; see *Results and Conclusions*), suggesting that any possible violations of assumption A4 would have little effect on estimates of predation rates at most colonies and years. Deposition rate data collected in year's past (when multiple estimates of deposition rates were measured for each species of avian predator over the course several time periods and years) showed no evidence of inter- or intra-annual trends in deposition probabilities (Assumption A5).

Assumption A6 relates to the use of the last date of live fish detection as a proxy for the date a PIT tag was deposited on a bird colony. This assumption needs to be only roughly true because detection efficiency did not change dramatically on a weekly basis (BRNW 2015; see *Results and Conclusions*). Assumption A7 relates to inference regarding the susceptibility of a PIT-tagged fish to consumption as it relates to all fish (tagged, untagged) from the same cohort (ESU/DPS, rear-type, etc.). There are few empirical data to support or refute assumption A7, except that the general run-timing and abundance of PIT-tagged fish is often in agreement with the run-timing and abundance of untagged fish passing dams in the Columbia River Basin. For some populations, fish were intentionally PIT-tagged in concert with the run passing a given dam to better ensure that a representative sample of fish were available for analyses (see Evans et al. 2014). For other groups of fish, however, individuals may have been culled for tagging based on their condition or size or only a small number or proportion of available fish were PIT-tagged of a given cohort or stock due to a lack of funding and/or due to shifting regional research, monitoring, and evaluation priorities each year.

RESULTS AND CONCLUSIONS

Predation by Double-crested Cormorants

PIT tag recovery: Following the nesting season, 26,967 PIT tags from 2015 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead 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 2015 was higher than the number recovered in 2014 (22,069 tagged smolts) and twice the average recovered annually during 2000-2013 (mean = 13,134 tagged smolts).

Control PIT tags sown to measure detection efficiency on the cormorant colony (n = 400) indicated that detection efficiency ranged from 69–85% for PIT tags deposited between 1 March and 31 August ([Table 2](#)). Detection efficiency estimates in 2015 were the highest ever recorded on the cormorant colony ([Figure 4](#)), an increase related in part to conducting three passes of the entire colony, instead of the two passes conducted prior to 2014 (Zamon et al. 2013). Other factors that may have contributed to higher detection efficiency include use of a larger scanning crew and continuous daily scanning until all three passes were completed, with limited weather delays or other interruptions that could have resulted in additional tag loss between passes (e.g., storms, flooding, or other events that could remove or damage tags). Additionally, management feasibility studies conducted on the cormorant colony during 2011-2013 resulted in several distinct nesting areas or zones on East Sand Island, areas that shifted from one year to the next (BRNW 2014). This shifting in the areas utilized by nesting cormorants likely reduced PIT tag densities from what would have been if the same areas would have been used in all years, resulting in a reduction in PIT tag collision and an increase in detection efficiency in recent years.

Based on previous studies that empirically measured deposition rates for double-crested cormorants nesting on East Sand Island, deposition was estimated to be 51% (95% CI = 34–70%; [Table 3](#) and Hostetter et al. 2015).

Predation rates: Predation rates on PIT-tagged smolts last detected passing Bonneville Dam on the Columbia River or Sullivan Dam on the Willamette River ranged from 2.4% (95% CI = 1.5–4.1%) on Snake River sockeye salmon to 14.5% (95% CI = 10.5–22.4%) on Snake River spring/summer Chinook salmon ([Table 4](#)). Predation rates in 2015 were generally higher than those observed in years past, particular predation on salmon populations ([Appendix A1, Table A1](#)). For instance, predation on Snake River spring/summer Chinook salmon, Snake River fall Chinook salmon (8.7%; 95% CI = 6.0–14.0%) and Upper Columbia River spring Chinook salmon (8.3%; 95% CI = 5.9–12.9) were the highest recorded since 2000. Predation on steelhead populations from the Snake River, Upper Columbia River, and Middle Columbia River DPSs were also elevated in 2015, but rates were not significantly higher than those observed in some other years during 2000-2014 ([Appendix A1, Table A1](#)). Predation rates on PIT-tagged smolts released from barges below Bonneville Dam (Snake River origin fish only) in 2015 were also generally higher than those observed in years past ([Appendix A1, Table A3](#)), with an estimated 7.8% (95% CI = 5.4–12.3%) and 16.1% (11.7–24.8%) of Snake River sockeye salmon and Snake River spring/summer Chinook salmon consumed, respectively ([Table 4](#)).

In 2015, predation rate estimates by East Sand Island double-crested cormorants were relatively similar between steelhead and salmon populations ([Table 4](#)). For instance, predation

rates on Snake River spring/summer Chinook salmon were not significantly different than those on Snake River steelhead (12.8%; 95% CI = 9.3–19.6%). As such, no clear and consistent preference for salmon ESUs versus steelhead DPSs were identified in 2015. Although no clear preference by ESU/DPS was identified in 2015, predation rates were highly variable over the course of last sixteen years, with significant differences by ESU/DPS observed in some but not all years (see [Appendix A1, Table A1](#)).

An investigation of smolt abundance indicates PIT-tagged smolt availability peaked in mid-May for most of the ESUs/DPSs evaluated, with the exception of Snake River fall Chinook salmon which peaked in early-June ([Figure 5](#)). In general, predation rates by double-crested cormorants increased in concert with the number of available PIT-tagged smolts, with the highest rates observed during the peak out-migration period for each ESU/DPS evaluated. Results indicate that as more fish became available, double-crested cormorants consumed a larger proportion; a finding that suggests larger numbers of prey were not able to swamp an individual fish's ability to avoid predation by double-crested cormorants (Ims 1990; Hostetter et al. 2012). Since double-crested cormorants are pursuit divers that can consume multiple fish in a single foraging bout (Hatch and Weseloh 1999), highly concentrated prey may be especially vulnerable to predation by cormorants. Relative trends in weekly predation rates (an increase or decrease) were similar amongst in-river and transported migrants ([Figure 5](#); see also Predation Impacts by rear-type and migration history below). Taken together, results indicate that susceptibility of salmonid ESUs/DPSs to double-crested cormorant predation in the Columbia River Estuary varied by week and year but that cormorants generally consumed fish in proportion to their relative availability (see also [Appendix D](#)).

Predation impacts by rear-type and migration history: Of those ESUs/DPSs with adequate sample sizes of available PIT-tagged fish, there were no consistent trends in the relative susceptibility of fish by rear-type (hatchery, wild) to East Sand Island double-crested cormorant predation during 2006-2015 ([Table 5](#)). Within the same ESU/DPS, results of odds-ratio comparisons were variable by week and year, with no statically significant result observed in most, but not all, cases ([Appendix B, Figure B1](#)). There was some evidence that wild Snake River steelhead were more susceptible to cormorant predation than their hatchery counterparts when data from all weeks and years are considered, but differences were inconsistent across the study period. For example, in 2015, there was some evidence that wild Snake River steelhead were more likely to be predated than hatchery-reared steelhead, but the opposite trend was observed during 2012-2014 ([Table 5](#)). Collectively, results indicate that hatchery and wild smolts last detected passing Bonneville Dam had no appreciable difference in susceptibility to double-crested cormorants nesting on East Sand Island, with no consistent preference based on a fish's rear-type observed from data collected over the course of the last decade. 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; BRNW 2015).

Similar to comparisons by rear-type, relative differences in predation impacts by double-crested cormorants by a fish's migration history (in-river, transported) varied by week, year, and salmonid ESU/DPS (*Appendix C, Figure C1*). Unlike rear-type comparisons, however, there was some evidence that transported Snake River fall Chinook salmon and transported Snake River sockeye salmon were more likely to be consumed by cormorants than in-river migrants (*Table 6*). Results were statistically significant for both species when data from all weeks and years are considered (*Appendix C, Figure C1*). There was also some evidence that in-river Snake River spring/summer Chinook salmon and in-river Snake River steelhead were more likely to be predated by cormorants compared with transported fish (*Table 6*). Odds-ratio results were less pronounced (closer to 1.0) and inconsistent (trends varied by week and year), however, in relation to those observed in Snake River fall Chinook salmon and Snake River sockeye salmon. For instance, results indicated that average predation odds were only slightly greater on in-river Snake River steelhead (16% greater odds of predation) compared with transported Snake River steelhead when data from all weeks and years are considered, but in some years the inverse trend was observed (e.g., in 2011 and 2013), whereby predation odds were greater on transported steelhead (*Table 6*). Sample sizes of available fish for these comparisons were large (generally > 20,000 PIT-tagged fish per year) and data were available in most weeks and years throughout the study period (see *Appendix B, Figure B1*). Given the robust datasets used for relative comparisons, tests were readily able to detect significant differences, even if the magnitude of difference in predation rates between in-river and transported fish were in some cases small (e.g., < 2% difference in predation rates). As such, although results were statistically significant in many cases, the biological implications of differences should be considered when interpreting results.

Collectively, results indicate that differences in fish susceptibility to cormorant predation by a fish's migration history (in-river, transported) were observed, although trends were not consistent within and across salmonid ESUs/DPSs, years and weeks. Trends were the most consistent and perhaps biologically relevant for transported Snake River fall Chinook salmon and transported Snake River sockeye salmon, with predation impacts higher on transported groups of fish in most weeks and years during 2006-2015. Factors that could potentially explain the greater susceptibility of transported fish to bird predation include differences in the relative health or condition (descaling, injuries, disease) of fish, the relative abundance of fish, and/or residency times and spatial distribution of fish in Columbia River Estuary. Hostetter et al. (2012) and BRNW (2014) observed that fish in degraded condition were more likely to be consumed by double-crested cormorants than seemingly healthy fish. Dietrich et al. (2009) observed higher rates of disease in transported Snake River fall Chinook salmon compared with in-river migrants of the same stock. As such, if in-river fish that survived out-migration to below Bonneville Dam were, on average, in better over-all condition or health than those fish that were transported, transported fish could disproportionately be consumed by cormorants. Given our finding that predation rates by cormorants on East Sand Island generally increase as more smolts become available, differences in predation impacts between in-river and transported fish may simply be related to difference in their relative abundance in Columbia River Estuary. Differences in abundance could occur if a large proportion of available Snake River fall Chinook

salmon were collected for transportation (FPC 2014) and/or if the survival of in-river fall Chinook salmon to Bonneville Dam were low in any given week or year (Hughes et al. 2013; Skalski et al. 2014; Evans et al. *in press*). Alternatively, or additionally, transported Snake River fall Chinook salmon may rear or reside within the lower estuary for greater periods of time (Marsh et al. 2007), a residency that would increase their availability to avian predators, particularly cormorants which are abundant in the estuary throughout the summer (BRNW 2015). Unfortunately, empirical data on the condition of fish, the residency times of fish, and the absolute abundance and spatial distribution of salmonids in the estuary is generally lacking; studies that thoroughly investigate these factors may be paramount in understanding how these mechanisms influence fish susceptibility to bird predation in the estuary (Hostetter et al. 2012; Sebring et al. 2013; Evans et al. *in press*).

Predation impacts prior to and following management actions: As noted in the *Methods*, 2015 was first year of management actions on the East Sand Island double-crested cormorant colony and activities (lethal culling of adults, oiling of eggs) were limited in scope and started after the majority of PIT-tagged fish used in this study had passed Bonneville and Sullivan's dams (see *Figure 5* for data on the run-timing and abundance of PIT-tagged smolts used in predation analyses in 2015). For instance, only 158 adult cormorants (< 1% of the estimated 12,150 nesting pairs; *Figure 6*) were culled between 22 May and 31 August 2015 (USACE 2015b), a number too small to influence predation rate estimates in a meaningful way, especially given the amount of natural variation observed in double-crested cormorant predation rates over the course of the last decade.

Predation by Caspian Terns

PIT tag recovery: Following the nesting season, 13,990 PIT tags from 2015 migration year smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead combined) were recovered on the East Sand Island Caspian tern colony (*Table 1*). Numbers of tags recovered from the Caspian colony in 2015 were similar to numbers recovered in 2014 ($n = 13,059$ tagged smolts) but substantially lower than the average during 2000-2013 (mean = 27,596 tagged smolts).

Recoveries of control PIT tags sown on the East Sand Island tern colony ($n = 300$) indicated that detection efficiency ranged from 63–94% for PIT tags deposited on-colony between 1 March and 31 August, 2015 (*Table 2*). Detection efficiency in 2015 was higher than that observed during 2010-2014 and similar to estimates during 2007-2009 (*Figure 4*). Recent increases in detection efficiency on the tern colony were likely due to efforts to physically remove PIT tags using a sweeper magnet towed behind an ATV. Physical recovery was conducted simultaneously with most electronic passes, with > 25,000 functional PIT tags removed during fall scanning efforts. Many of these removed PIT tags were from previous migration years (> 70% of all removed tags were from smolt migration years prior to 2015). This result indicates that PIT tags were gradually accumulating on the colony, that some PIT tags have remained on-colony since its inception, and that the large number of accumulated PIT tags likely contributes

to significant tag collision effects. Of the PIT tags from the 2015 smolt migration year detected on-colony, approximately 50% were physically recovered using sweeper magnets and removed from colony. This result was consistent for control tags (n = 300) with roughly 60% of tags sown in 2015 being subsequently recovered with sweeper magnets and removed. Notably, a greater proportion of post-season control tags (83%) were recovered when compared to pre- and mid-season releases of control tags, 30% and 66% respectively. Results from PIT tag removal efforts suggest that future efforts to physically remove PIT tags will improve on-colony detection efficiency by minimizing tag collision, as well as recover a significant number of PIT tags that were previously undetected, despite substantial efforts to electronically detect PIT tags in past years. Future management actions directed at the East Sand Island Caspian tern colony (i.e., further restrictions of colony area) will likely result in increased on-colony PIT tag collision due to greater nesting densities of terns, and will likely require the continual removal of PIT tags to ensure the precision of future estimates of predation rates derived from PIT tag recovery on East Sand Island.

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% CI = 51–89%; [Table 3](#) and Hostetter et al. 2015).

Predation rates: Estimated predation rates indicated that steelhead were the species of salmonid smolt most susceptible to predation by Caspian terns nesting on East Sand Island in 2015, with predation rates ranging from 7.8% (95% CI = 5.9–11.4%) on Middle Columbia River steelhead to 10.5% (95% CI = 8.2–15.0%) on Upper Columbia River steelhead ([Table 4](#)). By comparison, predation rates on salmon ESUs were significantly lower than those on steelhead DPSs, ranging from just 0.4% (95% CI = 0.1–1.5%) on Upper Willamette River spring Chinook salmon to 2.0% (95% CI = 1.5–2.9%) on Upper Columbia River spring Chinook salmon ([Table 4](#)). Results indicate that susceptibility to predation by East Sand Island Caspian terns was similar amongst ESUs of salmon, but that DPSs of steelhead were disproportionally consumed relative to their availability in the Columbia River Estuary, as indicated by the numbers of PIT-tagged smolts interrogated at Bonneville or Sullivan dams ([Figure 7](#)). Predation rates on transported smolts from the Snake River in 2015 were similar to those of in-river migrating smolts from the Snake River ([Table 4](#)), with predation rates highest on steelhead compared with salmon ESUs.

Differences in smolt susceptibility to tern predation in 2015 were very similar to those observed in years past, with Caspian tern predation rates on steelhead populations often 5 to 10 times greater than those on salmon ([Appendix A, Table A2](#)). Higher avian predation impacts by Caspian terns on juvenile steelhead compared with salmon is well documented in the published literature (Collis et al. 2001; Ryan et al. 2003; Evans et al. 2012; Evans et al. *in press*). Possible explanations for the greater susceptibility of juvenile steelhead to bird predation include differences in the size (length) and behavior of steelhead compared with other salmonid species. Beeman and Maule (2006) observed that steelhead smolts were more surface-oriented compared with salmon smolts. 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). Hostetter et al. (2012) and BRNW (2014) noted size-selectivity amongst avian predators, with larger fish typically predated at higher rate than smaller fish within the same species (see also *Predation impacts by rear-type and migration history*).

In comparison to predation impacts on salmon by double-crested cormorants nesting on East Sand Island, predation impacts on salmon smolts by Caspian terns were generally small, particularly during the last several years, where predation rates have been < 3% for each salmon ESU evaluated since 2011 (*Appendix A, Table A2*). An investigation of weekly predation rates by East Sand Island Caspian terns indicates that predation rates were generally lower when the largest number of PIT-tagged smolts were available as prey in estuary in 2015 (*Figure 7*). For instance, predation impacts on steelhead DPSs were the lowest during the peak of the run in mid-May and higher before (in late April) and after (early June) the peak. For most ESU/DPS evaluated, predation rates were particularly high during the last few weeks of the smolt run in 2015 (*Figure 7*). Trends in weekly predation rates observed in 2015 were very similar to those observed by Hostetter et al. (2012) and (BRNW 2015) and are consistent with the predatory-swamping hypothesis (Ims 1990). Weekly predation rate estimates by Caspian terns differs markedly from those observed in double-crested cormorants, where predation rates by cormorants were highest when the number of available smolts available was the highest (*Figure 5* and BRNW 2015).

Predation impacts by rear-type and migration history: There was evidence that hatchery Snake River spring/summer run Chinook salmon and hatchery Upper Columbia River spring Chinook salmon were more susceptible to predation by East Sand Island Caspian terns than their wild counterparts. For instance, the odds of predation were, on average, 62% and 66% greater for hatchery Snake River spring/summer Chinook salmon and hatchery Upper Columbia River spring Chinook salmon, respectively, over the course of the last decade (*Table 5*). Differences were consistent in all annual comparisons and in most weekly comparisons (*Appendix B, Figure B1*). Conversely, there was no evidence of a consistent difference in the relative susceptibility of hatchery and wild Snake and Upper Columbia River steelhead to tern predation (*Table 5* and *Appendix B, Figure B1*).

Data from other studies indicates that both behavior and physical traits associated with hatchery-raised salmonids may enhance susceptibility to predation (Olla and Davis 1989, Fritts et al. 2007, Hostetter et al. 2012). In the present study, the increased vulnerability of hatchery spring/summer Chinook salmon to Caspian tern predation was likely related to differences in the size (mm; fork length) of hatchery (mean = 144 mm) and wild (mean = 111 mm) spring/summer Chinook salmon last detected passing Bonneville Dam during the study period. An analysis of length data (based on lengths at release, limited to fish released within the same month they were interrogated passing Bonneville Dam) indicates that the odds of Caspian tern predation on spring/summer Chinook salmon increased by 12% (95% CI = 11.9–12.6%) for every 10 mm increase fork-length. Hostetter et al. (2012) also found evidence of size-selectivity in Caspian terns nesting at Crescent Island (Rkm 509), with larger PIT-tagged fish more likely to be

predated than smaller PIT-tagged fish within the same species. Interestingly, however, there has been no evidence of size-selectivity in double-crested cormorants on PIT-tagged juvenile salmonids (Hostetter et al. 2012, this study), perhaps due to differences in the foraging behavior of the two species (see *above*).

There was some evidence that Caspian terns disproportionately consumed in-river Snake River steelhead and in-river Snake River spring/summer Chinook salmon as compared with transported smolts from the Snake River. Similar to findings in double-crested cormorants, however, average odds-ratios were close to 1.0 (no preference) when comparisons are considered for all weeks and years, with odds of predation 15% and 14% greater amongst in-river Snake River spring/summer Chinook salmon and in-river Snake River steelhead, respectively (*Table 6*). Unlike predation by double-crested cormorants, there was no evidence of differential predation between in-river and transported Snake River fall Chinook salmon and Snake River sockeye salmon to Caspian tern predation, a finding driven in part by the low overall predation impacts on salmon by Caspian terns during 2006-2015. For example, predation rates by Caspian terns on Snake River fall Chinook salmon and Snake River sockeye salmon were < 3% for both groups (in-river and transported; see *Appendix A1, Table A2* for in-river predation rate estimates and *Appendix A1, Table A3* for transport predation rate estimates).

Ultimately, the probability of an individual fish surviving the juvenile life stage is determined by a complex set of interacting factors, including individual fish characteristics and environmental conditions (Skalski 1998, Muir et al. 2001, Zabel et al. 2005, Hostetter et al. 2011; Hostetter et al. 2012; this study). Differences in the size and condition of fish, the run-timing of fish, and the abundance and spatial distribution of fish have all been linked to susceptibility to bird predation. Due to lack of empirical data on the condition, absolute abundance, and spatial distribution of smolts in the Columbia River Estuary, additional research is needed to fully understand these factors and how (or if) they can be managed to reduce predation impacts, in addition to efforts to reduce the number of Caspian terns that reside in the Columbia River Estuary (see also *Appendix D*).

Predation impacts prior to and following management actions: Average annual predation rates on PIT-tagged smolts by Caspian terns were lower following management actions on East Sand Island during 2011-2015. For example, average annual predation rates on Snake River steelhead during 2000-2010 were 17.2% (95% CI = 15.7–19.3%), but were 9.9% (95% CI = 8.5–12.0%) following management actions to reduce the available tern nesting habitat on East Sand Island during 2011-2015 (*Table 7*). Similar reductions were observed in other steelhead and salmon DPSs/ESUs (*Table 7*). Reductions in Caspian tern predation rates during 2011-2015 coincided with proportional reductions in colony size at East Sand Island (*Figure 7*). This suggests that Caspian tern management initiatives aimed at reducing nesting habitat on East Sand Island are resulting in lower predation rates on many ESA-listed salmonid ESUs/DPSs. Reductions in the number of Caspian terns on East Sand Island (average of 6,656 nesting pairs during 2011-2015; *Figure 6*) have not, however, reached management target goals (between 3,125 and 4,375 nesting pairs) because Caspian terns have increased their nesting density on

East Sand Island in response to reduced nesting habitat (BRNW 2015). Thus, further reductions in the number of Caspian terns nesting on East Sand Island will likely further reduce Caspian tern predation rates. Finally, although an important factor, colony size is not the only factor linked to variation in East Sand Island Caspian tern predation rates on PIT-tagged smolts during the study period (see [Appendix D](#)).

Cumulative Predation Rates

Combined predation rate estimates by double-crested cormorants and Caspian terns (i.e., cumulative impacts) indicate that birds have annually consumed between 1.5–16.5% of available salmon ESUs and between 12.0–40.6% of available steelhead DPSs during 2000–2015 (depending on the ESU/DPS and year; [Figure 8](#)). On average, predation rates on steelhead populations have been greater than those on salmon populations ([Figure 8](#)), due largely to the greater susceptibility of steelhead to Caspian tern predation. For instance, of the estimated 40.6% of Snake River steelhead consumed by terns and cormorants combined in 2006, the majority (83%) was due to predation by Caspian terns only. Conversely, cumulative predation rates on salmon population in the estuary were largely driven by predation by double-crested cormorants.

Results from this and other studies (Lyons et al. 2013; Hostetter et al. 2015) indicated that losses of juvenile salmonids to bird predation in Columbia River Estuary are comparable to or greater than those of other, non-avian sources of mortality. For example, losses due to piscine predation (Northern Pikeminnow *Ptychocheilus oregonensis*, Smallmouth Bass *Micropterus dolomieu*) were estimated to be between 1–14% of available fish per reservoir (Rieman et al. 1991; Ward et al. 1995) and losses of fish passing individual dams were estimated to be between 1–7% of available fish (Muir et al. 2001; Ferguson et al. 2006; Skalski et al. 2009). Furthermore, it should be noted that smolt mortality associated with tern and cormorant predation in the estuary are on fish that have survived freshwater outmigration, including passage through the Federal Columbia River Power System, fish that presumably have a higher probability of surviving than those that are yet to complete outmigration (Roby et al. 2003). Finally, juvenile salmonids belonging to every ESA-listed ESU/DPS from the Columbia River Basin must pass through the Columbia River Estuary and are therefore susceptible to predation by piscivorous waterbirds nesting on East Sand Island (Roby et al. 2003).

Combined average annual predation rates by terns and cormorants nesting on East Sand Island have been relatively stable since 2000, especially when impacts are assessed at the level of steelhead DPS or salmon ESU ([Figure 8](#)). Interestingly, the total number of birds nesting on East Sand Island has also remained relative stable since 2000 ([Figure 6](#)), due in part because early increases in the size of double-crested cormorant colony were off-set by recent management-related decreases in the size of the Caspian tern colony on East Sand Island. In addition to Caspian terns and double-crested cormorants on East Sand Island, there are other nesting colonies for each species (e.g., Rice Island for terns and Astoria-Megler Bridge for cormorants) and other species of piscivorous waterbirds (e.g., pelicans and gulls) present in the Columbia

River Estuary. For instance, in 2015, Caspian terns attempted to nest on Rice Island (Rkm 34) in the upper estuary, a location where per capita (per bird) impacts to juvenile salmonid have been shown to be higher than those of birds nesting on East Sand Island (Collis et al. 2002; Roby et al. 2002; Lyons 2010). Increasing numbers of double-crested cormorants have also been observed nesting on Astoria-Megler Bridge in recent years (BRNW 2013), impacts that have not been quantified by this or other studies. Other piscivorous waterbirds (e.g., Brandt’s cormorants *P. penicillatus*, California brown pelicans *Pelecanus occidentalis californicus*, American white pelicans *P. erythrorhynchos*, and gulls *Larus* spp.) either nest or roost in the Columbia River Estuary, and their collective impacts on smolt survival are unknown. As such, the benefit of piscivorous waterbird management actions in the Columbia River Estuary as a whole will only be achieved if (1) reductions in the size and predation impacts of one colony are not off-set by increases in the size and predation impacts of other colonies and (2) reductions in the total number of piscivorous waterbirds that reside in the estuary are substantial and persistent through time.

Predation by Brandt’s Cormorants

PIT tag recovery: Following the 2015 nesting season, 991 PIT tags from 2015 migration year salmonid smolts (Chinook salmon, coho salmon, sockeye salmon, and steelhead combined) were recovered on the Brandt’s cormorant colony on East Sand Island ([Table 1](#)); an unmanaged species that colonized East Sand Island in 2006 ([Figure 6](#)). Due to the high degree of intermixing of the nests of the two cormorant species (Brandt’s cormorants, double-crested cormorants), recoveries of smolt PIT tags could not in all cases be definitively assigned to either a Brandt’s cormorant or double-crested cormorant (see [Methods](#)). This was more of an uncertainty in assigning PIT tags to Brandt’s cormorant predation than double-crested cormorant predation because multiple groups of Brandt’s cormorant nests were surrounded by or were replaced by double-crested cormorant nests during the course of breeding season.

Control PIT tags sown to measure detection efficiency on the Brandt’s cormorant colony – the same tags sown to measure detection efficiency on the double-crested cormorant colony – indicated that detection efficiency ranged from 69–85% for PIT tags deposited between 1 March and 31 August ([Table 2](#)). We also applied estimates of smolt PIT tag deposition rates from double-crested cormorants nesting on East Sand Island to deposition estimates by Brandt’s cormorants nesting on East Sand Island ([Table 3](#)), as there are currently no estimates of PIT tag deposition rates for Brandt’s cormorants. Although there are similarities in the foraging behavior, nesting behavior, and general life history between Brandt’s cormorants and double-crested cormorants (Couch and Lance 2004), it is possible that deposition rates differ between the two species and, if so, estimates of predation rates by Brandt’s cormorants may be biased to some degree. Given the lower predation impacts observed in 2015 (see [below](#)), however, differences in deposition rates between double-crested and Brandt’s cormorants would have little influence on the accuracy of Brandt’s cormorant predation rate estimates.

Predation rates: Of the PIT-tagged juvenile salmonids last detected passing Bonneville Dam or Sullivan Dam (*Map 1*), predation rates by Brandt's cormorants were $\leq 0.6\%$ for all ESUs/DPSs evaluated in 2015 (*Table 4*). Predation rates by Brandt's cormorants were so low that differences between salmonid species and ESUs/DPSs were not readily discernible, and likely not biologically meaningful.

Despite increases in the size of the Brandt's cormorant colony on East Sand Island during the last decade (from 44 breeding pairs in 2006 to 2,071 breeding pairs in 2015; *Figure 6*), predation rates on smolts by Brandt's cormorants have remain the lowest of the three piscivorous waterbird colonies evaluated on East Sand Island since studies on the Brandt's cormorant colony were first initiated in 2008 (BRNW 2010). Results provide evidence that Brandt's cormorants consume far fewer salmonid smolts per capita (per bird) than double-crested cormorants or Caspian terns nesting on East Sand Island (BRNW 2014). Several factors likely account for differences in predation impacts between these predator species. First, Brandt's cormorants are most commonly associated marine environments and usually forage for prey in the nearshore ocean, where non-salmonid prey types (e.g., anchovy, herring, smelt, and others) are more common (Couch and Lance 2004). Consequently, salmonids comprise a smaller proportion of the diet of Brandt's cormorants compared to that of double-crested cormorants and Caspian terns (BRNW 2014). Second, the nesting chronology of Brandt's cormorants differs slightly from that of double-crested cormorants and Caspian terns in the Columbia River Estuary, with colony attendance peaking in late June, compared to late-May for Caspian terns and double-crested cormorants (BRNW 2014). This difference in nesting chronology may be important because by June the peak of the salmonid run has usually passed, especially for the large groups of PIT-tagged steelhead and yearling Chinook salmon smolts used to evaluate predation rates in the present study. Finally, relative to double-crested cormorants, Brandt's cormorants are slightly smaller (by body mass) and thus presumably have lower daily food requirements (Ellis and Gabrielsen 2001).

Taken together, results from this and other studies (Couch and Lance 2004; BRNW 2014; BRNW 2015) indicate that Brandt's cormorants continue to pose little risk to juvenile salmonid survival in the Columbia River Estuary. As such, the continued presence of Brandt's cormorants on East Sand Island, especially in lieu of double-crested cormorants, should be of little concern to fisheries managers in the region.

Predation impacts by rear-type and migration history: Predation rate estimates on smolts by Brandt's cormorants were too low ($\leq 0.6\%$ per ESU/DPS; *Table 5*) to evaluate meaningful trends in the relative susceptibility of tagged smolts to bird predation by rear-type or migration history.

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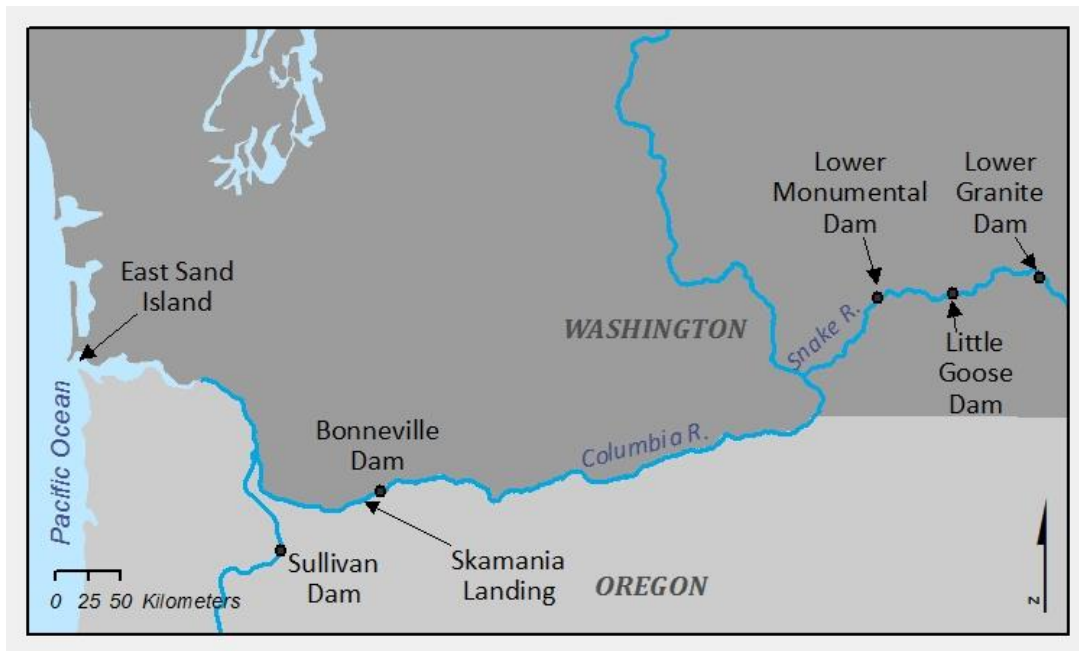
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MAPS



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 birds nesting on East Sand Island in the Columbia River estuary.

FIGURES



Figure 1: PIT tag detection equipment used on East Sand Island in 2015 included a hand-held portable system (top left), an eight coil flat-plate system attached to an ATV (top right), and a towable sweeper magnet attached to an ATV (bottom right and left).

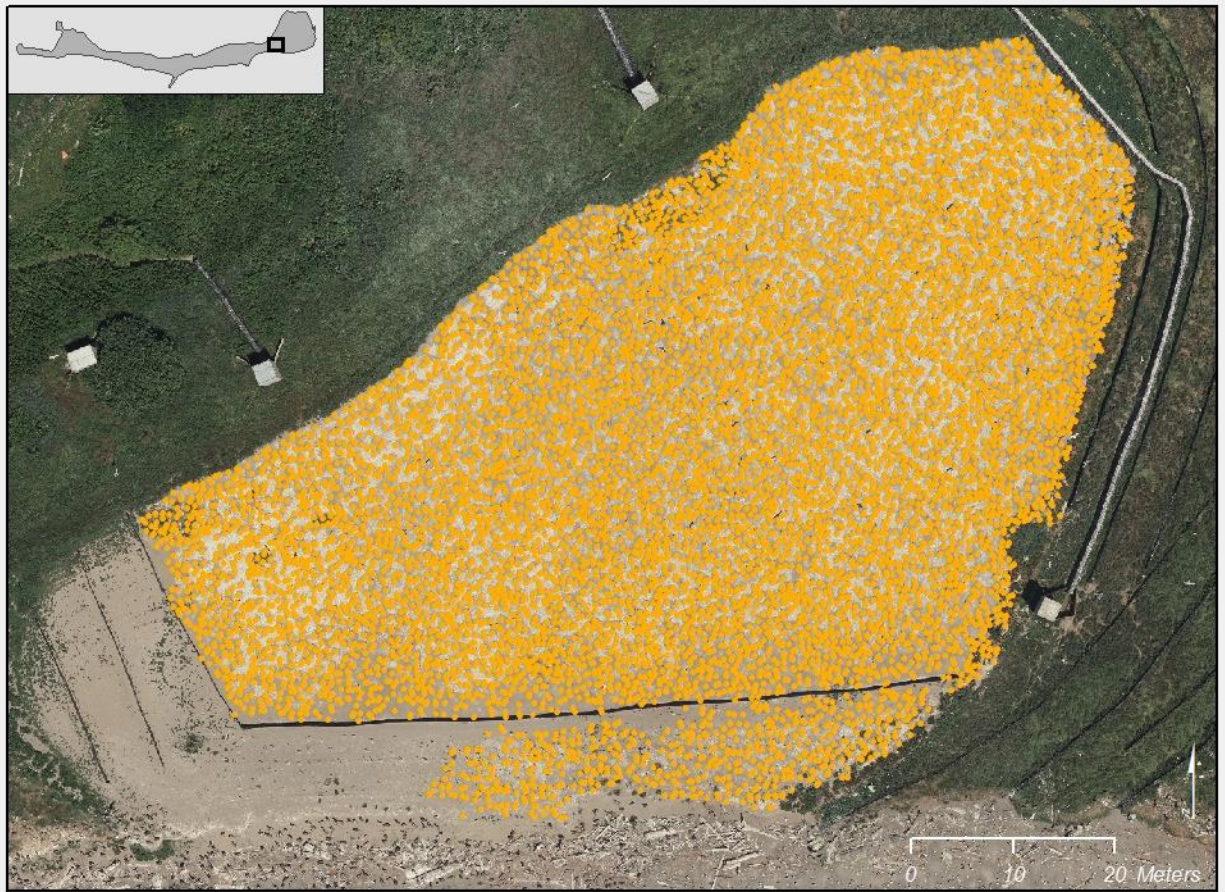
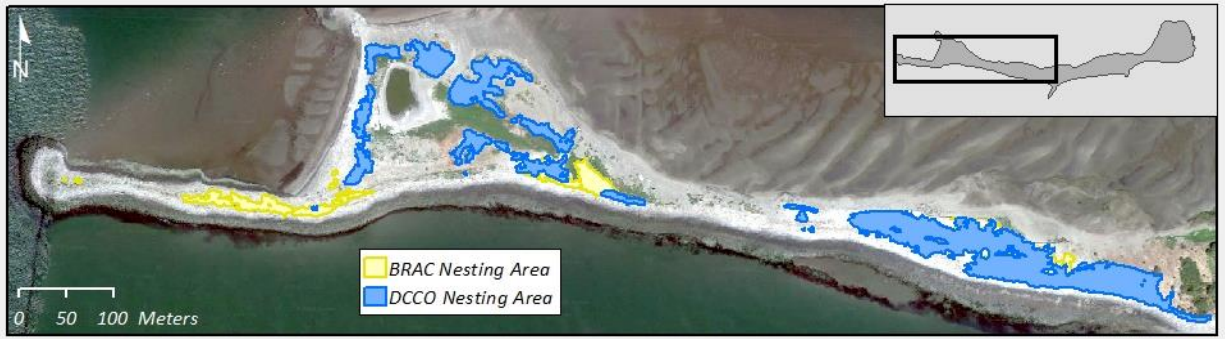


Figure 2: Areas scanned for PIT tags deposited by nesting double-crested cormorants (top), Brandt's cormorants (top), and Caspian terns (bottom) on East Sand Island in 2015.

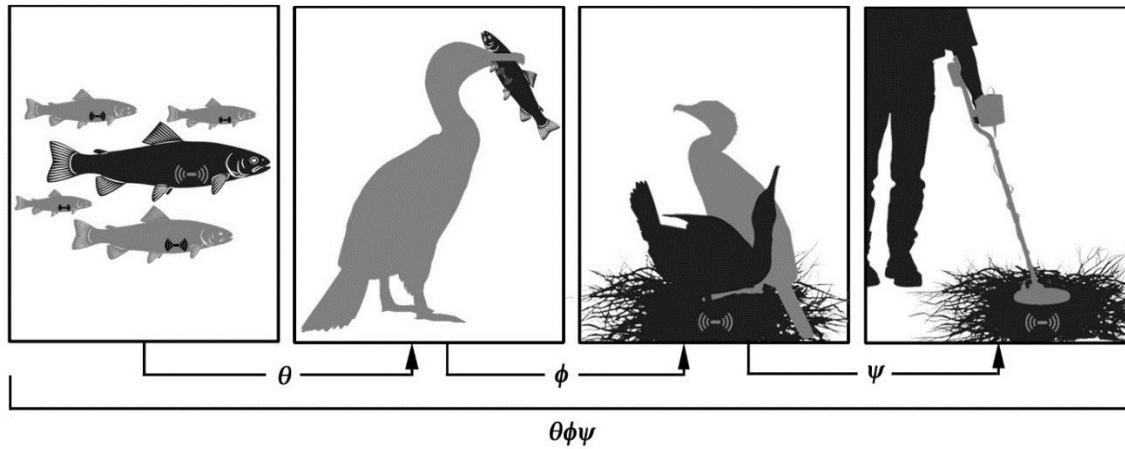


Figure 3: Conceptual model of the tag-recovery process in studies of avian predation. The probability of recovering a fish tag on a bird colony is the product of three probabilities: a fish was consumed (predation probability, θ), deposited on the nesting colony (deposition probability, ϕ), and detected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

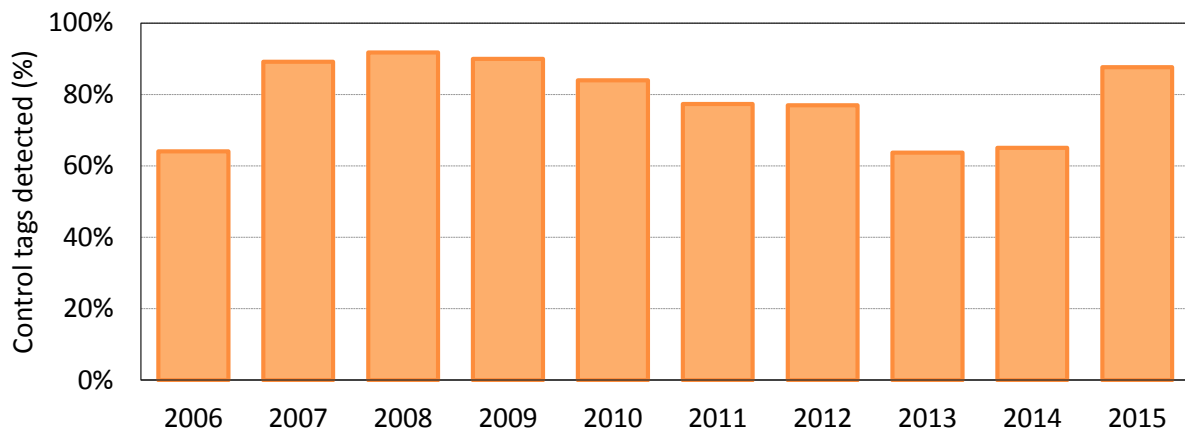
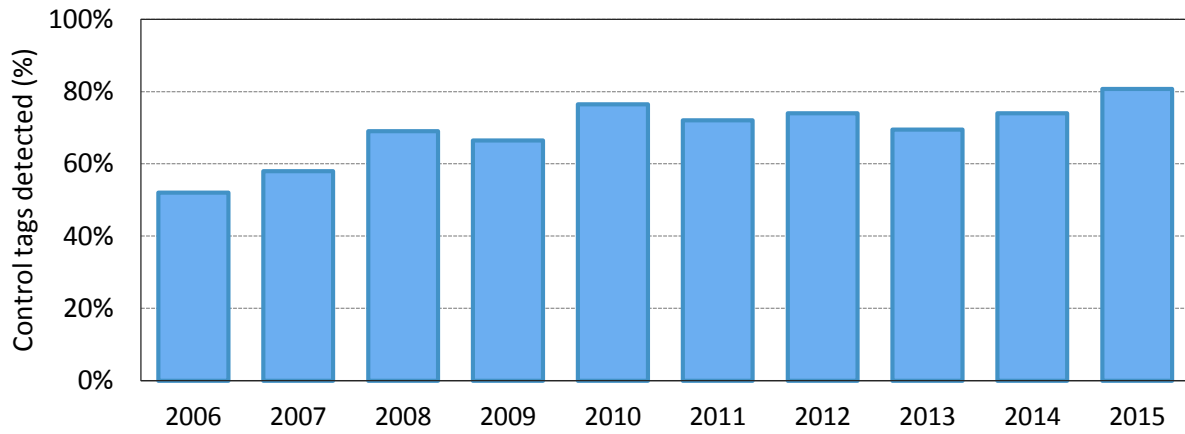


Figure 4: Average annual detection efficiency on the double-crested cormorant (top) and Caspian tern colonies (bottom) on East Sand Island during 2006-2015.



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 double-crested cormorants on East Sand Island during 2015. Error bars represent 95% credible intervals for predation rates.

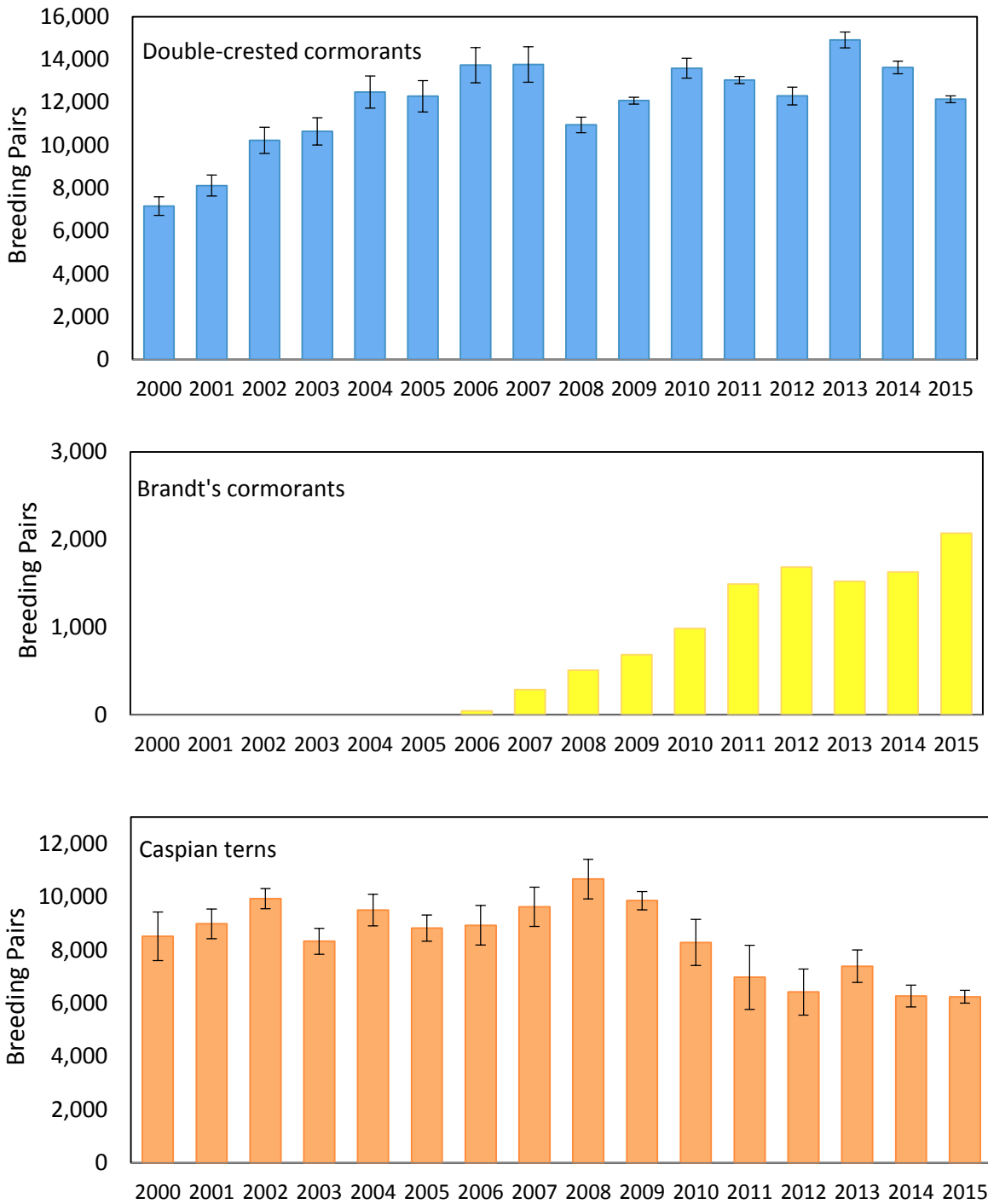


Figure 6. Estimated size (breeding pairs) of the double-crested cormorant, Brandt's cormorant, and Caspian tern colonies on East Sand Island during 2000-2015. Double-crested cormorant and Brandt's cormorant data in 2015 is from Smith and Associates (2016). Error bars represent 95% confidence intervals.

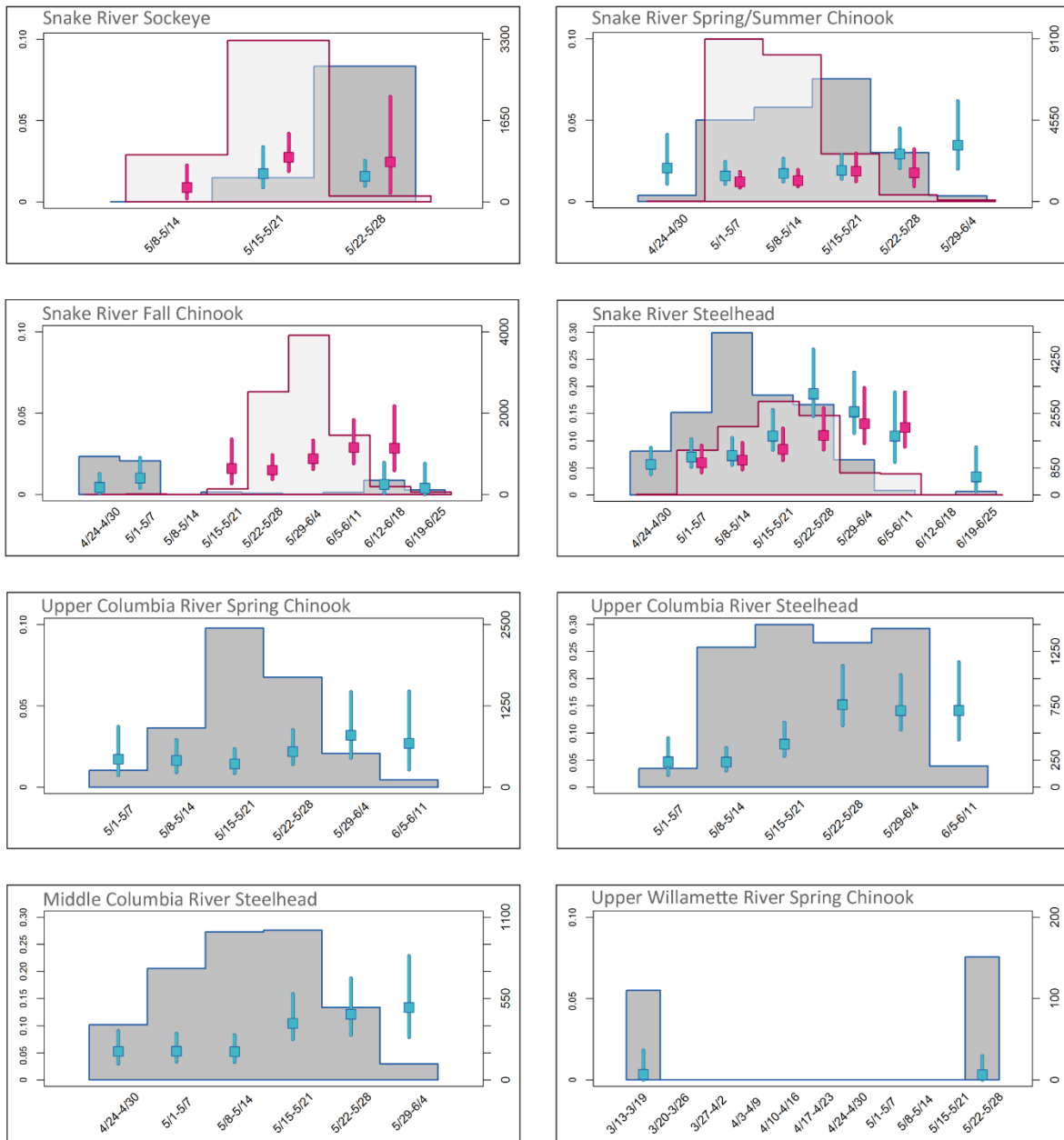


Figure 7. Estimated weekly predation rates (y_1 ; proportion of fish consumed) on in-river (blue squares) and transported (red squares) PIT-tagged juvenile salmonids last detected passing Bonneville or Sullivan dams (y_2 ; number available, dark gray bars) or transported from the lower Snake River (y_2^2 ; number available; light gray bars) by Caspian terns on East Sand Island during 2015. Error bars represent 95% credible intervals for predation rates.

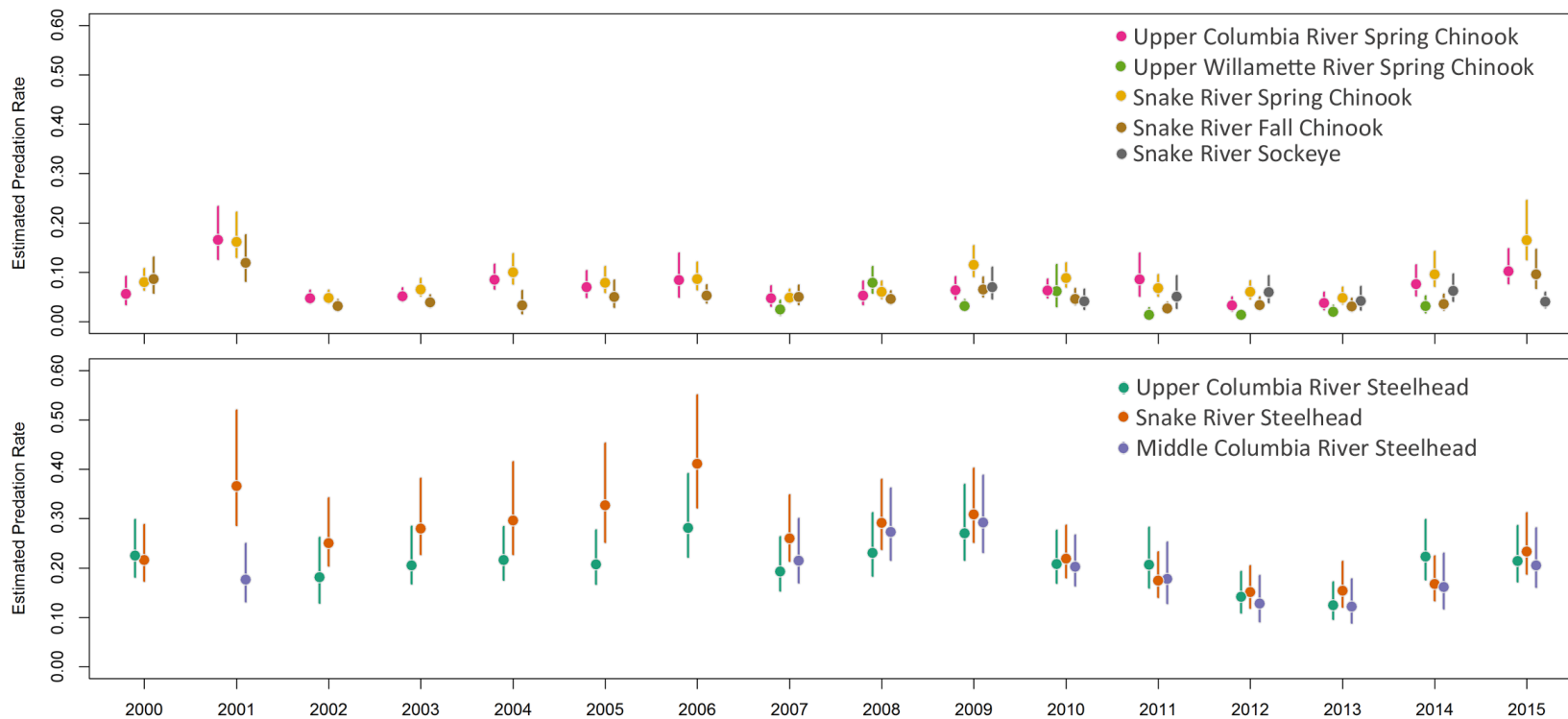


Figure 8. Estimated annual predation rates (proportion of fish consumed) on PIT-tagged juvenile salmonids last detected passing Bonneville or Sullivan dams by East Sand Island double-crested cormorants and Caspian terns combined during 2000-2015. Error bars represent 95% credible intervals for predation rates.

TABLES

Table 1. Number of 2015 migration year PIT-tagged juvenile salmonids (Chinook salmon, coho salmon, sockeye salmon, and steelhead combined) recovered (electronic and physical detections; see Methods and Figure 1) on bird colonies on East Sand Island following the 2015 breeding season.

River Segment	Location	Colony	PIT Tags Recovered
Estuary	East Sand Island	Double-crested cormorant	26,967
		Caspian tern	13,990
		Brandt's cormorant	991

Table 2. Range of detection efficiency estimates for PIT tags sown on bird colonies during the 2015 nesting season. Results were used to calculate the proportion of PIT tags deposited by birds on their nesting colony that were subsequently detected by researchers following the nesting season. Sample sizes (N) and the number of discrete release periods for sown tags are provided. Piscivorous waterbird species include double-crested cormorants (DCCO), Brandt’s cormorants (BRAC), and Caspian terns (CATE).

River Segment	Location	Colony	N (# of releases)	Date Range	Detection Efficiency
Estuary	East Sand Island	DCCO ¹	400 (2)	3/1 - 8/31	69 - 85%
		BRAC ¹			
		CATE	300 (3)	63 - 94%	

¹ The same detection efficiency values were used to estimate predation rates for both double-crested and Brandt’s cormorants (see Methods).

Table 3. Mean on-colony PIT tag deposition rate (DR [95% credible interval]) for nesting double-crested cormorants, Brandt’s cormorants, and Caspian terns on East Sand Island. Results were used to calculate the proportion of PIT tags consumed by birds that were subsequently deposited on their nesting colony. Sample sizes (N) of consumed PIT-tagged fish used to estimate deposition rate and the years when studies of deposition rates were conducted are provided. PIT-tagged fish were consumed during different periods of the day (morning, evening) and throughout the period of smolt out-migration in each study year (April to June; see Hostetter et al. [2015] for a detailed description of methods and results).

Species	Study Years	N	DR (95% CI)
Double-crested cormorants	2012-2013	428	51% (34-70%)
Brandt’s cormorants	DR for double-crested cormorants was applied		
Caspian terns	2004-2006	456	71% (51-89%)

Table 4. 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) by double-crested cormorants, Brandt's cormorants, or Caspian terns nesting on East Sand Island in 2015. Predation rates were adjusted to account for tag loss due to on-colony PIT tag detection efficiency (see Table 2) and deposition rates (see Table 3). 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.

ESU/DPS ¹	ESA ²	N		Double-crested cormorants		Caspian terns		Brandt's cormorants
		In-river	Transported	In-river	Transported	In-river	Transported	In-river
SR Sockeye	E	3,311	4,357	2.4% (1.5-4.1)	7.8% (5.4-12.3)	1.6% (1.0-2.6)	2.4% (1.7-3.6)	0.3% (0.1-0.7)
SR Spr/Sum Chinook	T	20,245	20,575	14.5% (10.5-22.4)	16.1% (11.7-24.8)	2.0% (1.5-2.9)	1.3% (1.0-2.0)	0.4% (0.2-0.7)
UCR Spr Chinook	E	5,943	-	8.3% (5.9-12.9)	-	1.9% (1.3-2.9)	-	0.4% (0.2-0.9)
SR Fall Chinook	T	2,629	8,347	8.7% (6.0-14.0)	5.3% (3.8-8.3)	0.8% (0.4-1.5)	2.1% (1.6-3.1)	0.4% (0.1-1.0)
UWR Spr Chinook	T	768	-	2.4% (0.9-5.2)	-	0.4% (0.1-1.5)	-	0.2% (<0.1-1.4)
SR Steelhead	T	16,451	10,461	12.8% (9.3-19.6)	9.3% (6.7-14.5)	10.2% (8.2-14.6)	8.9% (7.0-12.8)	0.4% (0.2-0.7)
UCR Steelhead	T	6,004	-	10.5% (7.6-16.2)	-	10.5% (8.2-15.0)	-	0.6% (0.3-1.2)
MCR Steelhead	T	3,927	-	12.4% (8.8-19.2)	-	7.8% (5.9-11.4)	-	0.5% (0.2-1.1)

¹ MCR = Middle Columbia River, SR = Snake River, UCR = Upper Columbia River, UWR = Upper Willamette River

² E = Endangered, T = Threatened

Table 5. Relative susceptibility of fish by rear-type to predation by double-crested cormorants and Caspian terns nesting on East Sand Island during 2006-2015. Values represent the odds-ratio of predation, with values < 1 indicating greater predation odds for hatchery fish and values > 1 indicating greater predation odds for wild fish (see Methods). Dashed lines denote years when insufficient sample sizes (< 500 PIT-tagged smolts of each rear-type) prevented comparisons. Comparisons within a given year were limited to weeks when both groups of fish (hatchery, wild) were available. An asterisks denotes statistical significance (see Appendix B, Figure B1 for weekly results with 95% confidence intervals). Salmonid populations included fish from the Snake River (SR) and Upper Columbia River (UCR), with runs of spring (Sp) and summer (Su) fish were evaluated.

	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2006-2015
Double-crested cormorants											
SR Sp/Su Chinook	0.75	1.11	1.16	1.03	1.17	0.65	0.72	0.93	0.60	0.93	0.95 (0.84-1.08)
UCR Sp Chinook	-	-	0.29	-	0.67	-	-	-	0.97	0.72	0.76 (0.55-1.04)
SR Steelhead	-	1.88	1.09	1.34	1.43*	1.34	0.89	0.85	0.93	1.30	1.25 (1.14-1.38)*
UCR Steelhead	-	-	-	-	0.74	-	-	-	-	1.02	1.01 (0.83-1.22)
Caspian terns											
SR Sp/Su Chinook	0.23*	0.45*	0.49*	0.34*	0.35*	0.34*	0.34*	0.46	0.28*	0.51*	0.38 (0.32-0.46)*
UCR Sp Chinook	-	-	0.18*	-	0.37*	-	-	-	0.21	0.70	0.34 (0.22-0.54)*
SR Steelhead		1.19	0.86	1.03	1.04	0.71	0.97	0.89	0.91	1.27*	1.02 (0.96-1.08)
UCR Steelhead	-	-	-	-	1.07	-	-	-	-	1.03	0.94 (0.80-1.10)

Table 6. Relative susceptibility of fish by migration history to predation by double-crested cormorants and Caspian terns nesting on East Sand Island during 2006-2015. Values represent the odds-ratio of predation, with values < 1 indicating greater predation odds for in-river fish and values > 1 indicating greater predation odds for transported fish (see Methods). Dashed lines denote years when insufficient sample sizes (< 500 PIT-tagged smolts of each migration history) prevented comparisons. Comparisons within a given year were limited to weeks when both groups of fish (in-river, transported) were available. An asterisks denotes statistical significance (see Appendix B, Figure B1 for weekly results with 95% confidence intervals). Salmonid populations were from the Snake River (SR) and include runs of spring (Sp), summer (Su), and fall (Fall) fish.

	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2006-2015
Double-crested cormorants											
SR Sp/Su Chinook	0.71	1.39	0.69*	0.77*	0.73*	0.82	0.63*	1.05	0.75*	0.83*	0.79 (0.76-0.83)*
SR Fall Chinook	0.68	0.76	1.53*	1.35*	1.32*	1.66*	2.03*	4.83*	1.54	1.14	1.52 (1.40-1.65)*
SR Sockeye	-	-	-	1.34	-	0.81	1.38	1.89	1.92	2.28	1.46 (1.15-1.87)*
SR Steelhead	0.69	0.72	0.56*	0.71*	0.90	1.22	0.86	1.56*	0.99	0.82*	0.80 (0.76-0.84)*
Caspian terns											
SR Sp/Su Chinook	0.85	0.68*	1.15	0.83*	0.82*	0.77	1.00	0.83	0.86	0.80	0.85 (0.81-0.90)*
SR Fall Chinook	0.83	1.07	0.77*	0.96	1.15	0.89	1.13	1.44	2.06	1.16	0.91 (0.83-1.01)
SR Sockeye	-	-	-	0.81	-	1.58	0.51	0.56	0.80	1.64	0.84 (0.58-1.21)
SR Steelhead	0.73*	0.76*	0.83*	0.92	0.95	0.81*	0.94	0.72*	1.07	0.72*	0.86 (0.83-0.88)*

Table 7. 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 that differences were statistically significant.

	Pre-management Period	Post-management Period
Salmonid ESU/DPS	2000-2010	2011-2015
Snake River Sockeye ¹	1.5% (0.9-2.2)	1.6% (1.2-2.2)
Snake River Spr/Sum Chinook	4.8% (4.3-5.4)	1.7% (1.4-2.1)*
Upper Columbia River Spr Chinook	3.9% (3.4-4.6)	1.4% (1.1-1.9)*
Snake River Fall Chinook	2.5% (2.2-3.0)	0.9% (0.7-1.2)*
Upper Columbia River Spr Chinook ²	2.5% (1.9-3.3)	0.9% (0.6-1.3)*
Snake River Steelhead	22.2% (20.3-24.8)	10.7% (9.2-12.8)*
Upper Columbia River Steelhead	17.2% (15.7-19.3)	9.9% (8.5-12.0)*
Middle Columbia River Steelhead ³	14.9% (13.1-17.6)	9.3% (7.7-11.4)*

¹ Predation rate estimates were not available in 2000-2008

² Predation rate estimates were not available in 2000-2006

³ Predation rate estimates were not available in 2000, 2002-2006

APPENDIX A: ANNUAL PREDATION RATES

This appendix provides annual PIT tag predation rate estimates for double-crested cormorants and Caspian terns nesting on East Sand Island. Predation rate estimates were based on the number (N) of PIT-tagged fish interrogated passing Bonneville Dam or Sullivan Dam (in-river migrants) during 2000-2015 or the number released from barges downstream of Bonneville Dam during 2006-2015. Predation rates were corrected for PIT detection and deposition probabilities (see *Methods*). Dashes denote insufficient sample sizes (< 500 PIT-tagged fish were available) for generating predation rates. 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.

Table A1. 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 2000-2015.

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2000	3.3% (2.3-5.3)	5.1% (2.9-9.3)	3.4% (1.6-6.8)	-	-	-	10.6% (7.5-16.8)	6.0% (3.9-10.0)
N	11,810	1,323	1,123				10,356	3,100
2001	2.2% (1.4-3.5)	5.5%(2.9-10.4)	3.3% (1.7-6.3)	-	-	2.5% (1.0-5.7)	2.8% (1.1-6.1)	-
N	8,845	807	1,230			872	774	
2002	1.8% (1.3-3.0)	1.4% (0.8-2.6)	2.2% (1.6-3.6)	-	-	-	3.1% (2.0-5.1)	3.7% (1.4-8.6)
N	30,617	4,899	20,493				7,331	561
2003	1.7% (1.2-2.7)	1.1% (0.7-2.0)	1.4% (0.9-2.1)	-	-	-	1.9% (1.2-3.0)	1.5% (1.0-2.4)
N	28,150	6,234	30,723				8,553	27,918
2004	5.1% (3.3-8.5)	1.9% (0.6-4.7)	4.7% (3.2-7.6)	-	-	-	3.6% (1.4-8.0)	7.4% (5.1-11.8)
N	4,816	929	9,533				803	6,040
2005	4.8% (3.2-7.9)	3.6% (1.8-6.9)	4.5% (2.8-7.8)	-	-	-	4.3% (2.0-8.6)	5.5% (3.7-8.8)
N	5,935	1,121	2,518				753	5610
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

Table A2. 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 2000-2015.

Year	SR Sp/Su Chinook	SR Fall Chinook	UCR Sp Chinook	UWR Sp Chinook	SR Sockeye	MCR Steelhead	SR Steelhead	UCR Steelhead
2000	4.6% (3.6-6.6)	3.3% (2.1-5.3)	2.2% (1.2-3.8)	-	-	-	10.5% (8.4-15.0)	6.2% (12.9-22.7)
N	11,810	1,323	1,123				10,356	3,100
2001	14.0% (11.1-20.0)	6.4% (4.2-10.0)	13.2% (9.9-19.5)	-	-	15.0% (11.1-21.9)	33.9% (26.3-49.1)	-
N	8,845	807	1,230			872	774	
2002	2.9% (2.3-4.1)	1.7% (1.2-2.6)	2.5% (1.9-3.5)	-	-	-	21.9% (17.6-31.0)	14.2% (10.1-21.3)
N	30,617	4,899	20,493				7,331	561
2003	4.7% (3.7-6.9)	2.7% (2.0-4.0)	3.7% (2.9-5.3)	-	-	-	26.0% (21.0-36.2)	19.0% (15.4-26.9)
N	28,150	6,234	30,723				8,553	27,918
2004	4.8% (3.6-7.0)	1.3% (0.6-2.6)	3.7% (2.9-5.4)	-	-	-	25.8% (19.7-37.3)	14.1% (11.3-19.8)
N	4,816	929	9,533				803	6,040
2005	3.0% (2.2-4.4)	1.3% (0.6-2.6)	2.4% (1.6-3.8)	-	-	-	28.3% (21.6-40.6)	15.1% (11.9-21.6)
N	5,935	1,121	2,518				753	5,610
2006	3.3% (2.4-5.0)	2.5% (1.7-3.9)	3.6% (1.8-6.6)	-	-	-	27.5% (21.0-39.1)	23.4% (18.1-34.1)
N	5,570	4,057	731				1,100	2,064
2007	3.1% (2.5-4.4)	3.4% (2.3-5.3)	1.9% (1.2-3.2)	1.4% (0.8-2.5)	-	18.7% (14.6-26.8)	22.6% (18.2-32.4)	15.7% (12.4-22.6)
N	23,830	2,005	2,268	1,505		2,234	6,391	3,042
2008	2.5% (1.9-3.6)	1.9% (1.5-2.7)	1.7% (1.0-2.9)	4.4% (3.2-6.7)	-	13.5% (10.6-19.2)	14.2% (11.5-19.9)	16.7% (13.1-24.2)
N	11,425	24,136	1,662	2,509		2,291	19,572	2,513
2009	4.7% (3.7-6.9)	2.0% (1.5-2.9)	3.7% (2.5-5.6)	1.7% (1.2-2.7)	1.3% (0.7-2.2)	14.1% (11.1-20.0)	14.5% (11.9-20.1)	20.0% (15.6-29.3)
N	17,396	16,314	2,064	5,573	1,845	2,700	23,311	2,265
2010	3.4% (2.7-4.8)	0.7% (0.5-1.1)	2.9% (2.2-4.3)	1.8% (0.6-4.4)	1.6% (0.8-2.9)	11.9% (9.4-17.4)	14.3% (11.3-20.4)	13.7% (11.0-19.3)
N	38,441	17,974	5,972	510	1,382	8,515	40,024	12,284
2011	2.5% (1.8-3.6)	0.7% (0.5-1.1)	2.9% (1.4-5.3)	0.9% (0.3-2.0)	0.4% (0.1-1.3)	9.6% (6.6-14.7)	12.0% (9.4-17.3)	9.1% (6.9-13.4)
N	6,557	12,327	704	1,119	826	865	7,028	2,419
2012	2.2% (1.7-3.3)	0.7% (0.5-1.1)	1.2% (0.7-2.1)	0.7% (0.4-1.3)	2.1% (1.2-3.7)	9.4% (6.5-14.4)	10.2% (7.7-14.9)	7.5% (5.6-11.3)
N	17,929	10,742	3,227	3,731	1,457	1,084	4,768	3,357
2013	1.2% (0.8-1.8)	0.9% (0.5-1.6)	0.7% (0.3-1.4)	1.0% (0.5-1.8)	0.8% (0.3-2.0)	9.9% (7.0-15.3)	12.7% (9.6-18.5)	8.9% (6.6-13.4)
N	16,167	4,465	3,112	2,629	1,454	1,865	8,516	4,473
2014	1.1% (0.8-1.7)	1.0% (0.5-1.9)	1.4% (0.7-2.5)	1.2% (0.5-2.5)	1.6% (0.8-3.0)	9.5% (6.5-14.5)	8.6% (6.7-12.5)	11.4% (8.5-16.8)
N	14,828	2,800	2,297	1,587	1,739	1,119	8,812	3,841
2015	2.0% (1.5-2.9)	0.8% (0.4-1.5)	1.9% (1.3-2.9)	0.4% (0.1-1.5)	1.6% (1-2.6)	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

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 2006-2015.

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

APPENDIX B: HATCHERY VERSUS WILD COMPARISONS

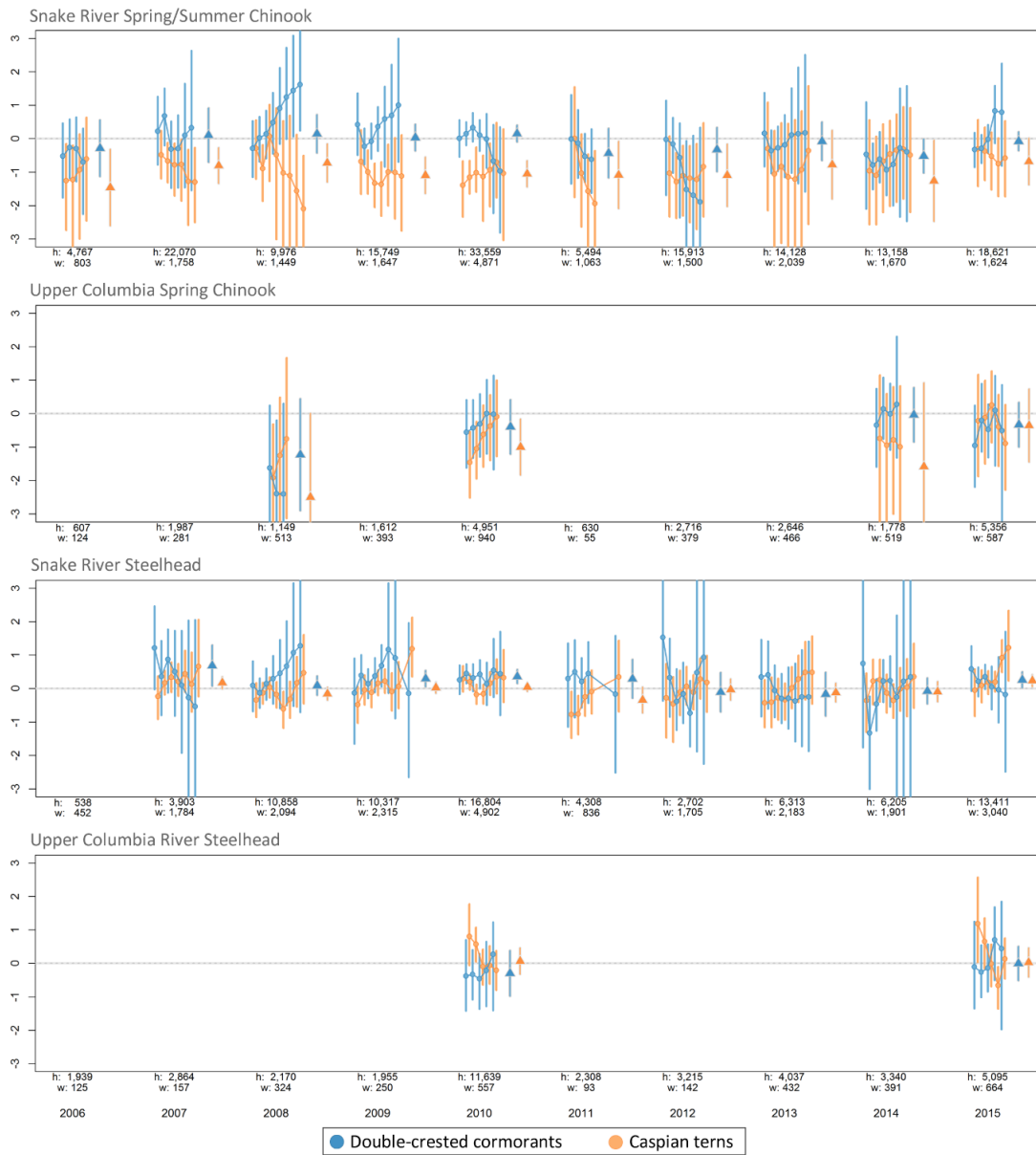


Figure B1. Relative susceptibility of fish by rear-type to predation by double-crested cormorants and Caspian terns nesting on East Sand Island during 2006-2015. Values represent the odds ratio of predation (y_1), with values < 0 indicating greater predation odds for hatchery fish and values > 0 indicating greater predation odds for wild fish. Error bars represent 95% confidence intervals, with values over-lapping 0 not statistical significant. Only years were > 500 PIT-tagged fish of each rear-type available are presented. Weekly estimates (circles) are followed by an annual estimate (triangles).

APPENDIX C: INRIVER VERSUS TRANSPORTED COMPARISONS

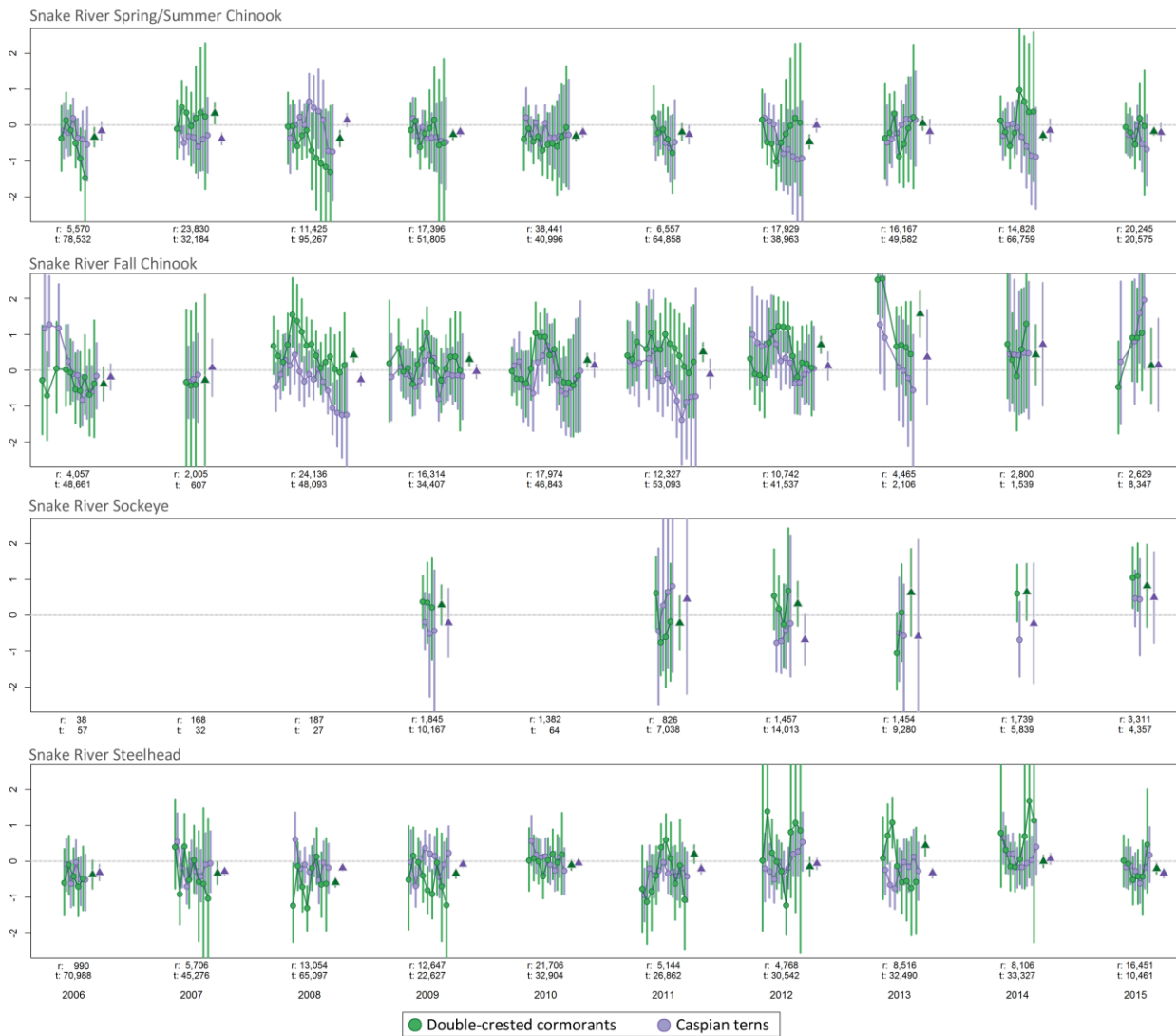


Figure C1. Relative susceptibility of fish by migration history to predation by double-crested cormorants and Caspian terns nesting on East Sand Island during 2006-2015. Values represent the odds ratio of predation (y_1), with values < 0 indicating greater predation odds for in-river fish and values > 0 indicating greater predation odds for transported fish. Error bars represent 95% confidence intervals, with values over-lapping 0 not statistical significant. Only years were > 500 PIT-tagged fish of each migration history available were presented. Weekly estimates (circles) are followed by an annual estimate (triangles).

APPENDIX D: FACTORS INFLUENCING PREDATION RATES

Executive Summary

Another objective of our research was to evaluate the relationship between predation rates and various biotic and abiotic conditions (covariates) in the Columbia River estuary using data collected since 2000. These data may be important in describing natural variation in predation impacts and identifying factors that influence a fish's susceptibility to bird predation in the estuary. Due to exploratory and ancillary aspects of this particular objective, data are presented as an appendix and results should be viewed as starting point for future analyses. For instance, additional data and model sleuthing are needed to more fully understand the dynamic and complex set of environmental and biological factors that influence fish susceptibility to avian predation in the Columbia River estuary.

Our analysis of abiotic and biotic factors potentially related to predation rates indicated that the historic variation in Caspian tern and double-crested cormorant predation rates were influenced by both manageable factors (i.e., colony size reductions and river operations) and environmental factors working at multiple scales (e.g., large scale climate indices such as the North Pacific Gyre Oscillation {NPGO} and regional upwelling). For terns, manageable factors drove approximately one quarter or less of the variability seen in predation rates during 2000-2015, while environmental factors were responsible for the largest share. For cormorants, the amount of variability in predation rates explained by management relevant factors varied substantially across highly ranked models but in many cases approached or exceeded 50%. Management relevant variables that were common to most of the highly ranked models generally fell into two categories: those describing bird colony size and productivity, and those related to operation of Bonneville Dam, particularly the percentage of water passed via spill. Further investigation of factors associated with the operation of Bonneville Dam would be helpful to confirm this preliminary result and clarify their potential influence on avian predation rates in the estuary. Presumably, environmental factors influence the availability of marine forage fish within the estuary which have been shown to be important in regulating predation on salmonids by both predators. This analysis adds to the evidence that predation on salmonids occurs within the context of a diverse forage fish community where factors that affect the availability of alternative prey species ultimately effect predation on salmonids, and factors that affect the availability of other salmonid ESU/DPSs ultimately affect predation on any particular salmon or steelhead population.

Background

It is well documented that biotic and abiotic conditions can play an important role in the survival of juvenile salmonids during outmigration to the ocean and after ocean entry (e.g., Petrosky and Schaller 2010, Burke et al. 2013, Peterson and Burke 2013; Evans et al. 2014). While levels of tern and cormorant predation on some populations of juvenile salmonids have been high on average, there has been substantial intra- and inter-annual variability in predation impacts (Lyons et al. 2014b). For instance, predation rates on the same salmonid population

can vary significantly by week (BRNW 2015) and year (Evans et al. 2012). Furthermore, even within the same salmonid population, differences in predation probabilities based on a fish's rear-type (hatchery, wild), migration history (in-river, transported), and run-timing have been observed (Ryan et al. 2003; Zamon et al. 2013; Lyons et al. 2014b; BRNW 2015). An exploratory analysis of the effects of biotic and abiotic conditions has previously been conducted for double-crested cormorants at East Sand Island (Lyons et al. 2014b), but is lacking for Caspian terns. Additionally, this previous cormorant analysis focused on the effects of strictly environmental variables after considering just one management relevant variable, colony size. An analysis investigating the potential effects of a more inclusive set of management relevant variables might identify additional factors that influence smolt susceptibility to avian predation.

Methods

To investigate the influence of biotic and abiotic factors on predation rates, we explored a suite of possible explanatory variables to explain variation in annual predation rates by Caspian terns and double-crested cormorants on Columbia River steelhead DPSs and Chinook salmon ESUs over a 16-year study period (2000-2015). The ultimate goal of this analysis was to assess the potential effects of “management relevant” factors or variables under some modicum of control by river managers. In order to accurately assess the effects of these variables, it was also necessary to assess the amount of variation in predation rates that was due to strictly environmental factors. Previous efforts to identify factors influencing cormorant predation rates have focused on environmental factors and a single manageable factor, colony size (Lyons et al. 2014b). In the present study, we included a larger suite of potential management relevant variables in our analyses (see list below). Ultimately, we sought to describe how much these manageable factors have influenced tern and cormorant predation rates to date, with the goal if potentially identify multiple management-relevant variables, in addition to colony size, that could reduce predation impacts by terns and cormorants nesting on East Sand Island.

Explanatory Variables Assessed: A list of possible explanatory variables of interest were compiled from previous research (Schaller and Petrosky 2007, Haeseker et al. 2012, Lyons et al. 2014b) and a Data Analysis Workshop coordinated by USACE (POC David Leonhardt) and attended by representatives from Real Time Research (RTR), U.S. Army Corps of Engineers (USACE), Bonneville Power Association (BPA), NOAA Fisheries (NOAA), Columbia River Inter-Tribal Fish Commission (CRITFC), U.S. Fish and Wildlife Service (USFWS), Fish Passage Center (FPC), Oregon Department of Fish and Wildlife (ODFW), Oregon State University (OSU), and Western Ecosystems Technology (WEST).

Variables describing river conditions included river discharge upstream of the estuary; average migration travel time; temperature, pool elevation, total discharge, and the smolt passage index as measured at Bonneville dam; and average smolt survival probability between McNary and Bonneville dams. River discharge and turbidity values were measured at the Beaver Army Terminal (rkm 87; obtained from the U.S. Geological Survey [<http://waterdata.usgs.gov>]). Travel time in the river was calculated as an annual average difference in days between successive observations of PIT-tagged fish at Bonneville Dam corner collector and juvenile bypass and at

the PIT tag pair-trawl in the estuary. Average estimates of survival from McNary dam to Bonneville dam were calculated using a CJS model using observations of PIT-tagged fish at McNary dam, Bonneville dam, and the Estuary Trawl (see Cormack 1964 for methodology). All PIT tag information was retrieved from PITAGIS (<http://www.ptagis.org/>). Metrics associated with Bonneville Dam, including daily estimates of spill, temperature, pool elevation, total discharge, and smolt indices, were obtained from the Columbia Basin Research website maintained by the University of Washington (<http://www.cbr.washington.edu/dart>). Smolt indices were used as a measure of total species-specific availability as well as relative availability (i.e., species total availability/sum of all species total availability).

Several variables describing the colonies themselves were also considered. Metrics of tern and cormorant colony size included annual peak colony size (number of adults), average weekly colony counts, and a cumulative sum of these weekly observations. A metric of productivity (total number of fledglings produced annually by each colony) was included as a means of quantifying the energy demands of the colony. Variables accessing the relative availability of marine forage versus fresh water and anadromous fish were calculated from diet data collected by Bird Research Northwest since 2000. The majority of this data was obtained from Bird Research Northwest (<http://www.birdresearchnw.org>). The 2015 double-crested cormorant colony size was obtained from David C. Smith and Associates, Inc. (Smith and Associates 2016).

Several larger scale environmental indices were considered including the Pacific Decadal Oscillation (PDO; obtained from the University of Washington Joint Institute for the Study of the Atmosphere and Ocean [<http://jisao.washington.edu/pdo>]), Multivariate El-Nino Index (MEI; obtained from the National Oceanic and Atmospheric Administration [NOAA] Earth System Research Laboratory [<http://www.cdc.noaa.gov>]), North Pacific Gyre Oscillation (NPGO; obtained from Emanuele Di Lorenzo [<http://www.o3d.org/npgo>]), Pacific Northwest Index (PNI; obtained from the Columbia Basin Research website). For each index, we averaged the monthly values associated with each year's tern and cormorant breeding season.

We further calculated several metrics to quantify the effects of upwelling in the Columbia River Estuary. Upwelling indices corresponding to the duration of the steelhead and yearling Chinook salmon smolt migration for each year were obtained from NOAA's Northwest Fisheries Science Center (<http://www.nwfsc.noaa.gov>). A cumulative sum and average daily values were used to measure the strength of the upwelling. Annual dates of the spring transition to upwelling were used to quantify upwelling timing.

Of the variables described above, we classified spill percentage at Bonneville dam, river discharge at Beaver Army Terminal, average smolt migration travel time from Bonneville to the Trawl, and all colony size and productivity metrics as "management relevant". We classified upwelling strength and timing; environmental indices; turbidity at Beaver Army Terminal; temperature, pool elevation and smolt indices at Bonneville dam; and colony diet data as "strictly environmental".

Response Variables and Model Structure: In order to simplify the analysis and interpretation we chose to combine similar ESU/DPS designated fish together, for each species. For steelhead, we modeled predation on Upper Columbia River, Snake River and Middle Columbia River DPSs combined. For Chinook, we combined the Upper Columbia River spring and Snake River spring/summer ESUs combined. These combined predation rate estimates were based on the availability of PIT-tagged smolts last detected passing Bonneville Dam, as described above, and were log transformed in order to correct for the asymmetry present in predation rate uncertainty intervals. The analysis of variation in predation rates was performed using multiple linear regression. Weighting was employed in order to account for the dissimilarity in the precision of predation rate estimates. Weights were defined to be the inverse of the squared standard error of the log predation rate estimates.

Modelling Process: We constructed a list of suitable models and instituted a two-stage model selection process, which reflected the objective of assessing the effects of management relevant variables after allowing for strictly environmental effects. In the initial stage, we evaluated how much variation in predation rates were explained by environmental variables alone (i.e. variables outside the control of managers) using an all subset approach through a custom written routine in R (R Development Core Team 2013). This routine accessed the relevance of the suite of environmental variables in explaining predation rates including all possible two way interactions equal to or less than eight degrees of freedom (half the number of years of data). We partitioned the full list of models by the number of total included parameters (the total number individual and interaction terms). We ranked models by their associated Bayesian Information Criterion (BIC) and selected the ten highest ranked models from each partition. The second stage of the modelling process expanded the all subsets routine to incorporate management relevant variables into each of the above selected models, including all possible interactions between management relevant and the strictly environmental variables. We used data from the years 2000-2012 to assemble this preliminary list. This enabled an independent evaluation of each models predictive abilities using data from 2013-2015. Furthermore, this approach enabled evaluation of several explanatory variables (turbidity, colony size characteristics for double-crested cormorants, etc.) for which data was not available past 2013.

We combined this final list of models and selected the hundred highest ranked models again using BIC. Each of the models of this list were subjected to additional evaluation to prevent spurious results due to influential data points (i.e., unusual explanatory values with large amounts of leverage) and to prevent overfitting (including an excessive number of explanatory variables to fit the sample data well but adding little value to forecasting future values). For each model we tested several influence-diagnostics (studentized residuals, leverage, and cook's distance [Chatterjee and Hadi 1986]), to reject any model overly influenced by outlier explanatory variables. We finally ranked this penultimate list by a combination of R^2 values, jackknife cross validation, and their ability to predict 2013-2015 predation rates. We compiled these rankings to ultimately select the five most defensible models of predation for each colony on each species collection. Finally, the total amount of variation explained by each models as

well as that explained by the “management relevant” and “strictly environmental” subset of variables (R-squared values) were calculated using the full 2000-2015 dataset.

Results and Conclusions

For all four analyses performed (tern predation on steelhead, tern predation on Chinook salmon, cormorant predation on steelhead, and cormorant predation on Chinook salmon), the five highest ranked models each included 5–8 variables (*Table D1* and *Table D2*). These parameter-rich models achieved excellent fits to the 16-year predation rate datasets, with the amount of variation explained (R^2 values) ranging from 76–91% for tern predation and 65–92% for cormorant predation. The highest ranked model for each analysis explained 82% and 91% of tern predation on steelhead and Chinook salmon, respectively, and 74% and 68% of cormorant predation on steelhead and Chinook salmon, respectively. These high ranking models, when based on data from 2000-2012, were able to predict predation rates for 2013-2015 reasonably well (*Figure D1*), indicating that model overfitting was not a significant problem.

For predation by terns, management relevant variables explained a smaller percentage of the total variation seen in predation rates compared with strictly environmental variables (*Table D1*). In the high ranking models for tern predation on steelhead, management relevant variables explained 3–10% of the variability, whereas environmental factors explained 67–75%. For 4 of the 5 models ranked highest for tern predation on Chinook salmon, management relevant variables explained 17–28% of the variability, while environmental factors explained 62–71%. One model of tern predation on Chinook anomalously flipped this relationship, however; 81% of the variability was explained by management factors and 9% by environmental factors.

Management relevant variables often explained a greater proportion of the variability in cormorant predation rates (*Table D2*). In the high ranking models for cormorant predation on steelhead, 0–53% of the variability was explained by management relevant variables and 36–75% explained by strictly environmental variables. Management relevant variables always explained a greater portion of the variability in cormorant predation rates on Chinook salmon, 43–63%, compared to that explained by environmental variables, 3–5%.

Management relevant variables that were common to multiple models generally fell into two categories: those describing bird colony size and productivity, and those related to operation of Bonneville Dam. Colony size metrics such as peak colony size, average colony counts across the season, and cumulative colony counts across the season appeared in all of the highly ranked tern predation models and several of the cormorant models. Colony productivity was present in a majority of cormorant models. The percentage of water spilled at Bonneville Dam (“percent spill”) was present in a majority of tern predation models. The pool elevation at Bonneville Dam and measures of discharge (mean and total discharge across the season) were present in a few models of predation by both species.

Several environmental variables were included in many of the highly ranked models. The North Pacific Gyre Oscillation large scale climate index was included in all of the tern predation on steelhead models, consistent with its importance in models of cormorant predation examined in Lyons et al. (2014b). Timing of the spring transition to coastal upwelling was also important in several highly ranked models. Presumably, these variables influence the availability of marine forage fish within the estuary. Marine forage fish have been shown to be important in regulating predation on salmonids for both species (Lyons 2010, Lyons et al. 2014b). Consistent with the demonstrated importance of alternative (non-salmonid) prey types, the availability of steelhead and/or Chinook salmon, and the availability of each relative to other salmonids, were important in many highly ranked models. A broad array of evidence has confirmed that both terns and cormorants are generalist predators with diverse diets that respond to fluctuating availability of a variety of prey species. Predation on salmonids occurs within the context of a diverse forage fish community (Weitkamp et al. 2012) where factors that affect the availability of alternative prey species ultimately effect predation on salmonids, and factors that affect the availability of other salmonid ESU/DPSs ultimately affect predation on any particular salmon or steelhead population.

The multivariate analyses of factors influencing predation rates indicated that reductions in tern colony size have not been the sole contributor to reducing tern predation rates on both steelhead and Chinook salmon. For example, percent spill was an important component of many highly ranked models and the two highest percent spill values occurred during the 2011-2015 management regime (in years 2011 and 2015). Fluctuations in large scale climate indices (NPGO, MEI) may have also created conditions where terns relied less on salmonids, presumably through the mechanism of regulating the abundance and availability of marine forage fish in the estuary. Management of the cormorant colony has not yet reduced the size of the colony, but our predation rate datasets spanned a period of initial colony growth followed by a decade of relative stability colony sizes with some natural fluctuations (*Figure D1*) and thus considered a range of cormorant colony sizes. Colony size was not consistently included in a majority of the highly ranked models of cormorant predation; however, colony productivity was. Perhaps counter-intuitively, in most models cormorant predation was negatively related to productivity, with low productivity associated with higher predation rates on steelhead and Chinook salmon. Lower productivity may be associated with poor marine forage fish (anchovy, herring, and others) availability (the primary prey species during the cormorant chick rearing period). Premature nest failure due to eagle disturbance or other factors early in their breeding season might also facilitate cormorants foraging more in the upper estuary in locations more distant from East Sand Island and where salmonids are the primary prey species. Both factors limiting cormorant (or potentially tern) productivity could be associated with greater predation on smolts. The relatively small role colony size had in explaining past variability in cormorant predation rates is due to highly variable environmental and river management conditions across the study period, coupled with only a few years of smaller colony size. The effect of smaller colony size in a few early years was apparently masked by coincident conditions that promoted cormorant predation on smolts. Consistent with Lyons et al. (2014b), the analyses presented here suggest that to observe reductions in predation due to reductions in tern or cormorant colony size, those reductions will need to be large and sustained.

The relationship observed between the percentage of water spilled at Bonneville and predation rates on smolts by Caspian terns in the estuary is a new finding and should be viewed as preliminary. Previously, river discharge has been shown to be related to smolt predation by Caspian terns (Lyons 2010). While under some management control, percent spill is unavoidably constrained by total river discharge, as are other aspects of the operation of Bonneville Dam (e.g., pool elevation, water discharge through the dam). It is possible that some of the mathematical relationship between tern predation rates and percent spill is driven by the documented effects of total river discharge on the estuary environment (e.g., Weitkamp et al. 2012) and smolt passage through the estuary. Additional analyses using a focused modelling approach designed to evaluate the specific effects spill (or perhaps in conjunction with the pool elevation and discharge at Bonneville Dam) and salmonid abundance in estuary (if accurate measures at various spatial and temporal scales can be obtained) with annual or within-season datasets might provide greater clarity than the exploratory analyses we report here are capable of.

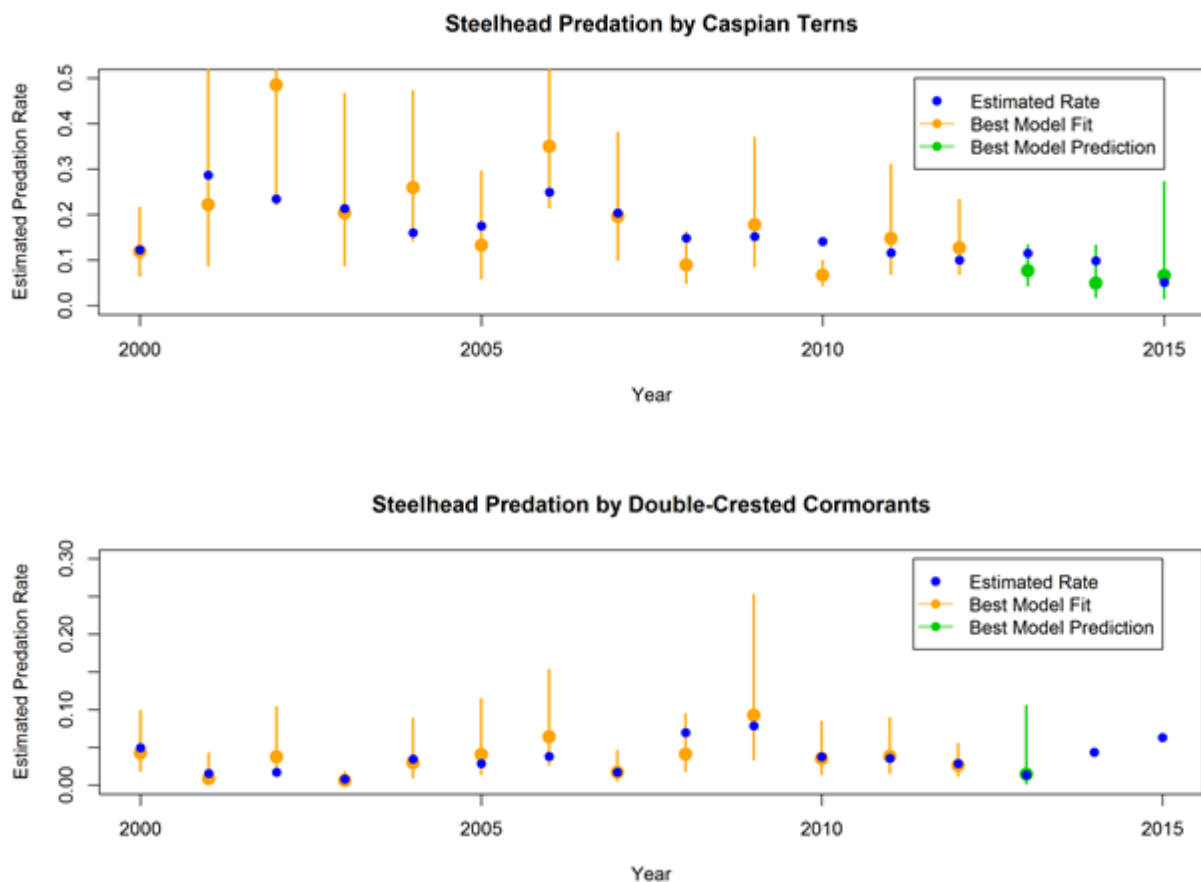


Figure D1. Annual predation rates on steelhead by East Sand Island Caspian terns and double-crested cormorants. Modeled predation rates are based on the highest ranked model for each species relating selected biotic and abiotic factors to predation rates during 2000 – 2012.

Table D1: The five highest ranked models explaining variation in Caspian tern predation rates on steelhead (Upper Columbia River, Snake River, and Middle Columbia River DPSs combined) and spring/summer Chinook salmon (Upper Columbia River and Snake River ESUs combined). Modeled values for the top model (in green) of steelhead consumption are graphed alongside observed values in Figure D1.

Best Models of Steelhead Predation by Caspian Terns on East Sand Island	Variation Explained (R ²)		
	Environmental	Management	All
Model			
log(predation rate) ~ total weekly colony size + transition week - mean MEI - mean NPGO + mean spill pct + mean weekly steelhead pct + total steelhead index	74.8%	7.2%	82.0%
log(predation rate) ~ total weekly colony size + transition week + mean NPGO + mean temperature + mean weekly steelhead pct - transition week*mean temperature - transition week*mean weekly steelhead pct	73.3%	6.4%	79.6%
log(predation rate) ~ mean colony size - transition week - mean MEI + mean NPGO + mean weekly steelhead index + mean spill pct - mean weekly steelhead pct	73.1%	6.4%	79.5%
log(predation rate) ~ -1/colony size - transition week - mean PDO + mean weekly steelhead index + mean spill pct	67.1%	9.7%	76.7%
log(predation rate) ~ -1/colony size + transition week + mean NPGO + mean temperature + mean weekly steelhead pct - transition week*mean temperature + transition week*mean weekly steelhead pct	73.3%	2.7%	75.9%
Best Models of Yearling Chinook Predation by Caspian Terns on East Sand Island	Variation Explained (R ²)		
	Environmental	Management	All
Model			
log(predation rate) ~ -transition week + mean MEI - mean spill pct - total Chinook index + max colony size - mean MEI*max colony size + mean spill pct*total Chinook index	71.4%	19.7%	91.1%
log(predation rate) ~ mean discharge at BON - transition week - mean spill pct - total Chinook index + max colony size + mean spill pct*total Chinook index - mean discharge at BON*max colony size	71.4%	19.5%	90.8%
log(predation rate) ~ colony productivity - mean upwelling + mean BON pool elevation + 1/sum weekly colony size - colony productivity*mean BON pool elevation + mean upwelling*1/sum weekly colony size - mean BON pool elevation*1/sum weekly colony size	9.1%	80.6%	89.7%
log(predation rate) ~ -mean spill pct + mean temperature + total Chinook index + max colony size + mean spill pct*total Chinook index - mean temperature*total Chinook index	63.8%	23.2%	87.0%
log(predation rate) ~ -transition week - mean spill pct - total Chinook index + max colony size + mean spill pct*total Chinook index	71.4%	15.3%	86.7%

Table D2: The five highest ranked models explaining variation in double-crested cormorant predation rates on steelhead (Upper Columbia River, Snake River, and Middle Columbia River DPSs combined) and spring/summer Chinook salmon (Upper Columbia River and Snake River ESUs combined). Modeled values for the top model (in green) of steelhead consumption are graphed alongside observed values in Figure D1.

Best Models of Steelhead Predation by double-crested cormorants on East Sand Island		Variation Explained (R ²)		
Model	Environmental	Management	All	
log(predation rate) ~ colony productivity + mean weekly steelhead index + mean spill pct - cumulative upwelling - total steelhead index - total discharge at BON + max colony size - mean weekly steelhead index*total steelhead index	75.2%	17.0%	92.2%	
log(predation rate) ~ -transition week - colony productivity - mean weekly steelhead index - mean temperature + mean weekly steelhead pct + mean BON pool elevation + max colony size + colony productivity*mean temperature	36.1%	53.2%	89.3%	
log(predation rate) ~ colony productivity - mean upwelling + mean PDO - mean weekly steelhead index + mean weekly steelhead pct + mean BON pool elevation + total steelhead index - mean PDO*mean BON pool elevation	55.4%	33.2%	88.6%	
log(predation rate) ~ colony productivity + mean NPGO - mean weekly steelhead index + mean weekly steelhead pct + mean BON pool elevation	36.6%	44.6%	81.1%	
log(predation rate) ~ colony productivity - mean PDO + mean weekly steelhead index + mean weekly steelhead pct + mean BON pool elevation - mean weekly steelhead index*mean weekly steelhead pct	54.9%	24.0%	78.8%	
Best Models of Yearling Chinook Predation by double-crested cormorants on East Sand Island		Variation Explained (R ²)		
Model	Environmental	Management	All	
log(predation rate) ~ colony productivity + mean weekly Chinook index + mean spill pct + total Chinook index - annual Chinook pct - mean BON pool elevation - mean weekly Chinook index*total Chinook index - colony productivity*annual Chinook pct + annual Chinook pct*mean BON pool elevation	56.7%	24.9%	81.6%	
log(predation rate) ~ colony productivity - transition week + mean upwelling + mean weekly Chinook index - total Chinook index + transition week*total Chinook index + mean upwelling*mean weekly Chinook index - mean upwelling*total Chinook index	45.6%	35.7%	81.3%	
log(predation rate) ~ -colony productivity - mean PDO + mean weekly Chinook pct - cumulative upwelling + total Chinook index - mean discharge at BON + colony productivity*mean discharge at BON	23.2%	53.6%	76.7%	
log(predation rate) ~ -colony productivity - mean upwelling - mean PDO + mean weekly Chinook pct + total Chinook index - mean discharge at BON + colony productivity*mean discharge at BON	25.1%	49.3%	74.4%	
log(predation rate) ~ colony productivity - transition week - mean upwelling + mean weekly Chinook index - total Chinook index + transition week*mean upwelling + transition week*total Chinook index + mean upwelling*mean weekly Chinook index	54.3%	17.7%	72.1%	