U.S. Army Corps of Engineers – Walla Walla District

Impacts of Avian Predation on Salmonid Smolts from the Columbia and Snake Rivers

2004-2009 Synthesis Report

2004-2009 SYNTHESIS REPORT

IMPACTS OF AVIAN PREDATION ON SALMONID SMOLTS FROM THE COLUMBIA AND SNAKE RIVERS

This 2004-2009 Synthesis Report has been prepared for the U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This report is not for citation without permission of the authors.

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EXECUTIVE SUMMARY

Populations of anadromous salmonids (*Oncorhynchus* spp.) in the Columbia River basin are currently the subjects of intense conservation activity following decades of decline. In recent years, avian predation across the basin has been considered a factor limiting recovery of these imperiled fish populations. Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), American white pelicans (*Pelecanus erythrorhynchos*), California gulls (*Larus californicus*), and ring-billed gulls (*L. delawarensis*) are native piscivorous colonial waterbirds with a history of nesting in the Columbia Plateau region. We investigated the impact on survival of juvenile salmonids from predation by piscivorous colonial waterbirds nesting in the Columbia Plateau region during 2004-2009.

Within the Columbia Plateau region, overall numbers of breeding Caspian terns remained relatively stable during the study period at between 800 and 1,000 breeding pairs at five colonies; the two largest breeding colonies were on Crescent Island in the mid-Columbia River and on Goose Island in Potholes Reservoir. Overall numbers of breeding double-crested cormorants in the Columbia Plateau region decreased during the study period, from about 1,500 breeding pairs to about 1,200 breeding pairs at four separate colonies; the largest breeding colony by far was at the north end of Potholes Reservoir. Numbers of breeding American white pelicans increased at the Badger Island colony on the mid-Columbia River, the sole breeding colony for the species in the State of Washington. Overall numbers of breeding gulls, the most numerous piscivorous colonial waterbirds in the region, declined during the study period. Potential limiting factors for piscivorous colonial waterbirds nesting in the Columbia Plateau region include human disturbance, mammalian predation, availability of suitable nesting habitat, inter-specific competition for limited nesting habitat, and food availability. Overall breeding numbers of Caspian terns and double-crested cormorants in the Columbia Plateau region are an order of magnitude less than the numbers of these two species nesting in the Columbia River estuary, whereas California gulls, ring-billed gulls, and American white pelicans are far more numerous in the Columbia Plateau region than in the estuary.

We used bioenergetics methods to estimate prey consumption by Caspian terns nesting at Crescent Island and double-crested cormorants nesting at Foundation Island, both located in the mid-Columbia River just below the confluence with the Snake River. Taken together, the Crescent Island tern colony and the Foundation Island cormorant colony consumed approximately one million juvenile salmonids annually during 2004 – 2009. Estimated annual consumption of smolts by Foundation Island cormorants ranged from 470,000 to 880,000, while that of Crescent Island terns ranged from 330,000 to 500,000. Consumption of salmon smolts by the Crescent Island tern colony declined during the study period, tracking a decline in colony size. Consumption of steelhead (*O. mykiss*) did not decline, however, perhaps reflecting greater steelhead availability in later years due to reduced transportation rates of Snake River steelhead. There was no apparent trend in smolt consumption by Foundation Island cormorants during the study period. Relative to salmonids, consumption of lamprey was minor, with fewer than 10,000 lamprey macropthalmia consumed per year by both colonies combined.

We recovered passive integrated transponder (PIT) tags from salmonid smolts on nine different piscivorous waterbird colonies in the Columbia River basin to evaluate avian predation on juvenile salmonids during 2004-2009. These nine bird colonies had the highest numbers of smolt PIT tags of any in the Columbia Basin. Minimum estimates of predation rates based on PIT tag recoveries were used to determine which salmonid stocks were most affected by avian

predation and which bird colonies had the greatest impact on smolt survival. This system-wide evaluation of avian predation indicated that Caspian terns and double-crested cormorants nesting on East Sand Island in the Columbia River estuary were consuming the highest proportions of available PIT-tagged smolts. However, Caspian terns and double-crested cormorants nesting at colonies in the Columbia Plateau region also had significant impacts on survival of specific salmonid stocks. Predation rates by Crescent Island terns on Snake River summer steelhead (7.7%) and by Goose Island terns on upper Columbia summer steelhead (10.0%) were substantial during the study period. Predation rates by Foundation Island cormorants on Snake River summer steelhead (2.0%) and Snake River sockeye (1.7%) were not as high, but notable. Predation rates by gulls and pelicans nesting in the Columbia Plateau region were minor (generally < 0.5% of available smolts) compared to smolt losses from inland tern and cormorant colonies. Hatchery smolts were often more susceptible to avian predation relative to their wild counterparts, although exceptions were numerous. Smolts out-migrating in June and July were often consumed at higher rates by birds than smolts of the same stock that out-migrated earlier (April or May).

Predation rates on PIT-tagged smolts that were adjusted for colony size (i.e., smolt consumption per bird) were substantially higher for terns and cormorants nesting at colonies in the Columbia Plateau region compared to those nesting in the estuary. Thus, while inland colonies of terns and cormorants are much smaller than their counterparts in the estuary, inland colonies can be more reliant on salmonids as a food source. This greater reliance on salmonids, coupled with lower diversity of available salmonid stocks compared to the estuary, is responsible for the unexpectedly high impact of some inland tern and cormorant colonies on specific stocks of salmonids, particularly steelhead. Current management efforts to increase smolt survival through reductions in tern and cormorant predation in the estuary could result in some terns and cormorants from estuary colonies recruiting to inland colonies, potentially resulting in higher predation rates on certain ESA-listed salmonid stocks. Recruitment from estuary colonies may result in small, but significant increases in numbers of these two species nesting in the Columbia Plateau region. Nesting habitat and food supply appear to limit Caspian tern numbers on the Columbia Plateau and the demographic connectivity between the double-crested cormorant colony in the estuary and those on the Columbia Plateau appears limited. Although the number of Caspian terns that could relocate from estuary colonies to colonies on the Columbia Plateau is likely small relative to numbers nesting in the Columbia River estuary (< 1,000 adults), the impact on specific steelhead stocks could be substantial and warrants monitoring.

We investigated factors that influence susceptibility of juvenile salmonids to avian predation using juvenile steelhead from the threatened Snake River stock. Steelhead smolts (n = 25,909) were captured, externally examined, marked with PIT tags, and released to continue out-migration during 2007-2009. Recoveries of steelhead PIT tags on the Crescent Island Caspian tern colony indicated that steelhead susceptibility to tern predation increased significantly with declining steelhead external condition, decreased water discharge, decreased water clarity, and increased steelhead length up to 202 mm (fork length), but decreased for larger steelhead. Recoveries of PIT tags on the Foundation Island double-crested cormorant colony indicated that steelhead susceptibility to cormorant predation increased significantly with declining steelhead external condition, plus steelhead of hatchery origin were more susceptible compared to their wild counterparts. These results indicate that steelhead susceptibility to avian predation is condition- and size-dependent and is influenced by both river conditions and rearing environment (hatchery vs. wild). These findings unequivocally demonstrate that at least a portion of the smolt mortality caused by avian predation in the mid-Columbia River is compensatory.

We also assessed the abundance, distribution, and diet of double-crested cormorants over-wintering on the lower Snake River in eastern Washington to investigate the potential for significant impacts from cormorant predation on survival of ESA-listed fall Chinook salmon that over-winter in the lower Snake River. A monthly average of 256 cormorants was observed on this reach of the lower Snake River. Overall diet composition of cormorants was highly variable and changed as winter progressed. The most prevalent prey types were centrarchids (34.3% by mass), followed by shad (15.0%). Fall Chinook salmon comprised an average of 3.4% by mass of the cormorant diet. Biomass consumption of all salmonids by overwintering cormorants was estimated at 3,100 to 11,000 kg, or about one third of the estimated salmonid biomass consumption by cormorants nesting at Foundation Island. The bulk of the diet of over-wintering cormorants, however, consisted of non-native fishes that compete with or depredate juvenile salmonids.

Based on the results of this study, the greatest potential for increasing survival of smolts from ESA-listed salmonid stocks by managing inland avian predators would be realized by focusing management efforts on Caspian terns nesting at colonies on Crescent Island, Goose Island, and the Blalock Islands. Reductions in the size of these tern colonies would enhance survival of upper Columbia River and Snake River steelhead stocks in particular. More limited enhancement of smolt survival for Snake River steelhead and Snake River sockeye could be achieved by managing the double-crested cormorant colony at Foundation Island. Management of other inland piscivorous waterbird colonies in the Columbia Plateau region would provide relatively small and perhaps undetectable increases in stock-specific smolt survival. Further work is necessary, however, to translate smolt consumption and predation rate estimates into assessments of the potential benefits for threatened and endangered salmonid populations of reducing avian predation in the Columbia Plateau region. The analysis of potential benefits from management of piscivorous waterbirds for restoring ESA-listed stocks of salmonids is key to informed decision-making, as resource managers consider management of specific waterbird colonies on the Columbia Plateau, and identifying management objectives.

CHAPTER 1

COLONY SIZE, NESTING SUCCESS, AND LIMITING FACTORS OF PISCIVOROUS COLONIAL WATERBIRDS IN THE COLUMBIA PLATEAU REGION, 2004-2009

This chapter has been prepared for U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This chapter summarizes colony size, nesting success, and limiting factors of Caspian terns, double-crested cormorants, American white pelicans, California gulls, and ring-billed gulls in the Columbia Plateau region during 2004-2009.

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SUMMARY

Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), American white pelicans (*Pelecanus erythrorhynchos*), California gulls (*Larus californicus*), and ring-billed gulls (*Larus delawarensis*) are piscivorous colonial waterbirds with a history of nesting in the Columbia Plateau region. The distribution and size of breeding colonies of these five species have been influenced, however, by human manipulation of river flows and nesting habitat, particularly during the last half of the 20th Century. More recently, fisheries managers in the region have grown concerned over the impact of predation by these piscivorous waterbirds on juvenile salmonids (*Oncorhynchus* spp.), some of which are listed under the Endangered Species Act. We investigated trends in colony size, nesting success, and potential limiting factors for colonies of piscivorous waterbirds during 2004-2009, with special emphasis on colonies with a history of salmonid depredation. Within the Columbia Plateau region, overall numbers of breeding Caspian terns remained relatively stable during the study period at between 800 and 1,000 breeding pairs at five colonies; the two largest breeding colonies were on Crescent Island in the mid-Columbia River and on Goose Island in Potholes Reservoir. Overall numbers of breeding double-crested cormorant in the Columbia Plateau region decreased slightly during the study period, from about 1,500 breeding pairs to about 1,200 breeding pairs at four separate colonies; the largest breeding colony by far was at the north end of Potholes Reservoir. Numbers of breeding American white pelicans increased at the Badger Island colony on the mid-Columbia River, the sole breeding colony for the species in the region and in the State of Washington, where the number of pelicans counted on-colony increased from about 900 individuals to about 1,800 individuals during the study period. Overall numbers of breeding gulls, the most numerous piscivorous colonial waterbirds in the Columbia Plateau region, declined during the study period, mostly because of the failure and abandonment of a large colony on Island 18 in the mid-Columbia River. Productivity of Caspian terns at Crescent Island and of double-crested cormorants at Foundation Island were lower compared to colonies of the respective species in the Columbia River estuary. Natal colony philopatry and colony site fidelity of Caspian terns at Crescent Island were lower than at East Sand Island in the Columbia River estuary. Some Caspian terns banded as chicks at Crescent Island were recruited to breed at East Sand Island. Potential limiting factors for piscivorous colonial waterbirds nesting in the Columbia Plateau region include human disturbance, mammalian predation, availability of suitable nesting habitat, inter-specific competition for limited nesting habitat, and food availability. Overall breeding numbers of Caspian terns and double-crested cormorants in the Columbia Plateau region are an order of magnitude less than the numbers of these two species nesting in the Columbia River estuary, while California gulls, ring-billed gulls, and American white pelicans are far more numerous in the Columbia Plateau region than in the estuary. Management to reduce the amount of nesting habitat for Caspian terns and double-crested cormorants in the Columbia River estuary would likely result in only a very small proportion of these displaced birds attempting to relocate to the Columbia Plateau region. Nesting habitat and food supply appear to limit Caspian tern numbers in the Columbia Plateau region and the demographic connectivity between the double-crested cormorant colony in the Columbia River estuary and those in the Columbia Plateau region appear limited.

INTRODUCTION

Predation by colonial waterbirds on juvenile salmonids (*Oncorhynchus* spp.) has become of concern to fisheries managers in the Columbia Basin, including the Columbia Plateau region (Ruggerone 1986, Schaeffer 1991, Jones et al. 1996, Collis et al. 2002, Roby et al. 2002, Roby et al. 2003, Antolos et al. 2005, Wiese et al. 2008). Some of the salmonids consumed by piscivorous colonial waterbirds are listed as threatened or endangered under the U.S. Endangered Species Act (Good et al. 2005). Piscivorous colonial waterbirds, specifically Caspian terns (*Hydroprogne caspia*), double-crested cormorants (*Phalacrocorax auritus*), American white pelicans (*Pelecanus erythrorhynchos*), California gulls (*Larus californicus*), and ring-billed gulls (*Larus delawarensis*), have a history of nesting on the Columbia Plateau (Brown 1926, Kitchin 1930, Decker and Bowles 1932, Hanson 1968, Conover et al. 1979, Thompson and Tabor 1981, Speich and Wahl 1989). Human manipulation of river flows and nesting habitat, however, have influenced the distribution and size of breeding colonies for these species in the region (Johnsgard 1956, Hanson 1963, Ackerman 1994).

Kitchin (1930) noted the first breeding record for Caspian terns in Washington State at Moses Lake in 1929; however, Gill and Mewaldt (1983) suggested that Caspian terns were established as a breeding species in inland Washington prior to 1929. The location of Caspian tern breeding colonies in this region shifted after the formation of Potholes Reservoir by the O'Sullivan Dam in the 1950s (Johnsgard 1956, Penland 1982) and the creation of Crescent Island from disposal of dredged materials in 1985 (Ackerman 1994). California and ring-billed gulls followed a similar pattern, shifting from breeding colonies at Moses Lake to Potholes Reservoir after its formation (Johnsgard 1956, Conover et al. 1979) and colonizing Crescent Island soon after the initial colonization by Caspian terns (Ackerman 1994). Both species of gulls nested on other islands in the mid-Columbia River created by dam impoundments during this time period (Broadbooks 1961, Hanson 1963, Thompson and Tabor 1981).

Double-crested cormorants were thought to breed in the Columbia Plateau region prior to 1932 and were common near the Tri-Cities in Washington up to 1953; however, nesting habitat was lost and numbers of nesting birds declined as a result of impoundment of the Columbia River behind McNary Dam beginning in 1954 (Hanson 1968). Double-crested cormorants were also known to nest on the Snake River upstream of Clarkston, Washington prior to dam impoundments (Weber and Larrison 1977, Smith et al. 1997).

The first documented breeding record for American white pelicans in the Columbia Plateau region was at Moses Lake in 1926 (Brown 1926). While white pelicans were observed in the region in the interim, their breeding status was unclear until 1994, when they were recorded nesting on Crescent Island (Ackerman 1994). American white pelicans shifted their breeding colony from Crescent Island to Badger Island, about 1 km up-river, in 1997.

The purposes of this study were fourfold: (1) estimate colony size for all known breeding colonies of Caspian terns, double-crested cormorants, American white pelicans, ring-billed gulls, and California gulls in the Columbia Plateau region during the study period (2004-2009); (2) assess productivity at these colonies, when feasible; (3) evaluate factors limiting the size and productivity of breeding colonies of these species in the Columbia Plateau region; and (4) assess the inter-colony movements of marked Caspian terns among colonies on the Columbia Plateau.

METHODS

Study Area

This study was conducted at Caspian tern, double-crested cormorant, American white pelican, California gull, and ring-billed gull breeding colonies in the Columbia Plateau region of Washington State during 2004-2009 (Table 1.1, Figure 1.1). Some data from the Caspian tern and double-crested cormorant colonies at East Sand Island, Oregon were included for comparison purposes. East Sand Island in the Columbia River estuary is home to the largest Caspian tern colony in the world, which is currently being managed to reduce the size of the colony and its impact on survival of juvenile salmonids from throughout the Columbia Basin (USFWS 2005, USFWS 2006). The double-crested cormorant colony at East Sand Island is the largest in western North America and, possibly, all of North America. The Columbia Plateau region in Washington State is delineated by the borders with Idaho and Oregon to the east and south, respectively, the Columbia River and mouth of the Okanogan River to the north, and the foothills of the Cascade Mountain Range to the west.

Colony Size and Productivity

Estimates of colony size for colonial waterbirds are generally obtained during the peak of the nesting season, when the greatest numbers of adults are aggregated at the nesting colonies (Bullock and Gomersal 1981, Gaston and Smith 1984), generally late in incubation. Aerial, boat, and road surveys were completed during this time period to identify active breeding colonies of the focal waterbird species. Although it is possible that small colonies (i.e., <12 breeding pairs) may have been missed during these surveys, we are confident that colonies of any consequence were identified.

Caspian terns– For the years 2004 and 2006-2009 the numbers of breeding pairs of Caspian terns nesting at Crescent Island were estimated by averaging six independent ground counts of all incubating terns on the colony near the end of the incubation period. Due to a temporary change in protocol in 2005, the number of Caspian tern breeding pairs nesting at Crescent Island in 2005 was estimated by averaging two independent colony counts that were corrected using ground counts of incubating and non-incubating terns on seven plots within the colony area. All counts were made from an observation blind situated near the edge of the tern colony. At other Caspian tern colonies in the Columbia Plateau region, colony size estimates were determined from either direct counts of attended nests in digital aerial photography or direct ground counts of attended nests (i.e., from an observation blind or a boat) around the peak of incubation[1]. When reported, productivity at a breeding colony was estimated by dividing the ground count of all fledglings present on the colony just prior to fledging by the number of breeding pairs.

Double-crested cormorants– The number of breeding pairs of double-crested cormorants nesting at Foundation Island was estimated using the peak count of attended nests based on weekly counts of the colony from late March through July during 2004-2009. Counts were conducted from an observation blind located in the water, approximately 25 m off the eastern shore of the island. Estimates of colony size should be considered minimums, however, as vegetation partially obscured some nests over the course of the breeding season. At other

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[1] 2005 Goose Island (Potholes Reservoir) colony size estimate is from Good et al. 2006.

cormorant colonies in the Columbia Plateau region, colony size estimates were determined from either direct counts of attended nests in digital aerial photography or direct ground counts of attended nests (i.e., from an observation blind or a boat) around the peak of incubation. Productivity at the Foundation Island cormorant colony was estimated from the number of chicks in monitored nests at 28 days post-hatching. Because of the distance of the blind from the colony and our vantage below the elevation of the nests, we assumed that chicks were approximately 10 days post-hatching when first observed.

American white pelicans– Badger Island was the site of the only known nesting colony of American white pelicans in the State of Washington during 2004-2009, and the species is listed as endangered by the State. Consequently, the island is closed to both the public and researchers in order to avoid human disturbance to nesting pelicans that might cause pelicans to abandon the colony. Aerial photography was taken of the Badger Island colony during the late incubation period in order to estimate colony size. Complete counts of the number of active pelican nests on Badger Island were not possible from the water because most nests were concealed by the thick, brushy vegetation on the island. Most, but probably not all, pelicans present on the island were visible in the aerial photography; however, we could not correct counts from aerial photography to estimate the number of breeding pairs because we were unable to obtain representative counts of incubating and non-incubating pelicans from the water. Counts of adult pelicans from aerial photography are, therefore, an index to the number of breeding pairs utilizing Badger Island, rather than a count of nesting pairs. During 2005-2007 one photo-count of adult pelicans on the colony was completed. During 2008-2009, we refined the photo-count process by using an in-house GIS workstation and conducted three independent counts of pelicans at the colony, reporting the mean. As it was only possible to obtain index counts of adults and juveniles at the Badger Island pelican colony, it was not possible to estimate productivity.

Gulls– In years when colony size estimates were reported, aerial photography was taken of the colony during the late incubation period. Most, but probably not all, gulls present on a colony were visible in the aerial photography; however, we could not correct counts from aerial photography to estimate the number of breeding pairs because representative counts of incubating and non-incubating gulls from the ground were not available. Counts of adult gulls from the aerial photography are an index to the number of breeding pairs utilizing the colony, rather than a count of nesting pairs. Three independent counts of gulls at each colony were conducted and the mean reported. Nesting success was not estimated but was confirmed by observing the presence of fledglings during ground and boat-based surveys.

Limiting Factors

In the Columbia Plateau region, quantitative data on limiting factors are restricted to kleptoparasitism and colony disturbance rates collected at the Caspian tern colony at Crescent Island. Our investigation of other factors that limit the size and productivity of tern colonies and other piscivorous waterbird colonies in the Columbia Plateau region (e.g., nest predation, availability of suitable nesting habitat, variable water levels, displacement by other colonial waterbirds, human disturbance, food availability, and disease) were based on qualitative observations, educated professional opinion, and other published reports.

Kleptoparasitism rates– Gulls that nest at the periphery of Caspian tern colonies in the Columbia Basin may have a negative effect on the survival of juvenile salmonids and the productivity of Caspian terns; some individuals kleptoparasitize (i.e., steal) juvenile salmonids, as well as other prey taxa, from terns as they return to the colony with fish in their bills to feed to mates and young. Gull kleptoparasitism rates (expressed as fractions) were calculated by dividing the number of fish delivered to the tern colony (i.e., bill loads) and subsequently stolen by gulls, by the total number of fish delivered to the tern colony with a known fate (i.e., usually consumed by a tern or stolen and consumed by a gull). This can be expressed as the following equation:

$$\text{kleptoparasitism rate} = \frac{\text{\# bill loads kleptoparasitized}}{\text{total \# bill loads with known fate}}$$

Data on bill load observations were only collected at the Crescent Island and East Sand Island Caspian tern colonies during the study period. Kleptoparasitism rates were calculated annually during all years of the study (2004-2009). For the years 2005-2007, kleptoparasitism rates were also calculated by size class of fish and by taxonomic group of fish at both tern colonies. Size classes were grouped as ≤ 10 cm, 11-14 cm, 15-18 cm, and ≥ 19 cm at both sites. Taxonomic groups were salmon, steelhead, centrarchids (bass and sunfish), and cyprinids (minnows and carp) at Crescent Island, and salmon, steelhead, anchovy, and clupeids (herring and sardines) at East Sand Island.

Disturbance rates– Daily disturbance rates were calculated for the Crescent Island Caspian tern colony for the years 2004-2009 and for the East Sand Island Caspian tern colony for the years 2004-2006 and 2009. Disturbance rates were calculated by dividing the number of disturbances observed in a day by the number of observation hours in that day. Our definition of disturbance follows Antolos et al. (2006) as "an episode where Caspian terns flushed from their nests, creating the potential for nest predation by opportunistic gulls". Disturbance rates were averaged across the season to get annual disturbance rates at both colonies. Disturbance rates were also averaged by stage of the breeding season at each colony site to investigate temporal trends and determine whether Caspian terns were more or less likely to be disturbed during certain stages of the breeding season. Stages of the breeding season were defined as pre-incubation (1 April – 15 April), incubation (16 April – 20 May), chick-rearing (21 May – 10 July), and post-chick rearing (11 July – 31 July).

Inter-colony Movement, Natal Philopatry, and Colony Site Fidelity

As part of this and other related studies beginning in 1997, adult and juvenile Caspian terns were color banded at several tern colonies throughout their range in the Pacific Coast region of North America in order to assess inter-colony movements and demography. As part of this study, breeding adult Caspian terns were banded at Crescent Island during 2003-2006 (n = 160) and at East Sand Island in 2001 and during 2004-2009 (n = 297). Caspian tern chicks were banded at Crescent Island during 2001-2009 (n = 1,092), at Goose Island (Potholes Reservoir) in 2006, 2007, and 2009 (n = 288), and at East Sand Island during 2001-2009 (n = 3,213). In this chapter we investigate the inter-colony movements, natal colony philopatry, and colony site fidelity of Caspian terns nesting at two colony sites located in the Columbia Plateau region (Crescent Island and Goose Island) and compare to similar data collected at the East Sand Island tern colony. See Suryan et al. (2004) for estimated survival rates of Caspian terns in the Pacific Coast population based on resightings of banded individuals.

Prior to 2005 for adults and prior to 2006 for chicks, Caspian terns were banded with a federal numbered metal leg band and a unique color combination of five plastic leg bands (the metal band and two plastic bands on one leg and three plastic bands on the other leg). Beginning in 2005 for adults and 2006 for chicks, terns were banded with a federal numbered metal leg band and two plastic, colored leg bands on one leg and a plastic leg band engraved with a unique alphanumeric code on the other.

Adult Caspian terns were captured using noose mats placed around active nests. Once captured, terns were immediately transferred to holding crates until they were banded and released. Tern chicks that were near fledging were captured on-colony by herding into holding pens and then transferred to holding crates until they were banded and released. Tern banding operations were conducted only during periods of moderate temperatures to reduce the risk of heat stress for captive terns.

Natal colony philopatry of Caspian terns banded as chicks at East Sand Island and Crescent Island was calculated as the number of birds returning to their natal colony for first reproduction as a proportion of all birds observed recruiting for first reproduction to a monitored colony. Colony site fidelity of Caspian terns banded as adults at East Sand Island and Crescent Island was estimated as the proportion of birds re-sighted solely at the colony where they had been banded (excluding the year they were banded) to all birds re-sighted in subsequent years, regardless of colony site.

Data included here are from color-banded terns re-sighted on breeding colonies by researchers as part of this study during the 2004-2010 breeding seasons. Data from 2010 were included as this was the only year in which a significant re-sighting effort was made at Goose Island in Potholes Reservoir. Philopatry and colony site fidelity calculations include data collected during 2002-2010. Equal re-sighting effort was not invested at all sites in all years.

RESULTS

Colony Size and Productivity

Caspian terns– The total number of Caspian terns nesting throughout the Columbia Plateau region ranged from 711 to 978 breeding pairs, with no overall population trend evident during 2004-2009 (Figure 1.2). The low of 711 breeding pairs occurred in 2007 and the high of 978 breeding pairs occurred in 2009.

The higher number of Caspian terns breeding in the Columbia Plateau region during 2009 was primarily due to the growth of the Goose Island colony in Potholes Reservoir, which increased by approximately 78% since we began estimating the size of this colony in 2004 (Figure 1.3). This is now the largest Caspian tern colony on the Columbia Plateau, eclipsing the colony at Crescent Island[2]. All Caspian tern nesting at Potholes Reservoir has occurred at Goose Island since 2005. In 2004, approximately 42 Caspian terns attempted to nest at Solstice Island; however, the colony failed due to flooding of the nesting area and no Caspian terns have nested at Solstice Island since 2004 (BRNW 2005; Table 1.2). Precise productivity estimates for

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[2] Fewer Caspian terns nested at the Goose Island colony in 2010 (416 breeding pairs) compared to 2009 (487 breeding pairs), but it remained the largest breeding colony in the Columbia Plateau region.

Caspian terns nesting at Potholes Reservoir are not available for most years, but observations indicate that the Goose Island tern colony successfully fledged young annually during 2004-2009[3].

During 2004-2009 the size of the Caspian tern colony on Crescent Island has trended downward (Figure 1.3), and ranged from a high of 530 breeding pairs in 2004 to a low of 349 breeding pairs in 2009. During this time period nesting success averaged 0.50 fledglings/breeding pair, with a high in productivity of 0.68 fledglings/breeding pair during 2007 and a low of 0.28 fledglings/breeding pair during 2008 (Figure 1.4).

Caspian tern nesting was first detected in the Blalock Islands in the mid-Columbia River (Figure 1.1) in 2005, when about 6 pairs of terns attempted to nest on Rock Island amidst a colony of ring-billed gulls and Forster's terns (*Sterna fosteri*). Caspian terns continued to initiate nesting at Rock Island during 2006-2009, with numbers of pairs attempting to nest ranging from a high of approximately 110 breeding pairs in 2006 to a low of approximately 43 breeding pairs in 2007 (Table 1.2). The Rock Island Caspian tern colony failed or nearly failed each year during 2006-2009; in 2006 due to mink predation, in 2007 due to avian predation, in 2008 due to unusually high water levels in John Day Pool during the incubation period, and in 2009 due to unknown causes[4].

We identified two additional Caspian tern colonies in the Columbia Plateau region (besides the colonies at Crescent Island, at Potholes Reservoir, and in the Blalock Islands), and both were small colonies off the Columbia and Snake rivers. One colony was at Twining Island on Banks Lake and the other was at Harper Island on Sprague Lake (Figure 1.1); both colonies were in the midst of much larger gull colonies. Tern colony size at Twining Island was estimated during 2005-2009, and ranged from a low of 13 breeding pairs (including seven pairs that nested on a nearby island) in 2005 to a high of 61 breeding pairs in 2009 (Table 1.2). Caspian terns at this colony were successful in fledging at least some young in all years of this study. Productivity estimates for this colony, however, are only available for 2008 and 2009, when an average of 0.33 fledglings/breeding pair were raised in both years. Colony size at Harper Island on Sprague Lake was estimated during 2005-2009 and ranged from a low of zero breeding pairs in 2007 to a high of 11 breeding pairs in 2008 (Table 1.2). We were unable to confirm nesting success at Harper Island in 2009; however, this colony failed to fledge any young in all other years of this study.

Double-crested cormorants– The total number of double-crested cormorants nesting throughout the Columbia Plateau region ranged from 1,196 to 1,554 breeding pairs. During 2005-2009, the overall population trend was stable or possibly declining (comparable estimates were not available from all colonies in 2004; Figure 1.5). The low of 1,196 breeding pairs occurred in 2009 and the high of 1,554 breeding pairs occurred in 2006.

The largest double-crested cormorant colony in the Columbia Plateau region during 2004-2009 was in trees within the North Potholes Reserve at Potholes Reservoir. During our study period, however, the numbers of cormorants nesting at this colony varied considerably. Between 300 and 500 breeding pairs nested at this site in 2004. Colony size peaked in 2006, when approximately 1,156 breeding pairs nested at the North Potholes colony, and then declined

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[3] The Goose Island Caspian tern colony failed in 2010 due to predation and disturbance by American mink (*Neovison vison*) and great horned owls (*Bubo virginianus*).

[4] No nesting attempts by Caspian terns were observed on Rock Island in 2010, but nesting Caspian terns were observed on another island in the Blalock Islands.

to about 809 breeding pairs by 2009 (Figure 1.6). Precise productivity estimates are not available for the North Potholes cormorant colony; however, this colony was successful in fledging young in all years of the study.

During 2004-2006 the Foundation Island cormorant colony on the mid-Columbia River (also in trees) gradually grew from approximately 300 breeding pairs to approximately 359 breeding pairs, before leveling off and then declining to around 309 breeding pairs in 2009 (Figure 1.6). The average productivity at the Foundation Island cormorant colony during 2005-2009 (no productivity estimate is available for 2004) was 1.99 fledglings/breeding pair, and ranged from a low of 1.37 fledglings/breeding pair in 2006 to a high of 2.3 fledglings/breeding pair in 2005 (Figure 1.7).

Double-crested cormorants nested at a small colony in trees near the mouth of the Okanogan River on the upper Columbia River during 2004-2009. The colony ranged in size from a low of 10 breeding pairs in 2007 to a high of 38 breeding pairs in 2005 (Table 1.2). Precise productivity estimates are not available for this site, but observations indicate that this colony fledged young in most years.

A double-crested cormorant colony first formed on the ground at Harper Island on Sprague Lake in 2008, when approximately 38 breeding pairs nested on the island. In 2009 an estimated 42 breeding pairs nested at this site[5]. Nesting success at this island, which is private property, is unknown.

Three ephemeral double-crested cormorant colonies were identified in the Columbia Plateau region during the study period. In 2006, five breeding pairs nested on rocks at Miller Rocks, an island group on the Columbia River just upstream of the Deschutes River, and two breeding pairs nested on a railroad trestle bridge amidst great blue heron nests on the lower Snake River near Lyons Ferry Hatchery (Figure 1.1). In 2007, eight breeding pairs nested in trees on the east bank of the Columbia River in the Wahluke Unit of Hanford Reach National Monument (Figure 1.1). The Hanford Reach colony failed due to wild fire, and nesting success at both the Miller Rocks and Lyons Ferry colonies is unknown. Double-crested cormorants have not nested at any of these three sites in subsequent years.

American white pelicans– Badger Island was the only known breeding colony of American white pelicans in the Columbia Plateau region during 2004-2009. This colony was also the only known breeding colony for the species in the entire State of Washington, where the species is listed as endangered, during the study period[6]. The count of 1,754 adult white pelicans recorded in 2009 was the highest total recorded during the study period, a 30% increase from 2008. Annual counts of adults in photographs have increased in all years since 2005, with the exception of 2007 (Figure 1.8; a comparable colony size estimate is not available for 2004)[7]. Precise productivity estimates are not available for the Badger Island pelican colony; however, observations indicate that pelicans at this site were successful in fledging young in all years of the study.

Gulls– Nesting gulls (California and ring-billed) were confirmed on 12 islands in the Columbia Plateau region during 2004-2009: Miller Rocks, Three Mile Canyon Island, Anvil and

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[5] The double-crested cormorant colony at Harper Island increased to 86 breeding pairs in 2010.

[6] In 2010, 31 pairs of American white pelicans attempted to nest on Crescent Island, about 1 km downstream of Badger Island; all these pelican nests subsequently failed.

[7] In 2010, 2,048 American white pelicans were counted from aerial photography of the Badger Island colony.

Rock islands in the Blalock Islands, Crescent Island, Island 18, Island 20, Goose and Solstice islands on Potholes Reservoir, Twining and Goose islands on Banks Lake, and Harper Island on Sprague Lake (Table 1.2, Figure 1.1). Nesting by gulls has not been recorded on Solstice Island since 2006. The gull colony on Island 18 was abandoned in 2008, due apparently to a combination of coyote (*Canis latrans*) predation and human disturbance, and was not re-colonized in 2009. Nesting was first documented on Anvil Island in the Blalock Islands during 2009. While breeding was documented at the remainder of these gull colonies in most years of the study, precise colony size estimates are only available for 2009 at most sites (Table 1.2).

In total, there were ca. 41,700 adult gulls counted at colonies on the mid-Columbia River from The Dalles Dam to Rock Island Dam in 2009, which is a 22% decrease in the number of gulls counted at colonies on the mid-Columbia River compared to 1998 (ca. 53,200), when the last comprehensive survey of gull colonies on the mid-Columbia River was conducted (Collis et al. 2002; Table 1.2). This decline was largely driven by the reduction in the number of gulls nesting on islands in the Tri-Cities area (Islands 18, 19, and 20 in the mid-Columbia River); ca. 35,000 gulls and ca. 19,300 gulls were counted at colonies on these islands in 1998 and 2009, respectively. Also, the gull colony at Three Mile Canyon Island declined from ca. 11,100 gulls in 1998 to ca. 6,200 gulls in 2009 (Collis et al. 2002; Table 1.2). Despite the overall decline in the number of gulls nesting at colonies on the mid-Columbia River from 1998 to 2009, three gull colonies increased in size during this time period. The number of gulls counted on-colony at the Miller Rocks colony increased from ca. 2,200 gulls in 1998 to ca. 6,000 gulls in 2009; the number of gulls counted at colonies in the Blalock Islands (Rock and Anvil islands) increased from 0 gulls in 1998 to ca. 1,600 gulls in 2009; and the number of gulls counted at Crescent Island increased from ca. 4,600 gulls in 1998 to ca. 8,600 gulls in 2009 (Collis et al. 2002; Table 1.2). No gull colonies were detected on the lower Snake River during the study period.

The total number of gulls nesting on the mid-Columbia River in 2009 was nearly equally divided between California gulls and ring-billed gulls. In addition, a total of ca. 25,900 gulls were counted on off-river colonies at Potholes Reservoir, Sprague Lake, and Banks Lake in 2009 (Table 1.2). Thus, our on-colony counts of all gull colonies in the Columbia Plateau region in 2009 totaled 67,600 gulls, and about 62% of the gulls nesting in the region were nesting on islands in the mid-Columbia River.

Precise productivity estimates are not available for these gull colonies; however, observations indicate that gulls nesting at colonies in the Columbia Plateau region were typically successful in fledging young during the study period.

Limiting Factors

Potential factors that could limit the size and productivity of breeding colonies of piscivorous waterbirds in the Columbia Plateau region include predation, human disturbance, food availability, water level fluctuations, availability of suitable nesting habitat, and disease. While the impact of most of these factors was not quantified at most colonies, incidental observations of the impact of these factors were recorded at most colonies. See the "Limiting Factors" section of the Discussion section for a more detailed evaluation of these factors.

Kleptoparasitism rates– Annual gull kleptoparasitism rates were higher at the Crescent Island Caspian tern colony (0.10 - 0.17) than at the East Sand Island tern colony (0.04 - 0.09) in all years of the study (2004-2009; Figure 1.9). Gull kleptoparasitism at the Crescent Island tern colony is by California gulls, while kleptoparasitism at the East Sand Island colony is by western/glaucous-winged gulls (*Larus occidentalis* X *L. glaucescens*).

When gull kleptoparasitism rates were separated by size class of stolen fish, rates were higher in the larger size classes in all years at the Crescent Island tern colony (Table 1.3). This trend was not as pronounced at the East Sand Island tern colony; however, in general kleptoparasitism rates followed the trend of higher rates with increasing fish size at East Sand Island as well (Table 1.3).

When gull kleptoparasitism rates were separated by taxonomic group, steelhead delivered to the tern colonies were kleptoparasitized at a higher rate than any other taxon of fish at both sites in all years. Kleptoparasitism rates on steelhead ranged from 0.26 to 0.31 at the Crescent Island tern colony and from 0.19 to 0.20 at the East Sand Island tern colony (Figures 1.10 and 1.11).

Disturbance rates– Disturbance rates ranged from 0 to 1.17 disturbances/hour (calculated on a daily basis) at Crescent Island and from 0 to 3.41 disturbances/hour at East Sand Island. Seasonal average disturbance rates were much lower at the Crescent Island tern colony (0.01 - 0.08 disturbances/hour) compared to the East Sand Island tern colony (0.10 - 0.38 disturbances/hour).

At East Sand Island there was a clear seasonal trend of decreasing average disturbance rates as the season progressed (i.e., average disturbance rates were highest during the pre-incubation period and lowest during the post-chick rearing period; Figure 1.12). This trend was not as distinct at Crescent Island, however. In general, average disturbance rates were highest at the Crescent Island tern colony during the pre-incubation period, decreased during the incubation and chick-rearing period, and increased again during the post-chick rearing period (Figure 1.13).

Inter-colony Movement, Natal Philopatry, and Colony Site Fidelity

Of the 257 banded Caspian terns re-sighted at the Crescent Island tern colony between 2004 and 2010, 241 (94%) were banded at Crescent Island (144 as adults and 97 as chicks), 12 (5%) were banded as chicks at East Sand Island, and the remaining four (1.6%) were banded as chicks at either Goose Island or Solstice Island in Potholes Reservoir. Of the 84 banded individuals re-sighted on Goose Island at Potholes Reservoir in 2010, 10 (12%) were banded as chicks at Goose Island, 62 (74%) were banded at Crescent Island (18 as adults and 44 as chicks), nine (11%) were banded as chicks at Solstice Island, two (2%) were banded as chicks at East Sand Island, and one was banded as an adult at Crump Lake in south-central Oregon.

Natal colony philopatry was 85% and 45% for Caspian terns banded as chicks at East Sand Island and Crescent Island, respectively. The proportion of terns banded as chicks that were recruited to East Sand Island from Crescent Island and confirmed breeding was 11%. Colony site fidelity for Caspian terns banded as adults was 95% for East Sand Island and 85% for Crescent Island. Of those terns banded as adults, approximately 5% from East Sand Island and approximately 15% from Crescent Island moved to other colonies. Individuals that were banded as adults at East Sand Island were not re-sighted at Crescent Island; however, some individuals that were banded as adults at Crescent Island were re-sighted at East Sand Island.

DISCUSSION

Colony Size and Productivity

The numbers of Caspian terns nesting in the Columbia Plateau region have remained approximately stable over the past decade, as have the numbers of Caspian terns nesting in the Columbia River estuary. Nevertheless, the number of Caspian terns nesting in the Columbia Plateau region is an order of magnitude less than that in the Columbia River estuary (BRNW 2010). While the Crescent Island tern colony declined during the study period, the Goose Island tern colony in Potholes Reservoir increased over the same time period. Since 2001, nesting success at the Crescent Island tern colony has also trended downward and productivity at this site has been low in most years compared to the East Sand Island tern colony (BRNW 2010; Figure 1.4).

In 2008, the U.S. Army Corps of Engineers began implementing the management actions outlined in the Final EIS (FEIS) and the Records of Decision (RODs) for Caspian tern management in the Columbia River estuary, a plan to redistribute a portion of the East Sand Island Caspian tern colony to alternative colony sites in interior Oregon and San Francisco Bay, California by 2015 (USFWS 2005, 2006). A substantial increase in the numbers of nesting Caspian terns along the mid-Columbia River as a result of management to reduce the numbers of Caspian terns nesting in the estuary is unlikely due to the paucity of suitable nesting habitat for terns in the Columbia Plateau region (Antolos 2004). Data from re-sighting of banded terns support this conclusion, as natal colony philopatry and colony site fidelity are low at Crescent Island compared to East Sand Island, and there was little apparent movement of terns from East Sand Island to Crescent Island during this study. Caspian terns banded as chicks at Crescent Island, however, have been confirmed breeding at East Sand Island in subsequent years. Data from Goose Island are limited, however, there appears to be limited movement from the Caspian tern colony at East Sand Island to the Caspian tern colony at Goose Island. There is apparent connectivity between the Caspian tern colonies at Crescent Island and Goose Island in Potholes Reservoir, as the majority of banded individuals re-sighted at Goose Island in 2010 had been previously banded at Crescent Island.

The size of the double-crested cormorant breeding population in the Columbia Plateau region has stabilized or even declined in the last few years despite unused suitable nesting habitat at both the Foundation Island and North Potholes colonies, suggesting that some other factor or factors have limited the size of the population. The number of double-crested cormorants nesting in this region is an order of magnitude less than the number of double-crested cormorants nesting at East Sand Island in the Columbia River estuary (BRNW 2010). Productivity at the Foundation Island cormorant colony was lower than at the East Sand Island cormorant colony in all years of the study except 2005, a year of exceptionally low nesting success at the East Sand Island cormorant colony (Figure 1.7).

The American white pelican colony at Badger Island appears to be steadily increasing and successfully producing young. Pelicans nesting at Badger Island have not utilized all available nesting habitat on the island, so it is feasible that this colony could continue to grow.

The trajectories in size of the various gull colonies along the mid-Columbia River were variable, with some gull colonies increasing dramatically over the last decade (i.e., Miller Rocks and Crescent Island) and others declining just as dramatically (i.e., Island 18, Three Mile Canyon Island). Overall, the breeding populations of ring-billed gulls and California gulls in the

Columbia Plateau region appear to have declined somewhat in the last decade, even though most colonies appear to be successfully fledging young.

Limiting Factors

Kleptoparasitism– Annual gull kleptoparasitism rates on Caspian terns at the Crescent Island colony were much higher than those at the East Sand Island tern colony. Additionally, gulls (primarily California gulls) that nested in close proximity to the tern colony at Crescent Island kleptoparasitized larger fish and steelhead at higher rates than they kleptoparasitized smaller fish and other taxa, including salmon. California gulls at Crescent Island are likely having a significant impact on the foraging efficiency and energetic demands of Caspian terns nesting at this site, as these terns are delivering fish to the colony to feed mates and chicks. When the fish that a tern delivers is stolen, that individual (or its mate) must compensate by spending more time and energy foraging.

Disturbance and predation– Colony disturbance, followed by intense gull predation pressure on unattended tern eggs and chicks, has been shown to be an important factor limiting Caspian tern nesting success in the Columbia River estuary (Collis et al. 1999). Based on observed disturbance rates, disturbance at the Crescent Island tern colony appears to be relatively low compared to the East Sand Island tern colony. Within a breeding season, average disturbance rates at the Crescent Island tern colony were typically higher during the pre-incubation period than during the incubation, chick-rearing, or post-chick rearing periods. This could be due to the larger numbers of bald eagles (*Haliaeetus leucocephalus*) in the area during the pre-incubation period, when large numbers of the eagles' primary prey (waterfowl) are present. Additionally, terns become more committed to their nests (i.e., less likely to flush from the nest) during the incubation and chick-rearing periods. Thus far, daytime disturbance and associated nest predation by gulls does not appear to be a primary limiting factor for the Crescent Island Caspian tern colony. Caspian terns are, however, highly sensitive to nocturnal disturbance and disturbance by mammalian predators. The Caspian tern colony at Three Mile Canyon Island was abandoned in 2000 due to mink predation (Antolos et al. 2004), the Rock Island tern colony failed in 2006 due to mink predation, and the Goose Island tern colony in Potholes Reservoir failed in 2010 due to predation and disturbance by great horned owls and mink (BRNW unpubl. data).

Disturbance and predation appear to be the primary limiting factors for the gull populations nesting in the Columbia Plateau region. A combination of disturbance by humans and predation by a coyote likely caused the large gull colony on Island 18 to abandon in 2008; the island was not re-colonized in 2009 or 2010. The large Three Mile Canyon Island gull colony has declined over the last decade, coinciding with the failure of the Caspian tern colony on the island due to predation and disturbance by mink in the early 2000s.

American white pelicans are highly susceptible to disturbance at the breeding colony, especially during the early stages of the nesting season (Knopf 2004). The potential effects of disturbance could be particularly dire for the breeding population in the Columbia Plateau region, as American white pelicans are currently only known to nest at one site in Washington State, at Badger Island.

Disease– American white pelicans and double-crested cormorants are susceptible to several infectious diseases. Large die-offs of American white pelicans from avian botulism have occurred at the Salton Sea in southern California (Rocke et al. 2004), avian cholera has caused large die-offs of hatch-year cormorants in the Canadian provinces of Alberta and Saskatchewan (Wildlife Health Centre Newsletter 1998 and 2005), and the West Nile virus has been diagnosed in an American white pelican at the North Potholes Reserve in the fall of 2010 (WDOH 2010), as well as an American white pelican from the Summer Lake Wildlife Area in south-central Oregon (ODFW 2007). Newcastle disease is the most common disease threat to double-crested cormorants and can cause high mortality rates among juvenile cormorants (Kuiken 1999). Newcastle disease was first diagnosed in juvenile double-crested cormorants from colonies in the Columbia River estuary and the Great Salt Lake in Utah during 1997 (Wildlife Health Centre Newsletter 1997, Kuiken 1999); cormorant fledglings from East Sand Island have since been diagnosed with the disease in multiple years (2003, 2005, 2007, 2009; BRNW unpubl. data). While these diseases have not been diagnosed in American white pelicans breeding at Badger Island or in double-crested cormorants breeding at Foundation Island or North Potholes Reserve, they are a potential threat to individuals nesting in the Columbia Plateau region.

Nesting habitat and forage fish availability– Antolos et al. (2004) suggested that a combination of availability of suitable nesting habitat and forage fish availability is the primary limiting factor for Caspian terns nesting in the Columbia Plateau region. Some Caspian terns in the region continue to nest on sub-optimal substrate, such as basalt rock with little or no sand at Twining Island on Banks Lake and Harper Island on Sprague Lake. Fluctuating water levels have caused breeding failure at colonies on Solstice and Rock islands when nesting areas were flooded. Caspian terns nesting at Goose Island in Potholes Reservoir continue to commute ≥ 35 km to forage on the mid-Columbia River (BRNW 2010), routinely exceeding the average foraging distance for Caspian terns at other colonies; average foraging distance for Caspian terns nesting on East Sand Island in the Columbia River estuary was ≤ 20 km (Anderson 2003). Lower productivity of Caspian terns nesting at Crescent Island relative to Caspian terns nesting at East Sand Island also suggests that food availability is a limiting factor for this colony. Food availability for Caspian terns nesting in the Columbia Plateau region may be particularly limited during the chick-rearing period because the primary prey type for colonies in the mid-Columbia River is salmonid smolts (Chapter 2), most of which have migrated past these colonies by this time (Chapter 4). Higher productivity at the Crescent Island tern colony in years when water conditions and salmonid smolt run timing increased the vulnerability of steelhead smolts during the chick-rearing period (e.g., 2004) also supports the hypothesis of limited food availability.

Double-crested cormorants and, in theory, American white pelicans breeding in the Columbia Plateau region may also face a lack of stable and predictable food resources due to severe drought in inland regions in some years (Carter et al. 1995). Based on the paucity of American white pelican breeding colonies, suitable undisturbed nesting habitat may also be a limiting factor for this species in the region. It is not evident that another suitable colony site for this species is currently available in the Columbia Plateau region.

Gulls in this region are plastic in their diets (Collis et al. 2002) and are less likely to be limited by food availability. However, encroaching vegetation in areas of Three Mile Canyon Island that were formerly occupied by nesting gulls may be a contributing factor causing that gull colony to decline. The available evidence suggests that the regional decline in numbers of breeding gulls is due to apparent declines in suitable sites for gull colonies, which generally support thousands of breeding pairs.

Inter-colony Movement, Natal Philopatry, and Colony Site Fidelity

Caspian terns– Our results indicate a high degree of connectivity between the Caspian tern colonies at East Sand Island in the Columbia River estuary and at Crescent Island on the mid-Columbia River and between the Caspian tern colonies at Crescent Island and at Goose Island in Potholes Reservoir. Movement between Crescent Island and East Sand Island, however, appears to be largely unidirectional. For instance, terns banded as adults at Crescent Island were re-sighted at East Sand Island. No individuals banded as adults at East Sand Island were re-sighted at Crescent Island during this study. However, previous research (Antolos et al. 2004) and more recent results (BRNW 2011) have documented small numbers of adult terns banded at colonies in the Columbia River estuary relocating to Columbia Plateau colonies to breed (< 5 individuals). This is not surprising, as colony site fidelity is higher at the East Sand Island tern colony (95%) compared to the Crescent Island tern colony (85%). Natal colony philopatry is also higher at the East Sand Island tern colony (85%) compared to the Crescent Island tern colony (45%) and, of the birds banded at Crescent Island and confirmed as first time breeders at any of the monitored sites, 11% were re-sighted at East Sand Island. It is more difficult to assess the pattern of movement between Crescent Island and Goose Island, as we have only one comprehensive year of re-sighting data at Goose Island (2010), no adult terns were banded at Goose Island during the study period, and 2006 was the first year we banded tern chicks at Goose Island. The majority of banded terns re-sighted at Goose Island in 2010, however, were banded at Crescent Island, either as adults or chicks. Lower colony site fidelity and natal philopatry at Crescent Island, plus greater apparent emigration from Crescent Island compared to immigration to the colony by adult and juvenile terns, are consistent with the downward trend in size of this colony during the study period.

These results suggest that the Crescent Island Caspian tern colony is declining due to competition with and kleptoparasitism from California gulls nesting on Crescent Island, and the tern colony at Goose Island is increasing as a consequence of immigration from Crescent Island. The size and number of Caspian tern colonies in the Columbia Plateau region, as well as the overall number of Caspian terns nesting in the region, appears to be limited by the availability of suitable nesting habitat on predator-free islands and the availability of forage fish within commuting distance of those islands.

Double-crested cormorants– Results from this study on the size and distribution of double-crested cormorant colonies in the Columbia Plateau region, combined with results from other studies of the genetic structure of the Western North America Population of double-crested cormorants (Mercer 2008) and the post-breeding dispersal of cormorants breeding at the large East Sand Island colony (Courtot et al., in prep.) indicate that, compared to Caspian terns, there is limited demographic connectivity between cormorant colonies in the Columbia Plateau region and those along the coast.

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LITERATURE CITED

Ackerman, S. M. 1994. American white pelicans nest successfully at Crescent Island, Washington. Washington Birds 3:44-49.

Anderson, S. K. 2003. Foraging ecology, colony attendance, and chick provisioning of Caspian terns *(Sterna caspia)* in the Columbia River estuary. M.S. Thesis, Oregon State University, Corvallis. Oregon.

Antolos, M., D. D. Roby, and K. Collis. 2004. Breeding ecology of Caspian terns at colonies on the Colubmia Plateau. Northwest Science 78:303-312.

Antolos, M., D. D. Roby, D. E. Lyons, K. Collis, A. F. Evans, M. Hawbecker, and B. A. Ryan. 2005. Caspian tern predation on juvenile salmonids in the Mid-Columbia River. Transactions of the American Fisheries Society 134:466-480.

Antolos, M., D. D. Roby, D. E. Lyons, S. K. Anderson, and K. Collis. 2006. Effects of nest density, location, and timing on breeding success of Caspian Terns. Waterbirds 29:465-472.

Bird Research Northwest (BRNW). 2005. Caspian tern research on the lower Columbia River: 2004 Final Season Summary. Real Time Research, Bend, OR and Oregon State University, Corvallis, OR. Available: www.birdresearchnw.org.

Bird Research Northwest (BRNW). 2010. Caspian tern research on the lower Columbia River: 2009 Final Season Summary. Real Time Research, Bend, OR and Oregon State University, Corvallis, OR. Available: www.birdresearchnw.org.

Bird Research Northwest (BRNW). 2011. 2011 Columbia Plateau weekly updates. Real Time Research, Bend, OR and Oregon State University, Corvallis, OR. Available: www.birdresearchnw.org.

Broadbooks, H. E. 1961. Ring-billed gulls nesting on Columbia River islands. Murrelet 42:7-8.

Brown, D. E. 1926. Birds observed at Moses Lake, Grant County, Washington. Murrelet 7:48-51.

Bullock, I. D., and C. H. Gomersal. 1981. The breeding populations of terns in Orkney and Shetland in 1980. Bird Study 28:187-200.

Carter, H. R., A. L. Sowls, M. S. Rodway, U. W. Wilson, R. W. Lowe, G. J. McChesney, F. Gress, and D. W. Anderson. 1995. Population size, trends, and conservation problems of the Double-crested Cormorant on the Pacific coast of North America. Colonial Waterbirds 18:189-215.

Collis, K., S. L. Adamany, D. D. Roby, D. P. Craig, and D. E. Lyons. 1999. Avian predation on juvenile salmonids in the lower Columbia River. 1998 Annual Report of the Oregon Cooperative Fish and Wildlife Unit at Oregon State University to Bonneville Power Administration and U.S. Army Corps of Engineers, Portland, Oregon, USA.

Collis, K., D. D. Roby, D. P. Craig, S. Adamany, J. Y. Adkins, and D. E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: Implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Conover, M. R., B. C. Thompson, R. E. Fitzner, and D. E. Miller. 1979. Increasing populations of ring-billed and California gulls in Washington State. Western Birds 10:31-36.

Decker, F. R. and J. H. Bowles. 1932. Two new breeding colonies for the State of Washington. Murrelet 13:53.

Gaston, A. J., and G. E. J. Smith. 1984. The interpretation of aerial surveys for seabirds: Some effects of behaviour. Canadian Wildlife Service Occasional Paper 53:1-20.

Good, T. P., R. S. Waples, and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commerce, NOAA Technical Memo. NMFS-NWFSC-66. 598 p.

Good, T. P., D. L. Harstad, C. A. Hall, B. A. Ryan, M. Carper, and C. J. Maranto. 2006. Predation on juvenile salmonids by Caspian terns nesting in Potholes Reservoir. Appendix in: Research, monitoring, and evaluation of avian predation on salmonid smolts in the Lower and Mid-Columbia River: Draft 2005 season summary (Roby, D. D., Collis, K., Adkins, J. Y., Couch, C., Courtot, B., Lord, R., Lyons, D. E., Suzuki, Y., Evans, A., Hawbecker, M. (2006). Available at: www.birdresearchnw.org.

Hanson, W. C. 1963. Census of ring-billed and California gull colonies in eastern Washington. Condor 65:123-164.

Hanson, W. C. 1968. Recent history of double-crested cormorant colonies in southeastern Washington. Murrelet 49:25-26.

Johnsgard, P. A. 1956. Effects of water fluctuation and vegetation change on bird populations, particularly waterfowl. Ecology 37:689-701.

Jones, S. T., G. M. Starke, and R. J. Stansell. 1996. Predation by birds and effectiveness of predation control measures at Bonneville, The Dalles, and John Day dams in 1995. U.S. Army Corps of Engineers, Portland District, Operations Division, CENPP-CO-SRF. 10 pp.

Kitchen, E. A. 1930. Nesting observations at Moses Lake in May. Murrelet 11:54-59.

Knopf, F. L. 2004. American White Pelican (*Pelecanus erythrorhynchos*). The Birds of North America Online. (A. Poole, Ed.) Ithaca: Cornell Laboratory of Ornithology; Retrieved from The Birds of North American Online database: http://bna.birds.cornell.edu/BNA/account/American_White_Pelican/.

Kuiken, T. 1999. Review of Newcastle disease in cormorants. Waterbirds 22:333-347.

Mercer, D. M. 2008. Phylogeography and population genetic structure of double-crested cormorants (*Phalacrocorax auritus*). Unpublished M.S. thesis, Oregon State University, Corvallis, Oregon.

Oregon Department of Fish and Wildlife (ODFW). 2007. West Nile Virus detected in pelican at Summer Lake Wildlife Area. ODFW News Release, 28 August 2007.

Roby, D. D., K. Collis, D. E. Lyons, D. P. Craig, J. Y. Adkins, A. M. Myers, R. M. Suryan. 2002. Effects of colony relocation on diet and productivity of Caspian terns. Journal of Wildlife Management 66:662-673.

Roby, D. D., K. Collis, and D. E. Lyons. 2003. Conservation and management for fish-eating birds and endangered salmon. Proceedings of the Third International Partners In Flight Conference, Asilomar, California.

Rocke, T. E., P. Nol, C. Pelizza, and K. K. Sturm. 2004. Type C botulism in pelicans and other fish-eating birds at the Salton Sea. Studies in Avian Biology 27:136-140.

Ruggerone, G. T. 1986. Consumption of migrating juvenile salmonids by gulls foraging below a Columbia River dam. Transactions of the American Fisheries Society 115:736-742.

Schaeffer, L. 1991. Predation study: Salmon hatchery smolts and survival. Oregon Department of Fish and Wildlife, Technical Report. 33 pp.

Smith, M. R., P. W. Mattocks, Jr., and K. M. Cassidy. 1997. Breeding birds of Washington State. Volume 4 *in* Washington State Gap Analysis – Final Report (K. M. Cassidy, C. E. Grue, M. R. Smith, and K. M. Dvornich, eds.). Seattle Audubon Society Publications in Zoology No. 1, Seattle, WA. 538 pp.

Speich, S. M., and T. R. Wahl. 1989. Catalog of Washington seabird colonies. U.S. Fish and Wildlife Service Biological Report 88(6). 510 pp.

Suryan, R. M., D. P. Craig, D. D. Roby, N. D. Chelgren, K. Collis, W. D. Shuford, and D. E. Lyons. 2004. Redistribution and growth of the Caspian tern population in the Pacific coast region of North America, 1981-2000. Condor 106:777-790.

Thompson, B. C., and J. E. Tabor. 1981. Nesting populations and breeding chronologies of gulls, terns, and herons on the Upper Columbia River, Oregon and Washington. Northwest Science 55:209-218.

USFWS (U.S. Fish and Wildlife Service). 2005. Caspian tern management to reduce predation of juvenile salmonids in the Columbia River estuary: Final Environmental Impact Statement, January 2005. Migratory Birds and Habitat Program, Portland OR.

USFWS (U.S. Fish and Wildlife Service). 2006. Caspian tern management to reduce predation of juvenile salmonids in the Columbia River estuary: Record of Decision, November 2006. Migratory Birds and Habitat Programs, Portland, Oregon.

Washington Department of Health (WDOH). 2010. West Nile virus: Grant County mallard and white pelican test positive. News Release (10-146), September 21, 2010. Available: http://www.doh.wa.gov/Publicat/2010_news/10-146.htm.

Weber, J. W., and E. J. Larrison. 1977. Double-crested Cormorant. Page 11 in Birds of Southeastern Washington, A distributional checklist of the birds of southeastern Washington with comments on the avifaunas of eastern Washington south and east of the Columbia and Spokane rivers and of adjacent Idaho. University Press of Idaho, Moscow, ID.

Wiese, F. K., J. K. Parrish, C. W. Thompson, and C. Maranto. 2008. Ecosystem-based management of predator- prey relationships: Piscivorous birds and salmonids. Ecological Applications 18:681-700.

Wildlife Health Centre Newsletter. 1997. Newcastle disease in cormorants, California, USA – May 1997. Vol. 4 (3): 2. Summer. Available: www.ccwhc.ca/newsletters/newsletter4-3.htm#USA.

Wildlife Health Centre Newsletter. 1998. Avian cholera in double-crested cormorants (*Phalacrocorax auritus*). Vol. 5 (3). Winter. Available: www.ccwhc.ca/newsletters/newsletter5-3.htm#mortality.

Wildlife Health Centre Newsletter. 2005. Avian cholera in double-crested cormorants in Saskatchewan. Vol. 11 (1). Fall. Available: www.ccwhc.ca/newsletters/CCWHC_Newsletter_Fall_2005.pdf.

Table 1.1. Characteristics of piscivorous waterbird colonies in the Columbia Plateau region. AWPE = American white pelican, BCNH = black-crowned night-heron, CATE = Caspian tern, CAGU = California gull, DCCO = double-crested cormorant, FOTE = Forster's tern, GBHE = great blue heron, GREG = great egret, and RBGU = ring-billed gull. Colony name corresponds with that used in text. Latitude and longitude in decimal degrees. Primary Nesting Species = species that nest at the location in relatively large numbers and/or in most years of the study; Primary Nesting Type = nesting substrate used by primary nesting species; Secondary Nesting Species = species that nest at the location in relatively small numbers and/or in few years of the study; Secondary Nesting Type = nesting substrate used by secondary nesting species.

Colony Name	Latitude	Longitude	Primary Nesting Species	Primary Nesting Type	Secondary Nesting Species	Secondary Nesting Type
Miller Rocks	45.657	-120.872	CAGU, RBGU	Ground	DCCO, CATE	Ground
Three Mile Canyon Island	45.817	-119.963	CAGU, RBGU	Ground	CATE	Ground
Rock Island (Blalock Islands)	45.910	-119.629	CATE, RBGU	Ground	FOTE	Ground
Anvil Island (Blalock Islands)	45.914	-119.619	RBGU	Ground		
Crescent Island	46.094	-118.938	CATE, CAGU	Ground	RBGU, GBHE, BCNH, GREG	Tree
Badger Island	46.110	-118.938	AWPE	Ground		
Foundation Island	46.159	-118.991	DCCO	Tree	GBHE, BCNH	Tree
Island 18	46.361	-119.263	RBGU, CAGU	Ground	GREG, GBHE	Tree
Island 20	46.313	-119.254	RBGU, CAGU	Ground		
Goose Island (Potholes Reservoir)	47.023	-119.290	CATE, RBGU, CAGU	Ground	FOTE	Ground
Solstice Island (Potholes Reservoir)	47.023	-119.353	CATE, RBGU, CAGU	Ground		
North Potholes	47.041	-119.403	DCCO	Tree	GREG, GBHE	Tree
Twining Island (Banks Lake)	47.624	-119.304	CATE, RBGU, CAGU	Ground		
Goose Island (Banks Lake)	47.647	-119.291	RBGU, CAGU	Ground	CATE	Ground
Harper Island (Sprague Lake)	47.241	-118.084	DCCO, RBGU, CAGU	Ground	CATE	Ground
Mouth of Okanogan River	48.093	-119.710	DCCO	Tree		
Lyons Ferry Railroad Trestle	46.589	-118.224	GBHE	Artificial structure	DCCO	Artificial structure
Hanford Reach	46.655	-119.417	GBHE, GREG	Tree	DCCO	Tree

Table 1.2. Estimates of numbers of Caspian terns, double-crested cormorants, American white pelicans, California gulls, and ring-billed gulls at breeding colonies in the Columbia Plateau region and at East Sand Island, Oregon, during 2004-2009. Estimates of terns and cormorants are number of breeding pairs; estimates of pelicans and gulls are number of adults on-colony.

General Area	Specific Location	Species	2004	2005	2006	2007	2008	2009
Lower Columbia River	East Sand Island	Caspian tern	9502	8822	8929	9623	10668	9854
		Double-crested cormorant	12480	12287	13738	13771	10950	12087
Mid-Columbia River	Miller Rocks	California and ring-billed gulls	B	B	B	3509	4443	6016
	Three Mile Canyon Island	California and ring-billed gulls	B	B	B	B	B	6161
	Rock Island (Blalock Is.)	Caspian tern	0	6	110	43	104	79
		Ring-billed gulls	–	B	B	B	B	940
	Anvil Island (Blalock Is.)	Ring-billed gulls	–	–	–	–	–	691
	Crescent Island	Caspian tern	530	476	448	355	388	349
		California and ring-billed gulls	B	B	B	5601	8567	8575
	Badger Island	American white pelican	B	1057	1310	913	1349	1754
	Foundation Island	Double-crested cormorant	300	315	359	334	357	309
	Island 18	California and ring-billed gulls	B	B	B	B	0	0
	Island 20	California and ring-billed gulls	B	B	B	B	20999	19356
Potholes Reservoir	Goose Island	Caspian tern	87	325 [a]	273	282	293	487
		California and ring-billed gulls	B	B	B	B	B	13022
	Solstice Island	Caspian tern	42	0	0	0	0	0
		California and ring-billed gulls	B	B	B	–	–	–
	North Potholes	Double-crested cormorant	B	865	1156	1015	1000 [b]	809
Banks Lake	Twining & Goose Islands	Caspian tern	–	B	23	31	27	61
		California and ring-billed gulls	–	B	B	B	B	6602
Sprague Lake	Harper Island	Caspian tern	–	7	7	0	11	4
		Double-crested cormorant	–	0	0	0	38	42
		California and ring-billed gulls	–	B	B	B	B	6302
Okanogan River	Mouth	Double-crested cormorant	25	38	32	10	33	36

[a] Good et al. (2006)

[b] Approximate estimate.

Table 1.3. Kleptoparasitism rates (expressed as fractions of fish delivered) by gulls on Caspian terns nesting at Crescent Island and East Sand Island during 2005-2007, broken down into categories of fish length.

	2005		2006		2007	
Length of Fish (cm)	Crescent	East Sand	Crescent	East Sand	Crescent	East Sand
≤ 10	0.01	0.02	0.02	0.01	0.01	0.02
11-14	0.10	0.05	0.12	0.05	0.07	0.02
15-18	0.21	0.07	0.18	0.15	0.18	0.05
≥ 19	0.30	0.19	0.33	0.13	0.26	0.12

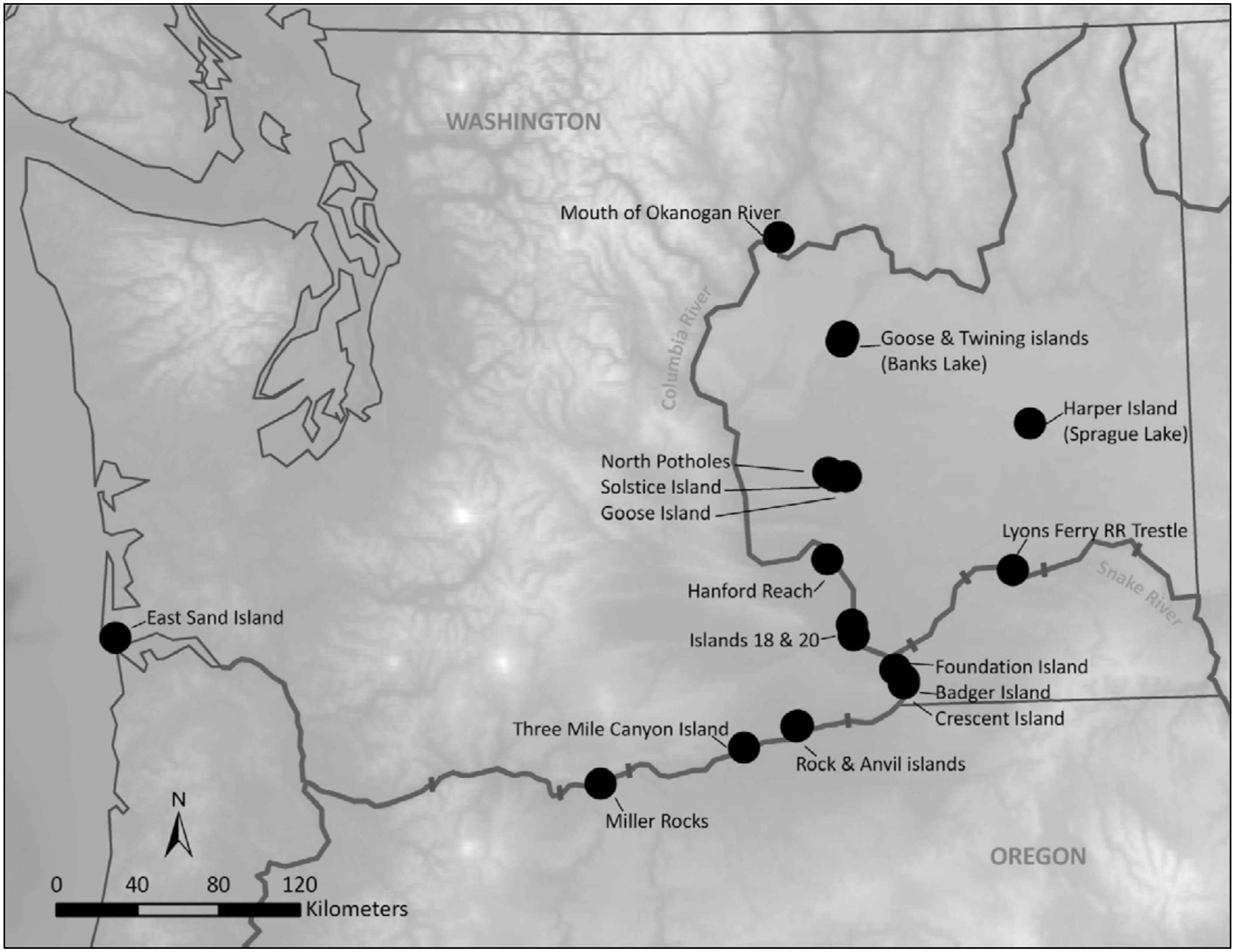

Figure 1.1. Map of the study area including the piscivorous waterbird colonies in the Columbia Plateau region and in the Columbia River estuary that are mentioned in the text. Eight Federal Columbia River Power System dams are represented by hatch marks along the Columbia and Snake rivers for reference.

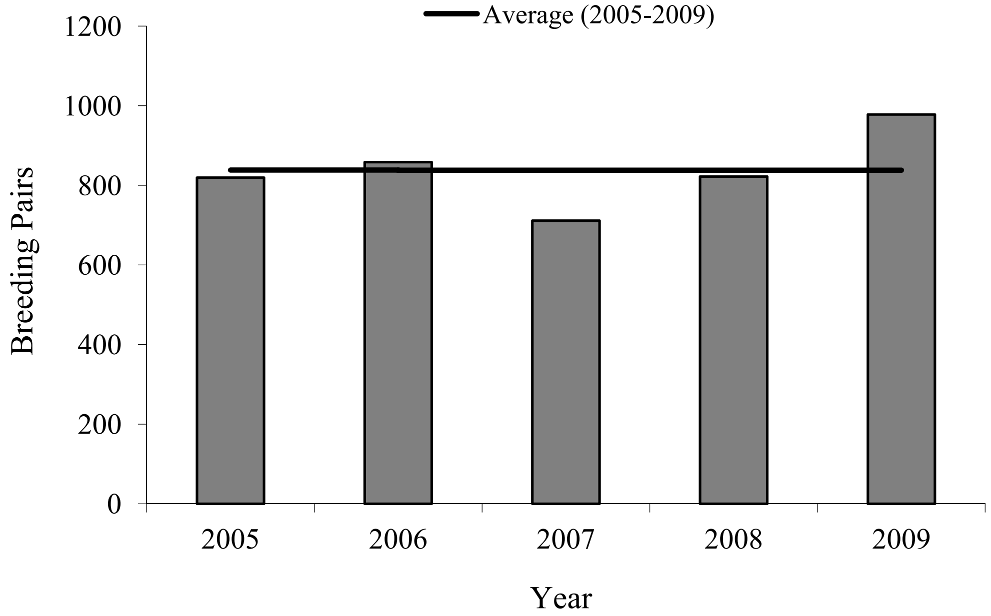

Figure 1.2. Number of Caspian terns nesting in the Columbia Plateau region, 2005-2009.

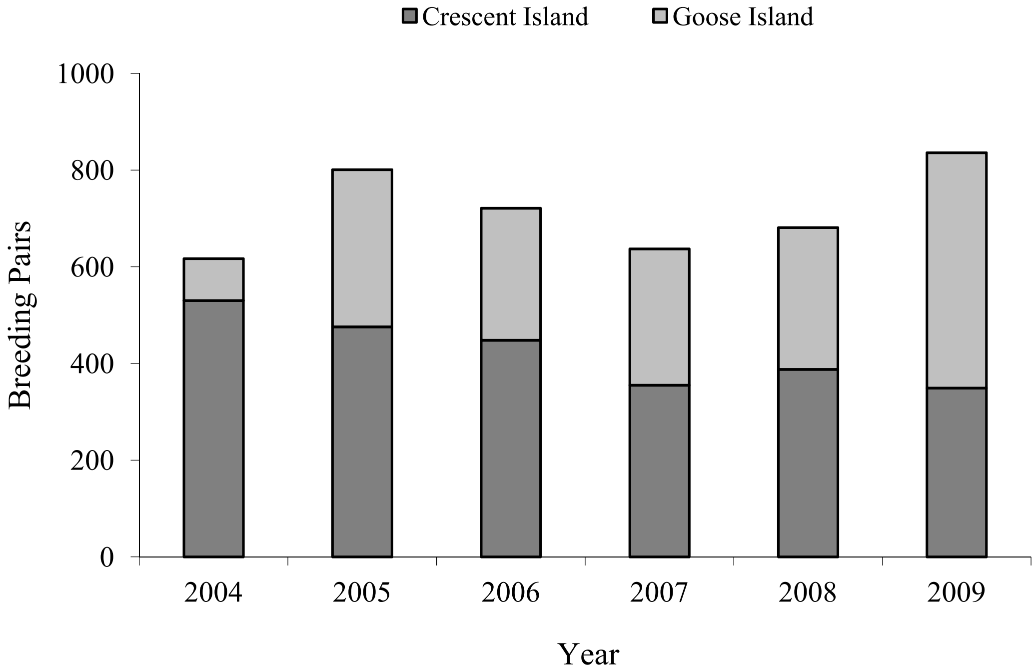

Figure 1.3. Numbers of Caspian terns nesting at the Crescent Island and Goose Island colonies in the Columbia Plateau region, 2004-2009.

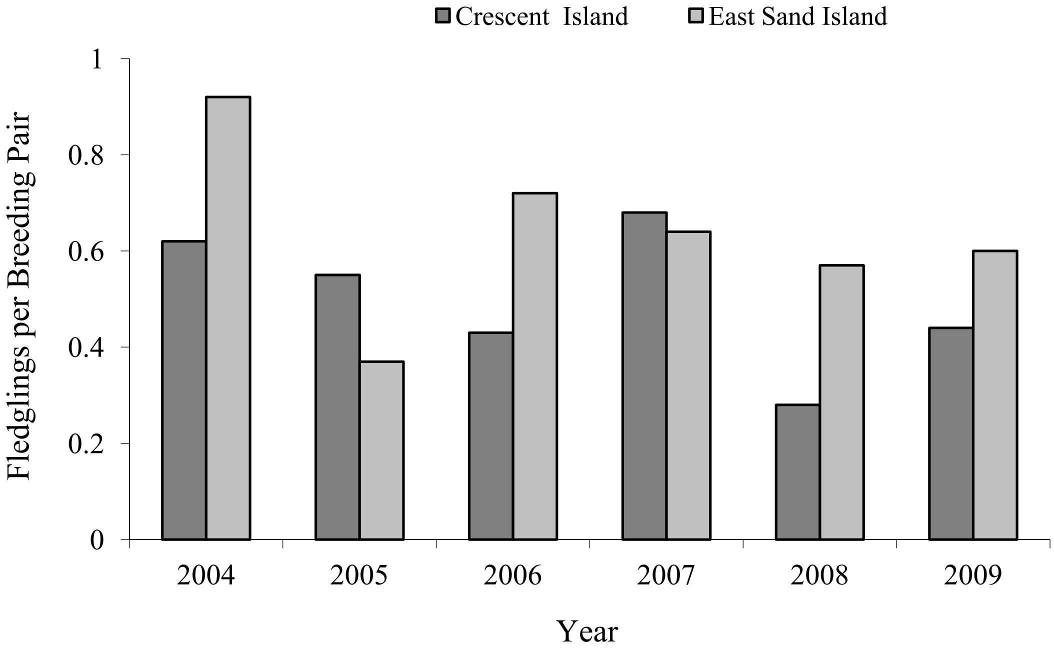

Figure 1.4. Productivity of the Caspian tern colony at Crescent Island in the Columbia Plateau region compared with the colony at East Sand Island in the Columbia River estuary during 2004-2009.

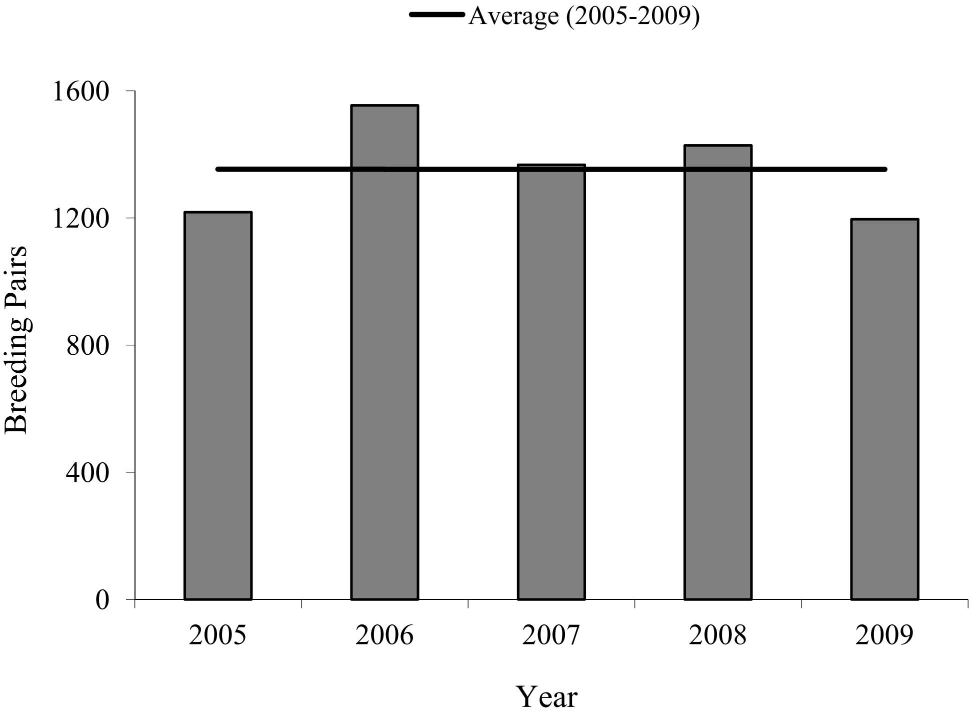

Figure 1.5. Number of double-crested cormorants nesting in the Columbia Plateau region, 2005-2009.

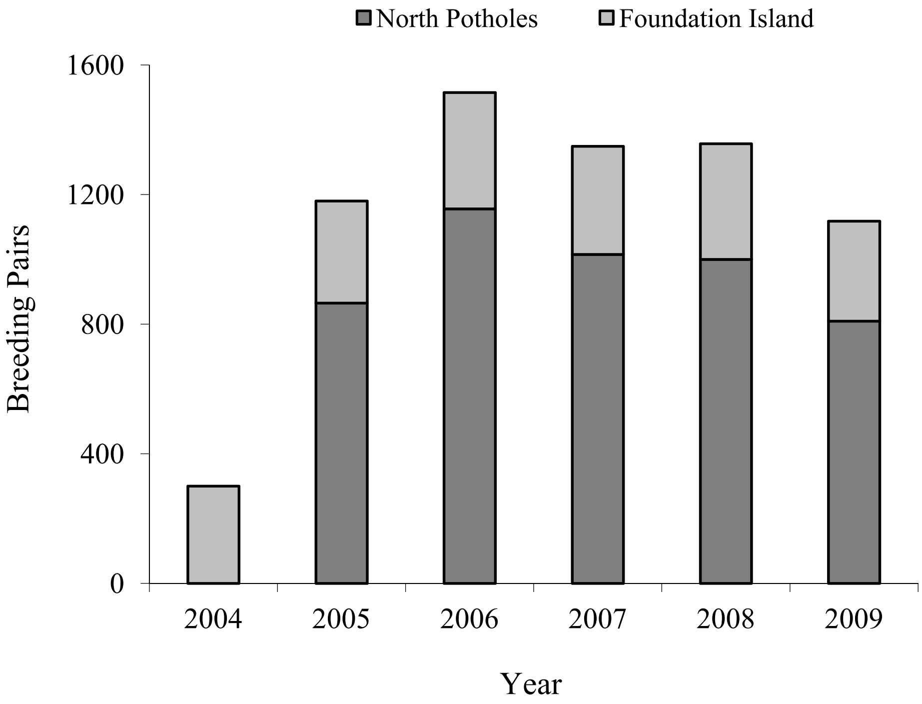

Figure 1.6. Numbers of double-crested cormorants nesting at the North Potholes and Foundation Island colonies in the Columbia Plateau region during 2004-2009. No comparable estimate of colony size was available for North Potholes in 2004.

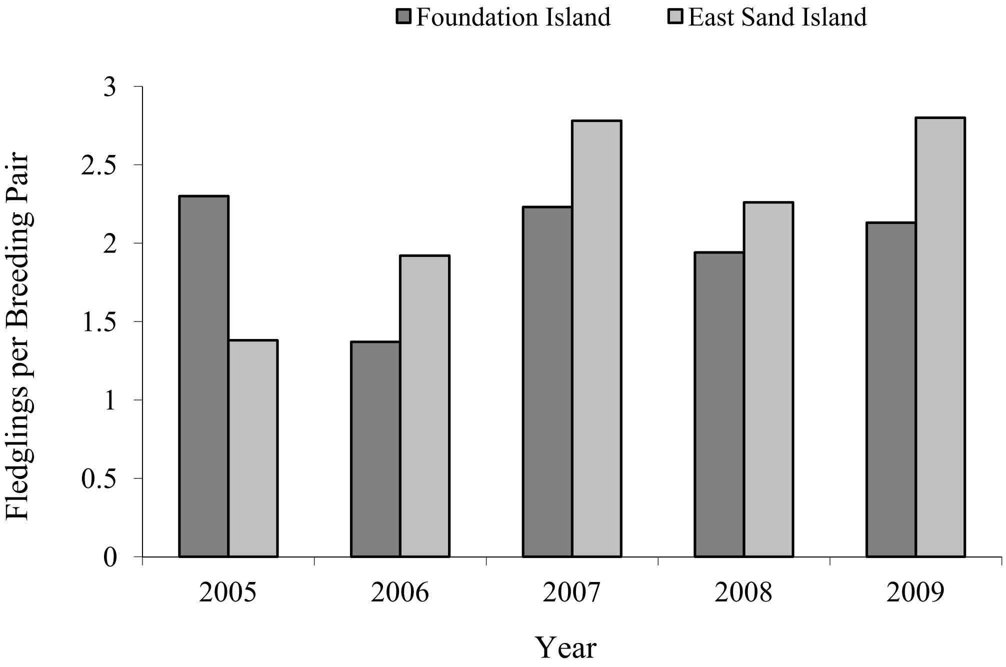

Figure 1.7. Productivity of the double-crested cormorant colony at Foundation Island in the Columbia Plateau region compared with the East Sand Island colony in the Columbia River estuary during 2005-2009.

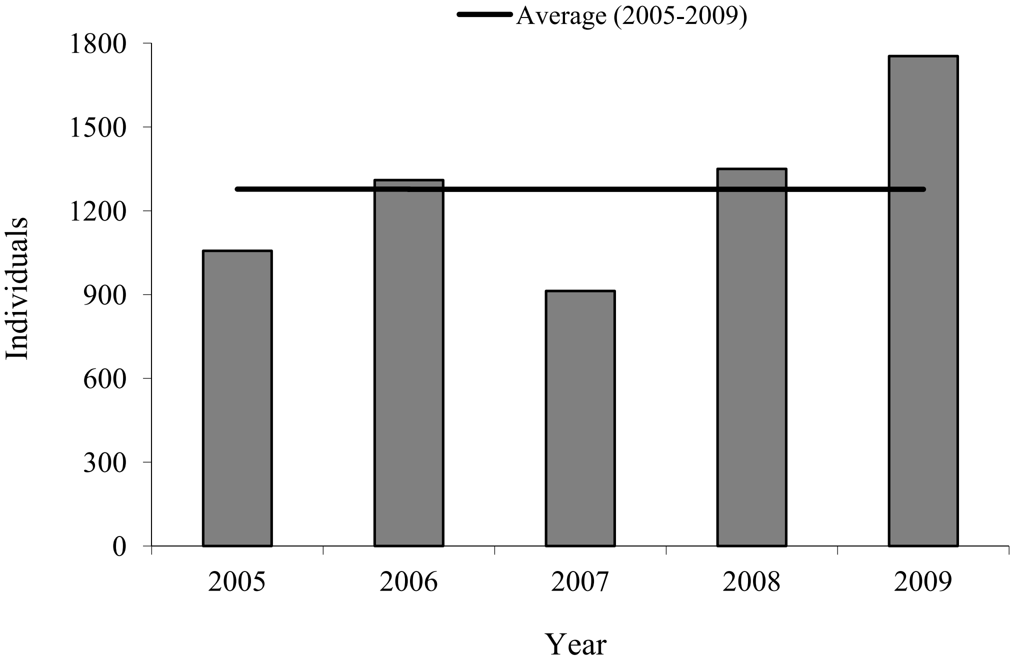

Figure 1.8. Number of individual American white pelicans counted in aerial photography of the breeding colony at Badger Island, mid-Columbia River during 2005-2009.

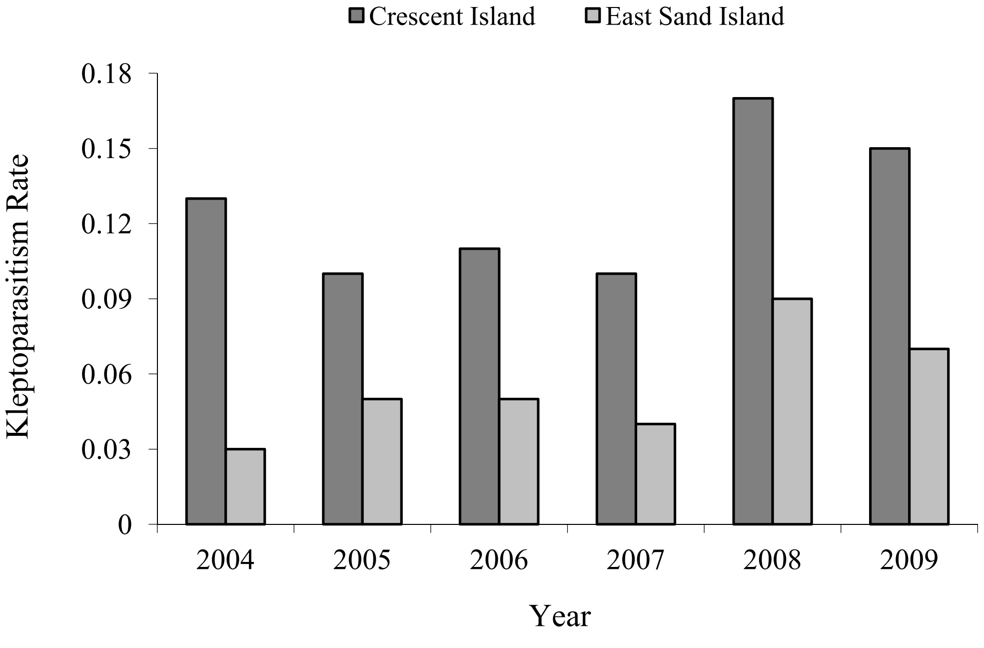

Figure 1.9. Annual kleptoparasitism rates of gulls on Caspian terns at the Crescent Island colony in the Columbia Plateau region compared to the East Sand Island colony in the the Columbia River estuary during 2004-2009. Kleptoparasitism rates are expressed as fractions of fish delivered to the colony by adult Caspian terns.

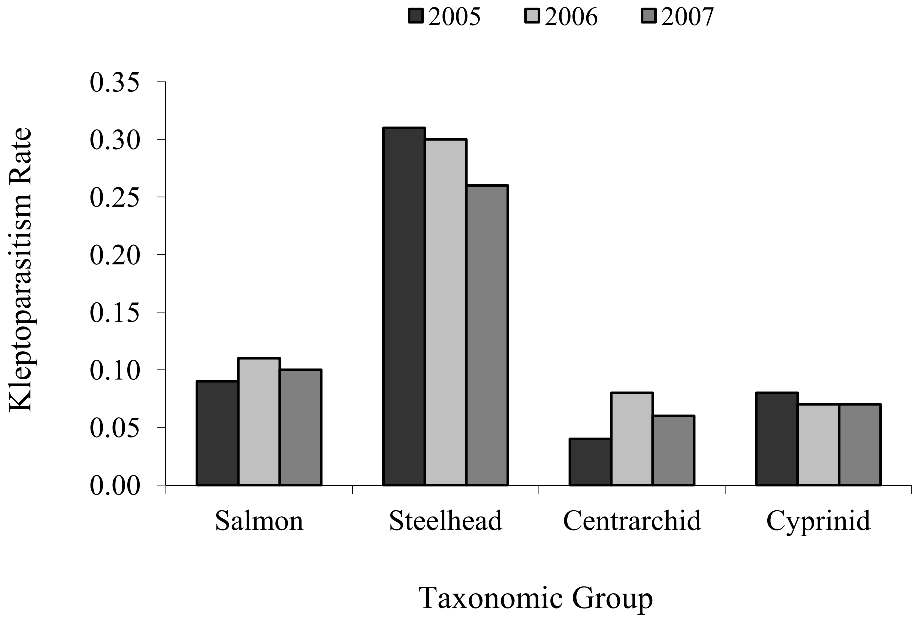

Figure 1.10. Kleptoparasitism rates by gulls on Caspian terns nesting at Crescent Island in the Columbia Plateau region during 2005-2007, separated by taxonomic group of fish. Kleptoparasitism rates are expressed as fractions of fish delivered to the colony by adult Caspian terns.

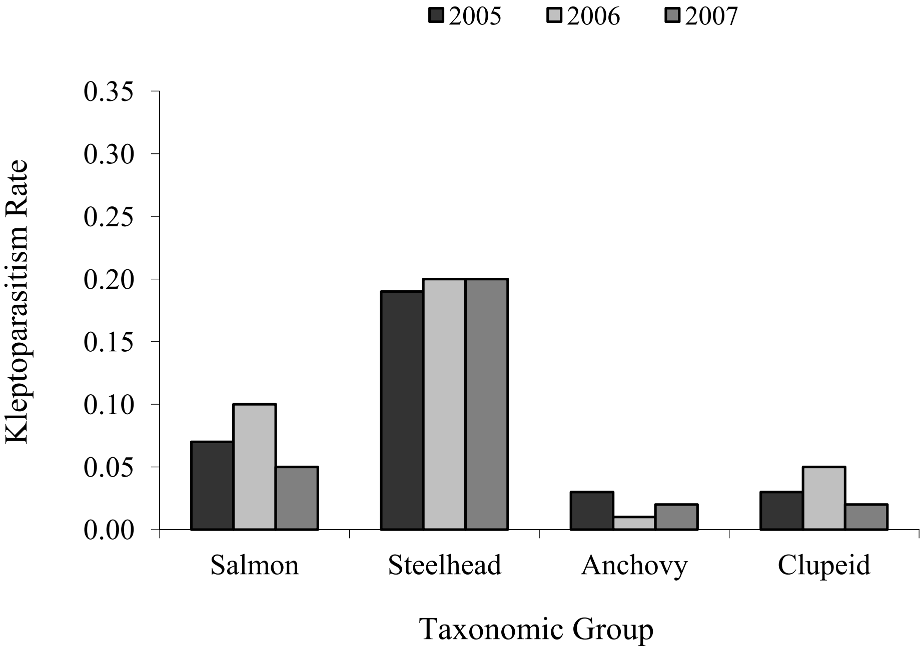

Figure 1.11. Kleptoparasitsm rates by gulls on Caspian terns nesting at East Sand Island in the Columbia River estuary during 2005-2007, separated by taxonomic group of fish. Kleptoparasitism rates are expressed as fractions of fish delivered to the colony by adult Caspian terns.

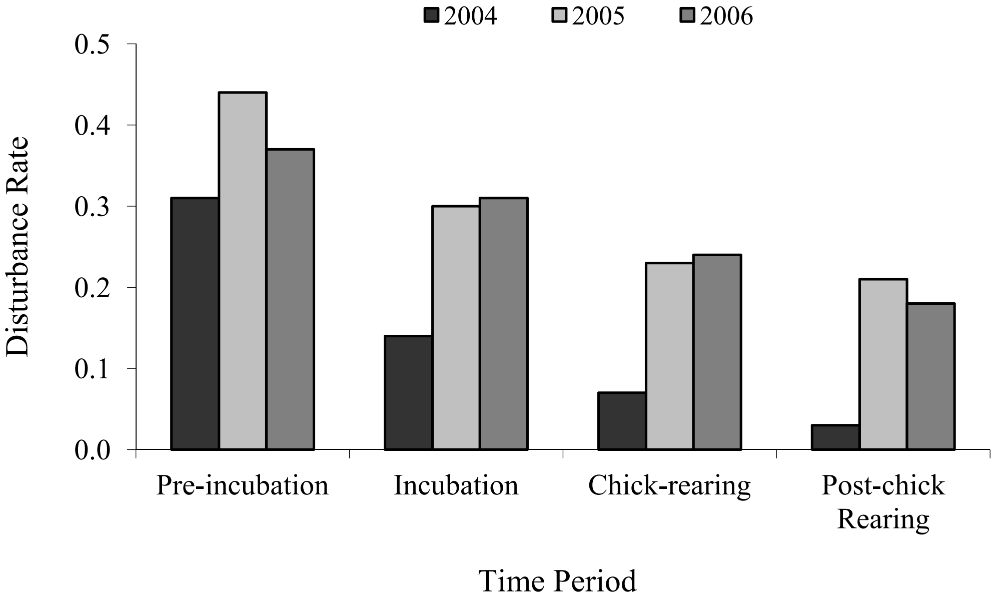

Figure 1.12. Average disturbance rate (disturbances/hour) of the Caspian tern colony at East Sand Island in the Columbia River estuary during the pre-incubation, incubation, chick-rearing, and post-chick rearing stages of the nesting period during 2004-2006.

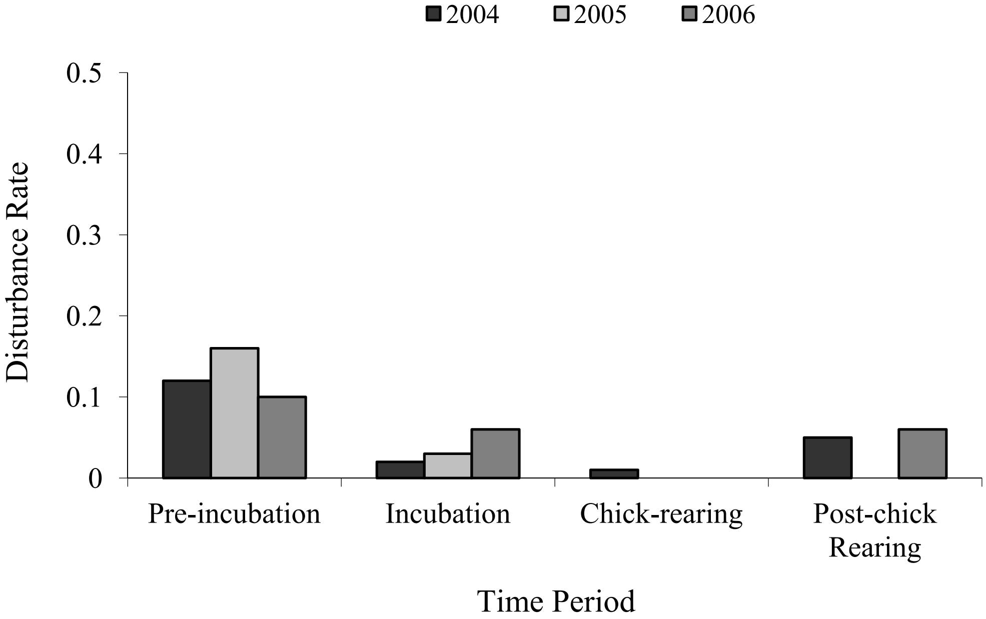

Figure 1.13. Average disturbance rate (disturbances/hour) of the Caspian tern colony at Crescent Island in the Columbia Plateau region during the pre-incubation, incubation, chick-rearing, and post-chick rearing stages of the nesting period during 2006-2008.

CHAPTER 2

IMPACTS OF PISCIVOROUS BIRDS ON NATIVE ANADROMOUS FISHES IN THE MID-COLUMBIA RIVER

This chapter has been prepared for the U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This chapter summarizes bioenergetic-based estimates of consumption of juvenile salmonids and lamprey by Crescent Island Caspian terns and Foundation Island double-crested cormorants during 2004-2009.

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SUMMARY

Anadromous salmonid (*Oncorhynchus* spp.) and Pacific lamprey (*Entosphenus tridentatus*) populations of the Columbia Basin are currently the subjects of intense conservation activity, following decades of decline. In recent years, avian predation across the Basin has been considered a possible factor limiting recovery of these imperiled fish populations. In this chapter we investigate whether two piscivorous waterbird colonies in the mid-Columbia River might be significant mortality factors for juvenile salmonids and lamprey by using bioenergetics methods to estimate prey consumption by Caspian terns (*Hydroprogne caspia*) nesting on Crescent Island and double-crested cormorants (*Phalacrocorax auritus*) nesting on Foundation Island.

Taken together, the Crescent Island Caspian tern colony and the Foundation Island double-crested cormorant colony consumed on the order of a million smolts annually during 2004 – 2009. Despite a somewhat smaller colony and less specialization on salmonids, cormorants consumed a greater biomass of salmonids than terns due primarily to their larger size and consequent greater individual energy requirements. Best estimates of salmonid consumption by Foundation Island cormorants ranged from ca. 470,000 to 880,000 smolts annually (depending on input variables and year). Estimates for Crescent Island terns ranged from ca. 330,000 to 500,000 smolts annually (depending on year). Consumption of salmon (coho [*O. kisutch*], sockeye [*O. nerka*], and Chinook salmon [*O. tshawytscha*] combined) by the Crescent Island tern colony declined during the study period, tracking a decline in colony size. Consumption of steelhead (*O. mykiss*) did not decline, however, perhaps reflecting greater steelhead availability in later years due to reduced transportation rates of Snake River steelhead. There was no trend in smolt consumption by cormorants during the study period, although our ability to detect inter-annual variability was constrained by insufficient data to investigate cormorant dietary differences between years. We found little evidence that birds from either colony were major predators of juvenile lamprey, with an estimate of fewer than 10,000 lamprey macropthalmia consumed per year by both colonies combined.

Further work is necessary to translate these bioenergetics-based estimates of juvenile salmonid consumption into predation rates (proportion consumed of those available) and to assess the potential benefits for threatened and endangered salmonid populations from reducing avian predation.

INTRODUCTION

In the Columbia River basin, avian predation is recognized as a factor that might limit recovery of salmonid (*Oncorhynchus* spp.) evolutionarily significant units (ESUs) listed under the U.S. Endangered Species Act (ESA; NOAA 2008). More than a century of anthropogenic declines in salmonid populations has resulted in the listing of 13 of 20 identified Columbia Basin ESUs as either threatened or endangered (Lichatowich 1999, Good et al. 2005). For more than a decade it has been known that many Basin populations of colonial-nesting piscivorous waterbirds prey upon juvenile salmonids (Collis et al. 2002). Research since that period of initial discovery has focused on identifying waterbird colonies that have potentially significant impacts on salmonids and assessing to what degree management to reduce predation on salmonids by those colonies would benefit ESA-listed stocks (Collis et al. 2001, Roby et al. 2003, Ryan et al. 2003, Antolos et al. 2003, Good et al. 2007, Maranto et al. 2008, Lyons 2010).

Similar to anadromous salmonids in western North America, anadromous Pacific lamprey (*Entosphenus tridentatus*) have experienced substantial declines in recent decades, mostly due to loss or degradation of freshwater habitats (Close et al. 2002). Pacific lamprey are currently considered a species of special concern by the U.S. Fish and Wildlife Service, a state sensitive species in Oregon and Washington, and a state-listed endangered species in Idaho (Brostrom et al. 2010). In the Columbia River basin, waterbirds have been documented for decades as predators of lamprey (e.g., Merrell 1959); however, the potential for birds to limit current lamprey recovery efforts has not been closely examined.

The largest colonies of strictly piscivorous waterbirds in the Basin reside on islands in the Columbia River estuary, and have received the most investigation, assessment, and consideration for management. Initial assessments concluded that reductions in predation by Caspian terns (*Hydroprogne caspia*), nesting in the estuary at the largest colony ever documented for the species (Suryan et al. 2004), might produce sufficient benefit to at least steelhead (*O. mykiss*) populations to warrant management (Roby et al. 2003, Good et al. 2007, Lyons 2010). Management efforts were initiated in 1999 and have been ongoing since that time (Roby et al. 2002, USFWS 2006). In recent years, management concern in the estuary has shifted from Caspian terns to double-crested cormorants (*Phalacrocorax auritus*), where the largest colony in western North America for this species also resides (Adkins and Roby 2010). Smolt consumption by the estuary cormorant population now exceeds that by Caspian terns (Lyons 2010).

In the Columbia Plateau region (interior Columbia Basin), impacts of piscivorous waterbirds on juvenile salmonids has been difficult to comprehensively assess because of the large numbers of waterbird species and colony sites across the region. Preliminary diet studies (Collis et al. 2002) and a subsequent more thorough investigation using recovery of salmonid passive integrated transponder (PIT) tags from bird colonies (Chapter 3) have indicated that, as in the estuary, Caspian tern and double-crested cormorant colonies are high priorities for further assessment. Consumption of juvenile salmonids by two Caspian tern colonies in the Columbia Plateau region has previously been estimated (Antolos et al. 2005, Maranto et al. 2008); however, both studies were of limited duration and did not characterize predation under the full range of river flows and river management regimes that are likely to occur in the future. Smolt consumption by double-crested cormorants in the mid-Columbia River has not been previously assessed.

Bioenergetics methods have been used to estimate prey consumption by piscivorous birds since the pioneering work of Wiens and Scott (1975) for seabirds along the Oregon coast. This approach has been applied numerous times to assess the impacts of waterbirds on salmonid populations of the Columbia River basin (Roby et al. 2003, Antolos et al. 2005, Wiese et al. 2008, Lyons 2010, Maranto et al. 2010), and has been an important component of management decision-making (e.g., USFWS 2005).

Our primary objective in this study was to quantify the consumption of juvenile salmonids and lamprey by Caspian terns nesting at Crescent Island and double-crested cormorants nesting at Foundation Island, on the mainstem Columbia River near Pasco, WA. For terns, we intended to compare our results to a previous characterization of predation by the Crescent Island colony (Antolos et al. 2005), and for both species to fit results into the context of avian predation occurring in the Columbia River estuary. Additionally, this work provides an independent line of evidence to corroborate PIT tag recovery efforts and analyses (Chapter 3), and will facilitate future analyses to quantify the potential benefits to at-risk fish populations if management to reduce avian predation is deemed appropriate.

METHODS

We characterized predation on both juvenile salmonids and lamprey by two species of piscivorous colonial waterbirds nesting near Pasco, WA, in Lake Wallula, the Columbia River reservoir impounded by McNary Dam. Specifically, we estimated prey consumption by Caspian terns nesting at Crescent Island and double-crested cormorants nesting at Foundation Island (Figure 2.1). Individuals of both species are known to forage in the mainstem Columbia and Snake rivers as well as in other smaller tributaries and nearby isolated surface waters. Crescent Island is an artificial island created in 1985 from the disposal of material dredged from a nearby channel. Caspian terns share the island with a large California gull (*Larus californicus*) colony and smaller numbers of ring-billed gulls (*Larus delawarensis*), great blue herons (*Ardea herodias*), black-crowned night-herons (*Nycticorax nycticorax*), and other waterbirds. Foundation Island was created by the rising waters of Lake Wallula following completion of McNary Dam in 1954. Cormorants nest in trees towards the downstream end of Foundation Island, along with great blue herons and black-crowned night-herons. As measured by passage through McNary Dam, the availability of steelhead trout and coho (*O. kisutch*), sockeye (*O. nerka*), and yearling Chinook (*O. tshawytscha*) salmon peaks in late April through early June of each year, with peak availability of sub-yearling Chinook salmon occurring in late June or early July (FPC 2011). Spikes in lamprey availability often occur in late May or early June, in association with high flow periods, with lower numbers observed at other times (FPC 2011).

Estimates of prey consumption were obtained using a bioenergetics modeling approach after Roby et al. (2003), as modified by Lyons (2010). Important input parameters for these calculations include: (1) number of consumers present (adults and chicks), (2) energy requirements of adults and chicks, (3) predator diet composition, and (4) energy content of the various prey types consumed. Data collection for terns and cormorants differed in some ways, so methodology for each will be described separately. For both species, data were collected across the breeding season (exceptions noted below), which for terns typically ran from early April until the end of July, and for cormorants, from early March until mid-July.

Caspian Tern Bioenergetics

The size of the Crescent Island Caspian tern colony was precisely estimated when most birds were nearing the end of the incubation period, which was when the peak colony size occurred each year. At least two simultaneous, independent counts were conducted of all active nests present at the colony multiple times per day during a 3-5 day period. The peak of these counts was considered to be the maximum colony size for the given year, with the maximum number of individual terns breeding at the colony taken to be double the number of active nests. To assess the number of adult terns associated with the colony at other times during the season, we conducted counts of all adults present on the colony multiple times per week across the breeding season. These counts were scaled to the number of adults present when the maximum colony size occurred.

The number of tern chicks that hatched was estimated by multiplying the peak colony size (number of active nests) by the average number hatched in a carefully monitored sample of nests. The number of chicks that fledged was estimated by counting the number of chicks on colony approximately one week after the first observation of a chick flying. Some chicks might have fledged and left the colony prior to this chick count but we assumed that the missed number of fledglings produced was balanced by other, younger chicks that were present and counted at this time but did not ultimately survive to fledge. To simplify consumption calculations, all chicks were considered to hatch synchronously at approximately the time we observed peak hatch. Chicks were assumed to leave the colony six weeks after hatch, consistent with a 35 day rearing period (Cuthbert and Wires, 1999) and a seven day residency following achievement of flight capability. Daily chick numbers were estimated by fitting a negative logarithmic function to the number hatched at the beginning of the chick-rearing period and the number estimated to have survived to fledging age (35 days).

Measured energy expenditure rates of free-ranging adult Caspian terns breeding in the Columbia River estuary (Roby et al. 2003) were used in conjunction with assimilation efficiencies measured in four week old Caspian tern chicks (Lyons and Roby 2011) to estimate energy requirements for tern adults. Chick energy requirements were drawn from energy consumption rates of captive-reared Caspian tern chicks fed an *ad libitum* diet (Lyons and Roby 2011).

Caspian terns carry whole fish back to the colony in their bill to feed mates and chicks. We determined diet composition by identifying these fish using binoculars from within a blind positioned at the edge of the colony. At least 100 prey items per week were identified to the taxonomic level of family. Salmonid prey were further specified as either steelhead trout or salmon (Chinook, coho, or sockeye salmon; see Antolos et al. 2005). It was not possible to further resolve the salmonid species composition of tern diets. We assumed all lamprey identified in tern bills were Pacific lamprey, the most common lamprey species found out-migrating in the mid-Columbia River (Wydoski and Whitney 2003). Energy content (i.e., average mass and energy density) of the various prey types were drawn from previous characterization of fish collected in the mid-Columbia River (Antolos et al. 2005).

Fish consumption was calculated for 2-week periods across each breeding season and summed to obtain annual totals. Calculations were performed using a routine written in Visual Basic 6.0 (Microsoft Corporation, Redmond, WA). A Monte Carlo calculation technique (1000 runs) was used to obtain confidence intervals for consumption estimates, with every run using independent, randomly drawn values for each of the input variables (Furness 1978). Each input parameter was assumed to be normally distributed and sampling errors in the input parameters were assumed to be uncorrelated. Annual variation in estimated juvenile salmonid and lamprey consumption by terns was a function of annual differences in colony size, seasonal attendance pattern, diet composition, and fledgling productivity. Calculations were performed in terms of numbers of prey consumed, consistent with tern diet composition data based on relative frequency of each prey type in the diet; biomass of each prey type consumed was also calculated using the average mass for each prey type from Antolos et al. (2005) to convert from numbers consumed.

Double-crested Cormorant Bioenergetics

The number of breeding double-crested cormorants at the Foundation Island colony was tracked across the season by counting the number of active nests visible from a blind positioned in shallow water off the northeast side of the island, typically three times per week. The number of adults associated with the colony was taken to be twice the number of active nests. The annual colony size reported here was the peak daily estimate of colony size in each year.

To account for the number of cormorant chicks present in the fish consumption calculations, we tracked the nest status of a sample of at least 50 focal nests each year. The number of chicks hatched per nest was taken to be the maximum number of chicks seen in a nest at any point following hatch, in the sample of focal nests. It was not possible to observe newly hatched chicks in nests directly because our vantage point in the blind was below the arboreal nests. Consequently, our hatchling estimates may slightly underestimate the actual number of hatchlings per nest if there was any mortality before chicks were large enough to stand in the nest and be visible to us. We also tabulated the number of chicks present in the sample of focal nests at approximately 28 days post-hatch. For the fish consumption calculations, as with terns, we assumed that all chicks hatched at the time of peak hatch and remained associated with the colony for 8 weeks. Daily chick numbers were estimated by fitting a negative logarithmic function to the number hatched (per nest, then multiplied by the colony size in breeding pairs) at the beginning of the chick-rearing period and the number estimated to have survived to 28 days post-hatch (per nest, then multiplied by the colony size). Chicks were considered to be associated with the colony, and included in the fish consumption calculations, until 56 days post-hatch.

Energy requirements of adult double-crested cormorants were derived from measurements of energy expenditure rates for adult cormorants in the Columbia River estuary (Lyons 2010), combined with measurements of assimilation efficiency in adult cormorants feeding on a variety of prey types (Brugger 1993). For chick energy needs, we used allometric predictions of total metabolizable energy requirements from hatching to fledging (Weathers 1992) and partitioned this total energy requirement into daily metabolizable energy requirements using the trend in daily requirements observed by Dunn (1975) for double-crested cormorants (see Lyons [2010] for details).

The taxonomic composition of cormorant diets was determined by lethally collecting adult cormorants returning to the Foundation Island colony and examining foregut contents. Diet composition, in percent biomass, was taken from the identification of all undigested soft tissue present in the foregut to the level of family (or to genus and species, when possible). Stomachs lacking any soft tissue (but possibly containing bones), and portions of gastro-intestinal tracts lacking any undigested soft tissue (e.g., bones in intestines), were excluded from the quantitative diet composition analysis. Soft tissue was identified to family using external features when possible or, when necessary, artificially digested to reveal diagnostic bones. Unidentifiable soft tissue lacking diagnostic bones was excluded from analysis. Because the colony was small (ca. 300 breeding pairs) it was not possible to collect enough cormorants in one year to sufficiently characterize diet for that year. Consequently, over the several years of data collection on diet composition (2005-2010), we sampled different periods of the cormorant breeding season to obtain a single, composite representation of Foundation Island cormorant diet, with greater sampling occurring during the period of juvenile salmonid out-migration from late April to early June (FPC 2011). Energy densities used for the various prey types were the same as for Caspian terns (Antolos et al. 2005).

Fish consumption was calculated as for Caspian terns, using 2-week periods across each breeding season and summed to obtain annual totals, with a Monte Carlo calculation technique (1000 runs) used to obtain confidence intervals. Annual variation in estimated juvenile salmonid and lamprey consumption by cormorants was a function of annual differences in colony size, seasonal attendance pattern, and fledgling productivity. As mentioned above, because of constraints on diet sampling, diet composition was assumed to be the same across all years. Calculations were performed in terms of prey biomass, consistent with cormorant diet composition data based on relative biomass of each prey type in the diet. Numbers of each prey type consumed were also calculated using the average mass for each prey type. Caspian tern prey mass data from Antolos et al. (2005) were used when appropriate; however, cormorants can consume larger sized prey than can terns. For prey where larger individuals were observed in cormorant stomachs than were in tern diets, we used the average prey mass obtained from whole or minimally digested fish from cormorant foregut samples.

We compared two techniques to convert salmonid biomass consumed by cormorants into numbers of juvenile salmonids consumed. The first approach used genetic identification of salmonid soft tissue recovered from the foreguts of collected adult cormorants. Salmonid samples were identified to species using PCR amplified genetic material (extracted from intact soft tissue or bone) by D.R.K. following procedures outlined in Purcell et al. (2004). The second approach presumed that cormorants took salmonids in proportion to availability. Availability estimates of smolts passing McNary Dam have recently become available based on counts of smolts conducted at the juvenile bypass facility (FPC 2011) combined with estimates of the proportion of all smolts passing through the dam that take this route (provided by B.P.S. and A.F.E. based on detection of smolts tagged with passive integrated transponders). The relative proportion of each salmonid species during each 2-week period of calculation was used, along with average mass, to partition the salmonid biomass consumed by cormorants into numbers of each species consumed. Given the uncertainties associated with either approach (small sample sizes for genetics analysis or presumption of take in proportion to availability), we report results here as total salmonids consumed, without further species/type resolution.

Temporal trends in input data and calculation outputs were examined using linear regression. Comparisons of early years (2004 – 2006) and late years (2007 – 2009) were sometimes made using Wilcoxon rank-sum tests. For output parameters, if 95% confidence intervals (CI) did not overlap, then differences were interpreted as significant.

RESULTS

The size of the Caspian tern colony on Crescent Island declined from a high of 530 breeding pairs in 2004 to a low of 349 pairs in 2009 ($P = 0.007$, Table 2.1). There was no directional trend in the size of the double-crested cormorant colony on Foundation Island, with the largest colony size recorded in 2006 at 359 breeding pairs and the smallest in 2004 at 300 pairs (Table 2.1). Initial egg-laying at the Crescent Island tern colony usually occurred in mid-April, with counts of adults on-colony peaking in the first half of May (Figure 2.2). Double-crested cormorants nesting at Foundation Island followed a slightly earlier nesting chronology, usually beginning egg-laying in March and peaking in colony size (counts of active nests) around the beginning of May (Figure 2.3). Activity at both colonies declined rapidly during June as birds either fledged young or experienced nest failure, and only small fractions of each colony were still present into July. The Caspian tern colony experienced variable productivity during the study period, with the highest level recorded in 2004, 0.62 fledglings per breeding pair, and the lowest in 2008, 0.26 fledglings per pair (Table 2.1). Cormorant productivity was higher but also varied substantially, from a low of 1.37 fledglings per breeding pair in 2006 to a high of 2.30 fledglings per pair in 2005 (Table 2.1).

Caspian terns relied on juvenile salmonids for 63 – 70% of their diet (by frequency) during 2004 – 2009 (Table 2.1). The proportion of the diet that was salmon (Chinook, coho, or sockeye salmon) declined from 63% in 2004 to 52% in 2009 (P = 0.02), while steelhead made up a greater proportion of the diet in later years: 10 – 14% in 2007 – 2009 versus 7 – 8% in 2004 – 2006 (P = 0.04). Salmonids (salmon and steelhead) often made up over 80% of the tern diet early in the season but declined to less than half the diet by the end of the nesting season in July (Figure 2.4). Juvenile lamprey (macropthalmia life stage) were a small component of tern diets (0.3 – 1.1% of annual diet; Table 2.1). Other important prey items for terns included centrarchids (bass [*Mircopterus* sp.], bluegill [*Lepomis macrochirus*], and others; 19 – 27% of annual diet) and cyprinids (peamouth [*Mylocheilus caurinus*], northern pikeminnow [*Ptychocheilus oregonensis*], and chiselmouth [*Acrocheilus alutaceus*]; 6 – 17% of annual diet).

Juvenile salmonids made up a smaller proportion of the diet of double-crested cormorants than for Caspian terns (Figure 2.5). For the composite expression of diet composition we derived from samples collected across the years during 2005 – 2010, salmonids were 22% of the cormorant diet (by biomass), averaged across semi-monthly periods from April to early July. The salmonid proportion of the cormorant diet was greatest during late April – late May, reaching a peak of 52% of the diet in early May. Juvenile lamprey were a very minor component (< 1%) of cormorant diets, and were only observed in samples collected during the second half of May. While adult cormorants are capable of capturing and consuming adult lamprey (authors' personal observation), we did not observe any adult lamprey taken by cormorants breeding at Foundation Island. Other important diet components included centrarchids (30%), ictalurids (channel catfish [*Ictalurus punctatus*], 14%), cyprinids (9%), and percids (yellow perch [*Perca flavescens*]; 8%).

Total prey consumption by the Crescent Island Caspian tern colony ranged between a minimum of 15,905 kg in 2008 (95% CI: 11,200 – 20,500 kg) and a maximum of 23,300 kg in 2004 (95% CI: 18,700 – 28,000 kg). Prey consumption by the Foundation Island double-crested cormorant colony averaged almost five times greater, between 82,000 kg in 2004 (95% CI: 64,100 – 99,800 kg) and 102,000 kg in 2007 (95% CI: 81,600 – 123,000 kg), despite the greater number of terns.

Consumption of juvenile salmonid biomass was also greater by the Foundation Island cormorant colony than by the Crescent Island tern colony in most years; however, differences were not as dramatic as for total prey consumption (Figure 2.6). Annual salmonid consumption by cormorants averaged 22,200 kg during 2004 – 2009, and ranged from a minimum of 19,600 in 2005 (95% CI: 15,000 – 24,200 kg) to a maximum of 24,700 kg in 2008 (95% CI: 20,300 – 29,200 kg). Terns consumed from 11,200 kg of salmonids in 2008 (95% CI: 7,900 – 14,500 kg) to 15,900 kg in 2004 (95% CI: 12,900 – 18,800 kg), and averaged 13,000 kg/year.

Salmonid biomass consumption by terns significantly declined over the period (P = 0.007), reflecting the decline in colony size; there was no apparent trend in salmonid consumption by cormorants during the study period. For both terns and cormorants, our estimates of salmonid biomass consumption peaked during late April and May (Figure 2.7); the pattern was particularly pronounced for cormorants, for whom salmonid consumption declined rapidly following the end of May. During that period of peak salmonid biomass consumption (May), our point estimates for consumption by cormorants always exceeded those for terns. At other times of the season, consumption by each colony was relatively similar.

Estimates of the total number of juvenile salmonids consumed by Caspian terns declined (P = 0.01; Figure 2.8) during the study from a high in 2004 of 497,000 smolts (95% CI: 448,000 – 547,000) to a low in 2008 of 334,000 (95% CI: 284,000 – 385,000). The trend in numbers of Chinook, coho, and sockeye salmon consumed by terns also declined over this period (P = 0.02) from 440,000 smolts in 2004 (95% CI: 348,000 – 532,000) to 271,000 in 2008 (95% CI: 189,000 – 353,000). Numbers of steelhead consumed by terns did not display a trend – the highest consumption was 73,800 in 2007 (95% CI: 52,100 – 95,600) and the lowest was 45,500 in 2005 (95% CI: 34,600 – 45,400).

Estimates of the number of juvenile salmonids consumed by double-crested cormorants depended greatly upon which approach was used to convert from biomass consumed to numbers of smolts consumed. Of the foregut samples used for diet analyses that contained salmonid soft tissue (n = 53), 70% had at least a portion of that salmonid soft tissue genetically identified as *O. mykiss*. In contrast, annually tabulated estimates of relative availability indicated that steelhead ranged from 8 – 20% of all available salmonids. Estimates of the total number of salmonids consumed by cormorants, using the genetics-based approach, ranged from 468,000 in 2004 (95% CI: 354,000 – 583,000) up to 589,000 in 2008 (95% CI: 448,000 – 729,000). Using the availability-based approach, comparable estimates ranged from 711,000 in 2005 (95% CI: 498,000 – 925,000) up to 883,000 in 2007 (95% CI: 620,000 – 1,146,000). Using either technique, there was no discernable trend in smolt consumption during the study period (Figure 2.8).

Annual consumption of juvenile lamprey was approximately two orders of magnitude less than consumption of juvenile salmonids (Figure 2.9). Point estimates of lamprey consumption by the Crescent Island Caspian tern colony averaged about 3,000 macropthalmia per year, with a range from 1,400 in 2007 (95% CI: 900 – 1,900) to 7,300 in 2005 (95% CI: 5,500 – 9,200). Double-crested cormorants from the Foundation Island colony consumed an average of about 2,000 macropthalmia per year, ranging from 1,800 in 2004 (95% CI: 800 – 2,000) to 2,300 in 2008 (95% CI: 1,100 – 3,400).

DISCUSSION

Our results indicate that piscivorous waterbirds nesting at two colonies near the confluence of the Snake and Columbia rivers are significant predators of small fish in the mid-Columbia River, and in particular may combine to consume on the order of a million juvenile salmonids during their annual out-migration. The Foundation Island double-crested cormorant colony consumed a much higher amount of prey biomass than did the Crescent Island Caspian tern colony, despite the somewhat smaller number of nesting cormorants. The greater total prey consumption by the cormorant colony reflected (1) the much greater energy requirements for individual cormorants (average adult energy expenditure rate of 3,423 kJ/day; Lyons 2010) compared to that for terns (1,040 kJ/day; Roby et al. 2003), (2) the greater energy requirements of cormorant broods containing more and larger chicks than tern broods, (3) a slightly longer (earlier beginning) breeding season for the Foundation Island cormorant colony, and (4) the cormorant diet consisting of greater proportions of somewhat lower energy density prey (e.g., centrarchids and perch).

Caspian terns consumed an estimated hundreds of thousands of smolts annually, but consumption was lower than the most recent previously documented level for this colony (690,000 smolts consumed in 2001 [95% CI: 533,000 – 825,000]; Antolos et al. 2005), and declined during our study period, 2004 – 2009. Estimates of smolt consumption levels by double-crested cormorants, not previously documented, were generally greater than for terns; differences between the colonies in salmonid biomass consumed were significant in most years (2006 – 2009). Steelhead were prominent in the diet of both species of avian predators.

The downward trend in estimated salmonid consumption by Caspian terns was driven by the decline in colony size during the study period. A number of factors might explain why the colony is in decline. First, a large California gull colony also exists on Crescent Island, and interactions between the two species have several negative consequences for terns. Gulls compete with terns for nesting space, kleptoparasitize the tern colony, stealing some portion of fish brought back by terns for their mates or chicks, and also are occasional nest predators of terns at this site, depredating eggs and/or chicks. Second, availability of nutritionally profitable juvenile salmonids, which comprise the majority of the tern diet in the mid-Columbia River, while high during the terns' incubation period (late April and May), declines rapidly during the chick-rearing period (late May and June for most terns), as the larger steelhead trout, yearling Chinook, coho, and sockeye salmon migrate out of the region (FPC 2011). The range of productivity seen at Crescent Island during this study (0.26 – 0.62 fledglings/pair) is lower than at regional Caspian tern colonies that are not in decline (Roby et al. 2002, Lyons 2010), and may be too low to sustain a stable population at the site (Cuthbert and Wires 1999, Suryan et al. 2004). The Crescent Island tern colony may be maintained by episodic recruitment and high productivity during particularly favorable years, such as when water conditions make steelhead particular vulnerable, especially during the chick-rearing period. For example, larger colonies and high productivity at this site have been seen in years of low flows (e.g., 2001 and 2004; Antolos et al. 2005, Chapter 4), when steelhead in particular were less likely to successfully migrate downstream.

The decline in total smolt consumption by Caspian terns at Crescent Island primarily reflects a decline in the number of Chinook, coho, and sockeye salmon consumed. No decline in steelhead consumption has been seen as the colony has declined in size. This may be attributable in part to tern preference for steelhead (Antolos et al. 2005, Chapter 3) and greater numbers of steelhead in the river in the later years of our study (2007 – 2009) due to the removal of fewer smolts from the river for transportation downstream in barges (FPC 2010).

Data limitations constrained our ability to quantify consumption levels by double-crested cormorants nesting at Foundation Island. Because of the small size of the colony, we were unable to collect sufficient information on diet composition to examine possible dietary differences between years. We instead collected samples across multiple years to assemble a single, composite description of cormorant diet. For this reason, our estimates of cormorant annual prey consumption likely underrepresented the actual degree of inter-annual variation. Our results do bound average conditions, however, and accurately portray the impact of annual changes in cormorant colony size and productivity on prey consumption.

We also have limited data to convert our estimates of salmonid biomass consumed by cormorants to numbers of smolts consumed. Genetically identified salmonids from the foreguts of cormorants suggest that *O. mykiss* was a major component of the salmonid proportion of the diet of Foundation Island cormorants. Using these data, or assuming that cormorants took salmonids in proportion to availability, the total number of smolts consumed by the Foundation Island cormorant colony was estimated to be similar or greater than that by the Crescent Island Caspian tern colony (Figure 2.8). This result is generally inconsistent with the number of passive integrated transponder (PIT) tags implanted in smolts and later detected on the ground beneath cormorant nesting trees on Foundation Island and on the nesting surface used by terns at Crescent Island, even after a correction for tag detection efficiency (Chapter 3). Estimates of the number of PIT tags present have consistently been greater for the tern colony.

At least two possibilities might explain the apparent contradiction between the similar bioenergetics-based estimates of salmonids consumed by each colony and the number of smolt PIT tags detected at each colony. First, cormorants might prey on groups of salmonids that are tagged at a low rate. For example, Chinook smolts are the most available salmonid species in the mid-Columbia River and some of these are not included in the PIT tag results (e.g., upriver brights originating downstream from Rock Island dam; see Chapter 3), but might be targeted by cormorants. In the Columbia River estuary, sub-yearling Chinook are sometimes substantially more susceptible to predation by cormorants from East Sand Island than by terns from the same site (S. Sebring, NOAA Fisheries, personal communication). Additionally, some proportion of the *O. mykiss* identified in stomachs could have been resident rainbow trout, not anadromous steelhead trout, stocked in nearby waters for angling opportunities. Each spring thousands of catchable (> 20 cm) rainbow trout are released into lakes and ponds within foraging distance of cormorants on Foundation Island (WDFW 2011). These stocked trout often exceed the size terns can efficiently prey upon, but would fall in a size range vulnerable to cormorant predation. A second reason bioenergetics-based consumption estimates might not be consistent with PIT tag results is that the number of tags detected beneath the nesting trees at Foundation Island might under-represent the number of PIT-tagged smolts actually consumed by cormorants. It is not known how PIT tags are expelled from cormorants once ingested. For example, PIT tags might be regurgitated, as are larger bones, or they might pass through the entire gastro-intestinal (GI) tract and be excreted with guano. If the former mode of excretion is most frequent, than a substantial proportion of egested PIT tags may not fall to the ground, but instead become incorporated in nest material. If the latter mode is frequent, then some proportion of ingested tags may be expelled away from the cormorant colony and thus undetected during PIT tag recovery (Chapter 3). Given the size difference between double-crested cormorants (ca. 2250 g body mass) and Caspian terns (ca. 650 g body mass), the proportion of tags passed completely through the GI tract and potentially expelled from the birds' body away from the nesting colony may be substantially different between the two species. This would result in the number of tags found at Foundation Island under representing the reliance of cormorants on all salmonids, including steelhead.

Given the uncertainty in our estimates of numbers of salmonids consumed by cormorants, it would be most conservative to assess the impact of Foundation Island cormorant predation on threatened and endangered salmonid populations based on comparisons of biomass consumption to other colonies being considered for management to reduce predation (e.g., Crescent Island Caspian terns). If estimates of the number of smolts consumed are deemed necessary to assess cormorant impacts, we advise that the uncertainties associated with such estimates be explicitly acknowledged (e.g., Figure 2.8).

Results for smolt consumption by Crescent Island Caspian terns and Foundation Island double-crested cormorants during 2004 – 2009 span a fortuitously representative range of river flow and transportation rate conditions. Average spring and early summer flows (March – July) through McNary Dam ranged from 20% below the 30-year average (1980 – 2009) in 2004 to 19% above average in 2006, with the 6-year (2004 – 2009) average just 3% below the longer term average (USACE 2011). Levels of transportation of Snake River smolts varied considerably during the study period, from an estimated high of 95% in 2004 to a low of 35% in 2007 (FPC 2010). These varying rates of fish removal had correspondingly profound impacts on the number of smolts migrating through the river and available to birds foraging in Lake Wallula. With respect to river flows and transportation rates, our results here are better predictors of the likely future impacts of the Crescent Island tern colony on salmonid populations than those of Antolos et al. (2005), which documented predation during 2001, a year of extremely low water (lowest flow year in 30-year period and 48% below average) and maximum transportation (> 98% of Snake River spring migrating smolts removed from the river; FPC 2010). The results of Antolos et al. (2005) for 2001 likely describe the maximum potential impact of the Crescent Island tern colony on salmonid smolt survival in the mid-Columbia River.

Consumption of juvenile salmonids by Crescent Island Caspian terns and Foundation Island double-crested cormorants during 2004 – 2009 is approximately an order of magnitude less than that by Caspian terns and double-crested cormorants nesting at colonies in the Columbia River estuary during the same period (6 – 17 million smolts annually; Lyons 2010). Mortality rates induced by these lower levels of smolt consumption can still be of potential concern for upper basin salmonid populations, however. Mortality rates for in-river migrating fish can still be significant, particularly for steelhead (occasionally in excess of 20%; see Chapter 3). Of course, mortality rates for in-river migrating fish do not directly correspond to mortality rates for entire evolutionarily significant units (ESUs) of salmonids for those ESUs where a portion of out-migrating smolts are transported around the islands of Lake Wallula.

As mentioned above, gulls at Crescent Island kleptoparasitize fish brought back to the Caspian tern colony by adult terns to provision their chicks and/or mates. This means that adult terns must capture additional fish to meet the energy demands of the parent-offspring unit. Our bioenergetics-based prey consumption calculations do not incorporate this additional demand; however, one might crudely estimate this additional demand using data on chick energy demands and kleptoparasitism rates. During the chick-rearing period, approximately 30% of the food required by the colony is that required by chicks. So approximately that proportion of the food captured by adult terns is brought back to the colony during that period and is subject to kleptoparasitism by gulls. At other times of the season, the proportion of food brought back to the colony, of that captured, would be less, as terns expend more effort provisioning chicks than mates (authors' personal observation); however, using 30% for the entire season would be conservative (i.e., would over-estimate the number of additional prey necessary to be captured). Kleptoparasitism rates seen at Crescent Island during the study period were approximately 30% for steelhead and 10% for other salmonids (Chapter 1). Consequently, one might very roughly estimate that at most, gulls might have stolen an amount equivalent to 9% of the steelhead we estimated to be consumed by the tern colony and 3% of the other salmonids, although the actual levels are probably lower for both prey types. These proportions could be used to conservatively estimate the additional impact of gull kleptoparasitism on salmonid mortality due to predation by Crescent island Caspian terns.

While our results demonstrate that Caspian terns and double-crested cormorants consume large numbers of juvenile salmonids, very few juvenile lamprey were consumed. Taken together, we estimated that the birds at these two colonies consume less than 10,000 lamprey macropthalmia per year, consistent with earlier results (3,000 – 6000 consumed per year by the tern colony alone in 2000-01; Antolos et al. 2005). A greater diet sampling frequency would be necessary to more precisely estimate lamprey consumption as it appears that use of lamprey as a prey resource is limited to occasional episodes during the birds' breeding season. Our diet sampling frequency for cormorants was too low to detect any cormorant consumption of adult lamprey, although it may occur.

In summary, piscivorous birds nesting at two colonies near the confluence of the Snake and Columbia rivers are significant predators of juvenile salmonids in the mid-Columbia River. The Crescent Island Caspian tern colony and the Foundation Island double-crested cormorant colony combined to consume on the order of a million smolts annually during 2004 – 2009, with cormorants consuming more salmonid biomass than terns. Consumption by the Crescent Island tern colony declined during the period, reflecting a decline in colony size; there was no trend in consumption by cormorants during the study period. We found little evidence that birds from either colony were consuming large numbers of juvenile lamprey. Further work is necessary to translate these bioenergetics-based estimates of juvenile salmonid consumption into predation rates (proportion consumed of those available) and to assess the potential benefits of reducing avian predation for threatened and endangered salmonid populations in the mid-Columbia River.

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REFERENCES

Adkins, J.Y., and D.D. Roby. 2010. A status assessment of the double-crested cormorant (*Phalacrocorax auritus*) in western North America: 1998 – 2009. U.S. Army Corps of Engineers, Portland, OR.

Antolos, M., D.D. Roby, D.E. Lyons, K. Collis, A.F. Evans, M. Hawbecker, M., and B.A. Ryan. 2005. Caspian tern predation on juvenile salmonids in the mid-Columbia River. Transactions of the American Fisheries Society 134:466-480.

Brostrom, J.K., C.W. Luzier, and K. Thompson. 2010. Best management practices to minimize adverse effects to Pacific lamprey (*Entosphenus tridentatus*). U.S. Fish and Wildlife Service, Portland, OR.

Brugger, K.E. 1993. Digestibility of three fish species by double-crested cormorants. Condor 95: 25-32.

Collis, K., D.D. Roby, D.P. Craig, B.A. Ryan, and R.D. Ledgerwood. 2001. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary: vulnerability of different salmonid species, stocks, and rearing types. Transactions of the American Fisheries Society 130:385-396.

Collis, K., D.D. Roby, D.P. Craig, S. Adamany, J.Y. Adkins, and D.E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Close, D.A., M.S. Fitzpatrick, and H.W. Li. 2002. The ecological and cultural importance of a species at risk of extinction, Pacific lamprey. Fisheries 27:19-25.

Cuthbert, F.J., and L.R. Wires. 1999. Caspian Tern. No. 403 *In* The Birds of North America. *Edited by* A. Poole and F. Gill. The Birds of North America, Inc., Philadelphia, PA.

Dunn, E.H. 1975. Caloric intake of nestling double-crested cormorants. Auk 92:553-565.

FPC (Fish Passage Center). 2010. The 2009 Fish Passage Center Annual Report. Portland, OR.

FPC (Fish Passage Center). 2011. Smolt Monitoring Program. Available at http://www.fpc.org [accessed 15 January 2011].

Furness, R.W. 1978. Energy requirements of seabird communities: A bioenergetics model. Journal of Animal Ecology 47:39-53.

Good, T.P., R.S. Waples, and P. Adams. (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. Technical memorandum NFMS-NWFSC-66, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, WA.

Good, T.P., M.M. McClure, B.P. Sanford, K.A. Barnas, D.M. Marsh, B.A. Ryan, and E. Casillas. 2007. Quantifying the effect of Caspian tern predation on threatened and endangered Pacific salmon in the Columbia River estuary. Endangered Species Research 3:11-21.

Lichatowich, J. 1999. Salmon without rivers: A history of the Pacific salmon crisis. Island Press, Washington, D.C.

Lyons, D.E. 2010. Bioenergetics-based predator-prey relationships between piscivorous birds and juvenile salmonids in the Columbia River estuary. Unpublished Ph.D. dissertation, Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR.

Lyons, D.E., and D.D. Roby. 2011. Validating growth and development of a seabird as an indicator of food availability: captive-reared Caspian Tern chicks fed ad libitum and restricted diets. Journal of Field Ornithology 82:88-100.

Maranto, C.J., T.P. Good, F.K. Wiese, and J.K. Parish. 2010. Impact of the Potholes Reservoir Caspian tern breeding colony on out-migrating juvenile salmonids in the mid-Columbia River. Transactions of the American Fisheries Society 139:362-381.

Merrell, T.R. 1959. Gull food habits on the Columbia River. Fish Commission of Oregon Research Briefs 7:82.

NOAA (National Oceanic and Atmospheric Administration). 2008. Biological opinion, Endangered Species Act section 7(a)(2) consultation: consultation on remand for operation of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(1)(A) permit for Juvenile Fish Transportation Program. Seattle, WA.

Purcell, M., G. Mackey, E, LaHood, H. Huber, and L. Park. 2004. Molecular methods for the genetic identification of salmonid prey from Pacific harbor seal (*Phoca vitulina richardsi*) scat. Fishery Bulletin 102:213-220.

Roby, D.D., K. Collis, D.E. Lyons, D.P. Craig, J.Y. Adkins, and A.M. Myers. 2002. Effects of colony relocation on diet and productivity of Caspian terns. Journal of Wildlife Management 66:662-673.

Roby, D.D., D.E. Lyons, D.P. Craig, K. Collis, and G.H. Visser. 2003. Quantifying the effect of predators on endangered species using a bioenergetics approach: Caspian terns and juvenile salmonids in the Columbia River estuary. Canadian Journal of Zoology 81:250-265.

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

Suryan, R.M., D.P. Craig, D.D. Roby, N.D. Chelgren, K. Collis, W.D. Shuford, and D.E. Lyons. 2004. Redistribution and growth of the Caspian tern population in the Pacific Coast region of North America, 1981-2000. Condor 106:777-790.

USACE (U.S. Army Corps of Engineers). 2011. Columbia Basin Water Management Division Interactive Database. Available at http://www.nwd-wc.usace.army.mil/index.html [accessed 15 February 2011].

USFWS (U.S. Fish and Wildlife Service). 2005. Caspian tern management to reduce predation of juvenile salmonids in the Columbia River estuary, Final Environmental Impact Statement. U.S. Fish and Wildlife Service, Migratory Birds and Habitat Programs, Portland, OR.

Washington Department of Fish and Wildlife (WDFW). 2011. Catchable Trout Planting Reports 2004-2009. Available at http://wdfw.wa.gov/fishing/plants/weekly/

Weathers, W.W. 1992. Scaling nestling energy requirements. Ibis 134:142-153.

Wiens, J.A., and J.M. Scott. 1975. Model estimation of energy flow in Oregon coastal seabird populations. Condor 77:439-452.

Wiese, F.K., J.K. Parrish, C.W. Thompson, and C. Maranto. 2008. Ecosystem-based management of predator-prey relationships: Piscivorous birds and salmonids in the mid-Columbia River. Ecological Applications 18:681-700.

Wydoski, R.S., and R.R. Whitney. 2003. Inland fishes of Washington. University of Washington Press, Seattle, WA, USA.

Table 2.1. Annually-varying input parameters for bioenergetic calculations of prey consumption by Caspian terns nesting at Crescent Island and double-crested cormorants nesting at Foundation Island near the confluence of the Snake and Columbia rivers during 2004-2009.

		2004	2005	2006	2007	2008	2009
Caspian Terns	Colony Size (breeding pairs)	530	476	448	355	388	349
	Productivity (fledglings/pair)	0.62	0.55	0.43	0.68	0.26	0.44
	Portion of Diet Salmon (%)	63%	58%	55%	55%	56%	52%
	Portion of Diet Steelhead (%)	7%	7%	8%	14%	10%	11%
	Portion of Diet Lamprey (%)	0.5%	1.1%	0.3%	0.3%	0.3%	0.3%
Double-crested Cormorants	Colony Size (breeding pairs)	300	315	359	334	357	309
	Productivity (fledglings/pair)	n/a*	2.30	1.37	2.23	1.94	2.13

*Cormorant productivity data for 2004 is not available. For the purposes of the bioenergetics calculations, productivity was assumed to be the average of the other years (1.99 fledglings/pair).

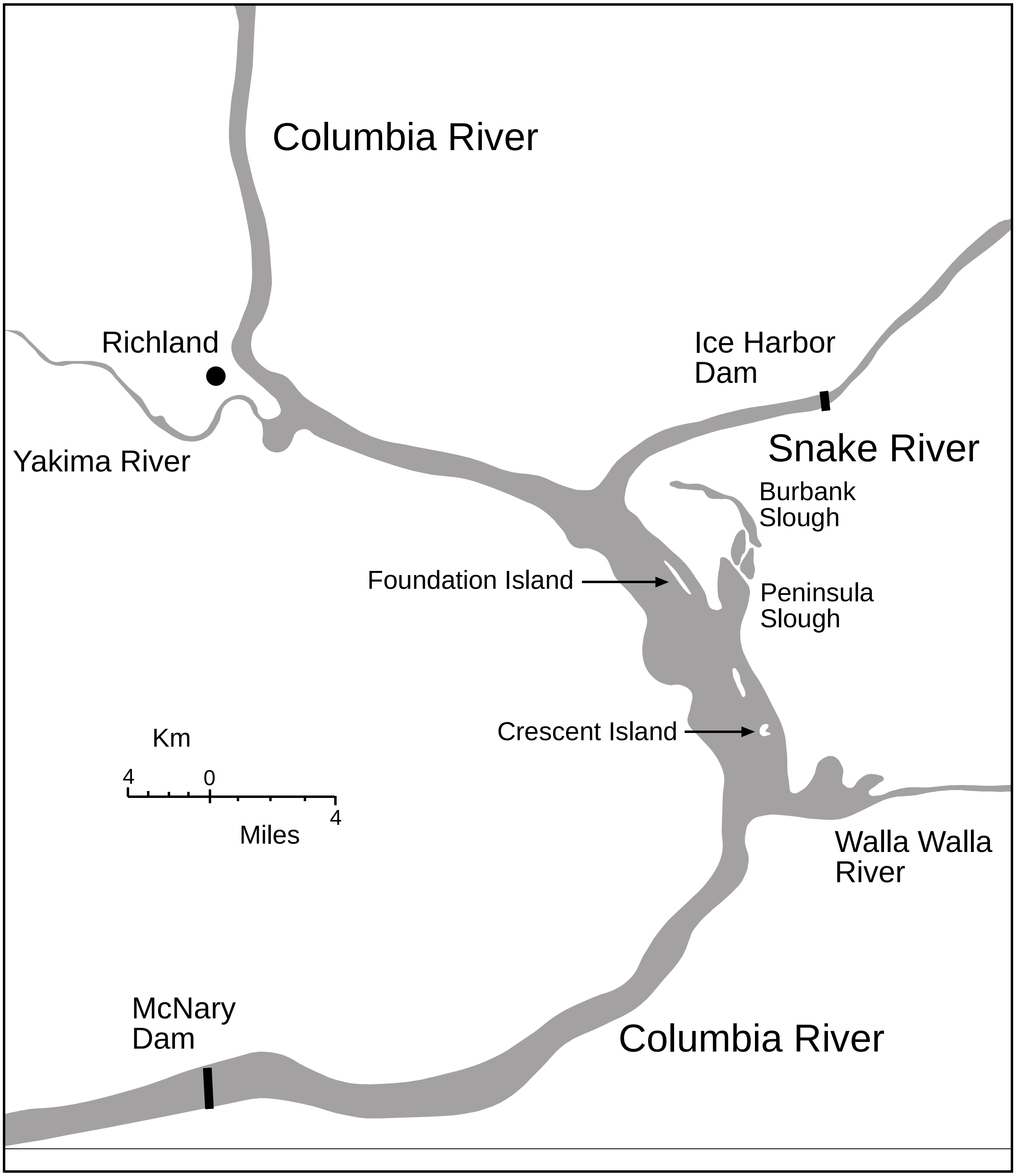

Figure 2.1. Map of the Lake Wallula impoundment formed by McNary Dam, showing the location of the Foundation Island double-crested cormorant colony and the Crescent Island Caspian tern colony on the mid-Columbia River.

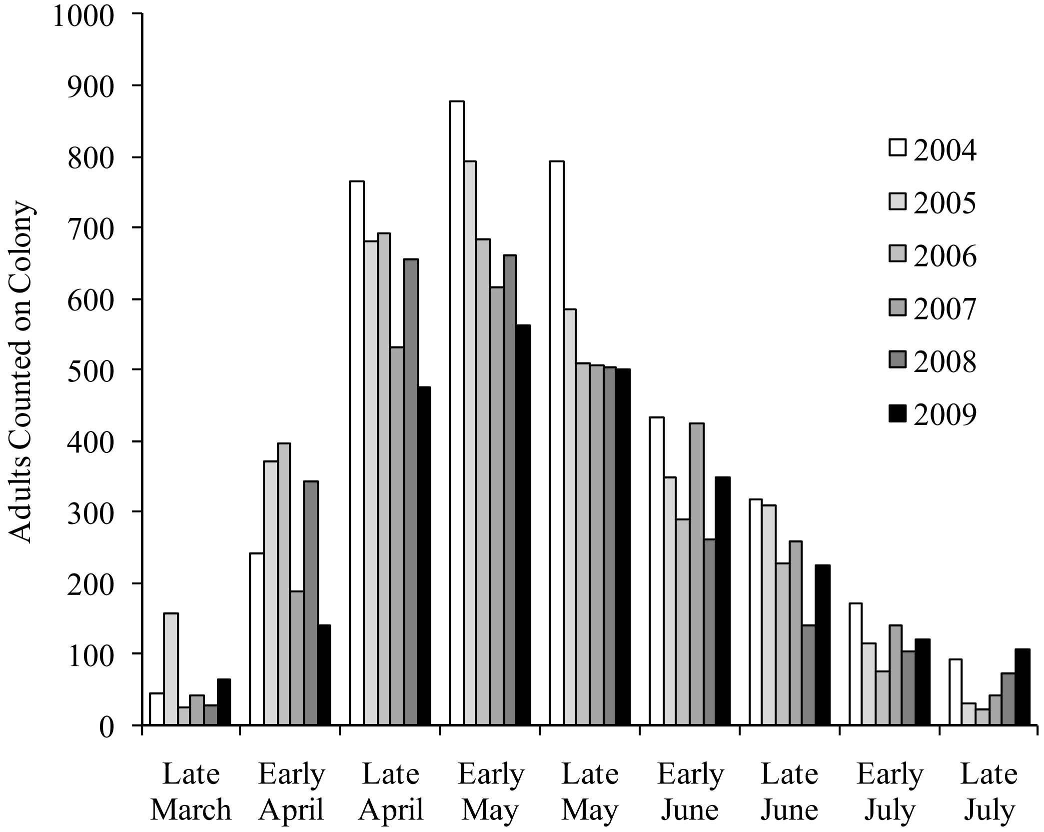

Figure 2.2. Within season trends in the size of the Crescent Island Caspian tern colony on the mid-Columbia River.

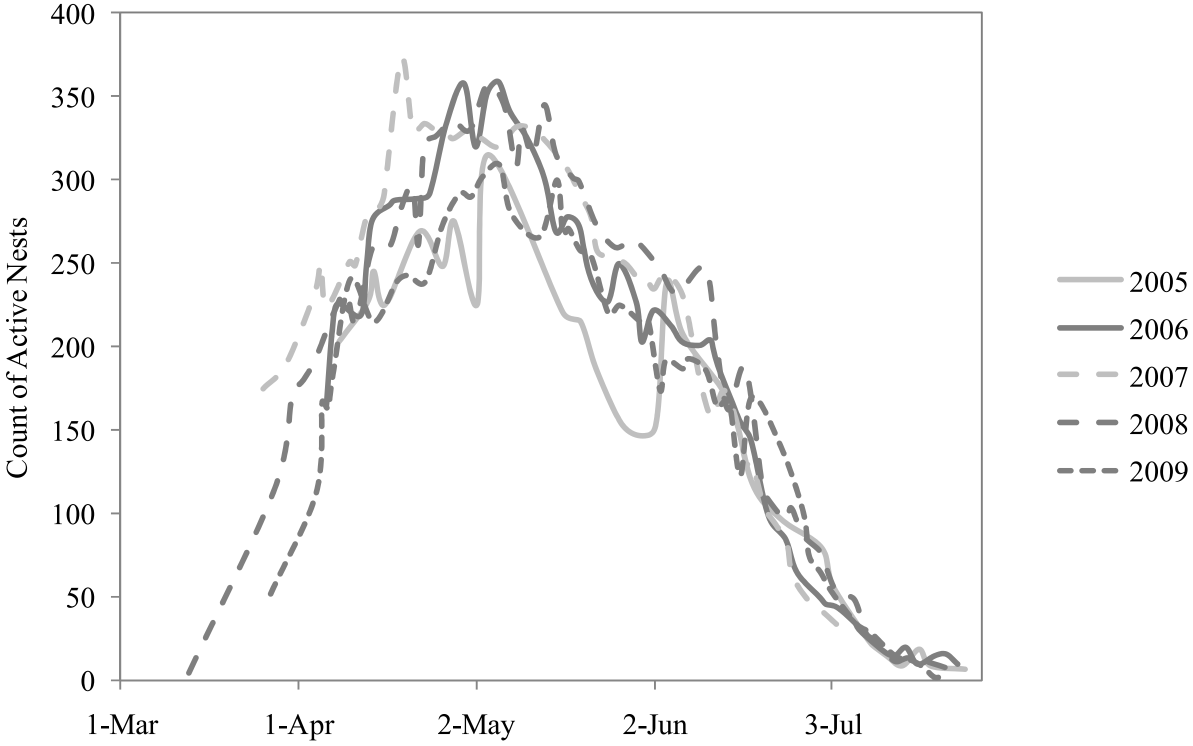

Figure 2.3. Within season trends in the size of the Foundation Island double-crested cormorant colony on the mid-Columbia River.

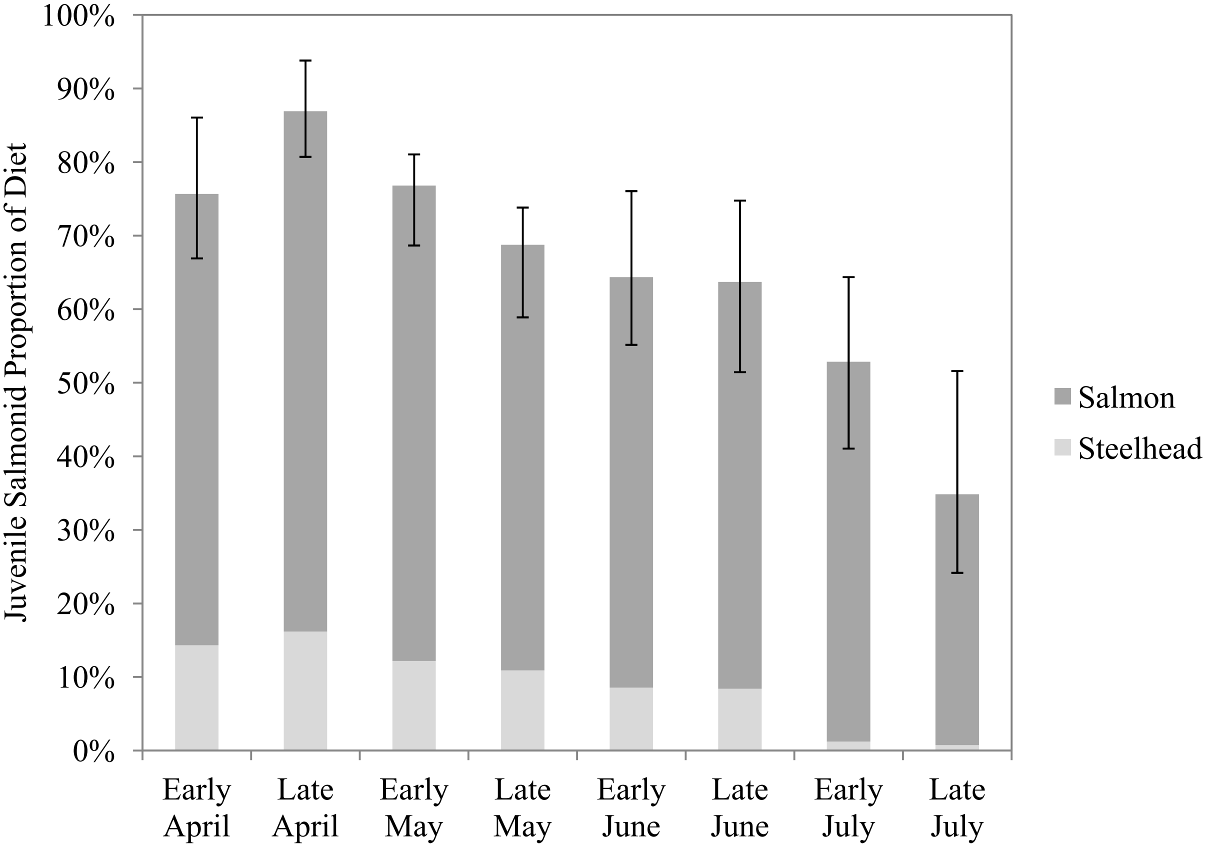

Figure 2.4. Seasonal pattern in the proportion of juvenile salmonids in the diet of Caspian terns nesting on Crescent Island (% frequency) broken into proportion of steelhead and salmon (Chinook, coho, and sockeye salmon) averaged across 2004 – 2009. Error bars represent the range of annual values for total proportion of salmonids in the diet.

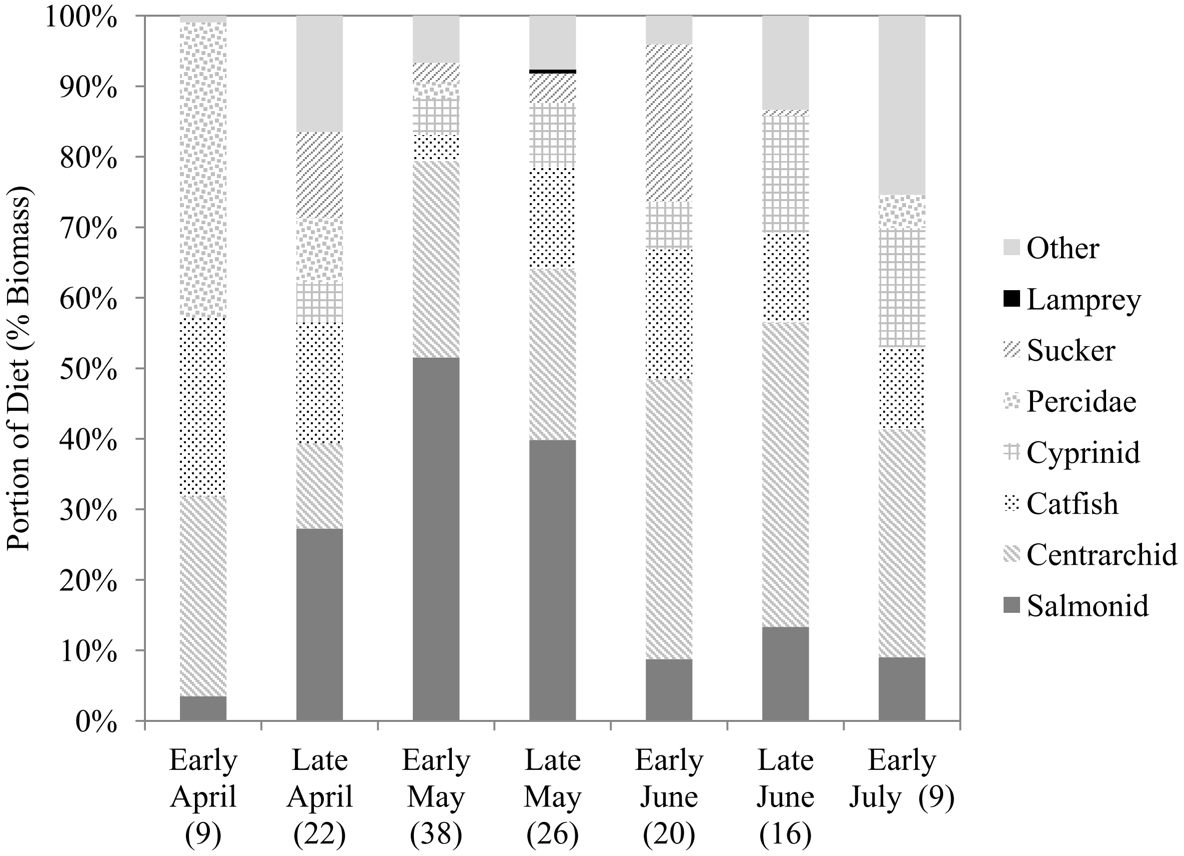

Figure 2.5. Seasonal pattern in diet composition (% biomass) of double-crested cormorants nesting at Foundation Island. Diet composition was based on identifiable soft tissue in foreguts of adult cormorants collected returning to the colony during 2005-2010. Sample sizes are indicated for each semi-monthly period. Prey types classified as "other" include sticklebacks, sculpins, crustaceans, and other unidentified non-salmonid fishes.

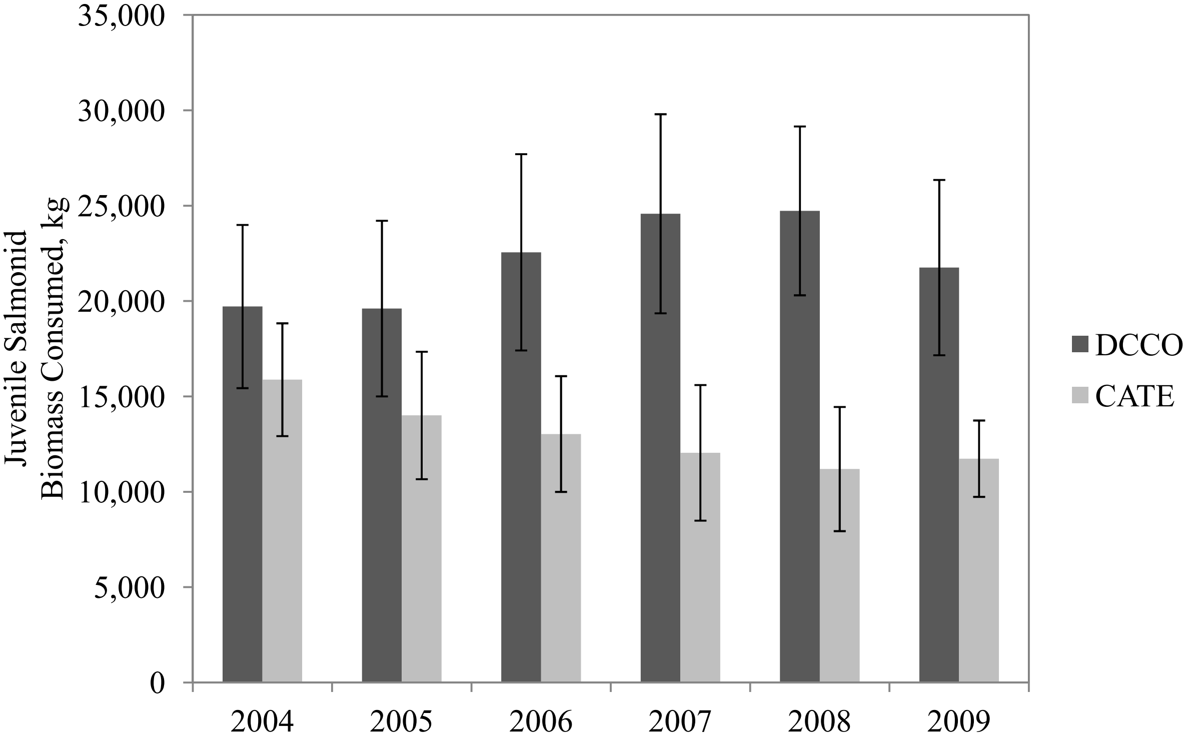

Figure 2.6. Annual biomass of juvenile salmonids consumed by Caspian terns (CATE) nesting at Crescent Island and double-crested cormorants (DCCO) nesting at Foundation Island in the mid-Columbia River during 2004 – 2009. Error bars represent 95% confidence intervals.

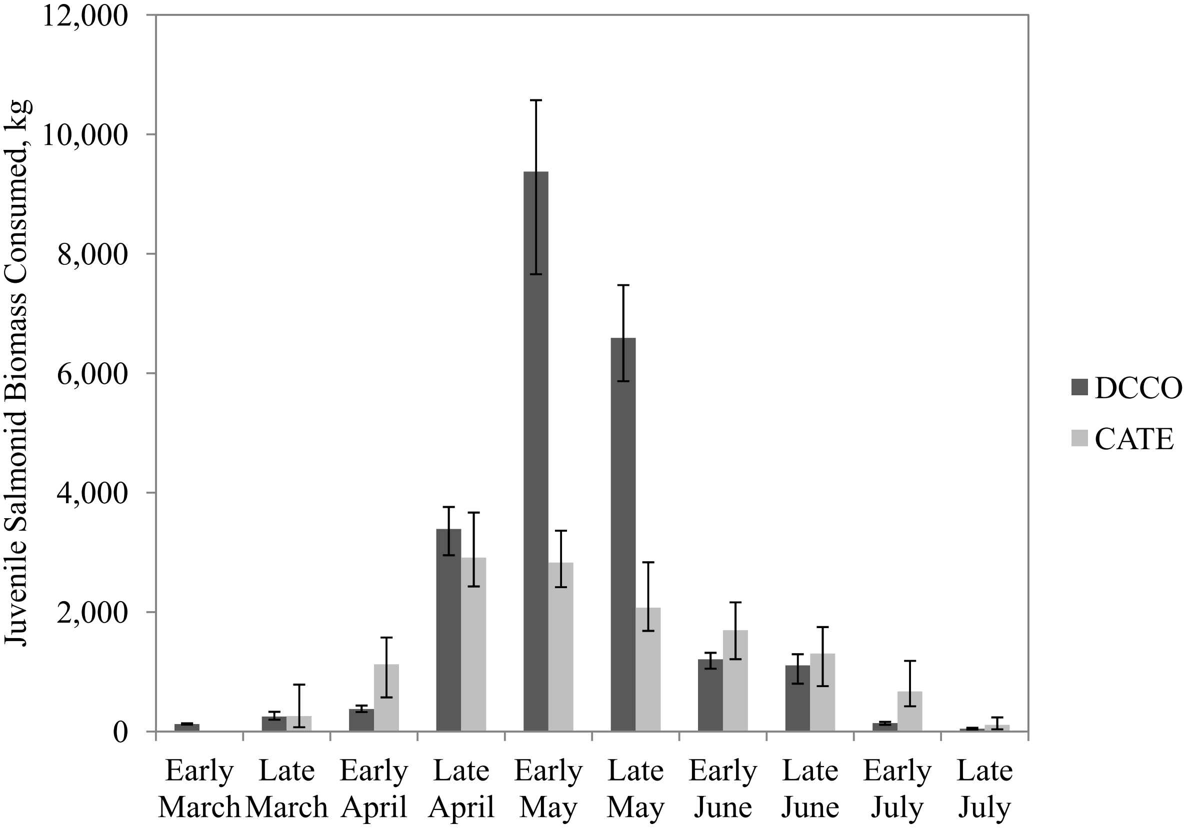

Figure 2.7. Average seasonal biomass of salmonids consumed by Foundation Island double-crested cormorants (DCCO) and Crescent Island Caspian terns (CATE) in the mid-Columbia River during 2004 – 2009. Error bars represent range in annual values for each semi-monthly period.

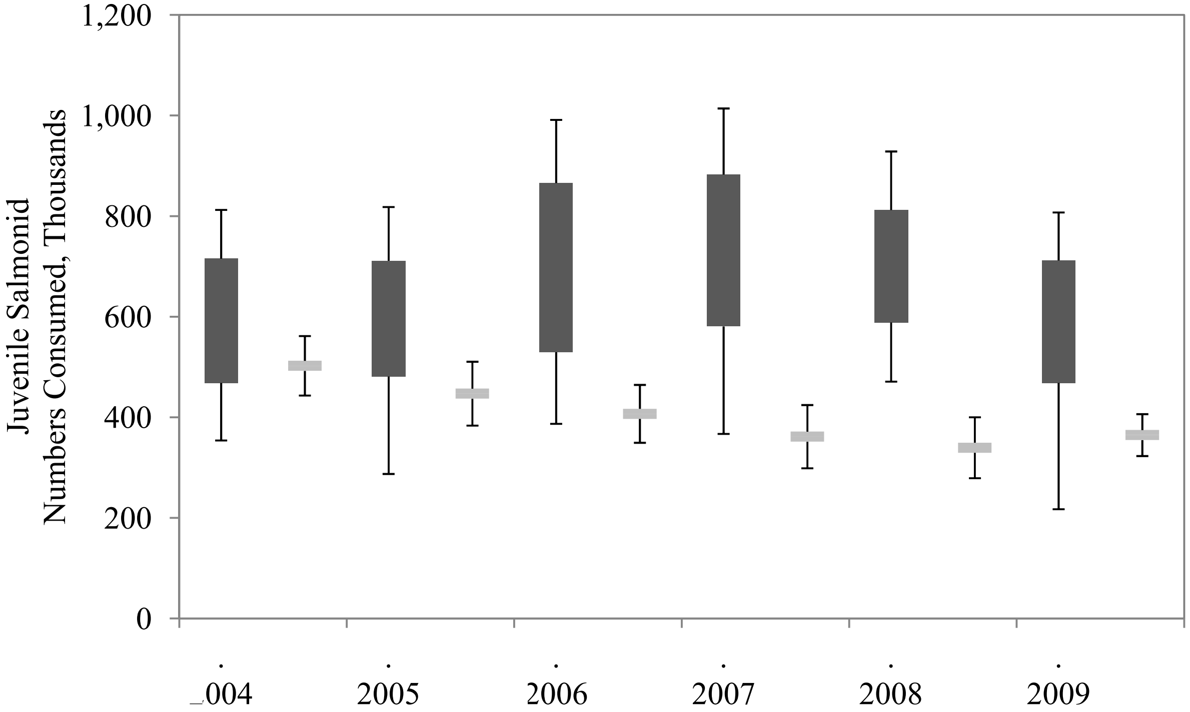

Figure 2.8. Annual numbers of juvenile salmonids consumed by Caspian terns nesting at Crescent Island (light shading) and double-crested cormorants nesting at Foundation Island (dark shading) in the mid-Columbia River during 2004 – 2009. Error bars for Caspian tern consumption represent 95% confidence intervals (CI) for the point (best) estimate. The number of salmonids consumed by cormorants represents two estimates: the first reflecting salmonid composition of the diet based on genetic identification of salmonids taken from cormorant stomachs (bottom of box and lower 95% CI bar) and the second reflecting salmonid composition of the diet based on salmonids taken in proportion to availability (top of box and upper 95% CI bar).

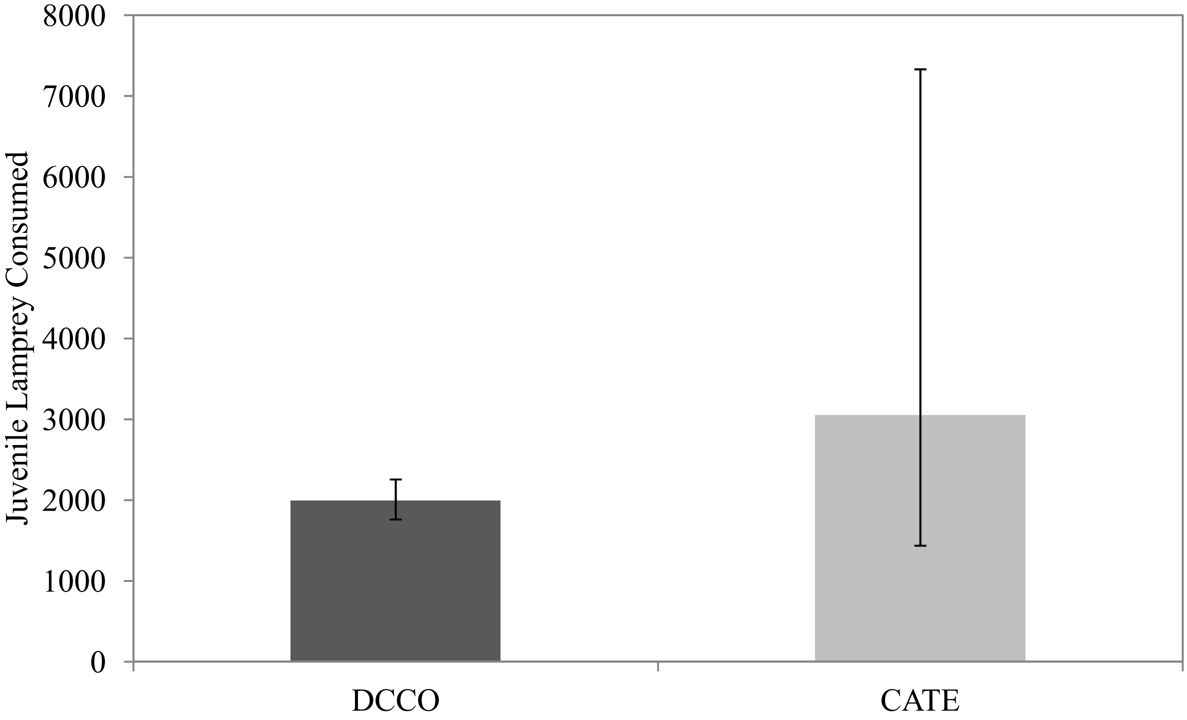

Figure 2.9. Average annual consumption of juvenile lamprey by Crescent Island Caspian terns (CATE) and Foundation Island double-crested cormorants (DCCO) in the mid-Columbia River during 2004-2009. Error bars represent the range in annual total lamprey consumption point (best) estimates.

CHAPTER 3

A SYSTEM-WIDE EVALUATION OF AVIAN PREDATION ON SALMONID SMOLTS IN THE COLUMBIA RIVER BASIN BASED ON RECOVERIES OF PASSIVE INTEGRATED TRANSPONDER (PIT) TAGS

This chapter has been prepared for the U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This chapter summarizes avian predation rate estimates based on data from salmonid PIT tags recovered on bird colonies during 2004-2009.

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SUMMARY

We recovered passive integrated transponder (PIT) tags on nine different piscivorous waterbird breeding colonies in the Columbia River basin to evaluate avian predation on juvenile salmonids (*Oncorhynchus* spp.) during 2004-2009. These nine bird colonies had the highest numbers of smolt PIT tags of any in the Columbia Basin. Bird colonies investigated included: Caspian tern (*Hydroprogne caspia*) colonies on East Sand Island (Columbia River estuary), the Blalock Islands (middle Columbia River), Crescent Island (confluence of the Columbia and Snake rivers), and Goose Island (Potholes Reservoir, WA); double-crested cormorant (*Phalacrocorax auritus*) colonies on East Sand Island and Foundation Island (confluence of the Columbia and Snake rivers); California gull (*Larus californicus*) and ring-billed gull (*L. delawarensis*) colonies on Miller Rocks (middle Columbia River) and Crescent Island; and the American white pelican (*Pelecanus erythrorhynchos*) colony on Badger Island (confluence of the Columbia and Snake rivers). Avian predation rates on smolts were calculated based on (1) the percentage of PIT-tagged smolts detected passing hydroelectric dams that were subsequently recovered on a downstream bird colony and (2) the probability of recovering PIT tags deposited on each bird colony (i.e., detection efficiency). Predation rates were not adjusted for off-colony deposition of PIT tags, resulting in minimum estimates of avian predation. Predation rates specific to salmonid species (Chinook salmon [*O. tshawytscha*], coho salmon [*O. kisutch*], sockeye salmon [*O. nerka*], and steelhead [*O. mykiss*]), river-of-origin (upper Columbia, middle Columbia, Snake, and Willamette), run-type (spring, summer, fall, winter), rear-type (hatchery, wild) and run-timing (per week) were evaluated to determine which salmonid stocks were most affected by avian predation and which bird colonies had the greatest impact on smolt survival during out-migration.

This system-wide evaluation of avian predation indicated that, among the nine piscivorous waterbird colonies investigated, Caspian terns and double-crested cormorants nesting on East Sand Island in the Columbia River estuary were consuming the highest proportions of available PIT-tagged smolts, with combined, minimum losses ranging from 2.6% for Willamette spring Chinook to 18.2% for Snake River summer steelhead during 2004-2009. Minimum predation rate estimates associated with the tern and cormorant colonies in the estuary were generally 2-5 times greater than for inland bird colonies. Due to the relatively high observed predation rates in the estuary, and because all anadromous salmonids must migrate through the estuary, our results indicate that the management of terns and cormorants nesting on East Sand Island near the mouth of the Columbia River has the greatest potential to enhance survival of juvenile salmonids from all Columbia Basin stocks combined.

Impacts to survival of specific stocks of salmonids from avian predation were also associated with some inland colonies of Caspian terns and double-crested cormorants. Minimum estimated predation rates by Crescent Island terns on Snake River summer steelhead (7.7%) and by Goose Island terns on upper Columbia summer steelhead (10.0%) were substantial during the study period. Predation rates on Snake River summer steelhead (2.0%) and Snake River sockeye (1.7%) by cormorants from the inland colony at Foundation Island were also relatively high compared to those of other inland bird colonies. Predation rates on smolts by terns nesting on the Blalock Islands, gulls nesting at colonies on Miller Rocks and Crescent Island, and by pelicans nesting at the colony on Badger Island were minor (generally < 0.5%). Of the gull and pelican colonies examined, predation rates on smolts were highest for the gull colony on Miller Rocks Islands.

Differences in smolt susceptibility to avian predation based on rear-type (hatchery vs. wild) and run-timing of smolts were also observed. Hatchery smolts were often more susceptible to avian predation relative to their wild counterparts, although numerous exceptions were documented during the six-year study. Smolts out-migrating in June and July were often consumed at higher rates by birds than smolts of the same stock that out-migrated early (April) or during the peak out-migration period (May). Trends in avian predation based on run-timing, however, were highly variable and differed based on salmonid stock, year, and bird colony.

Predation rates on PIT-tagged smolts that were adjusted for colony size (i.e., smolt consumption per bird) were substantially higher for terns and cormorants nesting at inland colonies compared to those nesting in the estuary. Total colony-wide predation rates, however, were much higher for the tern and cormorant colonies in the estuary compared to their inland counterparts. While inland colonies of terns and cormorants are much smaller than their counterparts in the estuary, these inland colonies can be more reliant on salmonids as a food source. This greater reliance on salmonids, coupled with lower diversity of available salmonid stocks compared to the estuary, is responsible for the unexpectedly high impact of inland tern and cormorant colonies on specific stocks of salmonids, particularly steelhead. Current management efforts to increase smolt survival through reductions in tern and cormorant predation in the estuary could result in higher predation rates on certain ESA-listed salmonid stocks if terns and cormorants dissuaded from nesting in the estuary recruited to inland colony sites.

Based on the results of this study, the greatest potential for increasing survival of smolts from ESA-listed salmonid stocks by managing inland avian predators would be realized by focusing management efforts on Caspian terns nesting at colonies on Crescent Island and Goose Island. Reductions in the size of these tern colonies would enhance survival of upper Columbia River and Snake River steelhead stocks in particular. Enhancement of smolt survival for Snake River stocks may also be achieved by managing the double-crested cormorant colony at Foundation Island. As inferred from PIT tags recovered on-colony, management of other inland piscivorous waterbird colonies on the Columbia Plateau would provide relatively small increases in stock-specific smolt survival.

INTRODUCTION

Predation on juvenile salmonids (*Oncorhynchus* spp.) during out-migration 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). Studies of avian predation in the Columbia River have focused on colonial waterbirds nesting in the estuary (Collis et al. 2001; Roby et al. 2003; Ryan et al. 2003; Lyons et al. 2010), where the largest known colonies of Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus)* in western North America currently reside (Lyons et al. 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 (ca. 5–17 million smolts annually; Lyons et al. 2010), including ESA-listed salmonid stocks. Breeding colonies of piscivorous colonial waterbirds, however, are not limited to the Columbia River estuary, but are distributed throughout the Columbia River basin. Over 100,000 piscivorous colonial waterbirds, representing five different species at 12 different colonies, were documented nesting at inland sites (upstream of the estuary) during 2004-2009 (see Chapter 1). Published research on the impacts of these inland bird colonies on survival of juvenile salmonids has been limited to the Caspian tern colonies on Crescent Island (Antolos et al. 2005) and on Goose Island (Maranto et al. 2010) on the Columbia Plateau.

Since 1987, passive integrated transponder (PIT) tags have been placed in juvenile salmonids from the Columbia River basin to study smolt behavior and survival during out-migration, and to measure return rates of adults. Smolt PIT tags were first discovered on colonies of piscivorous waterbirds in 1996 (Collis et al. 2001). Beginning in 1998, specially designed electronics (antennas and transceivers) were developed and used to recover PIT tags *in situ* on bird colonies (Ryan et al. 2003). PIT tags provide specific information on each fish, including species, stock, rear-type (hatchery or wild), run-timing, and temporal availability (based on detections of live fish passing hydroelectric dams during out-migration). Recoveries of smolt PIT tags on bird colonies have previously been used to determine smolt predation rates and as a means to measure the relative susceptibility of different salmonid populations to avian predation (Collis et al. 2001; Ryan et al. 2003; Antolos et al. 2005; Maranto et al. 2010). Previous estimates of avian predation based on PIT tag recoveries are consider minimum estimates of predation because not all tags deposited by birds on their nesting colony are subsequently found by researchers (Ryan et al. 2003) and because not all egested tags are deposited on nesting colonies; tags can be deposited off-colony at loafing or other areas used by the birds during the breeding season (Collis et al. 2007).

PIT tags recovered from the large tern and cormorant colonies in the Columbia River estuary revealed that steelhead (*O. mykiss*) were disproportionately consumed compared to other PIT-tagged salmonid species. Depending on the year, from 9% to 16% of the PIT-tagged steelhead smolts detected passing Bonneville Dam were subsequently recovered on the estuary tern and cormorant colonies (Collis et al. 2001; Ryan et al. 2003). PIT tags recovered on-colony also indicated that terns and cormorants nesting on East Sand Island generally preyed on hatchery-raised and wild steelhead in proportion to their availability (Collis et al. 2001; Ryan et al. 2003). Conversely, PIT-tagged Chinook salmon (*O. tshawytscha*) smolts were preyed on at much lower rates than steelhead, and hatchery-raised Chinook smolts were significantly more susceptible to avian predation than their wild counterparts (Collis et al. 2001; Ryan et al. 2003). Perhaps most significant from a management standpoint, juvenile salmonids listed as threatened or endangered under the ESA were equally susceptible to predation by terns and cormorants nesting in the Columbia River estuary as were smolts from unlisted stocks (Collis et al. 2001). Evaluation of similar trends in selection preference and overall impacts of avian predation from birds nesting at inland colonies is, with the few exceptions noted above, currently unknown.

Previous studies of the impacts of avian predation on survival of salmonid smolts from the Columbia Basin have focused on individual nesting colonies (Roby et al. 2003; Antolos et al. 2005; Maranto et al. 2010), as opposed to the cumulative effects of numerous colonies located on or near (i.e., within foraging range of) the Columbia and Snake rivers. Information on smolt losses to avian predation on larger spatial and temporal scales, however, is paramount in order to effectively manage avian predation and maximize potential benefits to ESA-listed salmonid populations from bird management initiatives. Furthermore, due to the ephemeral nature of many of the colony sites used by colonial piscivorous waterbirds and the observed frequency of inter-colony movements by avian predators (Conover et al. 1979; Cuthbert 1988; Quinn and Sirdevan 1998; Wires et al. 2001; Chapter 1), comprehensive and system-wide plans are needed to both investigate and manage avian predation in the Columbia River basin. Resource management agencies and conservation groups working in the Columbia River basin recognize the importance of addressing avian predation in efforts to restore salmonids (USFWS 2005; NOAA 2008). Plans to recover ESA-listed salmonid populations have been developed by the United States government, and specifically call for developing strategies to manage avian predation as a means to bolster in-river smolt survival (NOAA 2008). With the exception of Caspian terns nesting on East Sand Island in the Columbia River estuary (USFWS 2005), however, plans have not specified which bird colonies pose the greatest risks to smolt survival and what the potential benefits of bird management – in terms of increased smolt survival – will be if particular management initiatives are implemented.

The main objectives of this study were to use information collected from PIT-tagged salmonids to (1) determine colony-specific predation rates on smolts by various avian predators located on or near the Columbia River, (2) evaluate whether avian predation rates differ by salmonid species, stock, rear-type (hatchery, wild), or run-timing, (3) assess differences in predation rates based on the location of the bird colony (estuary versus inland), and (4) determine whether per capita predation rates (per bird smolt consumption) differ among bird species and among bird colonies. Objectives 1 and 2 address the paucity of knowledge regarding which stocks of salmonid smolts are most affected by avian predation. Objectives 3 and 4 were intended to help guide management efforts by identifying which bird species and colonies pose the greatest risk to salmonid populations in the region, and where reductions in avian predation would most enhance smolt survival.

METHODS

Study Area

Our study area encompassed breeding colonies of piscivorous waterbirds ranging from the mouth of the Columbia River to the upper Columbia River, a distance of approximately 730 river kilometers (Rkm) (Figure 3.1). Nine different bird colonies were selected for study because previous surveys of bird colonies for smolt PIT tags indicated that these nine had the highest numbers of smolt PIT tags of all colonies in the Basin (Ryan et al. 2003; Antolos et al. 2005; Maranto et al. 2010). This study did not attempt to recover PIT tags from off-colony loafing areas that colonial waterbirds may have visited during the nesting season. PIT tags were also not recovered from non-colonial or semi-colonial piscivorous waterbirds (e.g., mergansers, grebes, herons, and kingfisher) because of the impracticality of recovering tags from dispersed nests and our *a priori* assumption that these non-colonial species have far less potential to significantly impact the survival of salmonid smolts relative to bird species that nest in large colonies along the Columbia River (Wiese et al. 2008).

These nine colonies were scanned for the presence of salmonid PIT tags following dispersal of the birds from their nesting colonies at the end of the breeding season (Figure 3.1). Years in which PIT tags were recovered from bird colonies varied, with most of the nine study colonies scanned in most years during 2004-2009. However, some study colonies were only scanned in the later years of the study (2007-2009) due to the formation of new colonies or the expansion of existing colonies that warranted their inclusion in this study.

The nine bird colonies included in this study were located in the Columbia River estuary and inland along the middle Columbia River (between Bonneville Dam and McNary Dam), near the confluence of the Snake and Columbia rivers, and at an off-river location (Goose Island in Potholes Reservoir, WA; Figure 3.1). The specific breeding colonies scanned for PIT tags included Caspian tern colonies on East Sand Island (Rkm 8; estuary), the Blalock Islands (Rkm 445; middle Columbia), Crescent Island (Rkm 510; confluence), and Goose Island (Potholes Reservoir, WA); double-crested cormorant colonies on East Sand Island and Foundation Island (Rkm 518; confluence); gull colonies (California gulls [*Larus californicus*] and ring-billed gulls [*L. delawarensis*]) on Miller Rocks (Rkm 333; middle Columbia) and Crescent Island; and an American white pelican (*Pelecanus erythrorhynchos)* colony on Badger Island (Rkm 511; confluence) (Figure 3.1). In addition to these nine bird colonies, three additional colonies (double-crested cormorants in northern Potholes Reservoir, gulls on Three Mile Canyon Island in the middle Columbia, and gulls on Island 20 in the confluence) were also intermittently scanned for PIT tags during the study period but were not included in the study due to the paucity of smolt PIT tags detected during scanning.

PIT Tag Detections

PIT tag scanning was conducted at bird colonies using the methods of Ryan et al. (2003), whereby flat-plate and pole-mounted PIT tag antennas were used to recover PIT tags *in situ* after birds dispersed from their breeding colonies (August to November). PIT tags were detected at each colony by systematically scanning the area that was occupied by birds during the nesting season. The entire colony was scanned using PIT tag antennas (referred to as a “pass”). Numerous passes were then conducted until the number of previously undetected PIT tags found during a pass was ≤ 5% of the total number of PIT tags found during all previous passes.

PIT Tag Detection Efficiency

Not all PIT tags egested by birds on their nesting colony are subsequently found by researchers after the nesting season. For example, PIT tags can be blown off the colony during wind storms, washed away during high tides, rain storms, or other flooding events, or otherwise damaged or lost during the course of the nesting season. Furthermore, the detection methods used to find PIT tags are not 100% efficient, with some proportion of detectable tags missed by researchers during the scanning process (Ryan et al. 2003). To address these factors, we calculated PIT tag detection efficiency using methods similar to those of Evans et al. 2011, which measured the detection efficiency of coded wire tags recovered on a bird colony. Briefly, PIT tags with known tag codes were intentionally sown on the colony throughout the nesting season at each bird colony (hereafter referred to as “control tags”). Control tags were the same dimension and length as PIT tags used to mark smolts from the Columbia River basin (12 mm, 134.2 kHz full-duplex). The sowing of control tags was conducted during several discrete stages of the birds’ nesting season: (1) prior to the initiation of egg-laying (March to April), (2) during the egg incubation period (April to May), during the chick-rearing period (May to June), and immediately following the fledging of young (July to August). These periods were selected because they encompassed the time period when breeding birds consumed out-migrating salmonid smolts. The total number of control PIT tags sown varied by colony and year, with sample sizes ranging from a minimum of 100 to a maximum of 1,200 per colony, per year (median = 400 control tags). The number of discrete intervals or time periods when control tags were sown also varied, but was no less than one (at the beginning of the nesting season) and no more than four (median = 4). During each release, control tags were randomly sown throughout the entire area occupied by nesting birds during the breeding season. Priorities for sowing control tags were based on colony size (with larger colonies receiving the most control tags) and our *a priori* expectation of salmonid predation at that colony; tern and cormorant colonies generally received more control tags than gull or pelican colonies (Collis et al. 2002).

Predation Rates

We queried the regional salmonid PIT Tag Information System (PTAGIS), maintained by the Pacific States Marine Fisheries Commission, to acquire data on PIT-tagged smolts released in the Columbia River basin during 2004-2009. PIT-tagged smolts were grouped into stocks, with each stock representing a unique combination of species (Chinook, coho, sockeye, steelhead), run-type (spring, summer, fall, winter), and river-of-origin. River-of-origin was based on each smolt’s capture, tagging, and release location and included (1) the Willamette River, (2) the middle Columbia River (from the confluence with the Snake River downstream to Bonneville Dam), (3) the upper Columbia River (above the confluence with the Snake River), and (4) the Snake River (Figure 3.1). For example, Snake River summer-run steelhead represented a unique stock of fish for analysis.

Availability of PIT-tagged smolts to predation by birds nesting on different colonies was determined by detections of PIT-tagged smolts at the nearest upstream hydroelectric dam with juvenile fish interrogation capabilities. Smolts available to birds nesting at colonies on East Sand Island in the estuary were based on detections of PIT-tagged smolts at Bonneville Dam on the lower Columbia River (Rkm 225) or Sullivan Dam on the lower Willamette River (Rkm 206) (Figure 3.1). For bird colonies on Miller Rocks Islands and the Blalock Islands in the middle Columbia River, smolt availability was determined by detections of PIT-tagged smolts at McNary Dam (Rkm 470) (Figure 3.1). For bird colonies near the confluence of the Snake and Columbia rivers (Crescent, Badger, and Foundation islands), availability was determined by detections of PIT-tagged smolts at Lower Monumental Dam on the Snake River (Rkm 589) and Rock Island Dam on the upper Columbia River (Rkm 730) (Figure 3.1). For the off-river colony on Goose Island, Potholes Reservoir, availability was determined by detections of PIT-tagged smolts at Rock Island Dam because prior research indicated that terns nesting on Goose Island foraged on smolts in the Columbia River downstream of Rock Island Dam (Maranto et al. 2010).

The distance between the hydroelectric dam used to determine smolt availability and the downstream bird colony where PIT tags were recovered varied from a minimum of 25 Rkm (McNary Dam to the Blalock Islands) to a maximum of 220 Rkm (Rock Island Dam to Crescent

Island) (Figure 3.1). For most colonies in this study (N=6), the distance between the dam and the colony was beyond the foraging radius of birds, suggesting that birds rarely consumed smolts above detection locations at dams. Foraging distances reported for nesting Caspian terns, double-crested cormorants, California/ring-billed gulls, and American white pelicans indicate these species can forage up to 82 km (Maranto et al. 2010), 47 km (Anderson et al. 2004), 45 km (Baird 1976; Ryder 1993) and 300 km (Scoppettone et al. 2006) from their nesting colonies, respectively.

Predation rates on PIT-tagged smolts were calculated using a multi-step approach. First, for each salmonid stock, the proportion of PIT-tagged smolts consumed by avian predators on day *j* ($\widehat{q_j}$) was estimated by dividing the number of PIT-tagged smolts detected at a dam on day *j* that were subsequently recovered on a bird colony (recovered$_j$) by the total number of smolts detected at that dam on day *j* (available$_j$) (eq. 1).

$$(1)\ \ \widehat{q_j} = \frac{recovered_j}{available_j}$$

Second, we used logistic regression to estimate colony-specific daily detection efficiencies, whereby a binary response of detections (detected/not detected) was modeled as a function of time since control tags were placed on the bird colony:

$$(2)\ \ \widehat{p_j} = \frac{e^{(\beta_0 + \beta_1 t_j)}}{1 + e^{(\beta_0 + \beta_1 t_j)}}$$

where $\widehat{p_j}$ is the probability of detecting a control tag deposited on day *j*, β_0 is the regression intercept, β_1 is the regression slope, and t_j is the independent variable for deposition date. Next, to calculate colony-specific adjusted daily predation rates ($\widehat{r_j}$), the proportion of available PIT-tagged smolts recovered on a bird colony on day *j* ($\widehat{q_j}$) was corrected for colony-specific detection efficiency on day *j* ($\widehat{p_j}$) (eq. 3).

$$(3)\ \ \widehat{r_j} = \frac{\widehat{q_j}}{\widehat{p_j}}$$

To calculate weekly and seasonal predation rates, daily estimates of the total number of smolts consumed were summed and divided by the total number of smolts available within that same time period. Confidence intervals for predation rates were estimated by a bootstrapping simulation technique (Efron & Tibshirani 1986; Manly 1998). The bootstrapping analysis consisted of 2,000 iterations of the model calculations, with each iteration representing a unique bootstrap resample (random sample with replacement) of the observed detection efficiency and smolt PIT tag datasets. The 2.5th and 97.5th quartiles were used to represent the limits of a bootstrapped 95% confidence interval. Annual or seasonal predation rate estimates and 95% confidence intervals were calculated for each unique stock of PIT-tagged smolts consumed by a bird colony in each year. A multi-year estimate and 95% confidence interval was then generated for each bird colony using all available PIT-tagged smolts during 2004-2009 to evaluate colony-specific impacts on smolt survival during the entire study period. In all instances when a bird colony consumed < 0.1% of a given stock of salmonids, predation rates were noted as < 0.1%, and presented without confidence intervals due to the proximity of the estimate to zero.

To control for imprecise results that might arise from small sample sizes, estimates of predation rates were only calculated for stocks where ≥ 500 PIT-tagged smolts were interrogated while passing an upstream dam during a particular year of the study period. Additionally, only PIT-tagged smolts detected at a dam during the bird nesting season (1 March to 31 August for colonies in the estuary and 1 April to 31 July for inland colonies; see Chapter 1) were included in these analyses, as these smolts were known to be available to birds nesting at the colony. Analyses were conducted using R statistical software, with statistical significance set at $\alpha = 0.05$.

Rear-type and run-timing comparisons

We further investigated variation in stock-specific avian predation rates associated with smolt rear-type (hatchery or wild) and run-timing (by week). Comparisons were only evaluated when predation rates during the overall study period were ≥ 2.0% for any stock at a given bird colony. Comparisons of predation rates between rear-types were only attempted when ≥500 hatchery and ≥500 wild PIT-tagged smolts of the same stock were available in the same year. Comparisons of predation rates by run-timing were only attempted when ≥100 PIT-tagged smolts were available per week during ≥ five separate weeks in one year. These data restrictions enabled direct comparisons between predation rates based on rear-type and run-timing for salmonids when (1) substantial predation was noted on that salmonid stock and (2) adequate sample sizes where available for comparisons.

Colony size-adjusted predation rates

Predation rates adjusted for the differences in size of the breeding colony were generated for each bird colony and year to better address the management-relevant questions of which bird colonies pose the greatest risk to smolt survival during out-migration and how potential future changes in bird colony size would affect overall predation rates. Colony size-adjusted predation rates were calculated by dividing estimates of colony-wide predation rate by the number of adult birds present at each colony, in each year. The numbers of adult birds nesting at each colony in each year are presented in Chapter 1. The resulting estimates of per-bird predation rates were then multiplied by 1,000 adults (500 pairs) to scale up the results to those of a colony. Predation rates for the colony size-adjusted analysis were based on all PIT-tagged smolts (Chinook, coho, sockeye, and steelhead) detected passing the nearest upstream dam with PIT tag interrogation capabilities (i.e., Rock Island, Lower Monumental, McNary, Bonneville, or Sullivan), as a single measure of overall smolt availability to avian predators residing downstream of the dam.

Model assumptions

Results from our multi-step modeling procedure for estimating avian predation rates on PIT-tagged smolts rely on the following assumptions:

A1. Smolt release and detection information obtained from PTAGIS were complete and accurate.
A2. All PIT-tagged smolts detected passing an upstream dam were available to avian predators nesting downstream of that dam.
A3. The detection probability for control PIT tags was equal to that for PIT tags naturally deposited by birds.
A4. Off-colony PIT tag deposition rates (tags regurgitated or defecated by birds somewhere other than on their nesting colony) did not differ by bird species, colony, or year.

A5. PIT tags from consumed smolts were deposited on a bird colony the same day as they were detected passing the upstream dam.

To verify the first assumption (A1), irregular entries were either validated by specific coordinators or eliminated from the analysis. Assumption A2 concerns measurements of smolt availability. Detections of PIT-tagged smolts at dams upstream of bird colonies were deemed the most appropriate measure of smolt availability given the downstream movement of smolts, the ability to standardize data across all sites, and the ability to define unique groups of smolts by a known location and passage date. Detection efficiency estimates (A3) were generally high at all colonies (see Results); thus, possible violations of assumption A3 would have little effect on estimates of predation rates. Variation among bird species and colonies in the proportion of consumed PIT tags that are deposited off-colony (A4) could result in differences in the minimum predation rate estimates presented herein. At this time, however, off-colony deposition data are limited to Caspian terns on Crescent Island (Collis et al. 2007; see Discussion), so data to support or refute assumption A4 (i.e., relative differences in off-colony deposition rates by avian species, colony, and year) are not available. Assumption A5 relates to the use of the last date of live detection as a proxy for the date a PIT tag was deposited on a bird colony and needed only to be roughly true because detection efficiency did not change dramatically on a daily basis (see Results).

RESULTS

A total of approximately 13.5 million juvenile salmonids (Chinook, sockeye, coho, and steelhead combined) were PIT-tagged and released in the Columbia River basin during the study period. Of these, 1.7 million (12.6%) were detected during out-migration at one or more of the five dams used in this study to determine smolt availability to avian predators (Tables 3.1-3.3). The highest proportion of detected PIT-tagged smolts were detected passing McNary Dam (n = 740,431; Table 3.2) on the middle Columbia River, followed by Lower Monumental Dam (n = 440,001; Table 3.3) on the lower Snake River, and Bonneville Dam (n = 293,695; Table 3.1) on the lower Columbia River. The smallest numbers of PIT-tagged smolts were detected at Rock Island Dam (n = 181,898; Table 3.3) on the upper Columbia River and Sullivan Dam (n = 14,981; Table 3.1) on the lower Willamette River. A total of 18 unique salmonid stocks (based on a combination of species, river-of-origin, and run-type) were detected in sufficient numbers during out-migration for inclusion in this study (Tables 3.1-3.3). A lack of adequate in-river detection sites in the estuary precluded analysis of PIT-tagged salmonids originating from tributaries below Bonneville and Sullivan dams (i.e., lower Columbia River stocks).

Run-timing of PIT-tagged smolts nearly completely overlapped with the nesting seasons of the avian predators studied here, with 99.5% of all PIT-tagged smolts detected passing dams during the birds nesting season (see also Appendices C.1-C.35).

A total of 441,752 salmonid PIT tags (3.3% of the 13.5 million released) were recovered on the nine bird colonies during the study period (Table 3.4). Although numbers of PIT tags recovered were highly variable by colony and year, the largest numbers of smolt PIT tags were found on the East Sand Island tern colony ($\bar{\chi}$ = 33,692 PIT tags per year) and the East Sand Island cormorant colony ($\bar{\chi}$ = 17,553 PIT tags per year) (Table 3.4). The smallest numbers of PIT tags were found on the Crescent Island gull colony ($\bar{\chi}$ = 1,563 PIT tags per year), the

Blalock Islands tern colony ($\bar{\chi}$ = 1,267 PIT tags per year), and the Badger Island pelican colony ($\bar{\chi}$ = 1,143 PIT tags per year) (Table 3.4).

PIT Tag Detection Efficiency

Detection efficiency of control PIT tags on bird colonies – those intentionally sown by researchers during the bird nesting season – was unique to each bird colony and year. In general, detection efficiencies were high across the study colonies and years of the study period, with detection efficiency > 60% for 33 of the 42 (79%) yearly colony-specific detection efficiency estimates measured (Table 3.5). Detection efficiency estimates ranged from a low of 36.0% at the East Sand Island cormorant colony in 2004 to a high of 93.0% at the Blalock Islands tern colony in 2008 (Table 3.5). Within-season temporal differences in detection efficiency were also observed at some colonies, but varied by colony and year (Table 3.6). Logistic regression results indicated that, depending on the colony or year, estimated detection efficiency could increase, decrease, or remain stable through the nesting season (Table 3.6). The most common temporal trend was increasing, and this relationship was detected at the East Sand Island cormorant colony, the Crescent Island tern colony, the Crescent Island gull colony, and the Goose Island tern colony (Table 3.6).

Predation Rates by Bird Colony

Results indicated that avian predation on salmonid stocks varied by bird colony, salmonid stock, rear-type, and migration timing. The highest predation rates were often associated with steelhead, followed by coho, Chinook, and sockeye salmon (Figure 3.2-3.5). The highest smolt predation rates were associated with colonies of Caspian terns and double-crested cormorants, while smolt predation rates associated with gull and American white pelican colonies were relatively minor in comparison (Figures 3.2-3.5).

Predation rates over the entire study period indicated that, for the majority of salmonid stocks, avian predation was highest for estuary colonies, followed by confluence colonies, the off-river colony, and, finally, middle Columbia colonies (Figure 3.2-3.5). Estuary colonies (tern and cormorant colonies on East Sand Island) consumed between 2.6% – 18.2% (depending on stock) of the available PIT-tagged smolts detected passing Bonneville or Sullivan dams during the study period (Figure 3.2). Predation rates by middle Columbia colonies (gulls on the Miller Rocks Islands and terns on the Blalock Islands) were < 2.0% for any salmonid stock (Figure 3.3). Predation rates during the study period indicated that, in general, ~ 1% – 10% of PIT-tagged salmonid stocks detected passing Lower Monumental or Rock Island dams were consumed by avian predators nesting on islands near the confluence of the Snake and Columbia rivers, but again, predation rates varied by salmonid stock. For instance, the predation rate on Snake River summer steelhead by birds nesting at all the confluence colonies was 10.3%, but predation rates on stocks of sockeye and Chinook salmon were substantially lower (~1 – 4%; Figure 3.4). The only off-river bird colony evaluated, the Caspian tern colony on Goose Island, Potholes Reservoir, primarily affected survival of upper Columbia salmonid stocks, with study period (2006-2009) predation rates ranging from a high of 10.0% for upper Columbia summer steelhead to < 0.1% for any Snake River salmonid stock (Figure 3.5).

Estuary bird colonies-Evaluation of avian predation on salmonid stocks in the estuary was focused on two bird colonies: the East Sand Island double-crested cormorant colony and

East Sand Island Caspian tern colony (Figure 3.2). Tern predation rates were significantly higher for all stocks of steelhead (9.9 – 11.5%) and coho salmon (5.7%) compared to cormorant predation rates on the same stocks of steelhead (3.3 – 6.8%) and coho salmon (1.5%; Figure 3.2). In general, colony-specific predation rates on Chinook salmon were similar between the tern and cormorant colonies, as both colonies consumed ~ 1% – 3% of available Chinook stocks. Avian predation on middle Columbia fall Chinook was a notable exception to this generalization; the predation rate on this stock by cormorants nesting on East Sand Island (7.7%; 95% c.i.: 6.9 – 8.6%) was significantly higher than the predation rate by terns nesting on East Sand Island (1.2%; 95% c.i.: 0.9 – 1.5%) (Figure 3.2). Cormorant predation rates in the estuary on both upper Columbia and Snake River sockeye (2.5 – 2.7%) were significantly higher than tern predation rates on the same stocks (0.8% for both) (Figure 3.2).

There was significant inter-annual variation in the predation rates of East Sand Island cormorants and terns on PIT-tagged smolts in the estuary (Appendices A.1- A.2). During the study period (2004-2009), predation rates by East Sand Island cormorants were lowest in 2007, and highest in 2008 and 2009 (Appendix A.1). Predation rates by East Sand Island terns were also highly variable by study year but consistent inter-annual trends between stocks were not observed (see Appendix A.2).

Inland bird colonies- Evaluation of predation on smolts by birds nesting at inland colonies focused on seven bird colonies in three different areas: the Miller Rocks Islands gull colony and the Blalock Islands tern colony in the middle Columbia River (Figure 3.3); the Crescent Island tern colony, the Crescent Island gull colony, the Badger Island pelican colony, and the Foundation Island cormorant colony near the confluence of the Snake and Columbia rivers (Figure 3.4); and the Goose Island tern colony in Potholes Reservoir (Figure 3.5). Of these seven colonies, the Crescent Island tern colony was frequently associated with the highest smolt predation rates (0.4 – 7.4%, depending on species and stock) (Figure 3.4). Predation rates associated with the Goose Island tern colony were also substantial, but were primarily limited to upper Columbia salmonid stocks (0.2 – 10.0%, depending on species), while predation rates on all Snake River salmonid stocks were < 0.1% (Figure 3.5). Predation rates associated with the Foundation Island cormorant colony were also elevated relative to other inland colonies, but were less than those of the nearby tern colony at Crescent Island. Predation rates by Foundation Island cormorants were highest for Snake River stocks (0.3 – 2.0%, depending on species), and much lower for upper Columbia stocks (< 0.1% for all species). Predation rates associated with the remaining inland bird colonies (Miller Rocks Islands gull colony, Blalock Islands tern colony, Crescent Island gull colony, and Badger Island pelican colony) were generally minor (< 0.5%), regardless of salmonid species or stock (Figure 3.3-3.4).

Inter-annual trends in smolt predation rates were observed for some, but not all, of the inland bird colonies evaluated (see Appendices A.3-A.9). An inter-annual trend of declining salmonid predation rates was evident for the Crescent Island tern colony, with predation rates significantly higher during 2004-2005 compared to 2006-2009 (Appendix A.4). An inter-annual trend in predation rates was less evident for the Foundation Island cormorant colony, although a general increase in predation rates was observed across the study period. Inter-annual trends in predation rates for the gull and pelican colonies were not discernable, as predation rates were low in all years.

Predation Rates by Salmonid Species and Stock

Avian predation rates varied by salmonid stock, with differences noted among species, river-of-origin, and run-type (Figures 3.2-3.5). In general, avian predation rates were highest on steelhead smolts, compared to other salmonid species. Cumulative impacts of avian predation on salmonid stocks originating upstream from the confluence of Snake and Columbia rivers were often greater than on analogous stocks originating lower in the basin (e.g., middle Columbia or Willamette stocks), as upstream stocks were susceptible to predation by birds from several inland colonies that did not depredate lower river stocks (Figures 3.2-3.5). Run-type (i.e., spring, summer, fall, or winter) within a stock generally did not have as large an effect on avian predation rates as species or river-of-origin (Figures 3.2-3.5); however, there were notable exceptions (see below).

Steelhead trout- Upper Columbia summer steelhead and Snake River summer steelhead often suffered the greatest cumulative impacts from avian predation during their out-migration. Predation rates on upper Columbia summer steelhead by estuary colonies (14.7%) and middle Columbia colonies (1.5%) were similar to those on other summer steelhead stocks (e.g., middle Columbia summer steelhead; Figures 3.2 and 3.3). But upper Columbia summer steelhead were also susceptible to predation by terns nesting at Goose Island, Potholes Reservoir (off-river), and predation rates on this steelhead stock by terns from this colony were substantial (10.0%; 95% c.i.: 9.1 – 11.1%; Figure 3.5; Appendix A.9). Thus, the cumulative impact of avian predation on this threatened steelhead stock was quite high. Similarly, cumulative impacts of avian predation on Snake summer steelhead were greater than on middle or Willamette river steelhead stocks. Predation rates on Snake summer steelhead by birds in the estuary (18.2%) and by birds on the middle Columbia (1.9%) were similar to other steelhead stocks (Figures 3.2 and 3.3), but Snake summer steelhead suffered higher predation rates from birds nesting near the confluence of the Snake and Columbia rivers (10.3%) compared to other steelhead stocks (< 4.0%; Figure 3.4). Middle Columbia summer steelhead, middle Columbia winter steelhead, and Willamette summer steelhead suffered similar predation rates from birds nesting at estuary colonies as analogous steelhead stocks from further up-river (Figure 3.2). These results indicate that, in general, > 12.0% of all steelhead detected at Bonneville and Sullivan dams were consumed by birds in the estuary, regardless of river-of-origin.

Significant inter-annual variation in stock-specific steelhead predation rates was observed (Appendices A.1-A.9). For instance, in 2004, 22.2% (95% c.i.: 21.2 – 23.2%) of available Snake River summer steelhead were consumed by Crescent Island terns. In 2005, however, the predation rate by Crescent Island terns on Snake River summer steelhead was 10.0% (95% c.i.: 9.4 – 10.7%), and was < 7.0% in all subsequent years of the study (2006-2009; Appendix A.5). Inter-annual variation in avian predation was also observed for upper Columbia summer steelhead. In 2009, the predation rate by Goose Island terns on upper Columbia summer steelhead was 15.7% (95% c.i.: 13.6 – 18.2%), which was significantly higher than in all other years of the study (< 10.0%; Appendix A.9). Predation rates by East Sand Island terns were relatively similar for all steelhead stocks during those study years when multiple stocks were evaluated (8.6 – 16.0% during 2007-2009; Appendix A.2). Stock-specific predation rates by East Sand Island cormorants were highest on Snake summer steelhead (7.3 – 8.1%) and middle Columbia summer steelhead (6.5 – 6.6%) during 2008 and 2009 (Appendix A.1). Inadequate sample sizes of PIT-tagged Willamette and middle Columbia steelhead stocks during 2004-2006 precluded stock-specific comparisons of avian predation rates in the estuary during those years.

Chinook salmon- Stock-specific Chinook predation rates associated with the two estuary bird colonies were about 3.0 – 4.5% for all stocks examined, with only two exceptions; upper Columbia summer Chinook and middle Columbia fall Chinook (Figure 3.2). Predation rates on upper Columbia summer Chinook were 6.3%, mostly due to the relatively high predation rates on this stock by East Sand Island terns (4.0%; 95% c.i.: 3.7 – 4.4%; Figure 3.2, Appendix A.2). The higher predation rates on middle Columbia fall Chinook, however, were only associated with cormorant predation, as predation rates by terns were similar to other Chinook stocks (Figure 3.2). Predation rates by East Sand Island cormorants were significantly higher for middle Columbia fall Chinook (7.7%; 95% c.i.: 6.9 – 8.6%) compared to middle Columbia spring Chinook (2.0%; 95% c.i: 1.8 – 2.3%) or Snake fall Chinook (1.6%; 95% c.i.: 1.5 – 1.8%; Figure 3.2).

Predation rates on Chinook stocks associated with middle Columbia bird colonies were relatively minor, with predation rates < 1.0% for all stocks examined. Predation rates associated with bird colonies near the confluence were higher relative to those in the middle Columbia, with differences primarily associated with river-of-origin (Figure 3.4). Cumulative predation rates by colonies near the confluence indicated that predation rates were higher for Snake Chinook stocks (~ 2%) compared to upper Columbia Chinook stocks (< 1.0%; Figure 3.4). Lower predation rates on upper Columbia Chinook stocks were observed at both the Crescent Island tern colony and the Foundation Island cormorant colony (Figure 3.4; Appendices A.5 and A.8). Stock-specific Chinook predation rates by the Goose Island tern colony (off-river), however, indicated that predation rates varied by Chinook stock. For instance, Goose Island tern predation on upper Columbia spring Chinook (3.6%; 95% c.i.: 1.6 – 6.1) was significantly higher than all other Chinook stocks available to this colony (< 0.5% for all other stocks; Figure 3.5). It should be noted, however, that predation on upper Columbia spring and summer Chinook stocks were only evaluated in 2004 and 2009, the only two years when adequate sample sizes of PIT-tagged Chinook smolts from these stocks were detected at Rock Island Dam (Table 3.3).

General trends in stock-specific Chinook predation rates are presented in Figures 3.2-3.5; most colony-specific predation rates on Chinook stocks remained relatively constant across the study period. However, significant inter-annual variation in predation rates was detected at some bird colonies (Appendix A.1-A.9). Additionally, evaluation of differences in annual predation rates was not possible for some stocks due to inadequate detections of PIT-tagged Chinook smolts at various dams (e.g., Willamette stocks at Sullivan Dam or upper Columbia stocks at Rock Island Dam; Tables 3.1-3.3). Overall, results indicated that avian predation on Chinook stocks varied by stock, avian predator species, colony location, and year, with detailed results for each of these factors presented in Appendices A.1-A.9.

Coho salmon- Only two stocks of coho were available for analysis, upper Columbia coho and Snake coho (Table 3.1-3.3). Comparisons between these two stocks were difficult, however, as adequate detections at dams of Snake coho and upper Columbia coho were often not available in the same years (Tables 3.1-3.3); sufficient Snake coho were detected in 2004 and 2005, while sufficient upper Columbia coho were detected in 2006, 2008, and 2009 (Table 3.3). Across all years, results suggested that predation rates by birds nesting at the colonies near the confluence were stock-specific, with Snake coho (7.0%) consumed at a much higher rate than upper Columbia coho (1.4%; Figure 3.4).

Predation rates on upper Columbia coho associated with tern colonies were generally greater than those associated with cormorant, gull, or pelican colonies. Predation rates on coho were generally highest in the estuary (7.2%), followed by bird colonies near the confluence (1.4 – 7.0%, depending on stock), the off-river colony at Goose Island (2.4%), and finally the middle Columbia colonies (1.3%; Figures 3.2-3.5). Predation rates on coho stocks also displayed significant inter-annual variation. For instance, predation rates on upper Columbia coho by East Sand Island terns were significantly higher in 2004 (11.9%, 95% c.i.: 9.9 – 14.0%) compared to all other years (≤ 7.0% in any given year; Appendix A.2). An insufficient number of coho detections at Bonneville Dam, however, prevented stock-specific comparisons of predation rates associated with the estuary bird colonies between upper Columbia coho and other coho stocks (e.g., Snake or middle Columbia).

Sockeye salmon- PIT-tagged sockeye stocks evaluated by this study included stocks originating from the Snake and upper Columbia rivers. Avian predation rates on both Snake and upper Columbia sockeye were greatest in the estuary (3.4 – 3.6%), followed by the bird colonies near the confluence (0.4 – 3.1%), the bird colonies in the middle Columbia (0.4 – 1.3%), and finally the off-river colony at Goose Island (< 0.5%; Figures 3.2-3.5). In general, predation rates on Snake sockeye were higher than predation rates on upper Columbia sockeye for bird colonies near the confluence, as well as those in middle Columbia (Figures 3.3 and 4.3). Avian predation rates on these two sockeye stocks in the estuary (3.4 – 3.6%), however, were similar (Figure 3.2).

Inter-annual variation in avian predation rates on Snake sockeye could not be evaluated because in only one year (2009) were the numbers of detections at McNary and Bonneville dams sufficient to support estimation (Tables 3.1 and 3.2, Appendices A.1 – A.4). Annual upper Columbia sockeye predation rates were < 1.0% at all inland bird colonies (middle Columbia, confluence, off-river; Appendices A.1-A.9), but were as high as 3.5% (95% c.i.: 2.2 – 5.0%) at the East Sand Island cormorant colony. Inter-annual variation in upper Columbia sockeye predation rates were observed at both estuary colonies; predation rates by East Sand Island cormorants increased from a low of 1.5% in 2007 (95% c.i.: 0.4 – 2.7%) to a high of 3.5% in 2009 (95% c.i.: 2.2 – 5.0%; Appendix A.1), while during the same time period, predation rates by East Sand Island terns decreased from a high of 1.3% in 2007 (95% c.i.: 0.5 – 2.1%) to a low of 0.5% in 2009 (95% c.i.: 0.1 – 1.1%; Appendix A.2).

Comparisons by Rear-type

Predation rate comparisons based on rear-type were limited to tern and cormorant colonies because predation rates on PIT-tagged smolts by gull and pelican colonies were too small (< 2.0%) for inclusion in the analysis. Results from tern and cormorant colonies indicated that hatchery-raised smolts suffered significantly higher predation rates compared to their wild counterparts in 9 of the 16 stock-specific comparisons, where sample sizes of both hatchery and wild PIT-tagged smolts were sufficient (Table 3.7). Of the remaining 7 comparisons, 6 found no differences in predation rates between rear-types, and 1 comparison indicated that wild stocks suffered higher predation rates than their hatchery counterparts (Table 3.7).

Terns consistently consumed hatchery stocks at a higher rate than wild stocks. In the 9 comparisons involving tern predation, 7 of the comparisons indicated that hatchery stocks suffered significantly higher predation rates than wild stocks (Table 3.7). Predation rates on Snake summer steelhead by East Sand Island terns was the lone exception, as predation rates on wild Snake summer steelhead during the study-period (13.8%; 95% c.i.: 12.9 – 14.7%) were significantly higher than predation rates on their hatchery counterparts (10.6%; 95% c.i.: 10.3 – 11.0%). No statistical difference between hatchery and wild predation rates on Snake summer steelhead by Crescent Island terns were found (Table 3.7).

Predation rates on hatchery and wild fish of the same stock by cormorant colonies were more similar relative to comparisons from tern colonies (Table 3.7). For instance, of the 7 hatchery/wild comparisons involving cormorant predation, in 5 there was no significant difference between predation rates on hatchery and wild stocks, while in the remaining 2 comparisons predation rates were higher on hatchery fish compared to wild fish (Table 3.7).

Differences in avian predation rates associated with smolt rear-type exhibited inter-annual variation, although the general trend of higher predation rates on hatchery stocks was evident at most colonies in most years. Although differences in predation rates between rear-types were often statistically significant when pooled across the study years, within-year differences were more variable and were often not statistically significant. The lack of statistical significance within a year was often attributable to small sample sizes of wild fish. Results of predation rate comparisons by rear-type, colony, and year are presented in Appendices B.1-B.6.

Comparisons by Run-timing

Differences in avian predation rates based the run-timing of smolts were specific to each migration year. Some consistency across multiple stocks and years, however, was observed in the data. This consistency suggested that migration-timing may influence predation rates on steelhead stocks more than Chinook stocks, and that weekly predation rates varied more by bird species (i.e., tern vs. cormorant) than by colony location (i.e., inland or estuary). For instance, tern predation rates on steelhead migrating late in the season (i.e., June-July) were often significantly higher than those of steelhead migrating earlier in the season (April-May) for both the tern colony in the estuary and the tern colony near the confluence (Appendices C.18-C.22, C.23, C.24, and C. 26). Cormorant predation rates on steelhead, however, varied by year, and there was no consistent trend in predation rates as a function of migration-timing in some years and increased predation rates during the peak of out-migration (May) in others (Appendices C.7-C.11, and C.25). In the estuary, predation rates on steelhead by the tern and cormorant colonies combined were sometimes consistent throughout the year, even though weekly predation rates by terns or cormorants varied. This was due to increased predation rates by cormorants during the same time period (May) as predation rates by terns declined (see stock specific examples in appendices C.7 and C.18. or C.9 and C.20). This pattern in the estuary of higher cormorant predation rates during periods of lower tern predation rates was not, however, observed across all steelhead stocks or years (Appendices C.7-C.11 and C.18-C.22).

General trends in avian predation rates on Chinook stocks as a function of run-timing were not evident for either terns or cormorants; however, predation rates did vary among weeks in some years (Appendices C.1-C.6 and C.12-C.16). Overall, our results suggested that run-timing had no consistent affect on avian predation rates of Chinook smolts.

The highest avian predation rates recorded in this study were usually associated with tern predation on steelhead stocks migrating late in the season (June-July). In several instances, Caspian tern predation rates on late-season steelhead were > 15.0%, and even as high as 34.0%, regardless of steelhead stock (Appendices C.18-C.22, C.23, C.24, and C.26). Conversely, the lowest predation rates on steelhead by terns usually occurred during the peak of steelhead out-migration in May; indicating that when the numbers of in-river steelhead were highest, predation rates by terns were lowest (Appendices C.18-C.22, C.23, C.24, and C. 26). Weekly tern predation rates on Chinook smolts varied from < 0.1% to 12.3%, but did not display any within season trends that were consistent across stocks or years (Appendices C.12-C.16). Similarly, weekly cormorant predation rates on steelhead varied from < 0.1% to 14.6%, but temporal trends that were consistent across years and stocks were not evident (Appendices C.7-C.11 and C.25). In some years, predation on steelhead stocks by cormorants nesting East Sand Island increased during the peak of the steelhead out-migration (Appendices C.7-C.11), indicating that cormorants were consuming a larger proportion of available steelhead during weeks when more steelhead were present in-river. Increased cormorant predation rates during periods of increased steelhead abundance were not observed in all years or for all stocks (Appendices C.7-C.11). Weekly cormorant predation rates on Chinook smolts ranged from < 0.1% to 17.1%, but this variation was more associated with Chinook stocks than with migration-timing (Appendices C.1-C.6).

Upper Columbia coho was the only coho stock that met the predation rate and sample size criteria for evaluating the effects of run-timing. Our results indicated that, in the estuary, weekly tern predation rates on upper Columbia coho varied from ~1% to 11%, but did not display within season trends that were consistent across years (Appendix C.17). Upper Columbia and Snake sockeye interrogations at Bonneville Dam generally occurred only during a 2-4 week period, and therefore were not evaluated by this study, although predation rates on sockeye stocks by East Sand Island double-crested cormorants were > 2.0%.

Colony Size-adjusted Predation Rates

Once predation rates were adjusted for differences in colony size, colony size-adjusted predation rates (per bird predation rates) were highest for Crescent Island terns, Foundation Island cormorants, and Goose Island terns, all inland, not estuary, colonies. Annual smolt predation rates adjusted for colony size were an order of magnitude greater for Crescent Island terns compared to East Sand Island terns (Figure 3.6). Colony size-adjusted predation rates on smolts by gull and pelicans colonies, regardless of river reach or colony, remained low (< 0.1%).

DISCUSSION

This study is the first to document system-wide stock-specific predation rates on juvenile salmonids by multiple bird species nesting at different colonies in various reaches or segments of the same river system. In addition, we evaluated predation rates on salmonids associated with five previously unstudied piscivorous waterbird colonies (California and ring-billed gulls nesting at colonies on Miller Rocks Islands and Crescent Island, double-crested cormorants nesting on Foundation Island, Caspian terns nesting on the Blalock Islands, and American white pelicans nesting on Badger Island). Finally, we compared predation rates by birds nesting at these colonies to updated predation rates from four piscivorous waterbird colonies previously evaluated in the published literature: the Caspian tern and double-crested cormorant colonies on East Sand Island (Collis et al. 2001; Ryan et al. 2003), the Caspian tern colony on Crescent Island (Antolos et al. 2005), and the Caspian tern colony on Goose Island, Potholes Reservoir (Maranto et al. 2010).

PIT Tag Detections and Detection Efficiency

The largest numbers of smolt PIT tags were recovered on the East Sand Island tern colony, followed by the East Sand Island cormorant colony, the Crescent Island tern colony, and the Foundation Island cormorant colony. Recoveries of PIT tags on the remaining five bird colonies, all located upstream of Bonneville Dam (i.e., inland), were substantially less (< 15% of all recovered PIT tags). Differences in the number of PIT tags recovered, however, does not equate to differences in predation rates among colonies, as the proportion of tagged smolts available and PIT tag detection efficiency varied by both colony and year.

Variation in detection efficiency estimates were likely due to the unique characteristics of each island, including differences in substrate type (sand, rock, or soil-based nesting substrate) and weather conditions. The loss of PIT tags during the breeding season (e.g., to windstorms, rainstorms, or others factors) and the missed detections of functional PIT tags have not been incorporated into previously published studies (Collis et al. 2001; Ryan et al. 2003; Antolos et al. 2005; Maranto et al. 2010). Evans et al. (2011) documented substantial loss of salmonid coded wire tags on a Caspian tern colony in San Francisco Bay, CA. Similarly, our results demonstrate that if unaccounted for, predation rates not adjusted for detection efficiency can grossly underestimate smolt losses to avian predators. For example, predation rates by Caspian terns nesting on Crescent and Goose islands were highly influenced by within season differences in detection efficiency and relatively low detection efficiency estimates. Salmonid smolt out-migration occurs across several months; therefore, data on how detection efficiency of PIT tags deposited on bird colonies varies during these same time periods is required to make comparisons across stocks and years.

Even after adjustments for detection efficiency, an unknown number of smolt PIT tags consumed by birds are not accounted for due to off-colony deposition of PIT tags). An investigation of off-colony PIT deposition by Collis et al. (2007) estimated that between 24% and 37% of egested PIT tags by Caspian terns nesting on Crescent Island during 2004-2006 were deposited off-colony (somewhere other than the area occupied by nesting adults). Results were based on the percentage of PIT-tagged juvenile salmonids consumed by terns from net pens (n=265) and the number fed to captured terns (n=117) that were not recovered by researchers on Crescent Island following the nesting season (see Collis et al. 2007). Whether similar off-colony deposition rates exist in cormorant, gull, and pelican colonies is a critical unknown. Based on the results of Collis et al. (2007), however, predation rates by Crescent Island Caspian terns underestimate smolt losses by roughly 1/3rd. For example, using the off-colony PIT tag deposition rate of 0.37, predation rates on upper Columbia and Snake River steelhead – the two most impacted ESA-listed stocks identified herein – by Crescent Island Caspian terns increase from 2.3% to 3.7% for upper Columbia steelhead and from 7.4% to 11.7% for Snake River steelhead. Further study is needed to quantify off-colony PIT deposition rates in other bird species (gulls, cormorants, and pelicans) and colonies. Additionally, predation rate estimates associated with the Foundation Island cormorant colony may be further reduced relative to the other eight bird colonies evaluate during the study because Foundation Island cormorants are arboreal nesters and it's unknown what proportion of deposited tags remained in cormorant nests relative to the ground below them (where tags were detected by researchers). Thus, although adjustments for detection efficiency increase the accuracy of predation rate estimates, predation rates based on PIT tag recoveries are still considered minimum estimates of predation rates on PIT-tagged smolts.

Avian Predation Rates

Our results demonstrate that avian predation rates on salmonid smolts varied considerably by bird colony, salmonid species, stock, rear-type, and run-timing. Of the 18 unique salmonid stocks evaluated, 14 stocks (78%) suffered reach-specific predation rates in excess of 2.0% from at least one bird colony in the Columbia River basin. In general, the highest avian predation rates were on steelhead smolts, followed by coho, Chinook, and sockeye smolts. The salmonid stocks experiencing the highest proportional losses to avian predation were Snake River summer steelhead (predation rates of 1.9 to 18.2%, depending on reach) and upper Columbia summer steelhead (predation rates of 1.5 to 14.7%, depending on reach). In certain years, reach-specific avian predation rates on summer steelhead were in excess of 20% of available smolts. Information on avian predation rates on coho smolts was limited because many stocks could not be analyzed due to sample size limitations (e.g., Snake River and Middle Columbia stocks) or a lack of in-river PIT interrogation following release (i.e., Lower Columbia River stocks). When adequate sample sizes existed, predation rates on coho smolts were relatively high (1.3 to 7.1%, depending on reach) compared to other salmonid species. Of the various Chinook stocks evaluated, reach-specific avian predation rates were highest on middle Columbia fall Chinook in the estuary (8.9%), with the double-crested cormorants nesting on East Sand Island consuming the majority of fall Chinook smolts. Of the two sockeye stocks evaluated (Snake River and upper Columbia), reach-specific avian predation rates were highest on Snake River sockeye. High predation rates on sockeye where generally associated with predation by cormorants, rather than predation by terns, gulls, or pelicans.

Caspian terns consumed steelhead smolts disproportionately compared to other salmonid species, with tern predation rates on steelhead often 2 to 10 times greater than those on coho, Chinook, or sockeye. Cormorants also showed a preference for steelhead smolts relative to other salmonid species, but cormorant predation on smolts was more proportional to the relative availability of PIT-tagged smolts compared to tern predation. The higher susceptibility to avian predation of steelhead compared to other salmonid species in the Columbia River basin is well documented (Ryan et al. 2003; Antolos et al. 2004; Maranto et al. 2010). Possible reasons for the greater susceptibility of steelhead compared to salmon (Chinook, coho, or sockeye) may be related to differences in smolt behavior during out-migration and differences in the relative size (length) of smolts from these four species. Collis et al. (2001) hypothesized that the greater susceptibility of steelhead to avian predation was due to the larger size of steelhead smolts compared to Chinook, coho, and sockeye smolts. The positive association between average species specific salmonid length and avian predation rates described by Ryan et al. (2003) supported this hypothesis. Hostetter et al. (*in review*) provided new support for the hypothesis by demonstrating that individual smolt length influenced susceptibility to tern predation; steelhead smolt susceptibility to tern predation was greatest for steelhead around 200 mm for length, but decreased for longer and shorter smolts (see Chapter 4).

Predation associated with tern and cormorant colonies was almost always significantly higher than predation associated with gull and pelican colonies, regardless of the salmonid species and stock. Despite the large size of the gull and pelican colonies studied here (several thousand adults each; Chapter 1), predation rates never exceeded 2.0% of available PIT-tagged smolts, regardless of salmonid species, stock, or migration year.

Estuary avian predation- The highest reach-specific smolt predation rates occurred in the Columbia River estuary, with Caspian terns and double-crested cormorants nesting on East Sand

Island consuming the highest proportions of available smolts. Combined predation rates on PIT-tagged smolts last detected passing Bonneville or Sullivan dams ranged from 2.6% for Willamette spring Chinook to 18.2% for Snake summer steelhead. These results are similar, although somewhat higher, than those reported by Ryan et al. (2003) for terns and cormorants nesting on East Sand Island during 1998-2000. Ryan et al. (2003) reported a combined (tern and cormorant) average predation rate of 2.3% for all Chinook stocks and 12.6% for all steelhead stocks in the estuary. Differences in predation rates between those reported here and those reported by Ryan et al. (2003) are mostly due to the growing number of cormorants nesting on East Sand Island (from 6,825 pairs in 1998 to 13,771 pairs in 2007; Lyons et al. 2010), plus the lack of PIT tag detection efficiency corrections for predation rates estimated from PIT tag recoveries during 1998-2000.

Although large differences in avian predation rates were detected among salmonid species, some differences between run-types within a species were also observed. For example, predation rates on middle Columbia fall Chinook by cormorants nesting at East Sand Island were significantly higher than for all other Chinook stocks evaluated. Reasons for the greater stock-specific susceptibility of middle Columbia fall Chinook to cormorant predation in the estuary, a stock that originates closer to the estuary than many of the other stocks evaluated here, are unknown. Additional research will be needed to determine which factors (e.g., fish morphology, behavior, condition) lead to stock-specific differences in susceptibility to avian predation. It should also be noted that the stock-specific predation rates measured in this study and by others studying colonial waterbirds in the Columbia River estuary (Collis et al. 2001; Ryan et al. 2003) excluded smolts originating from the lower Columbia River due to a lack of adequate PIT tag detection sites below Bonneville and Sullivan dams. Data on avian predation rates on lower Columbia River salmonid stocks may be important to fisheries managers, particularly as they pertain to ESA-listed steelhead and coho populations (NOAA 2005).

When compared to the impact of avian predation associated with inland bird colonies, avian predation in the Columbia River estuary affects juvenile salmonids that have survived freshwater migration to the ocean and presumably have a higher probability of survival to return as adults compared to those fish that have yet to complete out-migration (Roby et al. 2003). Additionally, juvenile salmonids belonging to every ESA-listed stock 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. Management efforts focused on terns and cormorants in the Columbia River estuary will consequently benefit a greater diversity of salmonid populations (Roby et al. 2003; USFWS 2005; Lyons et al. 2010) compared to management of inland bird colonies.

Inland avian predation- The highest predation rates on PIT-tagged salmonid smolts by birds nesting at inland colonies were from Caspian terns nesting on Crescent Island and Goose Island and double-crested cormorants nesting on Foundation Island. Predation rates associated with these inland colonies were highest on summer steelhead, with Crescent Island terns disproportionately consuming Snake River summer steelhead and Goose Island terns disproportionately consuming upper Columbia summer steelhead. Variation in stock-specific predation rates was also observed for cormorants nesting on Foundation Island, with Snake River stocks significantly more susceptible to Foundation Island cormorant predation than upper Columbia River stocks. For example, predation rates on upper Columbia River stocks by Foundation Island cormorants were ≤ 0.1% (all salmonid species) compared to 0.3% – 2.0% (all species) for Snake River stocks. Predation rates by Foundation Island cormorants were highest on Snake River steelhead (2.0%), followed by Snake River sockeye (1.7%). For upper Columbia salmonid stocks, predation rates by Goose Island terns were the highest, with predation rates ranging from 0.2% for summer Chinook to an unexpectedly high 10.0% for summer steelhead during the study period. The predation rate estimate for steelhead was surprising because of the relatively small size of the Goose Island tern colony (< 500 breeding pairs; Chapter 1) and the location of the colony at least 35 km from the upper Columbia River and 73 km from the Snake River. It's important to note that results for most inland bird colonies exclude fish originating from the middle Columbia River due to a lack of in-river PIT tag detection sites below the confluence of the Snake and Columbia rivers, yet upstream of McNary Dam. As such, results for bird colonies in the Confluence reach are germane to Snake and upper Columbia stocks only.

Our estimates of predation rates on salmonid stocks, particularly steelhead stocks, by Caspian terns nesting on Goose Island in Potholes Reservoir differ considerably from those previously published by Maranto et al. (2010). Maranto et al. (2010) estimated an average predation rate of just 0.6% on upper Columbia summer steelhead by terns nesting at Potholes Reservoir during 2003-2006. Conversely, we measured a predation rate on upper Columbia steelhead by terns nesting at Potholes Reservoir of 10.0% (ranging from 5.5% to 15.6% per year) during 2006-2009. There are four possible explanations for this apparent discrepancy. First, the location of the Caspian tern colony in Potholes Reservoir shifted from Solstice Island in northern Potholes Reservoir to Goose Island in southern Potholes Reservoir (6 kilometers closer to the upper Columbia River) during 2003-2006. This move corresponded with a change in the birds' diet composition, with salmonid prey types more commonly observed in the diet of terns nesting on Goose Island (ca. 24% of prey items) compared to terns nesting on Solstice Island (ca. 2% of prey items) (Maranto et al. 2010). Second, the size of the Goose Island tern colony increased from an estimated 323 breeding pairs in 2006 (Maranto et al. 2010) to 487 breeding pairs in 2009 (Chapter 1). Third, no measure of PIT tag detection efficiency was available prior to 2006 and, consequently, predation rates presented by Maranto et al. (2010) were not corrected for on-colony PIT tag detection efficiency. This results in a substantial underestimation of predation rates because PIT tag detection efficiency on the Goose Island tern colony averaged just 54% during 2006-2009. Finally, smolt availability to Caspian terns nesting at Potholes Reservoir was calculated differently in the two studies. Maranto et al. (2010) based predation rate estimates on all PIT-tagged smolts released into the upper Columbia River. In our study, we limited our analysis to PIT-tagged smolts detected passing Rock Island Dam and subsequently recovered on the Goose Island tern colony. By restricting the proportion of the salmonid population that was exposed to predation by terns nesting at Potholes Reservoir, we calculated more realistic estimates of predation rates on available smolts. Rock Island Dam corresponds with the upper foraging range on the Columbia River of Caspian terns nesting at Potholes Reservoir (Maranto et al. 2010).

Avian predation rates on Snake River and upper Columbia stocks are relative to the numbers of smolts migrating past the inland bird colonies. Not all Snake River smolts, however, out-migrated past the inland bird colonies; a portion of smolts are collected at Snake River dams and are transported (via barges or trucks) to a release location below Bonneville Dam to reduce sources of in-river mortality during out-migration (Buchanan et al. 2006). Estimates of the percentage of Snake River smolts transported past inland bird colonies vary considerably by salmonid species and year, with estimates for Chinook and steelhead during the study period ranging from a high of approximately 95% in 2004 to a low of 35% in 2007 (FPC 2010).

Consequently, effects of avian predation associated with inland bird colonies on Snake River salmonid stocks apply only to that portion of the smolt population that is not transported around bird colonies (Antolos et al. 2005). Results presented here pertain only to the in-river or naturally out-migrating proportion of the run. All salmonids originating from the upper Columbia River, however, must out-migrate in-river past inland bird colonies in the Confluence reach. Similarly, because transported Snake River stocks were released below Bonneville Dam, all salmonid smolts must out-migrate past the bird colonies in the estuary.

Compared to predation rates associated with Caspian tern and double-crested cormorant colonies, predation rates by other inland colonies of piscivorous waterbirds (i.e., California gulls, ring-billed gulls, American white pelicans) were small. Of the gull and pelican colonies examined in this study, the Miller Rocks gull colony had the highest predation rates. Predation rates on PIT-tagged smolts by the pelican colony on Badger Island, the only breeding colony of white pelicans in Washington State (Ackerman 1997), were the smallest of the nine bird colonies investigated during the study. Pelican predation rates were ≤ 0.1% on 10 of the 11 salmonid stocks out-migrating through the confluence reach, with predation rates on Snake River summer steelhead only slightly higher (0.2%). Predation rates associated with the pelican colony remained low during the study period (2005-2009) despite a large increase in the size of the pelican colony (from ~ 1,100 adults in 2005 to ~ 1,800 adults in 2009; Chapter 1). Overall, impacts of pelican predation on out-migrating smolts from the upper Columbia and Snake rivers were marginal, especially when compared to nearby Caspian tern (Crescent Island) and cormorant (Foundation Island) colonies.

Extremely low predation rates on out-migrating salmonid smolts by white pelicans nesting on Badger Island may be due to several factors: a reliance on larger fish, a reliance on fish that congregate in shallow water habitats, and/or differences in foraging behavior that reduce the habitat overlap of Badger Island pelicans with out-migrating smolts. For instance, adult-sized fish up to 700 mm (fork-length) have been confirmed in the diet of American white pelicans (Scoppettone et al. 2006), including adult salmonids and common carp (*Cyprinus carpio)* found on the Badger Island pelican colony following the breeding season (author, unpublished data). Pelicans capture fish primarily from the top meter of the water column, limiting foraging to surface-oriented fish or to fish occupying shallow water habitats (Knopf and Evans 2004). In a study comparing American white pelicans and double-crested cormorants, Knopt and Kennedy (1981) attributed differences in predation on a population of tui chub (*Gila bicolor*) to differences in foraging techniques employed by cormorants and pelicans, with pelicans consuming fish near the surface. Results presented here are germane to actively out-migrating smolts from the Snake and upper Columbia rivers. Predation by pelicans on smolts in other rivers (e.g., Middle Columbia) or on adult salmonids may differ to an unknown degree from those on actively migrating smolts from the Snake and upper Columbia rivers. Finally, because the maximum reported foraging distance of American white pelicans (300 km; Scoppettone et al. 2006) is greater than that of Caspian terns, double-crested cormorants, California/ring-billed gulls (< 100 km; Baird 1976; Antolos et al. 2005; Maranto et al. 2010), it is possible pelicans spend larger amounts of time, on average, loafing off-colony. If true, off-colony PIT-tag deposition rates by pelicans could be greater than that of terns, cormorants, and gulls; a factor that could further underestimate predation rates on PIT-tagged smolts by Badger Island pelicans.

Susceptibility Based on Rear-type

In general, hatchery stocks were more susceptible than wild stocks, but this trend was not always consistent among avian predators or years. Several other studies in the Columbia River have noted that hatchery-reared salmonids were more susceptible to avian predation compared to their wild counterparts (Collis et al. 2001; Ryan et al. 2003; Kennedy et al. 2007). Increased susceptibility of hatchery-reared stocks has been attributed to a lack of innate and learned predator avoidance behaviors (Olla and Davis 1989; Berejikian 1995), greater surface orientation (Mason et al. 1967), physiological differences between hatchery and wild stocks (Kennedy et al 2007), and increased stress levels associated with handling (Schreck 1981; Olla and Davis 1989). The exact mechanism(s) that cause greater susceptibility of hatchery smolts compared to wild smolts, however, remains unclear. Although predation rates were generally greater for hatchery-reared smolts compared to their wild counterparts, the magnitude of differences in predation rates between rear-types were often small (≤ 1%). Small or non-existent differences in avian predation rates between hatchery and wild stocks imply that management efforts to reduce avian predation would benefit both wild and hatchery stocks.

Susceptibility Based on Run-timing

There is evidence that avian predation rates, particular those on steelhead, differ by smolt run-timing. Predation rates by terns were often significantly higher for steelhead late in the migration period (June-July), with predation rates 2 to 4 times higher than those on steelhead smolts passing earlier in the run (April-May). This trend was evident at both the Crescent Island and East Sand Island tern colonies, with late-season predation rates on steelhead stocks sometimes exceeding 25% of available smolts. Run-timing trends associated with cormorant predation, however, were often not consistent across years or colonies. General trends between run-timing and cormorant predation rates were not detected for any Chinook stocks. East Sand Island cormorant predation on steelhead, however, indicated that in some years the highest predation rates were during the peak of the steelhead out-migration (May).

Smolt survival is associated with a complex set of environmental factors and individual smolt characteristics (Raymond 1979; Gregory 1993; Gregory and Levings 1998; Korstrom and Birtwell 2006; Chapter 4). In Chapter 4, we noted that steelhead susceptibility to Caspian tern predation increased when steelhead abundance decreased and when discharge decreased. Large numbers of prey have been shown to swamp the short-term capacity of predators to consume them, which in turn can improve the chances of survival for individual prey (Ims 1990). Results presented in this Chapter also demonstrate that predation rates by terns were lowest during periods of high steelhead abundance, consistent with the predator swamping hypothesis. However, during the time periods when steelhead predation rates by terns in the estuary were lowest (May), cormorant predation rates were often the highest, which may have reduced any potential benefits from predator swamping. Interactions between different avian predators and colonies on the same stock of fish highlight the importance of a system-wide avian predation evaluation in the Columbia River basin.

Colony Size-adjusted Predation Rates

Predation rates on salmonid smolts that are adjusted for the large differences in size of the nine study colonies (i.e., per bird predation rates) were substantially higher at inland tern and cormorant colonies relative to their counterparts in the estuary. Colony size-adjusted predation rates on smolts associated with gull and pelican colonies were very low in comparison to those associated with terns and cormorant colonies. The higher colony size-adjusted predation rates of inland tern and cormorant colonies are due to the higher prevalence of juvenile salmonids in the diet of terns and cormorants nesting at inland colonies, compared with their estuary counterparts (Collis et al. 2002; Antolos et al. 2005; Lyons 2010; Chapter 2).

Previous research indicated that fish, and salmonids in particular, constituted a very small proportion of the diet of California and ring-billed gulls nesting on islands in the Columbia River (Collis et al. 2002). Gut content analysis of gulls nesting at Miller Rocks and Crescent Island by Collis et al. (2002) indicated that salmonid smolts comprised < 4% (by mass) of food biomass. In contrast, salmonids comprised 74% (by mass) of the diet of Caspian terns nesting on Crescent Island (Antolos et al. 2005). Differences in diet composition have also been attributed to colony location, as food availability often differs throughout the Columbia River basin (Collis et al. 2002; Roby et al. 2003; Lyons 2010). A comparison of the diet composition of Caspian terns nesting on Rice Island (Rkm 34) and on East Sand Island (Rkm 8) revealed that terns nesting on East Sand Island were more reliant on marine forage fishes (anchovy, herring, surf perch, etc.) than freshwater fishes (Roby et al. 2002). Collies et al. (*in review*) documented a similar trend in the diet composition of Caspian terns nesting in San Francisco Bay, with the percentage of salmonids found in the diet of terns directly related to the distance of the colony from the out-migration path of smolts. Similar differences in diet composition between inland and estuary cormorant colonies support this hypothesis, with salmonid smolts a larger proportion of the diet of the inland cormorant colony at Foundation Island relative to the colony in the estuary (Chapter 2).

Differences between colony-wide predation rates and colony size-adjusted predation rates indicate that current management efforts to increase smolt survival through reductions in avian predation in the estuary could be offset if birds nesting in the estuary relocated to inland sites in large numbers, as increases in colony size at inland sites could have a dramatic and negative impact on smolt survival for particular salmonid stocks, especially Snake River and upper Columbia steelhead. Although this scenario seems unlikely based on results presented in Chapter 1, it is possible that some, albeit not large numbers, of birds might relocate to the Columbia Plateau once management initiatives are implemented to reduce the size of estuary colonies.

Concluding Remarks

Evaluation of the impacts to salmonid stocks from avian predation on juvenile salmonids requires data on both the proportion of smolts consumed (predation rates) and the number of smolts consumed (Roby et al 2003; Antolos et al. 2005; Chapter 2). Predation rates based on PIT tag recoveries on bird colonies provide minimum estimates of the proportion of smolts consumed by avian predators and provide fisheries managers with specific information on when and where particular salmonid stocks are the most susceptible to predation by birds from different breeding colonies.

Assessments of avian predation by combining estimates of predation rates with estimates of consumption rates (see Chapter 2) allows managers to identify significant impacts on salmonid stocks from avian predation due either to large numbers of smolts consumed or high proportions of available smolts consumed. Further research is needed to determine whether reductions in smolt losses to avian predation translate into commensurate increases in smolt survival and adult salmonid recruitment. Alternatively, reductions in avian predation could be compensated for to some unknown degree by increases in other mortality factors (Schreck et al. 2006). For example, reductions in either the size or number of piscivorous waterbird colonies in the Columbia River basin will only result in higher recruitment of adult salmonids if reductions in avian predation are not compensated for by other smolt mortality factors (e.g., other predators, disease, starvation, etc).

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LITERATURE CITED

Ackerman, S.M. 1997. Update: American white pelican colony. Washington Ornithological Society News Letter 51. October/November. Available: http://www.wos.org/newsletterarchives.htm (February 2011).

Anderson, C.D., D.D. Roby, and K. Collis. 2004. Conservation implications of the large colony of double-crested cormorants on East Sand Island, Columbia River estuary, Oregon, USA. Waterbirds 27:155-160.

Antolos, M., D.D. Roby, D.E. Lyons, K. Collis, A.F. Evans, M. Hawbecker, and B.A. Ryan. 2005. Caspian tern predation on juvenile salmonids in the mid-Columbia River. Transactions of the American Fisheries Society 134:466-480.

Axel, G.A., E.F. Prentice, and B.P. Sandford. 2005. PIT-tag detection system for large-diameter juvenile fish bypass pipes at Columbia River basin hydroelectric dams. North American Journal of Fisheries Management 25:646-651.

Baird, P.A. 1976. Comparative ecology of California and ring-billed gulls (*Larus californicus* and *Larus delawarensis*). Diss. Abst. Inter. 37:5506.

Berejikian, B.A. 1995. The effects of hatchery and wild ancestry and experience on the relative ability of steelhead trout fry (*Oncorhynchus mykiss*) to avoid a benthic predator. Canadian Journal of Fisheries and Aquatic Sciences 52:2476-2482.

Buchanan, R.A., J.R. Skalski, and S.G. Smith. 2006. Estimating the effects of smolt transportation from different vantage points and management perspectives. North American Journal of Fisheries Management 26:460-472.

Collis, K., D.D. Roby, D.P. Craig, B.A. Ryan, and R.D. Ledgerwood. 2001. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary: vulnerability of different salmonid species, stocks, and rearing types. Transactions of the American Fisheries Society 130:385-396.

Collis, K., D.D. Roby, D.P. Craig, S. Adamany, J.Y. Adkins, and D.E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Collis, K. D.D. Roby, D.E. Lyons, Y. Suzuki, J.Y. Adkins, L. Reinalda, C. Hand, N. Hostetter, A.F. Evans, and M. Hawbecker. 2007. Research, monitoring, and evaluation of avian predation on salmonid smolts in the Lower and Mid-Columbia River: 2006 final season summary. Bird Research Northwest. Available on-line at www.birdresearchnw.org.

Collis, K., D.D. Roby, K.W. Larson, L.J. Adrean, S.K. Nelson, A.F. Evans, D.B. Battaglia, D.E. Lyons, T. Marcella, and A. Patterson. *In-review*. Trends in Caspian tern nesting and diet in San Francisco Bay: management implications. Submitted to Waterbirds

Conover, M.R, B.C. Thompson, R.E. Fitzner, and D.E. Miller. 1979. Increased populations of ring-billed and California gulls in Washington state. Western Birds 10:31-36.

Cuthbert, F.J. 1988. Reproductive success and colony-site tenacity in Caspian terns. Auk 105: 339–344.

Efron, B., and R.J. Tibshirani. 1986. Bootstrap methods for standard errors, confidence intervals, and other measures of statistical accuracy. Statistical Science 1:54-77.

Evans, A.F., D.D. Roby, K. Collis, B.M. Cramer, J.A. Sheggeby, L.J. Adrean, D.S. Battaglia, and D.E. Lyons. 2011. Recovery of coded wire tags on a Caspian tern colony in San Francisco Bay: a technique to evaluate impacts of avian predation on juvenile salmonids. North American Journal of Fisheries Management 31:79-87.

FPC (Fish Passage Center). 2010. The 2009 Fish Passage Center Annual Report. Fish Passage Center, Portland, OR. Available: http://www.fpc.org/documents/annual_FPC_report (February 2011)

Gregory, R.S. 1993. Effect of turbidity on the predator avoidance-behavior of juvenile Chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 50:241-246.

Gregory, R.S., and C.D. Levings. 1998. Turbidity reduces predation on migrating juvenile Pacific salmon. Transactions of the American Fisheries Society 127:275-285.

Hostetter, N.J., A.F. Evans, D.D. Roby, and K. Collis. In review. Susceptibility of juvenile steelhead (*Oncorhynchus mykiss*) to avian predation: The influence of individual fish characteristics and river conditions. Submitted to Canadian Journal of Fisheries and Aquatic Sciences.

Ims, R.A. 1990. On the adaptive value of reproductive synchrony as a predator-swamping strategy. American Naturalist 136:485-498.

Kennedy, B.M., W.L. Gale, and K.G. Ostrand. 2007. Relationship between smolt gill Na+, K+ ATPase activity and migration timing to avian predation risk of steelhead trout (*Oncorhynchus mykiss*) in a large estuary. Canadian Journal of Fisheries Aquatic Sciences 64:1506-1516.

Knoft, F.L, and J.L. Kennedy. 1981. Differential predation by two species of piscivorous birds. Wilson Bulletin 93:554-556.

Knopf, F., and R.M. Evans. 2004. American white pelican (*Pelecanus erythrorhynchos*). The Birds of North America Online (A. Poole, ed.). Ithaca: Cornell Lab of Ornithology. Available: <http://bna.birds.cornell.edu/bna/species/057>.

Korstrom, J.S., and I.K. Birtwell. 2006. Effects of suspended sediment on the escape behavior and cover-seeking response of juvenile Chinook salmon in freshwater. Transactions of the American Fisheries Society 135:1006-1016.

Lyons, D.E., D.D. Roby, and K. Collis. 2005. Foraging ecology of Caspian terns in the Columbia River Estuary, USA. Waterbirds 28:280-291.

Lyons, D.E. 2010. Bioenergetics-based predator-prey relationships between piscivorous birds and juvenile salmonids in the Columbia River estuary. Unpubl. Ph.D. dissertation, Oregon State University, Corvallis, Oregon. Available: http://www.birdresearchnw.org/Project-Info/publications-reports/graduate-student-theses (February 2011).

Manly, B.F.J. 1998. Randomization, bootstrap and Monte Carlo methods in biology. Second edition. Chapman & Hall, London.

Maranto, C.J., T.P. Good, F.K. Wiese, and J.K. Parrish. 2010. Impact of the Potholes Reservoir Caspian tern breeding colony on out-migrating juvenile salmonids in the mid-Columbia River. Transactions of the American Fisheries Society 139:362-381.

Mason, J.W., O.M. Brynildson, and P.E. Degurse. 1967. Comparative survival of wild and domestic strains of brook trout in streams. Transactions of the American Fisheries Society 96:313-319.

NOAA (National Oceanic and Atmospheric Administration). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. Available: http://www.nwr.noaa.gov/Publications/Biological-Status-Reviews/Salmon.cfm (September 2010).

NOAA (National Oceanic and Atmospheric Administration). 2008. Biological opinion, Endangered Species Act section 7(a)(2) consultation: consultation on remand for operation of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(1)(A) permit for Juvenile Fish Transportation Program. Seattle, Washington. Available: https://pcts.nmfs.noaa.gov/pls/pcts-pub/pcts_upload.summary_list_biop?p_id=27149. (September 2010).

Olla, B.L., and M.W. Davis. 1989. The role of learning and stress in predator avoidance of hatchery-reared coho salmon (*Oncorhynchus kisutch*) juveniles. Aquaculture 76:209-214.

Peven, C.M., and S. G. Hayes. 1989. Proportions of hatchery- and naturally produced steelhead smolts migrating past Rock Island Dam, Columbia River, Washington. North American Journal of Fisheries Management 9:53-59.

PSMFC (Pacific States Marine Fisheries Commission). 2010. PIT tag information system for the Columbia River Basin. Portland, Oregon. Available: http://www.ptagis.org/ptagis/ (September 2010).

Quinn, J.S., and J. Sirdevan. 1998. Experimental measurement of nesting substrate preference in Caspian terns, *Sterna caspia*, and the successful colonization of human constructed islands. Biological Conservation 85:63–68.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of juvenile Chinook salmon and steelhead from the Snake River, 1966 to 1975. Transactions of the American Fisheries Society 108:505-529.

Roby, D.D., K. Collis, D.E. Lyons, D.P. Craig, J. Adkins, A.M. Myers, and R.M. Suryan. 2002. Effects of colony relocation on diet and productivity of Caspian terns. Journal of Wildlife Management 66:662–673.

Roby, D.D., D.E. Lyons, D.P. Craig, K. Collis, and G.H. Visser. 2003. Quantifying the effect of predators on endangered species using a bioenergetics approach: Caspian terns and juvenile salmonids in the Columbia River estuary. Canadian Journal of Zoology 81:250-265.

Ruggerone, G.T. 1986. Consumption of migrating juvenile salmonids by gulls foraging below a Columbia River dam. Transactions of the American Fisheries Society 115:736-742.

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

Ryder, J.P. 1993. Ring-billed Gull (*Larus delawarensis*). The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology. Available: <http://bna.birds.cornell.edu/bna/species/033>.

Schreck, C.B. 1981. Stress and compensation in teleostean fishes: response to social and physical factors. Pp. 295-321 *in* Stress and fish. A.D. Pickering (ed.). Academic Press, New York.

Schreck, C.B., T.P. Stahl, L.E. Davis, D.D. Roby, and B.J. Clemens. 2006. Mortality estimates of juvenile spring-summer Chinook salmon in the lower Columbia River and estuary, 1992-1998: evidence for delayed mortality? Transactions of the American Fisheries Society 135:457-475.

Scoppettone, G.G., P.H. Rissler, D. Withers, and M.C. Fabes. 2006. Fish tag recovery from the American white pelican nesting colony on Anaho Island, Pyramid Lake, Nevada. Great Basin Birds 8:6-10.

USFWS (U.S. Fish and Wildlife Service). 2005. Caspian tern management to reduce predation of juvenile salmonids in the Columbia River estuary: Final Environmental Impact Statement, January 2005. Migratory Birds and Habitat Programs, Portland, OR.

Wertheimer, R.H. 2007. Evaluation of a surface flow bypass system for steelhead kelt passage at Bonneville Dam, Washington. North American Journal of Fisheries Management 27:21–29.

Wiese, F.K., J.K. Parish, C.W. Thompson, and C.M. Maranto. 2008. Ecosystem-based management of predator-prey relationships: piscivorous birds and salmonids. Ecological Applications 18:681-700.

Wires, L.R., and F.J. Cuthbert. 2000. Trends in Caspian tern numbers and distribution in North America: a review. Waterbirds 23:388–404.

Table 3.1. Numbers of PIT-tagged smolts interrogated passing Bonneville Dam (BON) on the lower Columbia River or Sullivan Dam (SUL) on the lower Willamette River. These numbers were used to generate predation rate estimates for bird colonies in the Columbia River estuary during smolt migration years 2004-2009. Dashed lines denote PIT-tagged stocks with too few interrogations for analyses (< 500 detections per year).

				Migration Year						
Dam	River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Total
BON	Middle Columbia	Chinook	Spring	2,074	2,484	1,508	3,926	6,607	5,687	22,286
			Summer	-	-	-	-	-	-	-
			Fall	-	-	-	-	3,043	3,166	6,209
		Coho		-	-	-	-	-	-	-
		Steelhead	Summer	-	-	-	2,263	2,388	3,511	8,162
			Winter	-	-	-	952	2,258	1,308	4,518
	Snake	Chinook	Spring	3,411	4,117	4,387	19,460	9,134	13,338	53,847
			Summer	1,405	1,818	1,183	4,370	2,291	4,058	15,125
			Fall	929	1,121	4,057	2,005	24,136	16,314	48,562
		Coho		-	-	-	-	-	-	-
		Sockeye		-	-	-	-	-	1,845	1,845
		Steelhead	Summer	803	753	1,100	6,391	19,571	23,310	51,928
	Upper Columbia	Chinook	Spring	10,417	3,330	1,660	3,965	3,440	4,634	27,446
			Summer	9,764	2,579	-	-	-	1,222	13,565
			Fall	-	-	-	1,301	1,862	1,218	4,381
		Coho		1,064	619	-	3,155	2,766	3,943	11,547
		Sockeye		-	-	-	867	768	1,010	2,645
		Steelhead	Summer	6,060	5,622	2,078	3,056	2,541	2,272	21,629
SUL	Willamette	Chinook	Spring	-	-	-	1,505	2,509	5,573	9,587
		Steelhead	Summer	-	-	-	1,437	2,104	1,853	5,394
			Winter	-	-	-	-	-	-	-

Table 3.2. Numbers of PIT-tagged smolts interrogated passing McNary Dam (McN). These numbers were used to generate predation rate estimates for bird colonies in the middle Columbia River during smolt migration years 2004-2009. Dashed lines denote PIT-tagged stocks with too few interrogations for analyses (< 500 detections per year).

				Migration Year						
Dam	River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Total
McN	Middle Columbia	Chinook	Spring	573	-	807	1,232	1,179	1,564	13,598
			Fall	-	-	-	-	-	-	-
		Coho		-	-	-	-	-	-	-
		Steelhead	Summer	-	-	-	983	1,158	1,861	4,002
	Snake	Chinook	Spring	17,390	20,874	31,975	63,281	21,723	45,821	201,064
			Summer	5,980	8,392	4,871	11,624	5,565	14,334	50,766
			Fall	7,785	10,572	27,467	7,374	36,857	43,461	133,516
		Coho		-	-	-	-	-	-	-
		Sockeye		-	-	-	-	-	2,088	2,088
		Steelhead	Summer	4,860	9,299	11,995	7,680	15,447	29,877	79,158
	Upper Columbia	Chinook	Spring	35,092	15,665	10,384	12,979	9,574	11,017	94,711
			Summer	33,982	11,968	-	-	598	2,845	49,393
			Fall	2,428	3,948	2,244	5,154	4,980	3,967	22,721
		Coho		3,209	2,558	3,488	3,268	4,028	4,066	20,617
		Sockeye		-	-	518	1,319	1,042	2,430	5,309
		Steelhead	Summer	17,849	39,887	4,304	3,146	3,254	3,291	71,731

Table 3.3. Numbers of PIT-tagged smolts interrogated passing Lower Monumental Dam (LMN) on the Snake River or Rock Island Dam (RIS) on the upper Columbia River. These numbers were used to generate predation rate estimates for bird colonies located below the confluence of Snake and Columbia rivers or off-river at Potholes Reservoir, WA during smolt migration years 2004-2009. Dashed lines denote PIT-tagged stocks with too few interrogations for analyses (< 500 detections per year).

				Migration Year						
Dam	River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Total
LMN	Snake	Chinook	Spring	9,810	18,855	39,970	20,002	24,725	15,325	128,687
			Summer	4,174	7,466	5,409	2,728	5,417	5,428	30,622
			Fall	8,563	7,090	25,789	2,147	22,968	27,198	93,755
		Coho		658	580	-	-	-	-	1,238
		Sockeye		-	-	695	-	767	2,651	4,113
		Steelhead	Summer	23,339	28,151	32,104	17,120	28,652	52,220	181,586
RIS	Upper Columbia	Chinook	Spring	553	-	-	-	-	738	1,291
			Summer	87,625	43,347	-	-	-	1,413	132,385
		Coho		-	-	513	-	546	550	1,609
		Sockeye		1,083	887	3,600	2,074	1,960	2,099	11,703
		Steelhead	Summer	4,856	7,334	3,971	3,781	7,742	7,226	34,910

Table 3.4. Numbers of smolt PIT tags detected on Caspian tern (tern), double-crested cormorant (cormorant), American white pelican (pelican), and California and ring-billed gull (gull) colonies in the Columbia River basin during 2004-2009. PIT tag recoveries include Chinook, coho, sockeye, and steelhead combined. Dashed lines denote colonies that were not scanned for PIT tags following a given smolt migration year. Values are ranked from highest to lowest based on the average number of PIT tags detected during the study period.

		Migration Year						
Island	Bird colony	2004	2005	2006	2007	2008	2009	Average
East Sand	tern	36,437	31,356	28,303	25,383	42,331	38,339	33,692
East Sand	cormorant	16,056	14,978	20,063	7,635	21,317	25,270	17,553
Crescent	tern	23,197	17,650	5,950	3,778	7,146	8,153	10,979
Foundation	cormorant	3,882	5,553	4,735	6,611	7,237	7,375	5,899
Miller Rocks	gulls	-	-	-	3,579	3,394	4,135	3,703
Goose	tern	-	-	1,396	1,337	2,021	2,948	1,926
Crescent	gulls	-	-	-	1,421	1,443	1,825	1,563
Blalocks	tern	-	-	-	1,003	1,530	1,269	1,267
Badger	pelican	-	863	751	923	1,427	1,752	1,143

Table 3.5. Detection efficiency of PIT tags sown by researchers on Caspian tern (tern), double-crested cormorant (cormorant), American white pelican (pelican), and California gull/ring-billed gull (gull) colonies in the Columbia River basin during 2004-2009. In-season variation (standard deviation in parentheses) was calculated when multiple releases of PIT tags were sown during the bird nesting season. Sample sizes are presented in Table 3.6. Dashed lines denote colonies that were not scanned for PIT tags in a given year.

Island	Bird colony	2004	2005	2006	2007	2008	2009
East Sand	tern	92.2%	83.3%	64.1%	89.1%	91.8%	90.0%
		(5.4)	(5.9)	(10.4)	(6.8)	(5.1)	(7.0)
East Sand	cormorant	36.0%	55.0%	52.0%	58.0%	69.0%	70.0%
		(NA)*	(NA) *	(NA) *	(9.9)	(21.2)	(12.7)
Miller Rocks	gulls	-	-	-	86.5%	82.5%	78.0%
					(4.9)	(2.1)	(12.7)
Blalock	tern	-	-	-	88.0%	93.0%	84.0%
					(0.0)	(9.9)	(22.6)
Crescent	tern	79.2%	70.7%	47.4%	69.8%	62.0%	71.0%
		(16.7)	(28.9)	(35.1)	(28.7)	(28.1)	(36.6)
Crescent	gulls	-	-	-	63.0%	73.5%	72.5%
					(50.9)	(29.0)	(17.7)
Badger	pelican	-	58.0%	64.5%	64.5%	68.0%	85.0%
			(NA)*	(0.7)	(10.6)	(8.5)	(12.7)
Foundation	cormorant	63.0%	67.8%	67.3%	68.0%	74.3%	72.8%
		(NA)*	(9.0)	(9.3)	(11.3)	(5.9)	(9.8)
Goose	tern	-	-	53.2%	53.0%	63.5%	46.5%
				(24.3)	(55.2)	(20.6)	(22.3)

* No standard deviation calculated as only one colony-wide release of PIT tags occurred during the nesting season

Table 3.6. Temporal trends in the detection efficiency of PIT tags sown by researchers on Caspian tern (tern), double-crested cormorant (cormorant), American white pelican (pelican), and California gull/ring-billed gull (gull) colonies in the Columbia River basin during 2004-2009. Regression coefficients (number of PIT tags sown in parentheses) are from logistic regression models used to predict associations between the date a PIT tag was deposited on a bird colony and the likelihood it was recovered in that year. Positive regression coefficients indicate that detection efficiency increased with Julian date of deposition, while negative regression coefficients indicate that detection efficiency decreased with Julian date of deposition. Years where detection efficiency was significantly ($P \leq 0.05$) associated with the date a PIT tag was deposited are bolded. Dashed lines denote colonies that were not scanned for PIT tags in a given year.

Island	Bird colony	2004	2005	2006	2007	2008	2009
East Sand	tern	**-0.0098** (1,100)	0.0005 (1,200)	0.0012 (1,200)	-0.0011 (600)	**0.0097** (600)	**0.0070** (600)
East Sand	cormorant	NA* (600)	NA* (800)	NA* (600)	**0.0038** (200)	**0.0087** (600)	0.0020 (600)
Miller Rocks	gulls	-	-	-	-0.0052 (200)	0.0017 (200)	**0.0092** (200)
Blalocks	tern	-	-	-	0.0000 (200)	0.1550 (100)	0.1650 (100)
Crescent	tern	**0.0192** (960)	**0.0266** (960)	**0.0270** (960)	**0.0258** (800)	**0.0238** (800)	**0.0372** (400)
Crescent	gulls	-	-	-	**0.0466** (200)	**0.0216** (200)	**0.0105** (200)
Badger	pelican	-	NA*	0.0002 (200)	**0.0031** (200)	0.0026 (200)	**0.0066** (200)
Foundation	cormorant	NA*	-0.0039 (400)	-0.0036 (400)	-0.0024 (400)	-0.0023 (400)	**-0.0050** (400)
Goose	tern	-	-	**0.0080** (600)	**0.0600** (100)	**0.0176** (400)	**0.0138** (400)

* No standard deviation calculated as only one colony-wide release of PIT tags occurred during the nesting season

Table 3.7. Estimated predation rates (95% confidence intervals) of hatchery and wild PIT-tagged salmonid smolts by birds nesting at colonies in the Columbia River basin during 2004-2009. Predation rates during the study period were calculated if predation rates for a specific stock were ≥ 2% (see Figure 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in the same years.

East Sand Island cormorants

			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Mid-Columbia	Chinook	Spring	2.1% (1.7 - 2.6)	1.7% (0.8 - 2.7)	ND
Snake			2.0% (1.9 - 2.2)	2.5% (1.8 - 3.2)	ND
Upper Columbia			1.6% (1.2 - 2.0)	1.6% (0.8 - 2.4)	ND
Snake		Summer	1.9% (1.5 - 2.3)	1.8% (1.1 - 2.5)	ND
Mid-Columbia	Steelhead	Summer	5.4% (4.7 - 6.3)	4.4% (3.5 - 5.4)	ND
Snake			7.1% (6.6 - 7.6)	5.7% (4.9 - 6.6)	H

East Sand Island tern

Mid-Columbia	Chinook	Spring	2.5% (2.2 - 2.9)	1.0% (0.4 - 1.6)	H
Snake			2.7% (2.5 - 2.9)	1.2% (0.8 - 1.6)	H
Upper Columbia			2.2% (1.9 - 2.6)	0.9% (0.4 - 1.4)	H
Snake		Summer	2.4% (2.1 - 2.7)	1.3% (0.8 - 1.7)	H
Mid-Columbia	Steelhead	Summer	11.5% (10.6 - 12.5)	9.7% (8.5 - 10.8)	H
Snake			10.6% (10.3 - 11.0)	13.8% (12.9 - 14.7)	W

Crescent Island terns

Snake	Steelhead	Summer	7.4% (7.2 - 7.7)	7.5% (7.2 - 8.0)	ND
Upper Columbia	Steelhead	Summer	2.5% (2.3 - 2.7)	1.8% (1.5 - 2.2)	H

Foundation Island cormorants

Snake	Steelhead	Summer	2.1% (2.0 - 2.2)	1.6% (1.4 - 1.7)	H

Goose Island terns

Upper Columbia	Steelhead	Summer	10.6% (9.6 - 11.9)	8.4% (7.3 - 9.9)	H

[a] Significance levels:
H = Predation rate of hatchery smolts was significantly higher than wild smolts
W = Predation rate of wild smolts was significantly higher than hatchery smolts
ND = Difference in predation rates between hatchery and wild smolts was not significant

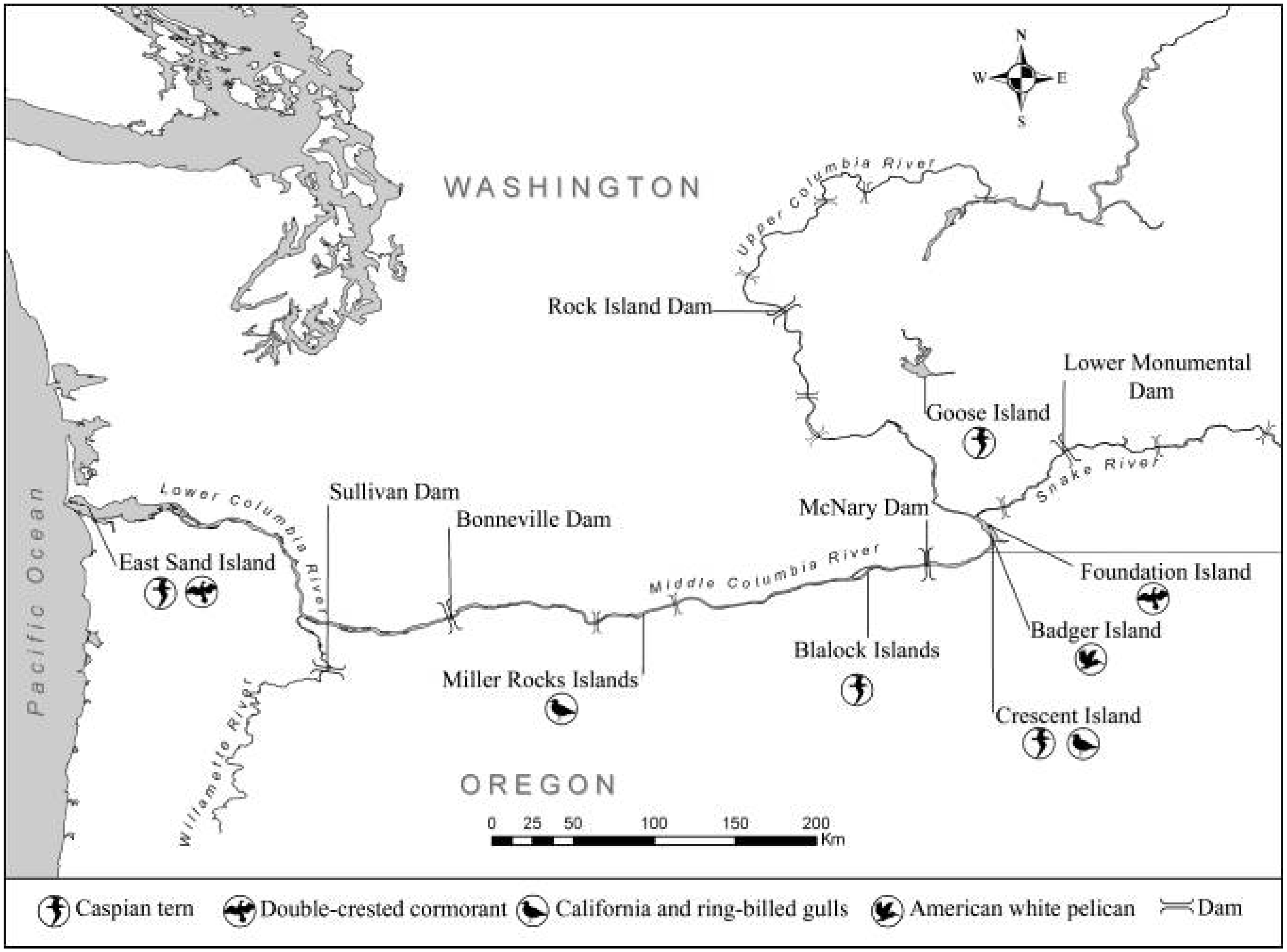


Figure 3.1. Map of the Columbia River basin showing the nine bird nesting colonies scanned for smolt passive integrated transponder (PIT) tags, the major river systems used to designate smolt river-of-origin, and the hydroelectric dams used to determine smolt availability (via PIT tag detections) during 2004-2009.

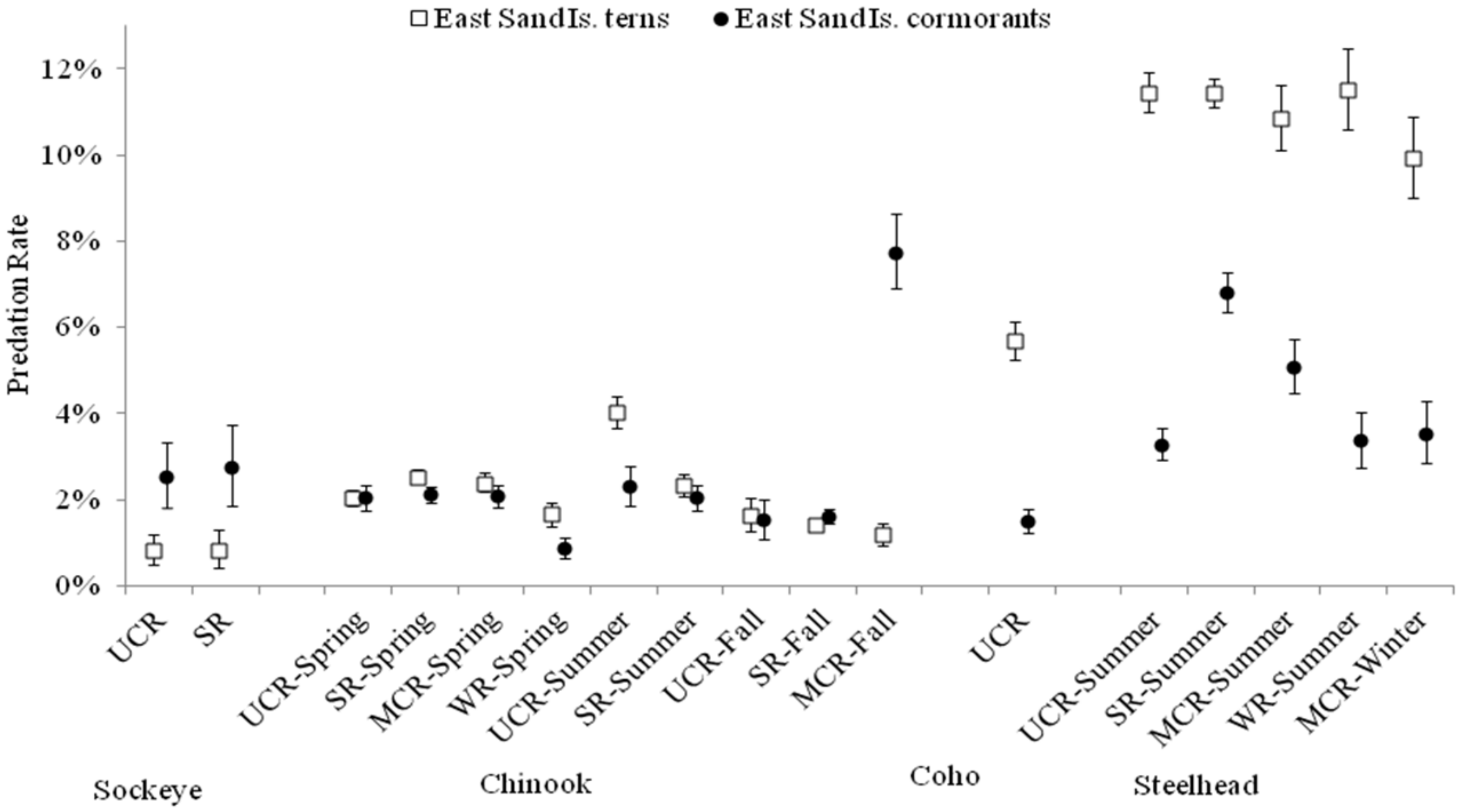


Figure 3.2. Estimated predation rates of PIT-tagged smolts last interrogated passing Bonneville Dam on the lower Columbia River or Sullivan Dam on the lower Willamette River (WR) by Caspian terns (terns) and double-crested cormorants (cormorants) nesting on East Sand Island in the Columbia River estuary during 2004-2009. The salmonid species (sockeye, Chinook, coho, steelhead), run-type (spring, summer, fall, winter), and river-of-origin (UCR for upper Columbia River, SR for Snake River, MCR for middle Columbia River, WR for Willamette River) for each PIT-tagged stock are provided. Only stocks with ≥ 500 PIT-tagged individuals interrogated in any given year (see Table 3.1) are presented. Error bars represent 95% confidence intervals.

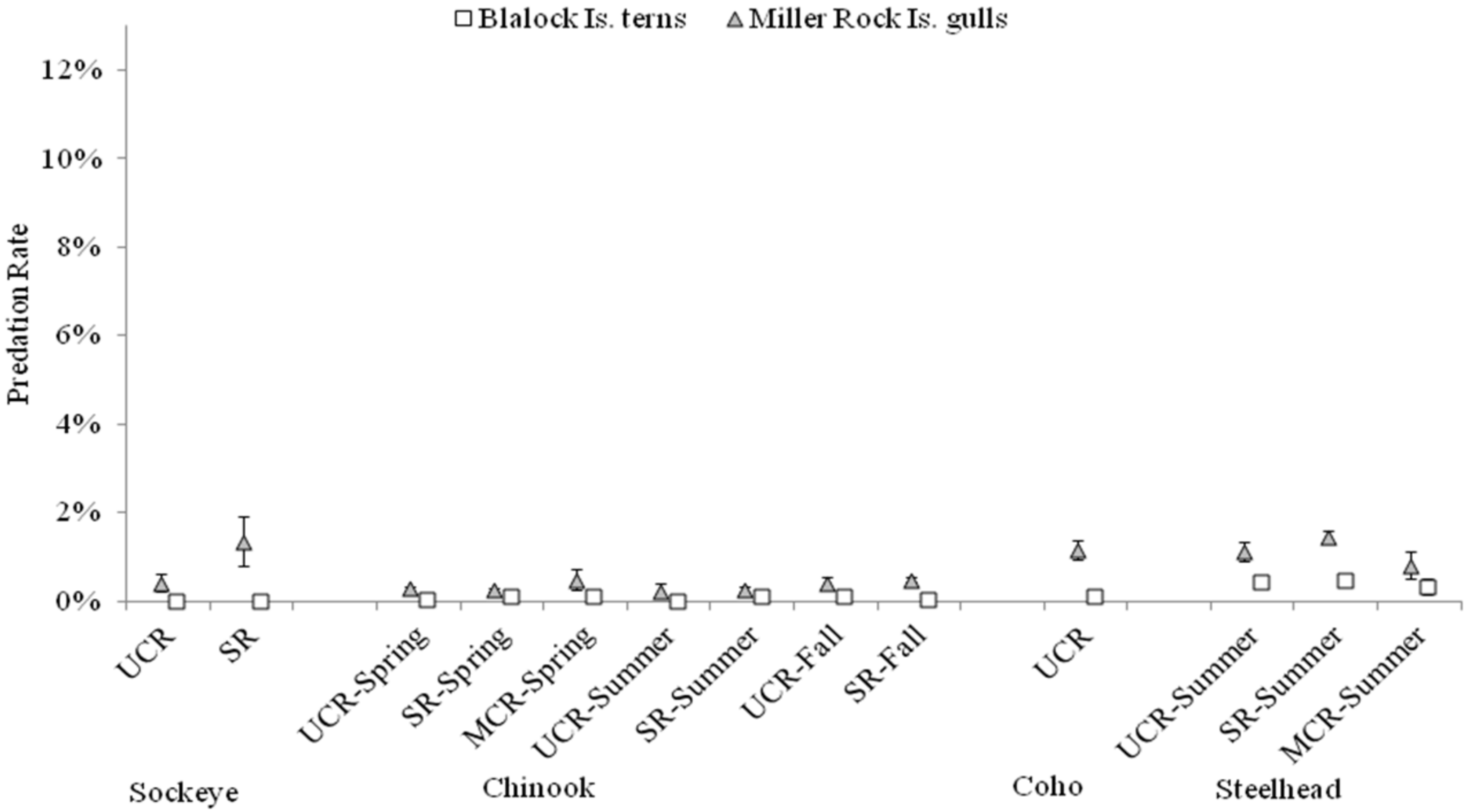


Figure 3.3. Estimated predation rates of PIT-tagged smolts last interrogated passing McNary Dam on the middle Columbia River by Caspian terns (terns) and California and ring-billed gulls (gulls) nesting on islands (Blalock, Miller Rocks) in the middle Columbia River during 2007-2009. The salmonid species (sockeye, Chinook, coho, steelhead), run-type (spring, summer, fall), and river-of-origin (UCR for upper Columbia River, SR for Snake River, and MCR for middle Columbia River) for each PIT-tagged stock are provided. Only stocks with ≥ 500 PIT-tagged individuals interrogated in any given year (see Table 3.2) are presented. Error bars represent 95% confidence intervals.

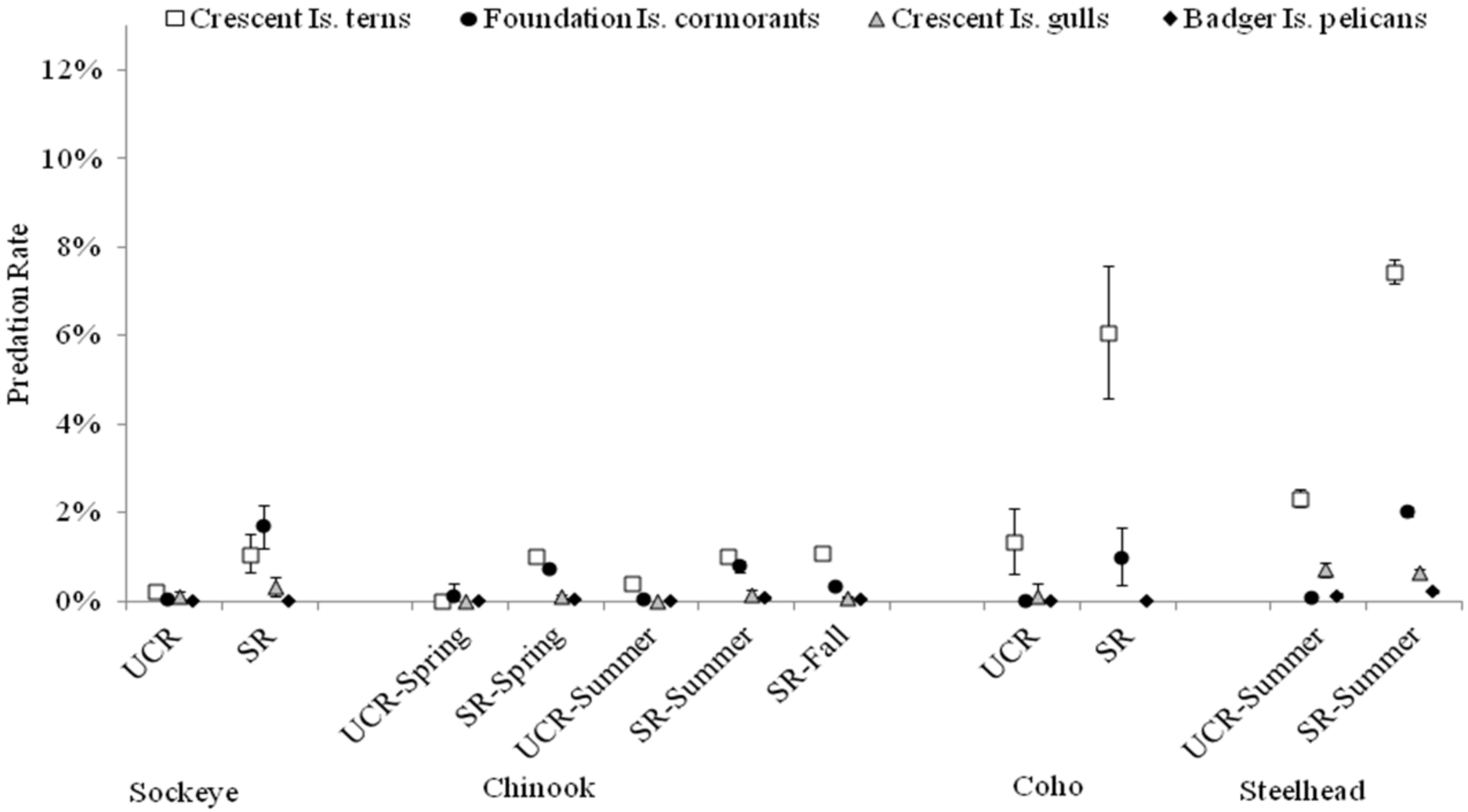


Figure 3.4. Estimated predation rates of PIT-tagged smolts last interrogated passing Lower Monumental Dam on the Snake River or Rock Island Dam on the upper Columbia River by Caspian terns (terns), double-crested cormorants (cormorants), California and ring-billed gulls (gulls), and American white pelicans (pelicans) nesting on islands (Crescent, Foundation, Badger) near the confluence of the Snake and Columbia rivers during 2004-2009. The salmonid species (sockeye, Chinook, coho, steelhead), run-type (spring, summer, fall), and river-of-origin (UCR for upper Columbia River, SR for Snake River) for each PIT-tagged stock are provided. Only stocks with ≥ 500 PIT-tagged individuals interrogated in any given year (see Table 3.3) are presented. Error bars represent 95% confidence intervals.

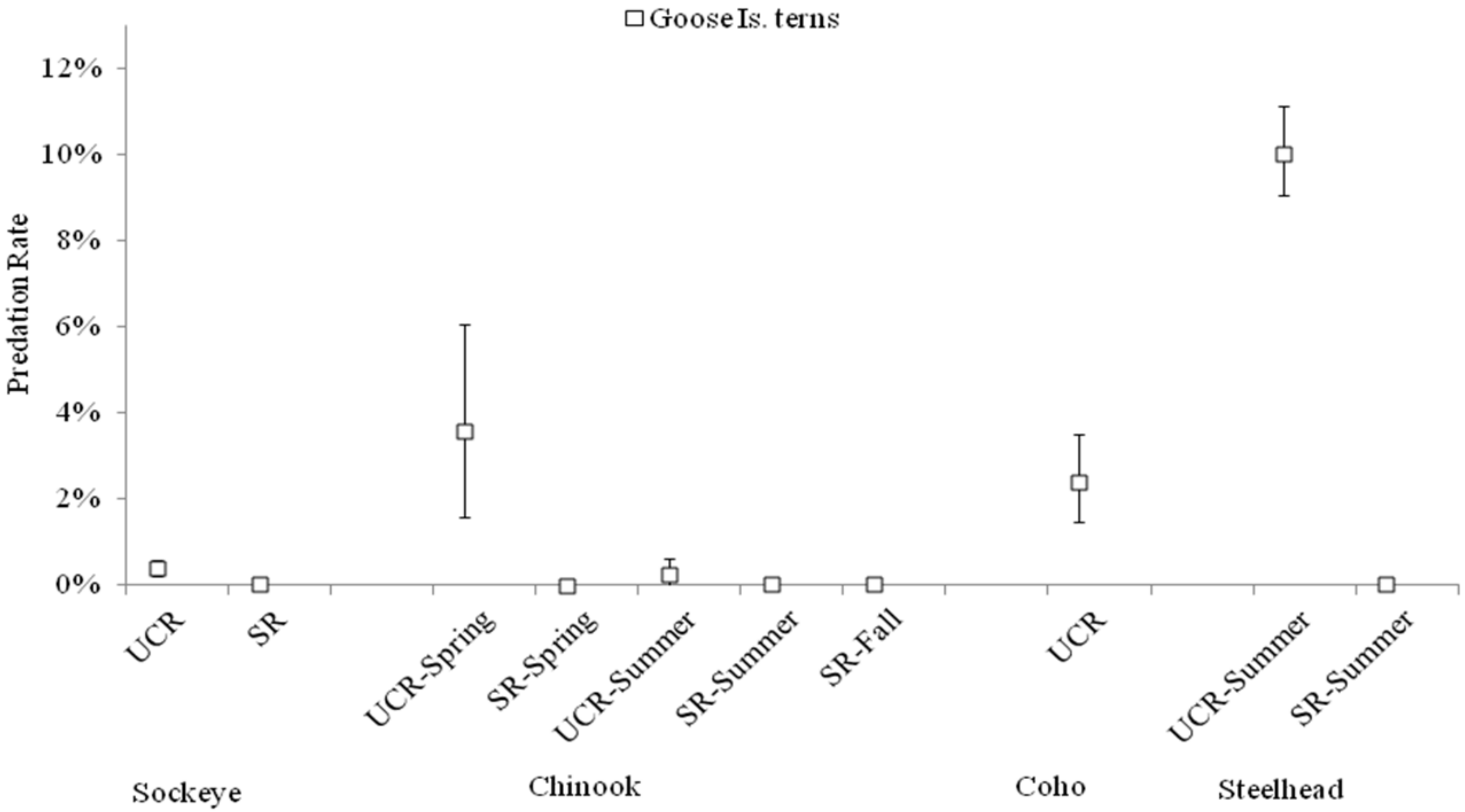


Figure 3.5. Estimated predation rates of PIT-tagged smolts last interrogated passing Lower Monumental Dam on the Snake River or Rock Island Dam on the upper Columbia River by Caspian terns (terns) nesting on Goose Island in Potholes Reservoir during 2006-2009. The salmonid species (sockeye, Chinook, coho, steelhead), run-type (spring, summer, fall), and river-of-origin (UCR for upper Columbia River, SR for Snake River) for each PIT-tagged stock are provided. Only stocks with ≥ 500 PIT-tagged individuals interrogated in any given year (see Table 3.3) are presented. Error bars represent 95% confidence intervals.

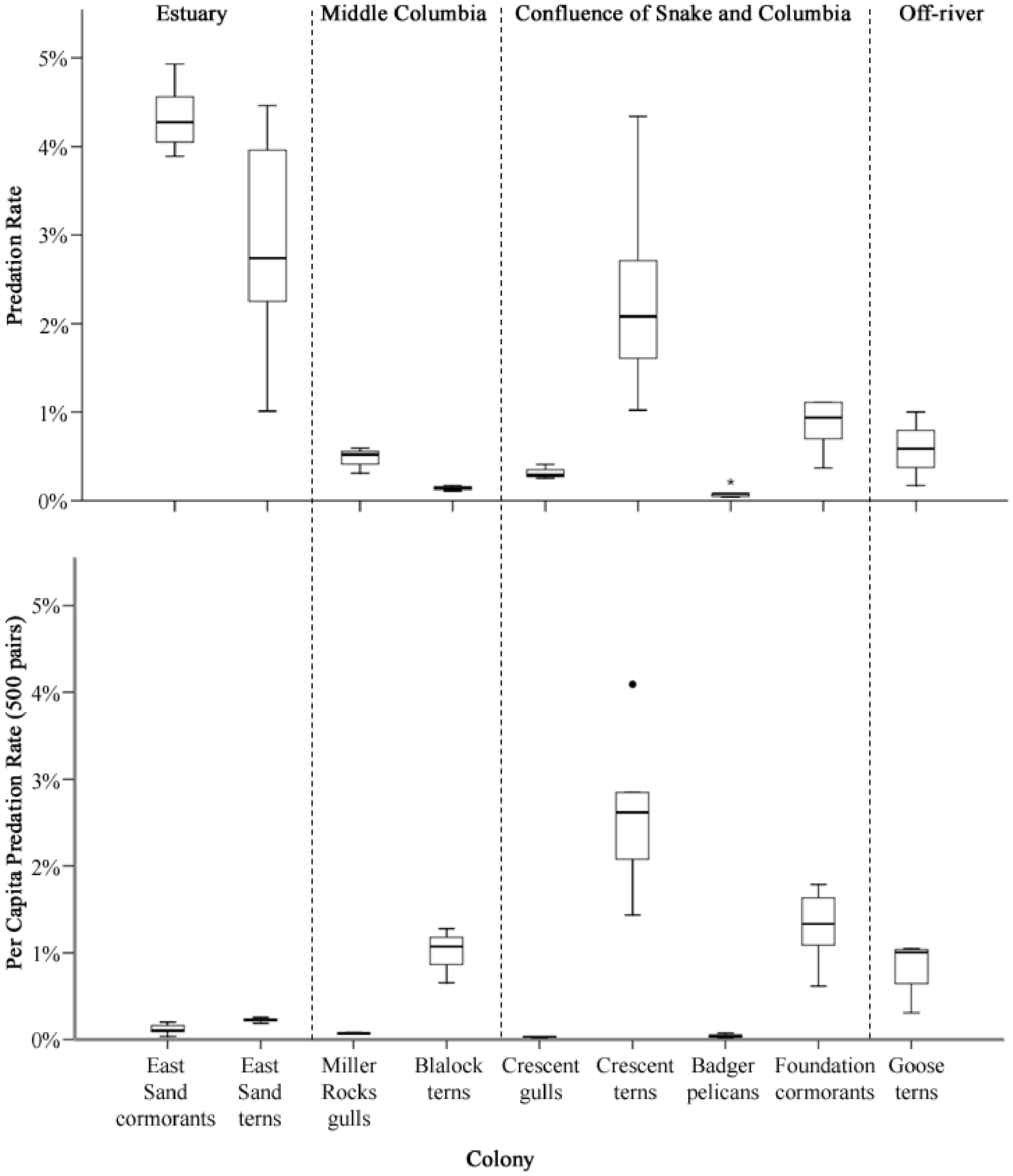


Figure 3.6. Box-and-whiskers plots of colony-wide predation rates (top) and per capita predation rates (bottom) on PIT-tagged smolts (Chinook, coho, sockeye and steelhead combined) by Caspian terns (terns), double-crested cormorants (cormorants), California and ring-billed gulls (gulls), and American white pelicans (pelicans) nesting at nine colonies in the Columbia River basin during 2004-2009. Per capita predation rates are based on a colony of 500 nesting pairs. Smolt availability is based on the number of PIT-tagged smolts last detected passing Bonneville and Sullivan dams for colonies in the estuary, McNary Dam for colonies in the middle Columbia River, and Lower Monumental and Rock Island dams for colonies near the confluence of the Snake and Columbia rivers or off-river (Goose Island in Potholes Reservoir).

Appendix A.1. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by double-crested cormorants nesting on East Sand Island during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at Sullivan Dam or Bonneville Dam that were subsequently consumed by cormorants. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year.

			East Sand Island Double-crested Cormorants						
River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Willamette	Chinook	Spring	-	-	-	0.4% (0.1-0.9)	1.6% (1.0-2.3)	0.7% (0.4-0.9)	0.9% (0.6-1.1)
		Fall	-	-	-	-	-	-	-
		Unknown	-	-	-	-	0.7% (0.2-1.4)	-	0.7% (0.2-1.4)
	Steelhead	Summer	-	-	-	0.6% (0.1-1.3)	4.3% (3.2-5.5)	4.4% (3.3-5.8)	3.4% (2.7-4.0)
		Winter	-	-	-	-	-	-	-
Middle Columbia	Chinook	Spring	1.3% (0.5-2.2)	1.0% (0.4-1.5)	2.4% (1.4-3.6)	0.5% (0.2-0.8)	3.1% (2.5-3.7)	2.5% (2.0-3.0)	2.0% (1.8-2.3)
		Summer	-	-	-	-	-	-	-
		Fall	-	-	-	-	7.7% (6.5-9.0)	7.8% (6.6-9.0)	7.7% (6.9-8.6)
		Unknown	-	-	-	-	-	-	-
	Coho	All	-	-	-	-	-	-	-
	Steelhead	Summer	-	-	-	1.3% (0.7-2.0)	6.6% (5.3-8.1)	6.5% (5.4-7.7)	5.1% (4.5-5.7)
		Winter	-	-	-	0.4% (0.1-1.0)	4.8% (3.6-6.1)	3.6% (2.4-5.0)	3.5% (2.8-4.3)
		Unknown	-	-	-	-	-	-	-
Snake	Chinook	Spring	3.7% (2.7-5.0)	2.6% (1.9-3.3)	3.0% (2.2-3.8)	0.9% (0.7-1.1)	1.7% (1.3-2.1)	3.3% (2.9-3.8)	2.1% (1.9-2.3)
		Summer	2.0% (0.8-3.3)	2.8% (1.8-3.8)	2.3% (1.1-3.6)	0.6% (0.3-0.9)	1.8% (1.2-2.6)	3.3% (2.6-4.0)	2.0% (1.7-2.3)
		Fall	1.2% (0.3-2.5)	1.9% (0.9-3.1)	1.5% (1.0-2.1)	0.7% (0.3-1.3)	1.3% (1.1-1.5)	2.2% (1.9-2.5)	1.6% (1.5-1.8)
		Unknown	3.3% (1.7-5.2)	1.8% (1.2-2.4)	2.6% (2.1-3.1)	0.9% (0.7-1.2)	1.7% (1.4-2.1)	3.6% (3.2-4.0)	2.3% (2.1-2.5)
	Coho	All	-	-	-	-	-	-	-
	Sockeye	All	-	-	-	-	-	2.7% (1.9-3.7)	2.7% (1.9-3.7)
	Steelhead	Summer	2.1% (0.7-4.0)	2.2% (0.9-3.7)	7.0% (4.9-9.2)	1.7% (1.3-2.3)	7.3% (6.6-8.1)	8.1% (7.4-8.8)	6.8% (6.4-7.3)
Upper Columbia	Chinook	Spring	2.9% (2.3-3.5)	2.0% (1.4-2.6)	2.3% (1.4-3.4)	1.0% (0.6-1.5)	1.9% (1.3-2.5)	1.4% (1.0-1.9)	2.1% (1.8-2.4)
		Summer	2.6% (2.0-3.2)	1.8% (1.1-2.6)	-	-	-	1.1% (0.5-1.9)	2.3% (1.9-2.8)
		Fall	-	-	-	1.2% (0.5-2.0)	1.7% (1.0-2.4)	1.6% (0.8-2.6)	1.5% (1.1-2.0)
		Unknown	-	-	-	-	-	-	-
	Coho	All	0.8% (0.1-1.8)	5.0% (2.8-7.4)	-	0.8% (0.4-1.3)	2.3% (1.6-3.0)	1.1% (0.7-1.5)	1.5%(1.2-1.8)
	Sockeye	All	-	-	-	1.5% (0.4-2.7)	2.5% (1.2-4.0)	3.5% (2.2-5.0)	2.5% (1.8-3.3)
	Steelhead	Summer	4.6% (3.7-5.7)	3.0% (2.4-3.7)	2.5% (1.6-3.5)	1.6% (1.1-2.3)	3.1% (2.2-4.0)	3.5% (2.6-4.6)	3.3% (2.9-3.7)

Appendix A.2. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by Caspian terns nesting on East Sand Island during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at Sullivan Dam or Bonneville Dam that were subsequently consumed by terns. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year.

			East Sand Island Caspian Terns						
River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Willamette	Chinook	Spring	-	-	-	1.0% (0.5-1.5)	3.1% (2.4-3.8)	1.2% (0.9-1.5)	1.7% (1.4-1.9)
		Fall	-	-	-	-	-	-	-
		Unknown	-	-	-	-	1.6% (0.8-2.5)	-	1.6% (0.8-2.5)
	Steelhead	Summer	-	-	-	12.6% (10.8-14.5)	13.2% (11.7-14.9)	8.6% (7.3-10.1)	11.5% (10.6-12.5)
		Winter	-	-	-	-	-	-	-
Middle Columbia	Chinook	Spring	1.1% (0.7-1.5)	1.6% (1.0-2.1)	1.8% (1.0-2.7)	0.9% (0.6-1.2)	4.0% (3.5-4.5	2.6% (2.1-3.0)	2.3% (2.2-2.6)
		Summer	-	-	-	-	-	-	-
		Fall	-	-	-	-	0.6% (0.3-0.9)	1.7% (1.3-2.2)	1.2% (0.9-1.5)
		Unknown	-	-	-	-	-	-	-
	Coho	All	-	-	-	-	-	-	-
	Steelhead	Summer	-	-	-	13.0% (11.5-14.6)	9.9% (8.6-11.2)	10.1% (9.1-11.2)	10.8 (10.1-11.6)
		Winter	-	-	-	10.6% (8.5-12.8)	8.6% (7.4-9.9)	11.8% (9.9-13.8)	9.9% (9.0-10.9)
		Unknown	-	-	-	-	-	-	-
Snake	Chinook	Spring	3.1% (2.5-3.7)	2.0% (1.6-2.5)	2.4% (1.8-3.0)	2.3% (2.0-2.5)	1.8% (1.5-2.1)	3.5% (3.2-3.8)	2.5% (2.4-2.7)
		Summer	3.9% (2.9-5.0)	2.2% (1.5-3.0)	2.1% (1.2-3.2)	2.0% (1.6-2.5)	1.5% (1.0-2.0)	2.7% (2.1-3.2)	2.3% (2.1-2.6)
		Fall	0.8% (0.2-1.5)	0.9% (0.3-1.5)	1.7% (1.3-2.2)	2.3% (1.6-3.0)	1.3% (1.2-1.5)	1.4% (1.2-1.6)	1.4% (1.3-1.5)
		Unknown	2.2% (1.4-3.1)	2.0% (1.6-2.6)	1.8% (1.5-2.2)	1.9% (1.6-2.2)	2.0% (1.7-2.3)	3.7% (3.3-4.0)	2.4% (2.3-2.6)
	Coho	All	-	-	-	-	-	-	-
	Sockeye	All	-	-	-	-	-	0.8% (0.4-1.3)	0.8% (0.4-1.3)
	Steelhead	Summer	18.1% (15.2-20.9)	20.0% (17.0-23.1)	19.4% (16.3-22.9)	16.0% (15.0-17.2)	10.1% (9.6-10.7)	10.4% (9.9-10.9)	11.4% (11.1-11.8)
Upper Columbia	Chinook	Spring	2.5% (2.2-2.8)	1.4% (1.0-1.9)	2.2% (1.4-3.2)	1.3% (0.9-1.7)	1.4% (1.0-1.8)	2.4% (2.0-2.9)	2.0% (1.9-2.2)
		Summer	5.0% (4.6-5.5)	1.4% (0.9-1.9)	-	-	-	1.8% (1.1-2.6)	4.0% (3.7-4.4)
		Fall	-	-	-	1.7% (1.0-2.5)	1.4% (0.9-2.0)	2.0% (1.2-2.9)	1.6% (1.3-2.1)
		Unknown	-	-	-	-	-	-	-
	Coho	All	11.9% (9.9-14.0)	7.0% (4.8-9.3)	-	5.9% (5.-6.8)	4.1% (3.3-4.8)	4.8% (4.1-5.5)	5.7% (5.2-6.1)
	Sockeye	All	-	-	-	1.3% (0.5-2.1)	0.7% (0.1-1.3)	0.5% (0.1-1.1)	0.8% (0.5-1.2)
	Steelhead	Summer	9.9% (9.1-10.7)	10.5% (9.6-11.4)	16.3% (14.2-18.6)	11.0% (9.9-12.2)	11.5% (10.2-12.9)	13.8% (12.4-15.4)	11.4% (11.0-11.9)

Appendix A.3. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by California and ring-billed gulls nesting on Miller Rocks during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at McNary Dam that were subsequently consumed by gulls. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year. NA denotes years PIT tags were not recovered following the nesting season.

Miller Rocks Island California and Ring-billed Gulls

River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Middle Columbia	Chinook	Spring	NA	NA	NA	0.3% (0.0-0.7)	0.7% (0.2-1.3)	0.4% (0.1-0.8)	0.5% (0.2-0.7)
		Summer	NA	NA	NA	-	-	-	-
		Fall	NA	NA	NA	-	-	-	-
	Coho	All	NA	NA	NA	-	-	-	-
	Steelhead	Summer	NA	NA	NA	0.6% (0.1-1.1)	0.7% (0.3-1.3)	1.0% (0.5-1.5)	0.8% (0.5-1.1)
		Winter	NA	NA	NA	-	-	-	-
		Unknown	NA	NA	NA	-	-	-	-
Snake	Chinook	Spring	NA	NA	NA	0.2% (0.1-0.2)	0.4% (0.3-0.4)	0.3% (0.3-0.4)	0.3% (0.2-0.3)
		Summer	NA	NA	NA	0.1% (0.1-0.2)	0.3% (0.1-0.5)	0.3% (0.2-0.4)	0.2% (0.2-0.3)
		Fall	NA	NA	NA	0.5% (0.3-0.7)	0.4% (0.3-0.5)	0.6% (0.5-0.6)	0.5% (0.4-0.5)
		Unknown	NA	NA	NA	0.2% (0.1-0.2)	0.4% (0.3-0.5)	0.4% (0.3-0.4)	0.3% (0.2-0.3)
	Coho	All	NA	NA	NA	-	-	-	-
	Sockeye	All	NA	NA	NA	-	-	1.3% (0.8-1.9)	1.3% (0.8-1.9)
	Steelhead	Summer	NA	NA	NA	1.5% (1.2-1.8)	1.4% (1.2-1.6)	1.5% (1.3-1.7)	1.4% (1.3-1.6)
Upper Columbia	Chinook	Spring	NA	NA	NA	0.2% (0.2-0.3)	0.3% (0.2-0.4)	0.3% (0.2-0.5)	0.3% (0.2-0.3)
		Summer	NA	NA	NA	-	<0.1%	0.3% (0.1-0.5)	0.2% (0.1-0.4)
		Fall	NA	NA	NA	0.3% (0.2-0.5)	0.4% (0.2-0.7)	0.5% (0.2-0.7)	0.4% (0.3-0.5)
		Unknown	NA	NA	NA	-	-	-	-
	Coho	All	NA	NA	NA	1.2% (0.8-1.6)	0.7% (0.4-0.9)	1.7% (1.2-2.1)	1.2% (0.9-1.4)
	Sockeye	All	NA	NA	NA	0.3% (0.1-0.7)	0.5% (0.1-1.0)	0.4% (0.2-0.7)	0.4% (0.2-0.6)
	Steelhead	Summer	NA	NA	NA	1.3% (0.9-1.7)	1.1% (0.7-1.5)	1.0% (0.6-1.4)	1.1% (0.9-1.4)

Appendix A.4. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by Caspian terns nesting on the Blalock Islands during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at McNary Dam that were subsequently consumed by terns. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year. NA denotes years PIT tags were not recovered following the nesting season.

Blalock Island Caspian Terns

River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Middle Columbia	Chinook	Spring	NA	NA	NA	0.2% (0.1-0.5)	0.1% (1.0-0.3)	0.1% (1.0-0.3)	0.1% (0.1-0.2)
		Summer	NA	NA	NA	-	-	-	-
		Fall	NA	NA	NA	-	-	-	-
		Unknown	NA	NA	NA	-	-	-	-
	Coho	All	NA	NA	NA	-	-	-	-
	Steelhead	Summer	NA	NA	NA	0.1% (0.1-0.4)	0.3% (0.1-0.7)	0.4% (0.2-0.8)	0.3% (0.2-0.5)
		Winter	NA	NA	NA	-	-	-	-
Snake	Chinook	Spring	NA	NA	NA	0.1% (0.0-0.2)	0.1% (0.1-0.1)	0.2% (0.2-0.2)	0.1% (0.1-0.1)
		Summer	NA	NA	NA	< 0.1%	0.1% (0.0-0.2)	0.2% (0.1-0.3)	0.1% (0.1-0.2)
		Fall	NA	NA	NA	0.1% (0.0-0.1)	0.1% (0.0-0.1)	< 0.1%	< 0.1%
		Unknown	NA	NA	NA	0.1% (0.1-0.1)	0.1% (0.0-0.1)	0.2% (0.1-0.2)	0.1% (0.1-0.1)
	Coho	All	NA	NA	NA	-	-	-	-
	Sockeye	All	NA	NA	NA	-	-	< 0.1%	< 0.1%
	Steelhead	Summer	NA	NA	NA	0.6% (0.4-0.8)	0.5% (0.4-0.6)	0.4% (0.3-0.5)	0.5% (0.4-0.5)
Upper Columbia	Chinook	Spring	NA	NA	NA	< 0.1%	< 0.1%	0.1% (0.0-0.1)	0.1% (0.0-0.1)
		Summer	NA	NA	NA	-	< 0.1%	< 0.1%	< 0.1%
		Fall	NA	NA	NA	0.1% (0.0-0.2)	0.2% (0.1-0.4)	0.1% (0.0 - 0.1)	0.1% (0.1-0.2)
		Unknown	NA	NA	NA	-	-	-	-
	Coho	All	NA	NA	NA	0.1% (0.0-0.2)	0.1% (0.0-0.2)	0.1% (0.2-0.2)	0.1% (0.0-0.2)
	Sockeye	All	NA	NA	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%
	Steelhead	Summer	NA	NA	NA	0.7% (0.4 - 1.0)	0.4% (0.2 - 0.6)	0.3% (0.1 - 0.5)	0.4% (0.3-0.6)

Appendix A.5. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by Caspian terns nesting on Crescent Island during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at Lower Monumental Dam or Rock Island Dam that were subsequently consumed by terns. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year.

			Crescent Island Caspian Terns						
River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Snake	Chinook	Spring	2.0% (1.7-2.3)	0.8% (0.7-1.0)	1.5% (1.2-1.8)	0.3% (0.2-0.4)	0.6% (0.5-0.8)	1.0% (0.8-1.3)	1.0% (0.9-1.1)
		Summer	3.0% (2.4-3.6)	0.6% (0.4-0.8)	0.9% (0.4-1.4)	0.2% (0.1-0.5)	0.7% (0.4-1.0)	1.0% (0.7-1.4)	1.0% (0.9-1.2)
		Fall	1.9% (1.6-2.2)	1.8% (1.4-2.1)	1.0% (0.9-1.3)	0.6% (0.3-1.0)	1.1% (0.9-1.3)	0.7% (0.6-0.9)	1.1% (1.0-1.2)
		Unknown	2.0% (1.6-2.5)	1.0% (0.8-1.2)	1.3% (1.1-1.6)	0.2% (0.1-0.3)	0.5% (0.4-0.6)	1.3% (1.1-1.6)	1.1% (1.0-1.2)
	Coho	All	9.1% (6.8-11.7)	2.6% (1.1-4.2)	-	-	-	-	6.1% (4.6-7.6)
	Sockeye	All	-	-	2.8% (1.0-5.0)	-	1.0% (0.2-2.0)	0.6% (0.3-1.0)	1.1% (0.7-1.5)
	Steelhead	Summer	22.2% (21.2-23.2)	10.0% (9.4-10.7)	6.9% (6.0-8.1)	2.8% (2.5-3.1)	4.1% (3.8-4.6)	3.2% (2.9-3.5)	7.4% (7.2-7.7)
Upper Columbia	Chinook	Spring	< 0.1%	-	-	-	-	< 0.1%	< 0.1%
		Summer	0.5% (0.5-0.6)	0.2% (0.1-0.2)	-	-	-	0.1% (0.0-0.3)	0.4% (0.4-0.5)
		Fall	-	-	-	-	-	-	-
		Unknown	0.3% (0.1-0.5)	0.2% (0.1-0.4)	< 0.1%	0.1% (0.0-0.2)	0.3% (0.1-0.6)	-	0.2% (0.1-0.2)
	Coho	All	-	-	0.5% (0.0-1.7)	-	2.6% (1.1-4.5)	0.8% (0.0-1.8)	1.3% (0.6-2.1)
	Sockeye	All	0.8% (0.2-1.5)	0.3% (0.0-0.8)	0.1% (0.1-0.4)	0.3% (0.0-0.6)	< 0.1%	0.1% (0.0-0.3)	0.2% (0.1-0.3)
	Steelhead	Summer	3.6% (3.0-4.2)	3.0% (2.5-3.4)	2.1% (1.4-2.9)	1.7% (1.2-2.2)	2.0% (1.6-2.5)	1.6% (1.2-1.9)	2.3% (2.1-2.5)

Appendix A.6. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by California and ring-billed gulls nesting on Crescent Island during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at Lower Monumental Dam or Rock Island Dam that were subsequently consumed by gulls. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year. NA denotes years PIT tags were not recovered following the nesting season.

			Crescent Island California and Ring-billed Gulls						
River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Snake	Chinook	Spring	NA	NA	NA	0.1% (0.0-0.1)	0.1% (0.1-0.2)	0.1% (0.1-0.2)	0.1% (0.1-0.1)
		Summer	NA	NA	NA	0.2% (0.0-0.4)	0.2% (0.1-0.3)	0.1% (0.0-0.3)	0.2% (0.1-0.2)
		Fall	NA	NA	NA	< 0.1%	0.1% (0.1-0.1)	0.1% (0.1-0.1)	0.1% (0.1-0.1)
		Unknown	NA	NA	NA	0.2% (0.1-0.3)	0.1% (0.1-0.2)	0.2% (0.1-0.2)	0.2% (0.1-0.2)
	Coho	All	NA	NA	NA	-	-	-	-
	Sockeye	All	NA	NA	NA	-	0.2% (0.0-0.5)	0.4% (0.1-0.7)	0.3% (0.1-0.6)
	Steelhead	Summer	NA	NA	NA	0.6% (0.4-0.8)	0.6% (0.5-0.7)	0.7% (0.6-0.8)	0.7% (0.6-0.7)
Upper Columbia	Chinook	Spring	NA	NA	NA	-	-	< 0.1%	< 0.1%
		Summer	NA	NA	NA	-	-	< 0.1%	< 0.1%
		Fall	NA	NA	NA	-	-	-	-
		Unknown	NA	NA	NA	< 0.1%	< 0.1%	-	< 0.1%
	Coho	All	NA	NA	NA	-	< 0.1%	0.2% (0.1-0.8)	0.1% (0.0-0.4)
	Sockeye	All	NA	NA	NA	0.2% (0.0-0.4)	0.1% (0.0-0.3)	0.0% (0.1-0.2)	0.1% (0.0-0.2)
	Steelhead	Summer	NA	NA	NA	0.9% (0.5-1.3)	0.4% (0.2-0.5)	1.0% (0.7-1.3)	0.7% (0.6-0.9)

Appendix A.7. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by American white pelicans nesting on Badger Island during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at Lower Monumental Dam or Rock Island Dam that were subsequently consumed by pelicans. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year. NA denotes years PIT tags were not recovered following the nesting season.

Badger Island American White Pelicans									
River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Snake	Chinook	Spring	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	0.2% (0.1-0.2)	< 0.1%
		Summer	NA	< 0.1%	< 0.1%	< 0.1%	0.1% (0.0-0.2)	0.2% (0.1-0.3)	0.1% (0.0-0.1)
		Fall	NA	< 0.1%	< 0.1%	< 0.1%	0.1% (0.0-0.1)	0.1% (0.0-0.1)	0.1% (0.0-0.1)
		Unknown	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	0.1% (0.1-0.2)	< 0.1%
	Coho	All	NA	< 0.1%	-	-	-	-	< 0.1%
	Sockeye	All	NA	-	< 0.1%	-	< 0.1%	< 0.1%	< 0.1%
	Steelhead	Summer	NA	0.2% (0.1-0.2)	0.1% (0.1-0.2)	0.3% (0.2-0.4)	0.2% (0.1-0.2)	0.3% (0.3-0.4)	0.2% (0.2-0.3)
Upper Columbia	Chinook	Spring	NA	-	-	-	-	< 0.1%	< 0.1%
		Summer	NA	< 0.1%	-	-	-	< 0.1%	< 0.1%
		Fall	NA	-	-	-	-	-	-
		Unknown	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	-	< 0.1%
	Coho	All	NA	-	< 0.1%	-	< 0.1%	< 0.1%	< 0.1%
	Sockeye	All	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
	Steelhead	Summer	NA	0.1% (0.0-0.2)	< 0.1%	0.1% (0.0-0.2)	0.1% (0.0-0.2)	0.3% (0.1-0.4)	0.1% (0.1-0.2)

Appendix A.8. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by double-crested cormorants nesting on Foundation Island during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at Lower Monumental Dam or Rock Island Dam that were subsequently consumed by cormorants. Dashed lines denote PIT-tagged stocks excluded from analysis due to < 500 individuals interrogated in that year. NA denotes years PIT tags were not recovered following the nesting season.

Foundation Island Double-crested Cormorants

River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Snake	Chinook	Spring	0.4% (0.2-0.5)	0.8% (0.6-1.0)	0.5% (0.4-0.6)	0.7% (0.6-0.9)	1.0% (0.8-1.1)	0.9% (0.8-1.1)	0.7% (0.7-0.8)
		Summer	0.6% (0.3-0.9)	0.7% (0.5-0.9)	0.8% (0.5-1.1)	0.7% (0.3-1.1)	1.0% (0.7-1.4)	0.8% (0.5-1.1)	0.8% (0.7-0.9)
		Fall	0.1% (0.1-0.2)	0.1% (0.1-0.3)	0.1% (0.0-0.1)	0.9% (0.4-1.4)	0.4% (0.3-0.5)	0.5% (0.4-0.6)	0.3% (0.2-0.4)
		Unknown	0.5% (0.3-0.7)	0.9% (0.7-1.1)	0.5% (0.4-0.6)	0.9% (0.7-1.1)	1.0% (0.9-1.2)	0.9% (0.8-1.1)	0.8% (0.7-0.8)
	Coho	All	0.5% (0.0-1.2)	1.5% (0.5-2.8)	-	-	-	-	1.0% (0.4-1.6)
	Sockeye	All	-	-	0.9% (0.2-1.8)	-	1.1% (0.3-2.0)	2.1% (1.4-2.7)	1.7% (1.2-2.2)
	Steelhead	Summer	1.7% (1.5-2.0)	1.9% (1.7-2.1)	2.3% (2.1-2.6)	2.3% (2.0-2.7)	2.3% (2.1-2.6)	1.7% (1.6-1.9)	2.0% (1.9-2.1)
Upper Columbia	Chinook	Spring	0.3% (0.0-0.9)	-	-	-	-	< 0.1%	0.1% (0.0-0.4)
		Summer	< 0.1%	0.1% (0.0-0.1)	-	-	-	< 0.1%	0.1% (0.0-0.1)
		Fall	-	-	-	-	-	-	-
		Unknown	0.2% (0.0-0.4)	0.1% (0.0-0.2)	0.1% (0.0-0.2)	< 0.1%	0.1% (0.0-0.2)	-	0.1% (0.0-0.1)
	Coho	All	-	-	< 0.1%	-	< 0.1%	< 0.1%	< 0.1%
	Sockeye	All	< 0.1%	< 0.1%	< 0.1%	0.1% (0.0-0.4)	< 0.1%	< 0.1%	< 0.1%
	Steelhead	Summer	< 0.1%	0.1% (0.0-0.1%)	0.1% (0.0-0.3%)	< 0.1%	0.1% (0.1-0.3)	0.1% (0.0-0.2)	0.1% (0.1-0.1)

Appendix A.9. Annual predation rates (95% confidence intervals) of PIT-tagged salmonid stocks by Caspian terns nesting on Goose Island in Potholes Reservoir during 2004-2009 and across all years evaluated (study-period). Predation rates are based on the number of smolts interrogated at passing Lower Monumental Dam or Rock Island Dam that were subsequently consumed by terns. Dashed lines denote PIT-tagged stocks excluded from analysis due to < <500 individuals interrogated in that year. NA denotes years PIT tags were not recovered following the nesting season.

			Goose Island (Potholes Reservoir) Caspian Terns						
River of Origin	Species	Run	2004	2005	2006	2007	2008	2009	Study-period
Snake	Chinook	Spring	NA	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
		Summer	NA	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
		Fall	NA	NA	< 0.1%	0.1% (0.0-0.6)	< 0.1%	< 0.1%	< 0.1%
		Unknown	NA	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
	Coho	All	NA	NA	-	-	-	-	-
	Sockeye	All	NA	NA	< 0.1%	-	0.2% (0.0-0.6)	< 0.1%	< 0.1%
	Steelhead	Summer	NA	NA	< 0.1%	< 0.1%	< 0.1%	< 0.1%	< 0.1%
Upper Columbia	Chinook	Spring	NA	NA	-	-	-	3.6% (1.6-6.1)	3.6% (1.6-6.1)
		Summer	NA	NA	-	-	-	0.2% (0.0-0.6)	0.2% (0.0-0.6)
		Unknown	NA	NA	0.4% (0.1-0.7)	0.2% (0.0-0.6)	0.6% (0.3-1.0)	-	0.4% (0.2-0.6)
	Coho	All	NA	NA	2.6% (0.8-4.8)	-	2.3% (0.8-4.1)	2.3% (0.7-4.4)	2.4% (1.5-3.5)
	Sockeye	All	NA	NA	0.3% (0.1-0.6)	0.6% (0.2-1.3)	0.4% (0.1-0.8)	0.2% (0.0-0.6)	0.4% (0.2-0.6)
	Steelhead	Summer	NA	NA	5.5% (4.4-6.7)	9.1% (6.3-14.0)	7.5% (6.5-8.5)	15.7% (13.6-18.2)	10.0% (9.1-11.1)

Appendix B.1. Predation rates (95% confidence intervals) of hatchery and wild PIT-tagged smolts by avian predators in the Columbia River basin during 2004. Predation rates were calculated if study-period predation rates for a specific stock were ≥ 2.0% (see Figures 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in 2004. Dashed lines denote stocks that had ≤ 500 PIT-tagged individuals interrogated in 2004, but were evaluated in at least one year during the 2004-2009 study period (Table 3.7).

East Sand Island cormorants

			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Middle Columbia	Chinook	Spring	-	-	
Snake			3.9% (2.7 - 5.3)	3.1% (1.0 - 5.8)	ND
Upper Columbia			-	-	
Snake		Summer	-	-	
Middle Columbia	Steelhead	Summer	-	-	
Snake			-	-	

East Sand Island terns

River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Middle Columbia	Chinook	Spring	-	-	
Snake			3.6% (2.9 - 4.4)	0.9% (0.2 - 1.7)	H
Upper Columbia			-	-	
Snake		Summer	-	-	
Middle Columbia	Steelhead	Summer	-	-	
Snake			-	-	

Crescent Island terns

River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Snake	Steelhead	Summer	23.3% (22.3 - 24.4)	19.3% (18.1 - 20.6)	H
Upper Columbia	Steelhead	Summer	3.6% (3.0 - 4.4)	3.3% (2.2 - 4.7)	ND

Foundation Island cormorants

River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Snake	Steelhead	Summer	1.9% (1.6 - 2.2)	1.2% (0.9 - 1.6)	H

Goose Island terns

River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Upper Columbia	Steelhead	Summer	-	-	

[a] Significance levels:
H = Predation rate of hatchery smolts was significantly higher than wild smolts
W = Predation rate of wild smolts was significantly higher than hatchery smolts
ND = Difference in predation rates between hatchery and wild smolts was not significant

Appendix B.2. Predation rates (95% confidence intervals) of hatchery and wild PIT-tagged smolts by avian predators in the Columbia River basin during 2005. Predation rates were calculated if study-period predation rates for a specific stock were ≥ 2.0% (see Figures 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in 2005. Dashed lines denote stocks that had < 500 PIT-tagged individuals interrogated in 2005, but were evaluated in at least one year during the 2004-2009 study period (Table 3.7).

East Sand Island cormorants

			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Middle Columbia	Chinook	Spring	1.1% (0.5 - 1.8)	0.6% (0.0 - 1.6)	ND
Snake			-	-	
Upper Columbia			-	-	
Snake		Summer	-	-	
Middle Columbia	Steelhead	Summer	-	-	
Snake			-	-	

East Sand Island terns

Middle Columbia	Chinook	Spring	1.9% (1.3 - 2.6)	0.2% (0.0 - 0.6)	H
Snake			-	-	
Upper Columbia			-	-	
Snake		Summer	-	-	
Middle Columbia	Steelhead	Summer	-	-	
Snake			-	-	

Crescent Island terns

Snake	Steelhead	Summer	10.6% (9.9 - 11.3)	7.9% (7.1 - 8.8)	H
Upper Columbia	Steelhead	Summer	3.1% (2.6 - 3.7)	2.1% (1.2 - 3.2)	ND

Foundation Island cormorants

Snake	Steelhead	Summer	2.0% (1.7 - 2.2)	1.8% (1.4 - 2.2)	ND

Goose Island terns

Upper Columbia	Steelhead	Summer	-	-	

[a] Significance levels:

H = Predation rate of hatchery smolts was significantly higher than wild smolts
W = Predation rate of wild smolts was significantly higher than hatchery smolts
ND = Difference in predation rates between hatchery and wild smolts was not significant

Appendix B.3. Predation rates (95% confidence intervals) of hatchery and wild PIT-tagged smolts by avian predators in the Columbia River basin during 2006. Predation rates were calculated if study-period predation rates for a specific stock were ≥ 2.0% (see Figures 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in 2006. Dashed lines denote stocks that had < 500 PIT-tagged individuals interrogated in 2006, but were evaluated in at least one year during the 2004-2009 study period (Table 3.7).

East Sand Island cormorants

			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Middle Columbia	Chinook	Spring	-	-	
Snake			3.1% (2.4 - 3.9)	2.1% (0.7 - 3.9)	ND
Upper Columbia			-	-	
Snake		Summer	-	-	
Middle Columbia	Steelhead	Summer	-	-	
Snake			-	-	

East Sand Island terns

Middle Columbia	Chinook	Spring	-	-	
Snake			2.6% (1.9 - 3.2)	1.1% (0.3 - 2.5)	H
Upper Columbia			-	-	
Snake		Summer	-	-	
Middle Columbia	Steelhead	Summer	-	-	
Snake			-	-	

Crescent Island terns

Snake	Steelhead	Summer	7.7% (6.7 - 9.0)	4.9% (3.9 - 6.1)	H
Upper Columbia	Steelhead	Summer	2.5% (1.6 - 3.6)	1.4% (0.4 - 2.6)	ND

d. Foundation Island cormorants

Snake	Steelhead	Summer	2.8% (2.5 - 3.1)	1.3% (1.0 - 1.6)	H

Goose Island terns

Upper Columbia	Steelhead	Summer	6.6% (5.3 - 8.2)	2.9% (1.6 - 4.3)	H

[a] Significance levels:
H = Predation rate of hatchery smolts was significantly higher than wild smolts
W = Predation rate of wild smolts was significantly higher than hatchery smolts
ND = Difference in predation rates between hatchery and wild smolts was not significant

Appendix B.4. Predation rates (95% confidence intervals) of hatchery and wild PIT-tagged smolts by avian predators in the Columbia River basin during 2007. Predation rates were calculated if study-period predation rates for a specific stock were ≥ 2.0% (see Figures 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in 2007. Dashed lines denote stocks that had < 500 PIT-tagged individuals interrogated in 2007, but were evaluated in at least one year during the 2004-2009 study period (Table 3.7).

East Sand Island cormorants					
			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Middle Columbia	Chinook	Spring	-	-	
Snake			0.8% (0.7 - 1.1)	1.4% (0.4 - 2.5)	ND
Upper Columbia			-	-	
Snake		Summer	0.6% (0.3 - 1.0)	0.6% (0.0 - 1.4)	ND
Middle Columbia	Steelhead	Summer	1.0% (0.4 - 1.8)	1.7% (0.6 - 3.1)	ND
Snake			1.4% (0.9 - 2.0)	2.6% (1.6 - 3.7)	W
East Sand Island terns					
Middle Columbia	Chinook	Spring	-	-	
Snake			2.3% (2.1 - 2.5)	1.1% (0.4 - 1.9)	H
Upper Columbia			-	-	
Snake		Summer	2.2% (1.7 - 2.7)	1.5% (0.8 - 2.4)	ND
Middle Columbia	Steelhead	Summer	13.2% (11.2 - 15.0)	12.8% (10.4 - 15.4)	ND
Snake			15.4% (14.2 - 16.6)	17.7% (15.8 - 19.7)	W
Crescent Island terns					
Snake	Steelhead	Summer	2.8% (2.4 - 3.1)	2.7% (2.1 - 3.4)	ND
Upper Columbia	Steelhead	Summer	1.6% (1.1 - 2.2)	1.9% (1.0 - 2.9)	ND
Foundation Island cormorants					
Snake	Steelhead	Summer	2.3% (2.0 - 2.7)	2.3% (1.7 - 2.9)	ND
Goose Island terns					
Upper Columbia	Steelhead	Summer	9.4% (6.3 - 14.4)	8.6% (4.9 - 13.9)	ND

[a] Significance levels:

H = Predation rate of hatchery smolts was significantly higher than wild smolts
W = Predation rate of wild smolts was significantly higher than hatchery smolts
ND = Difference in predation rates between hatchery and wild smolts was not significant

Appendix B.5. Predation rates (95% confidence intervals) of hatchery and wild PIT-tagged smolts by avian predators in the Columbia River basin during 2008. Predation rates were calculated if study-period predation rates for a specific stock were ≥ 2.0% (see Figures 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in 2008. Dashed lines denote stocks that had < 500 PIT-tagged individuals interrogated in 2008, but were evaluated in at least one year during the 2004-2009 study period (Table 3.7).

East Sand Island cormorants

			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Difference[a]
Middle Columbia	Chinook	Spring	-	-	
Snake			1.6% (1.3 - 2.0)	2.4% (1.2 - 3.8)	ND
Upper Columbia			2.1% (1.4 - 2.9)	1.4% (0.5 - 2.3)	ND
Snake		Summer	1.9% (1.2 - 2.8)	1.5% (0.5 - 2.8)	ND
Middle Columbia	Steelhead	Summer	7.1% (5.3 - 8.9)	5.8% (3.9 - 8.0)	ND
Snake			7.4% (6.6 - 8.3)	6.0% (4.7 - 7.6)	ND

East Sand Island terns

Middle Columbia	Chinook	Spring	-	-	
Snake			1.9% (1.5 - 2.2)	1.5% (0.7 - 2.4)	ND
Upper Columbia			1.6% (1.1 - 2.1)	0.9% (0.3 - 1.6)	ND
Snake		Summer	1.6% (1.0 - 2.3)	1.0% (0.3 - 1.9)	ND
Middle Columbia	Steelhead	Summer	10.7% (9.0 - 12.4)	10.8% (9.6 - 12.1)	ND
Snake			10.1% (9.5 - 10.6)	10.0% (9.5 - 10.6)	ND

Crescent Island terns

Snake	Steelhead	Summer	3.7% (3.3 - 4.1)	5.5% (4.8 - 6.2)	W
Upper Columbia	Steelhead	Summer	2.2% (1.8 - 2.8)	1.4% (0.8 - 2.1)	H

Foundation Island cormorants

Snake	Steelhead	Summer	2.5% (2.2 - 2.8)	1.7% (1.4 - 2.1)	H

Goose Island terns

Upper Columbia	Steelhead	Summer	8.1% (7.0 - 9.3)	5.7% (4.4 - 7.2)	H

[a] Significance levels:

H = Predation rate of hatchery smolts was significantly higher than wild smolts
W = Predation rate of wild smolts was significantly higher than hatchery smolts
ND = Difference in predation rates between hatchery and wild smolts was not significant

Appendix B.6. Predation rates (95% confidence intervals) of hatchery and wild PIT-tagged smolts by avian predators in the Columbia River basin during 2009. Predation rates were calculated if study-period predation rates for a specific stock were ≥ 2.0% (see Figures 3.2-3.5) and ≥ 500 PIT-tagged smolts of both rearing-types from that stock were available in 2009. Dashed lines denote stocks that had < 500 PIT-tagged individuals interrogated in 2009, but were evaluated in at least one year during the 2004-2009 study period (Table 3.7).

East Sand Island cormorants

			Predation rate		
River of Origin	Species	Run	Hatchery	Wild	Significance[a]
Middle Columbia	Chinook	Spring	2.5% (2.0 - 3.1)	2.8% (1.2 - 4.6)	ND
Snake			3.3% (2.9 - 3.8)	3.3% (1.9 - 4.9)	ND
Upper Columbia			1.3% (0.9 - 1.7)	1.9% (0.8 - 3.5)	ND
Snake		Summer	3.3% (2.5 - 4.1)	3.4% (1.8 - 5.1)	ND
Middle Columbia	Steelhead	Summer	7.0% (5.7 - 8.4)	5.3% (3.8 - 7.2)	ND
Snake			8.1% (7.4 - 8.9)	7.8% (6.4 - 9.4)	ND

East Sand Island terns

Middle Columbia	Chinook	Spring	2.7% (2.3 - 3.2)	1.8% (0.8 - 3.0)	ND
Snake			3.7% (3.3 - 4.0)	1.1% (0.5 - 1.9)	H
Upper Columbia			2.7% (2.1 - 3.2)	0.8% (0.2 - 1.6)	H
Snake		Summer	3.0% (2.4 - 3.6)	1.2% (0.4 - 2.0)	H
Middle Columbia	Steelhead	Summer	11.0% (9.7 - 12.4)	8.1% (6.4 - 9.9)	H
Snake			10.6% (9.3 – 12.0)	13.5% (12.1 - 15.2)	W

Crescent Island terns

Snake	Steelhead	Summer	2.9% (2.6 - 3.2)	4.7% (4.1 - 5.4)	W
Upper Columbia	Steelhead	Summer	1.6% (1.2 - 2.0)	1.6% (1.0 - 2.3)	ND

Foundation Island cormorants

Snake	Steelhead	Summer	1.8% (1.6 - 1.9)	1.7% (1.4 - 2.1)	ND

Goose Island terns

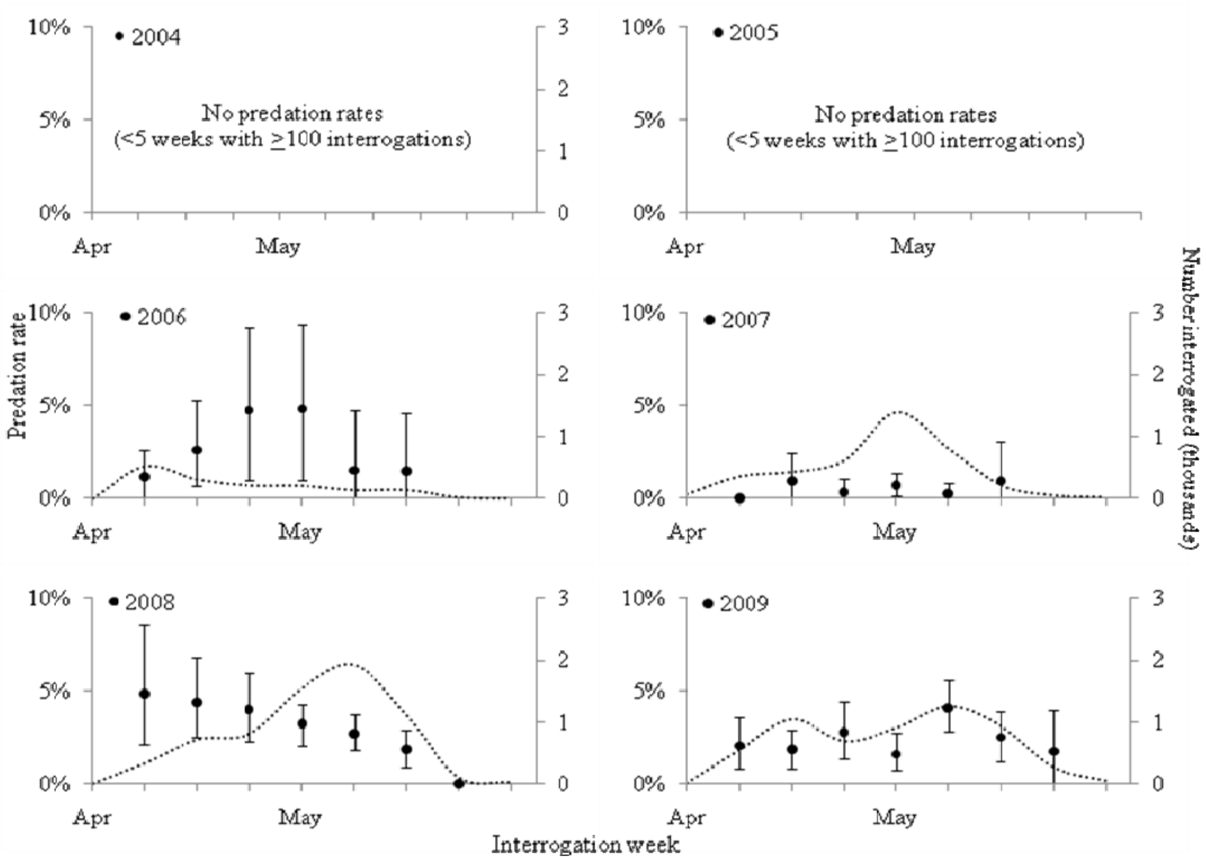
Upper Columbia	Steelhead	Summer	16.1% (13.8 - 18.8)	14.7% (11.8 - 18)	ND

[a] Significance levels:

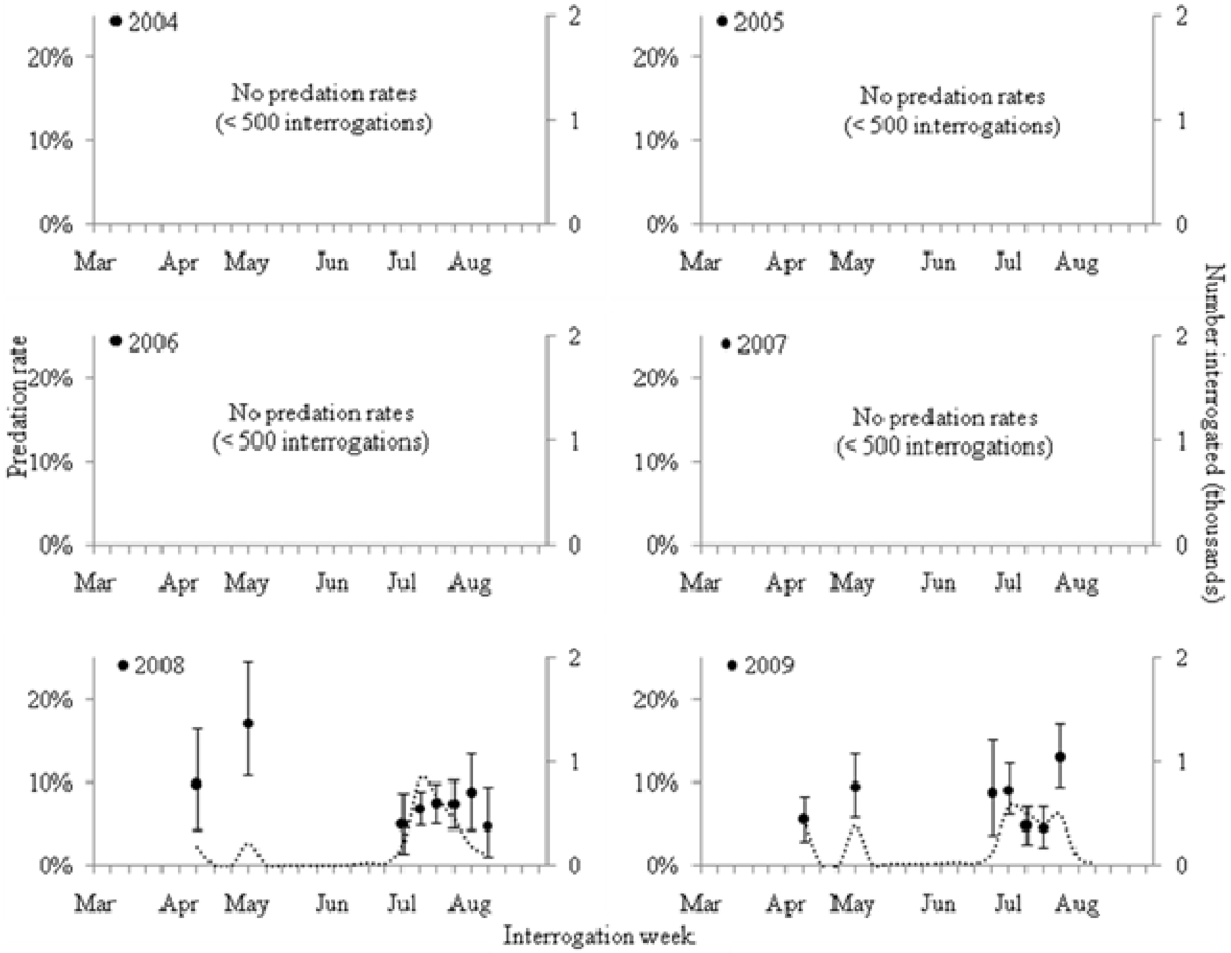
H = Predation rate of hatchery smolts was significantly higher than wild smolts

W = Predation rate of wild smolts was significantly higher than hatchery smolts

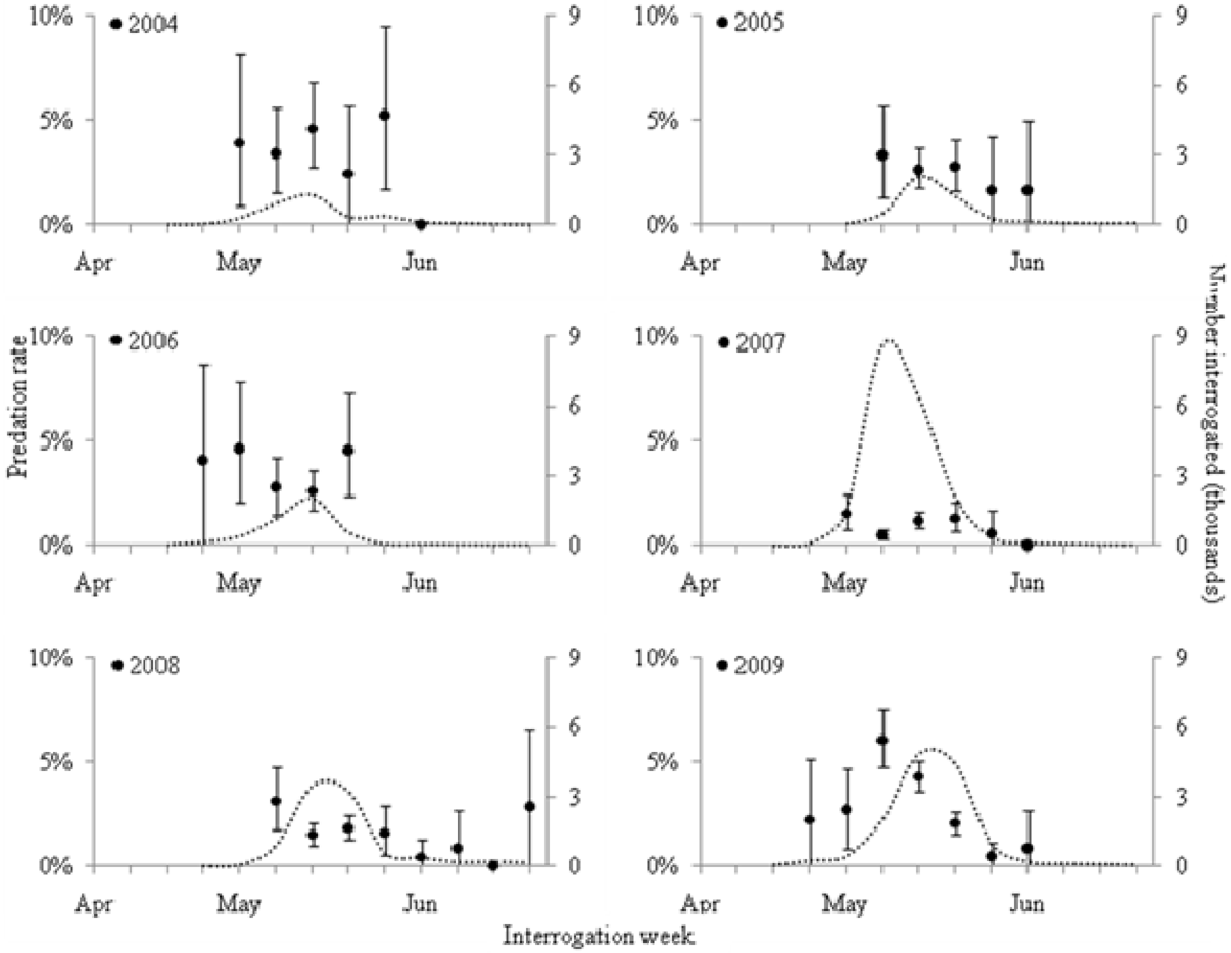
ND = Difference in predation rates between hatchery and wild smolts was not significant



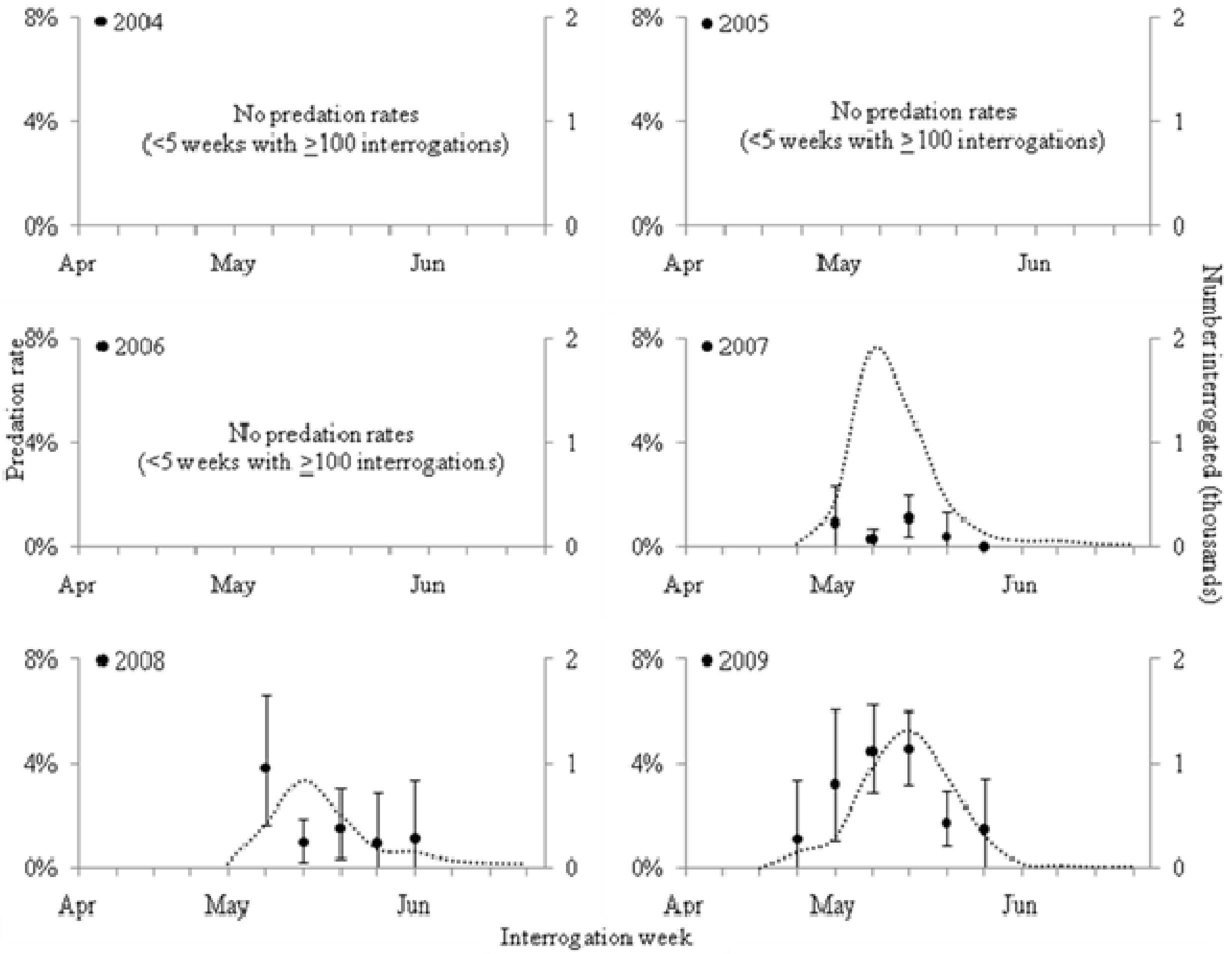
Appendix C.1. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia spring Chinook smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged spring Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia spring Chinook smolts detected at Bonneville Dam.



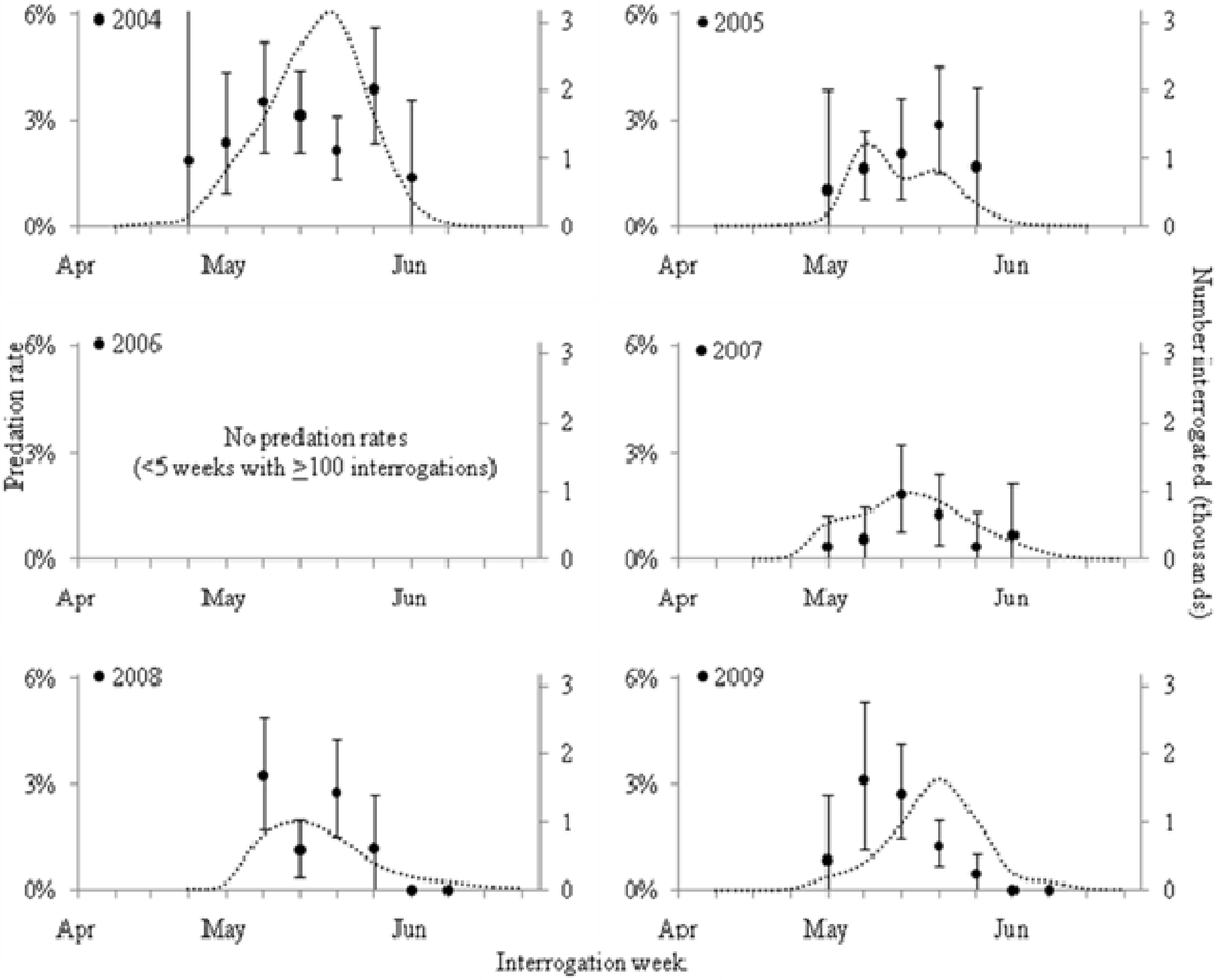
Appendix C.2. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia fall Chinook smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged fall Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia fall Chinook smolts detected at Bonneville Dam.



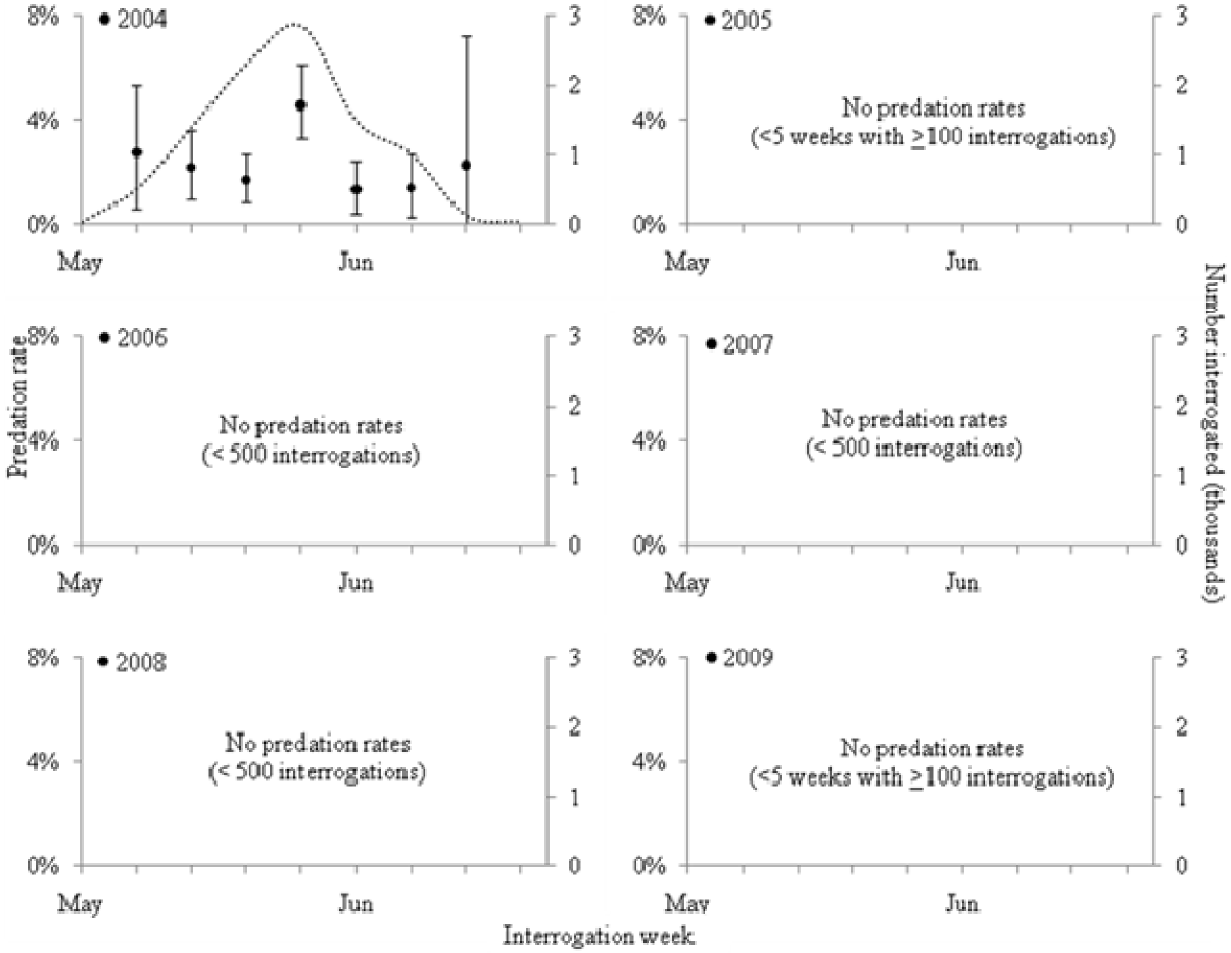
Appendix C.3. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River spring Chinook smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged spring Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged Snake River spring Chinook smolts detected at Bonneville Dam.



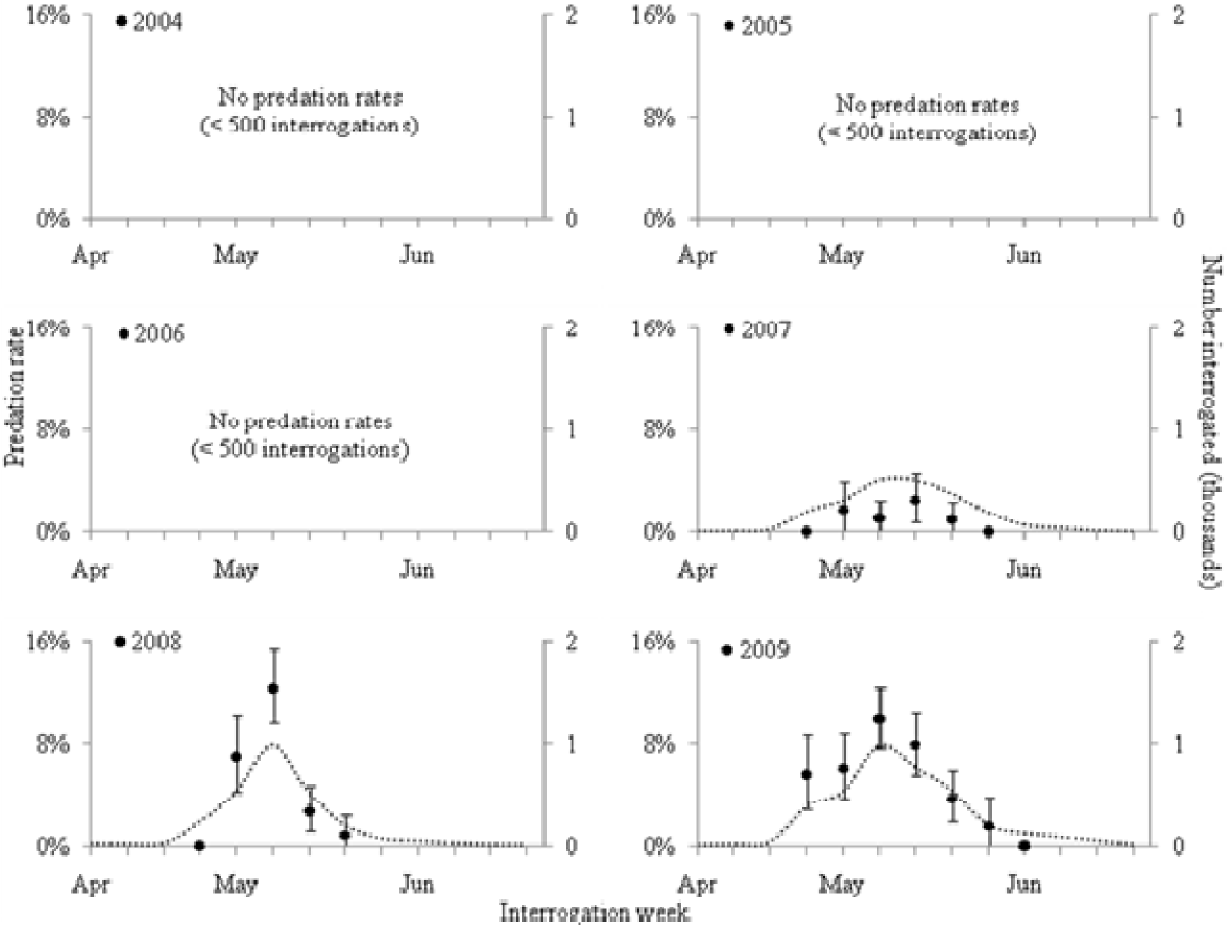
Appendix C.4. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River summer Chinook smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥ 100 PIT-tagged summer Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged Snake River summer Chinook smolts detected at Bonneville Dam.



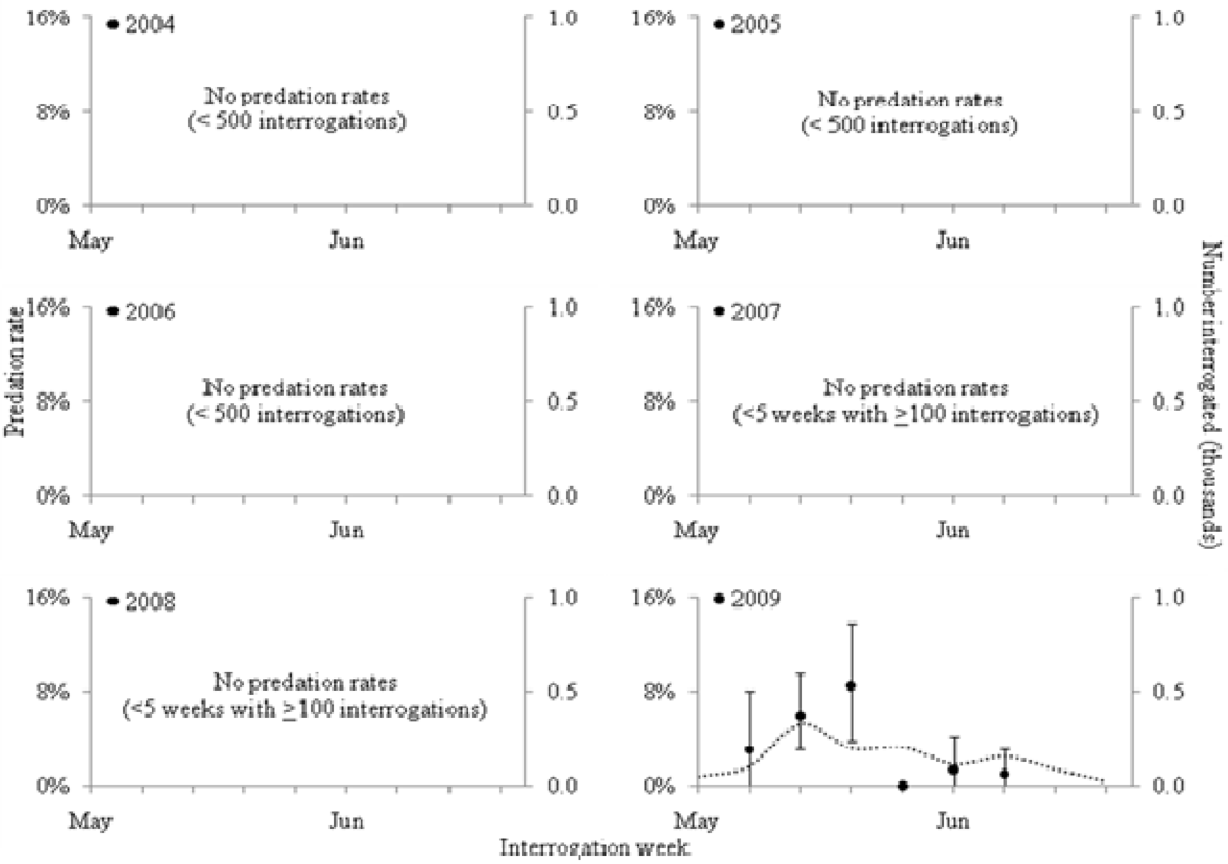
Appendix C.5. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia spring Chinook smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged spring Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia spring Chinook smolts detected at Bonneville Dam.



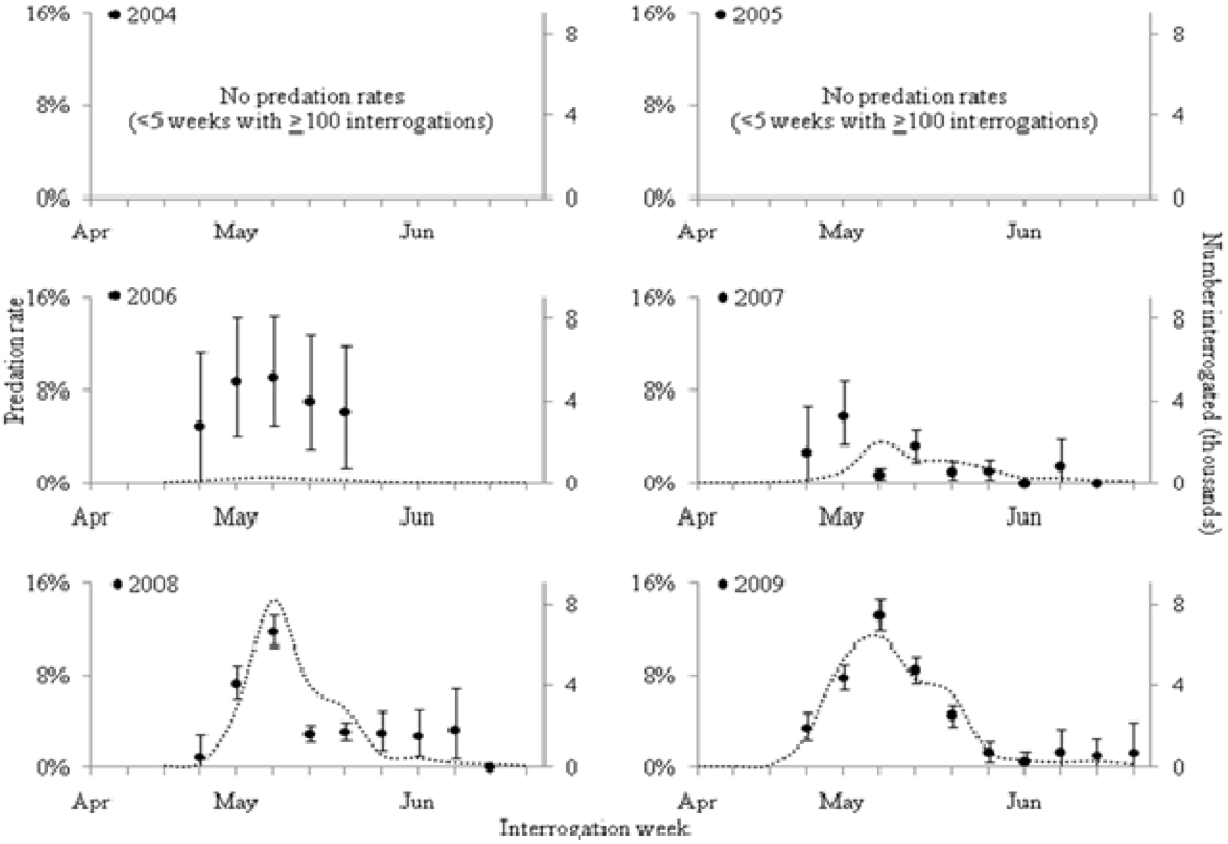
Appendix C.6. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia summer Chinook smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia summer Chinook smolts detected at Bonneville Dam.



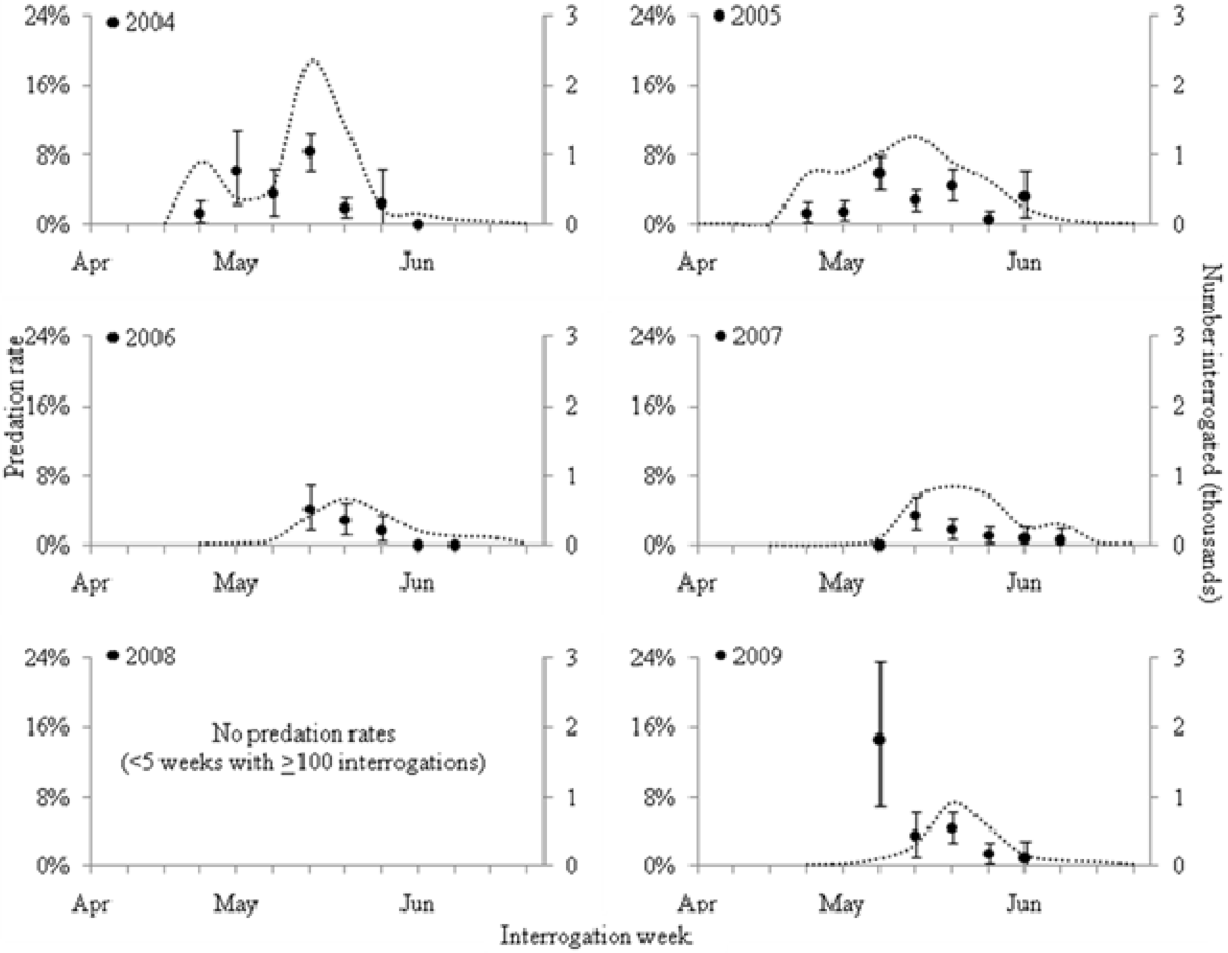
Appendix C.7. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia summer steelhead smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia summer steelhead smolts detected at Bonneville Dam.



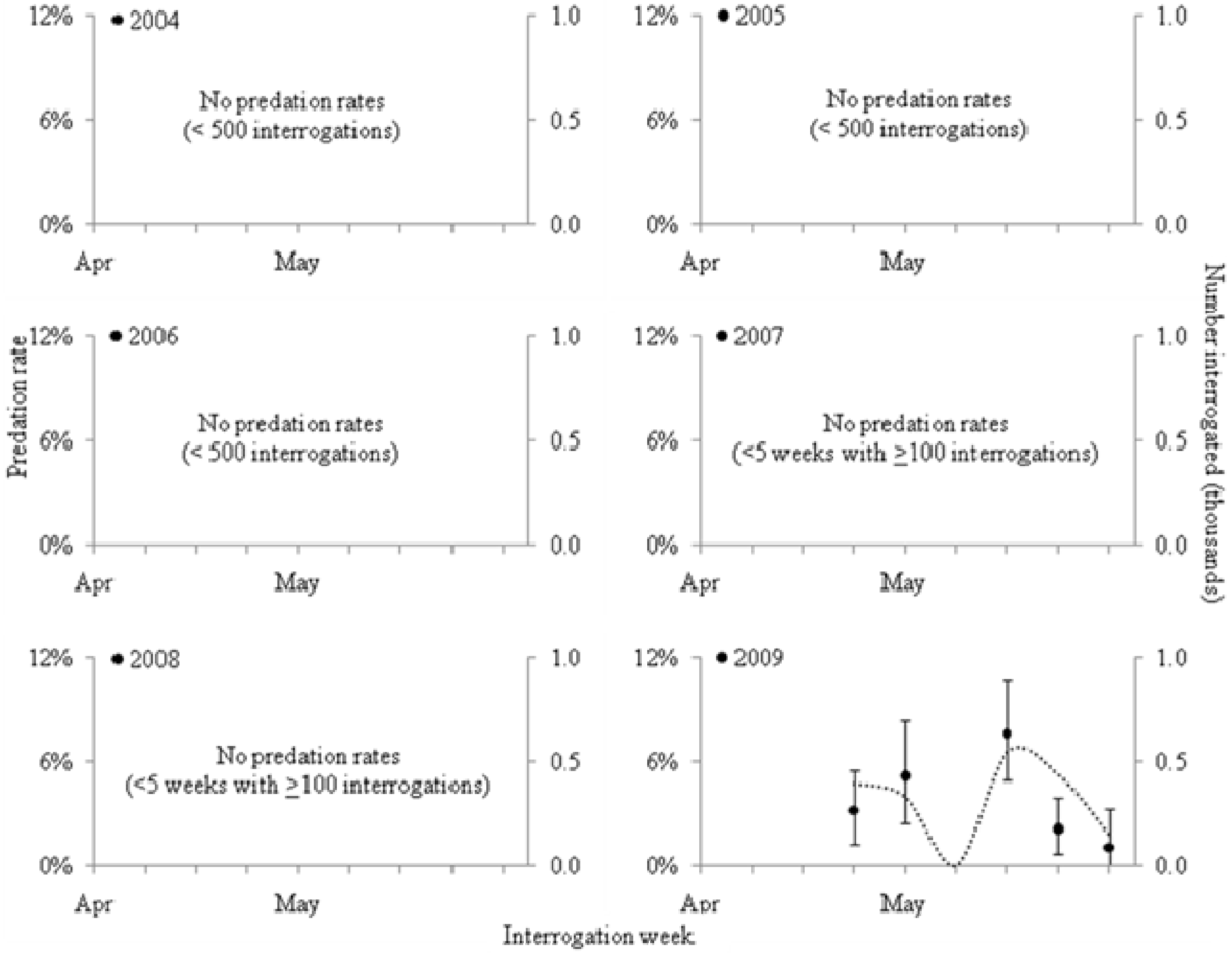
Appendix C.8. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia winter steelhead smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥ 100 PIT-tagged winter steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia winter steelhead smolts detected at Bonneville Dam.



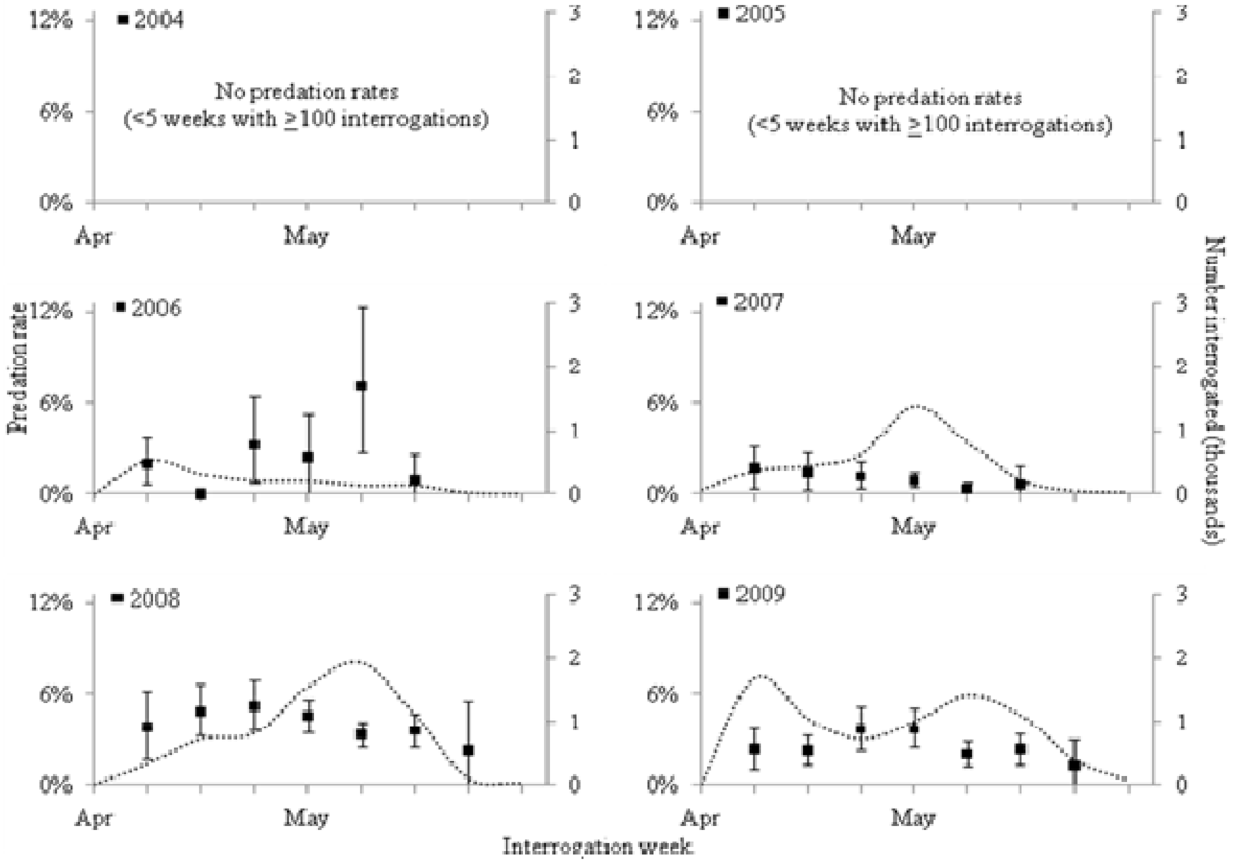
Appendix C.9. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River summer steelhead smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged Snake River summer steelhead smolts detected at Bonneville Dam.



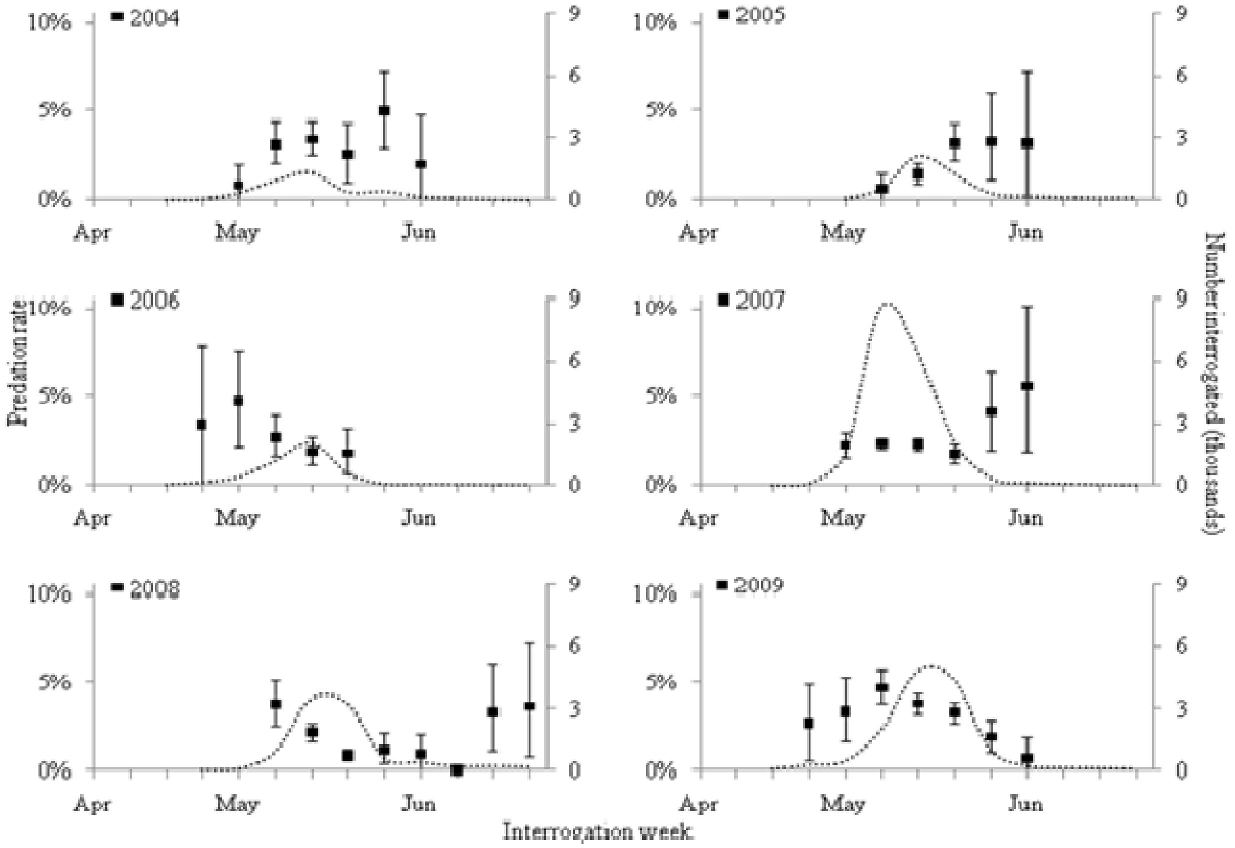
Appendix C.10. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia summer steelhead smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia summer steelhead smolts detected at Bonneville Dam.



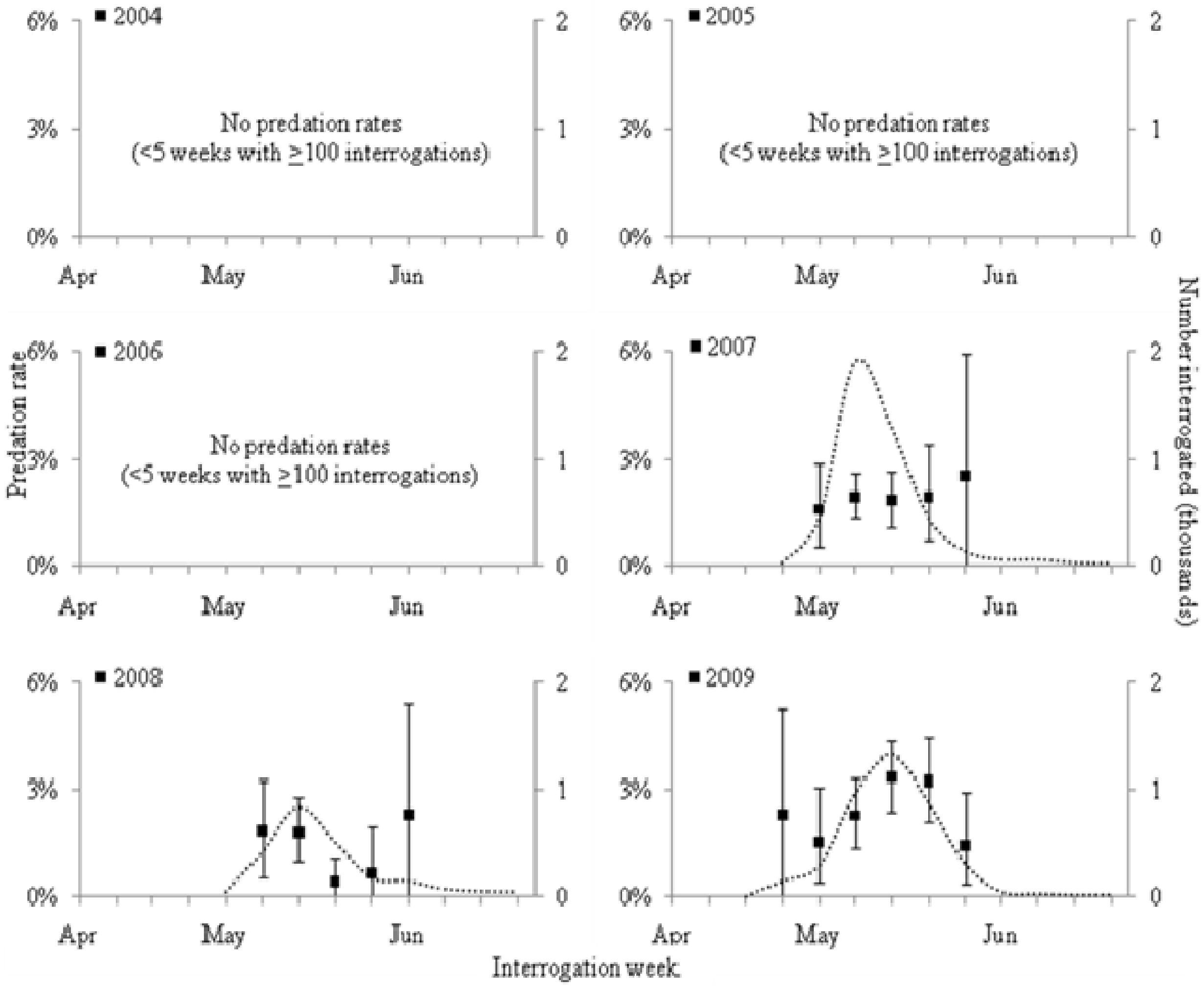
Appendix C.11. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Willamette summer steelhead smolts by double-crested cormorants nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Sullivan Dam were included. Dashed lines represent the number of PIT-tagged Willamette summer steelhead smolts detected at Sullivan Dam.



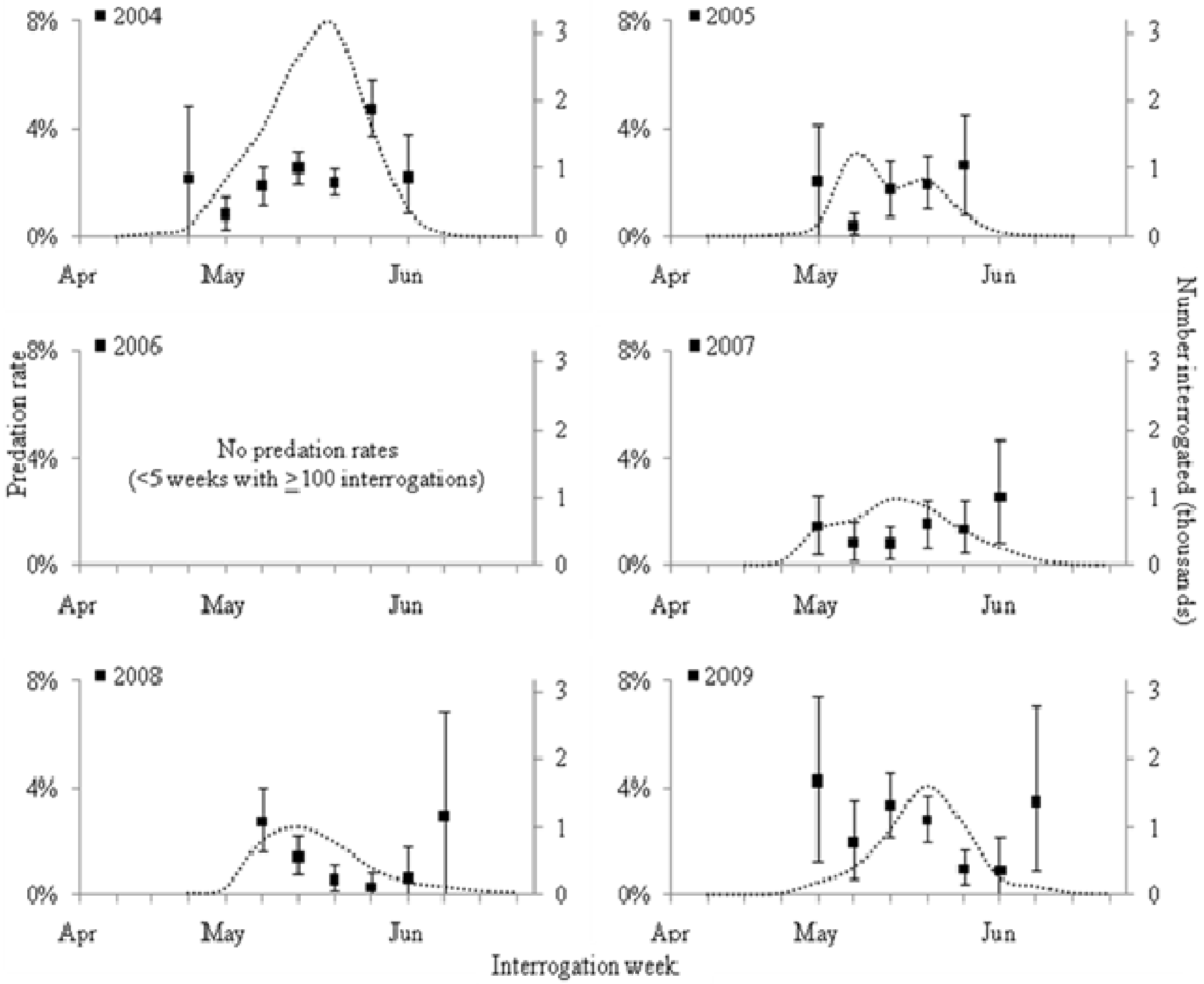
Appendix C.12. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia spring Chinook smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged spring Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia spring Chinook smolts detected at Bonneville Dam.



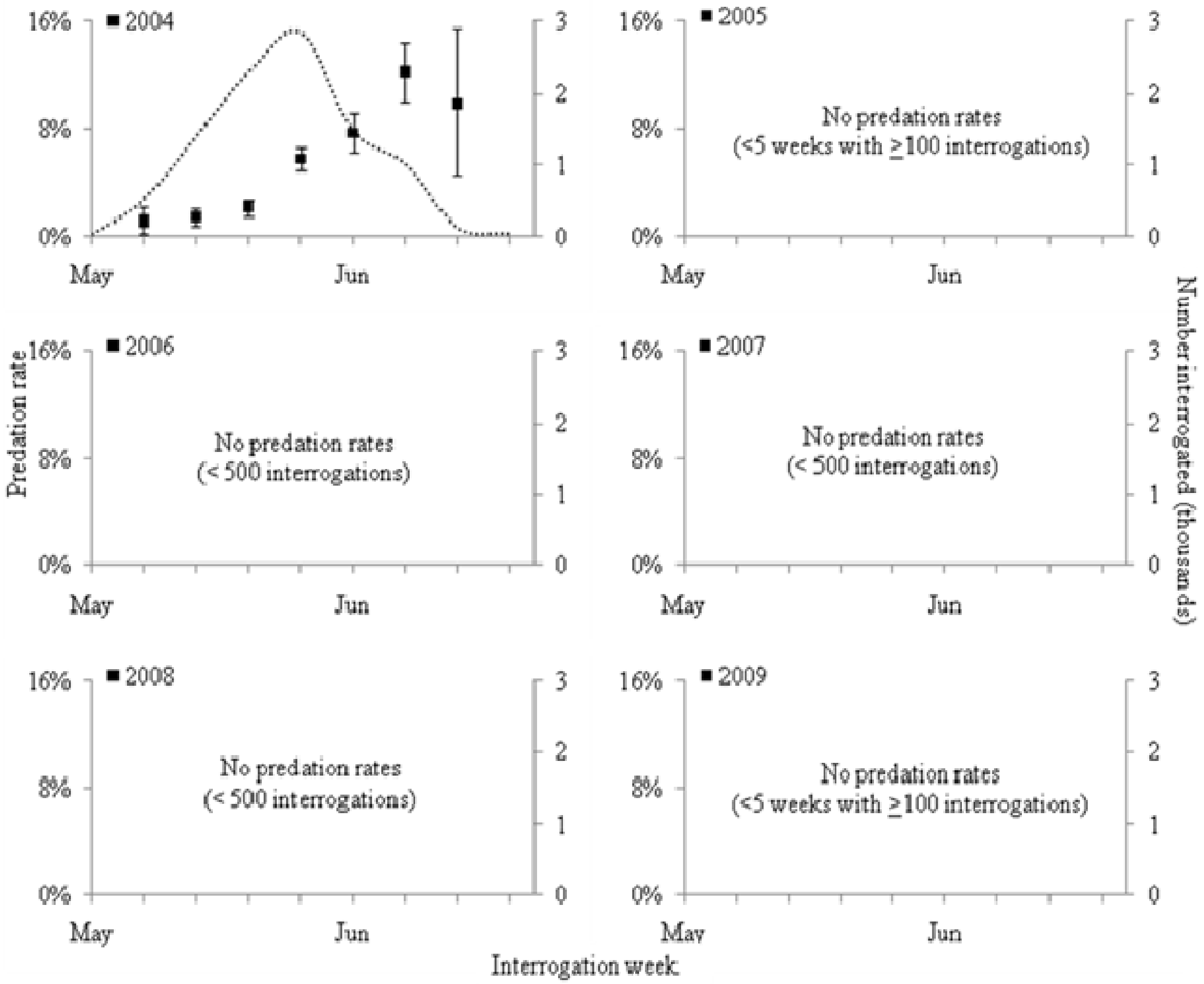
Appendix C.13. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River spring Chinook smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged spring Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged Snake River spring Chinook smolts detected at Bonneville Dam.



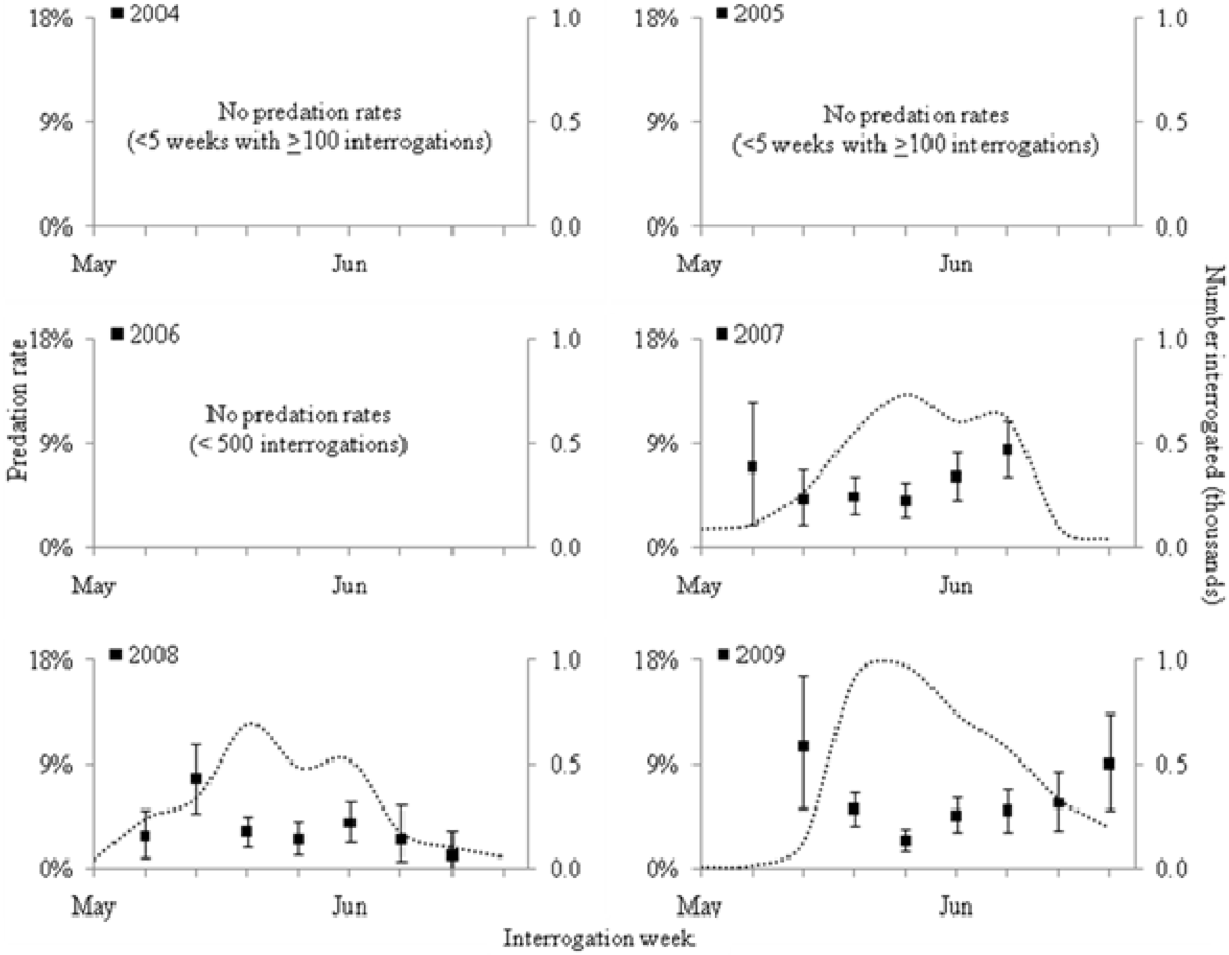
Appendix C.14. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River summer Chinook smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged Snake River summer Chinook smolts detected at Bonneville Dam.



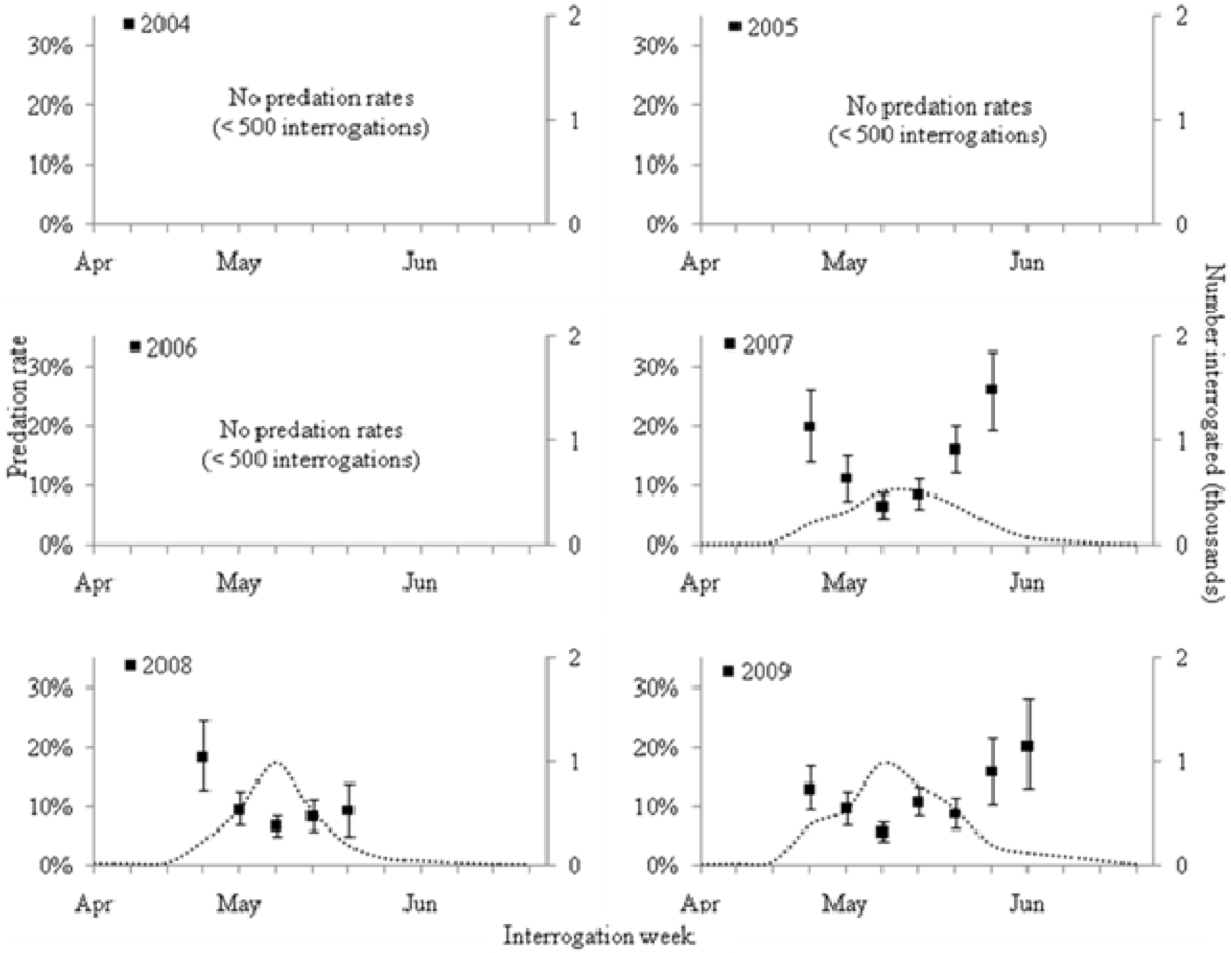
Appendix C.15. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia spring Chinook smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged spring Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia spring Chinook smolts detected at Bonneville Dam.



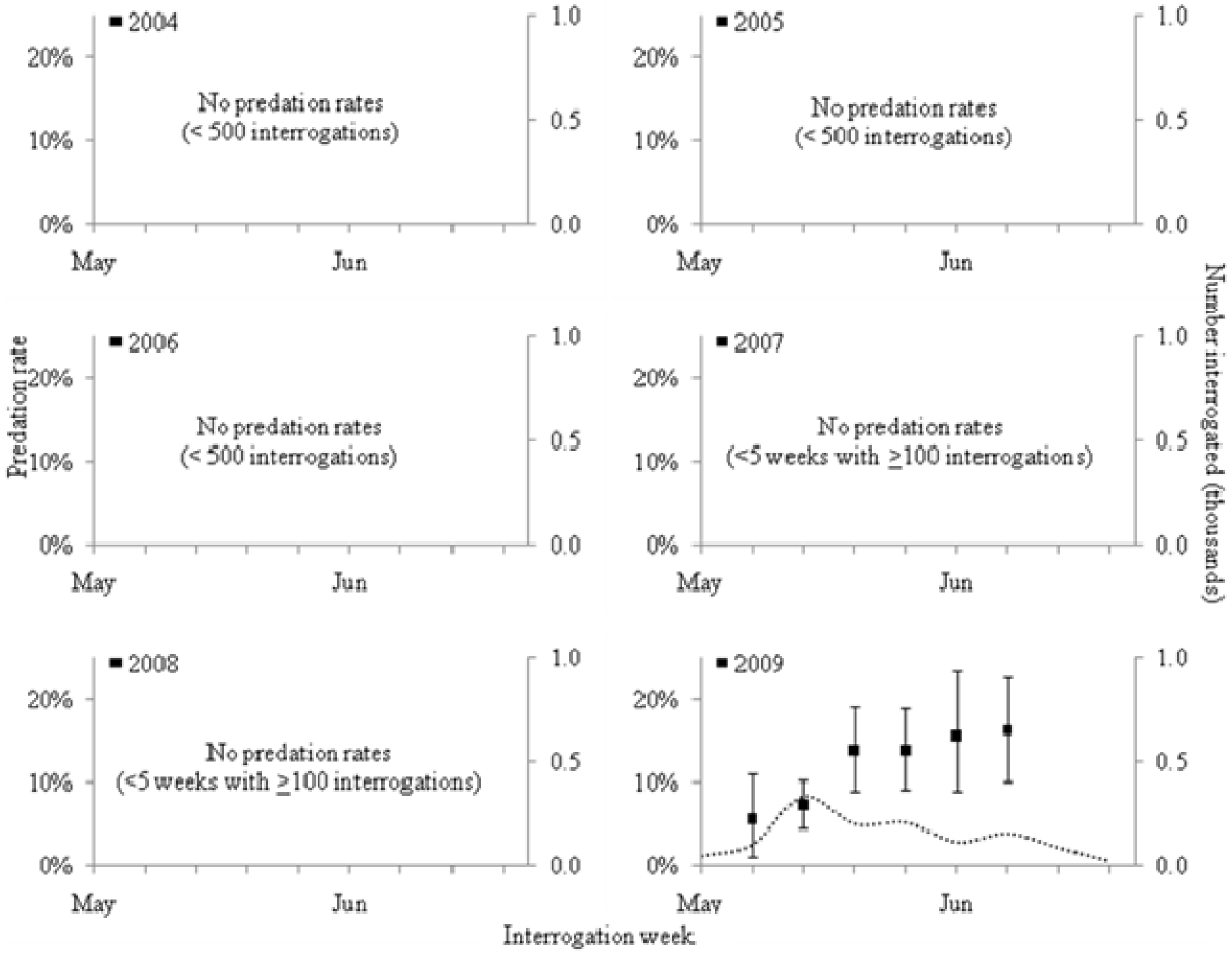
Appendix C.16. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia summer Chinook smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer Chinook were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia summer Chinook smolts detected at Bonneville Dam.



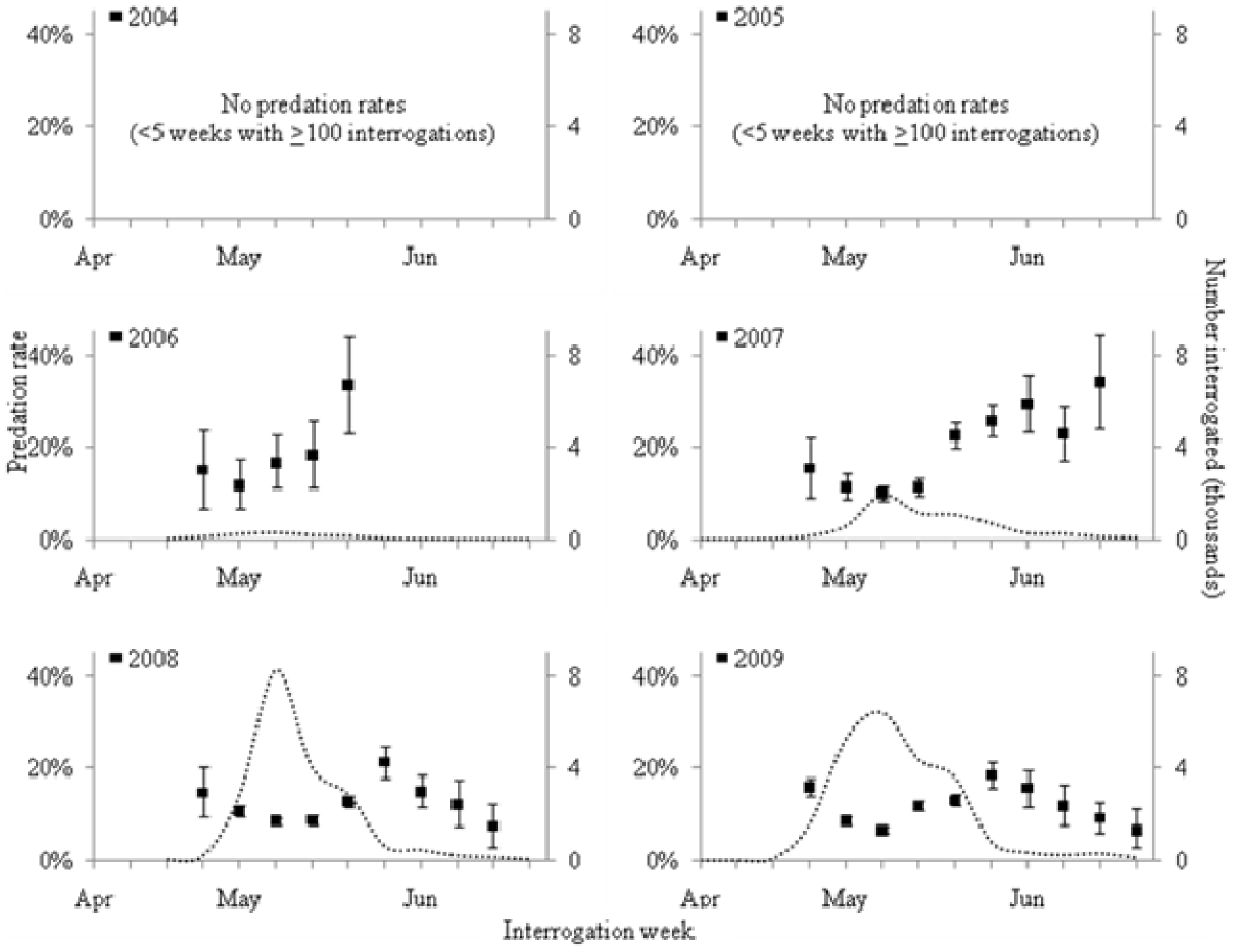
Appendix C.17. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia coho smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged coho were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia coho smolts detected at Bonneville Dam.



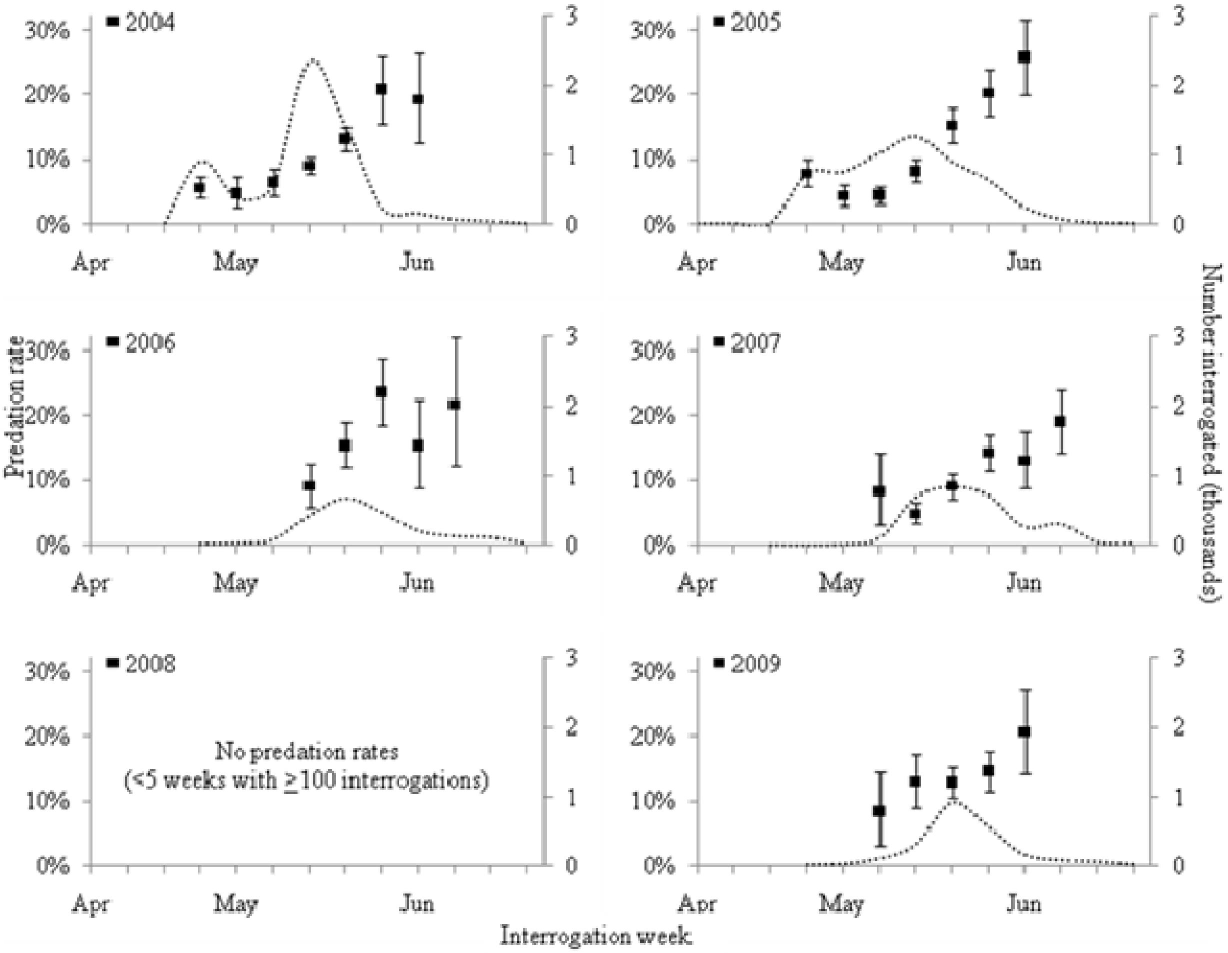
Appendix C.18. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia summer steelhead smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia summer steelhead smolts detected at Bonneville Dam.



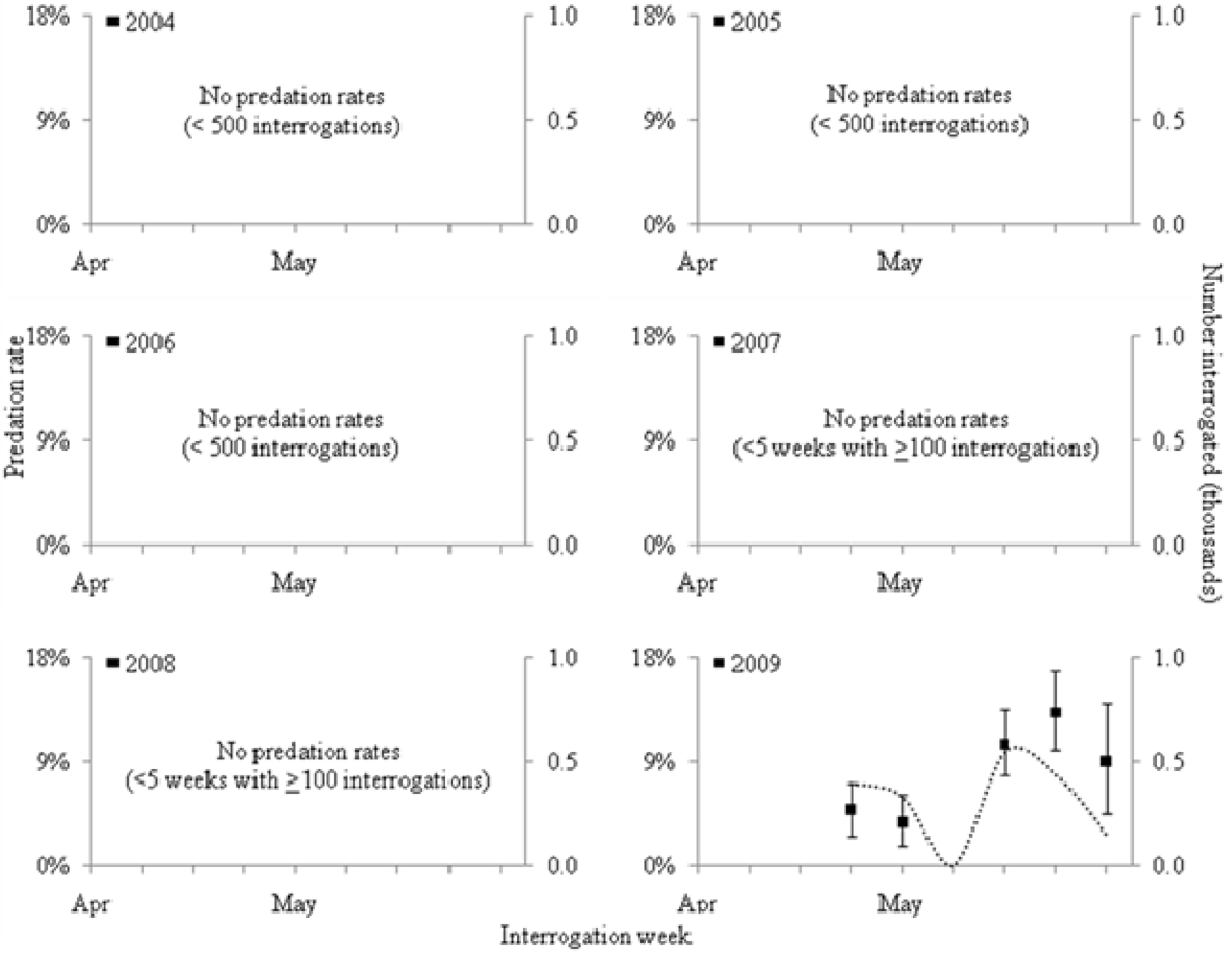
Appendix C.19. Weekly predation rates (with 95% confidence intervals) of PIT-tagged middle Columbia winter steelhead smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged winter steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged middle Columbia winter steelhead smolts detected at Bonneville Dam.



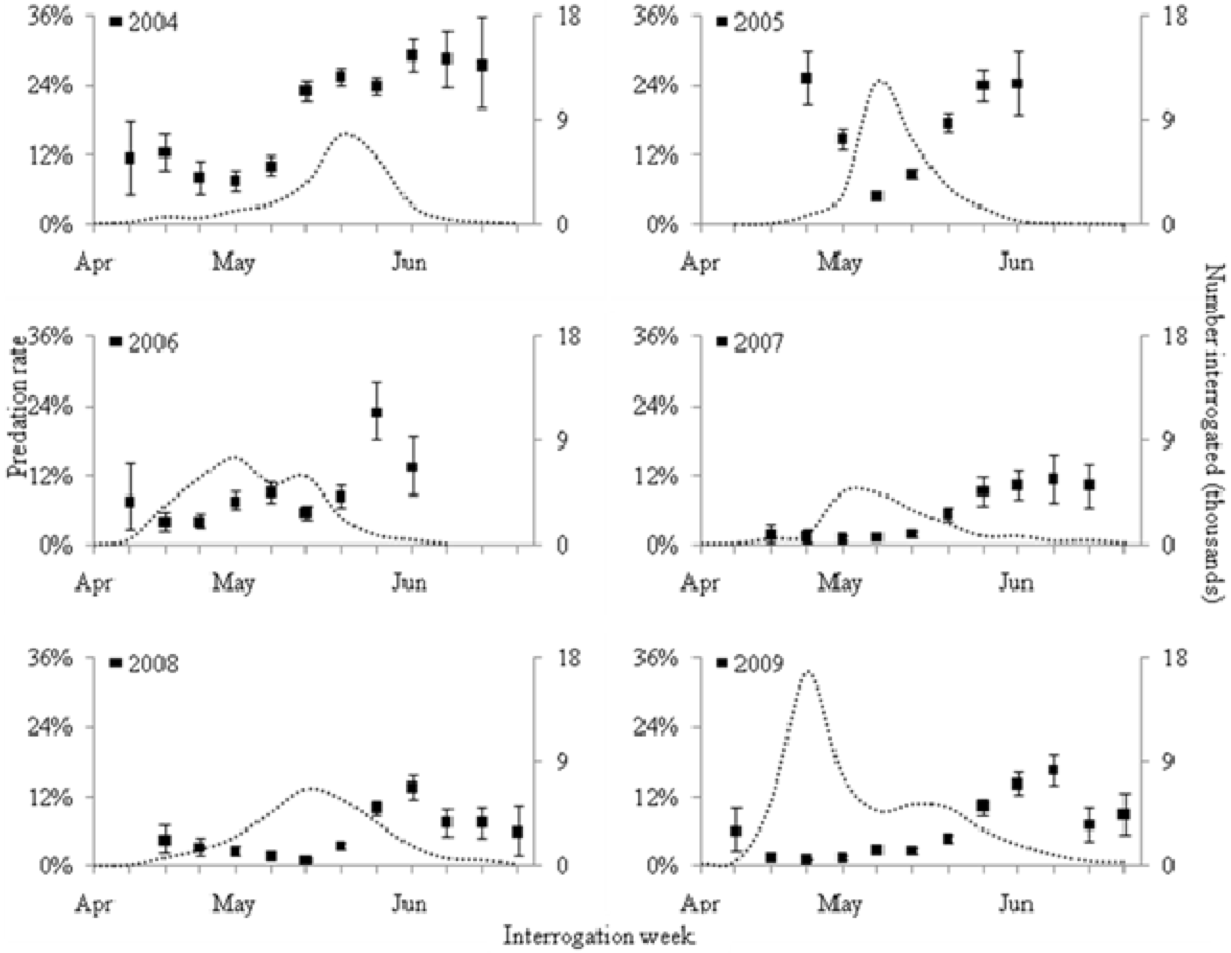
Appendix C.20. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River summer steelhead smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged Snake River summer steelhead smolts detected at Bonneville Dam.



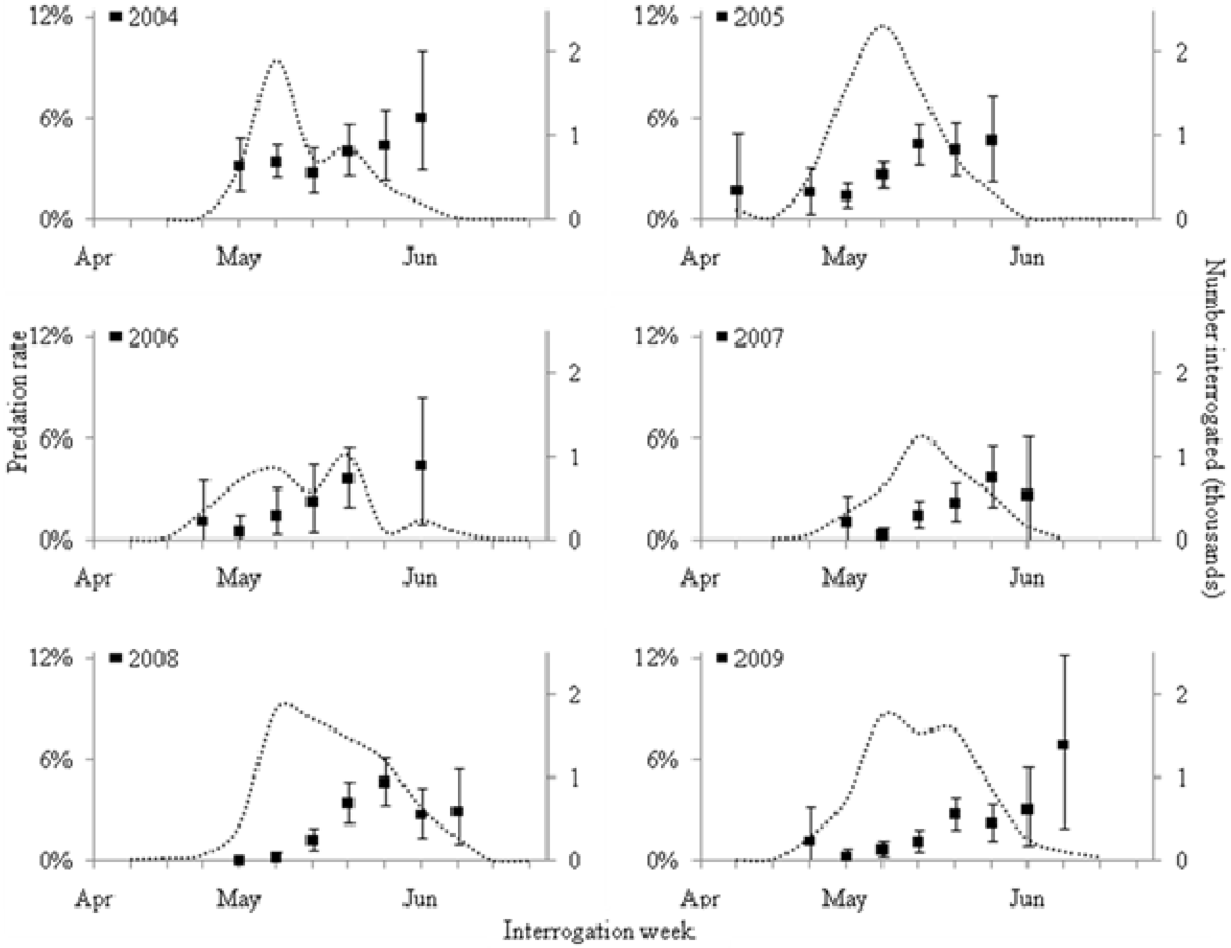
Appendix C.21. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia summer steelhead smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Bonneville Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia summer steelhead smolts detected at Bonneville Dam.



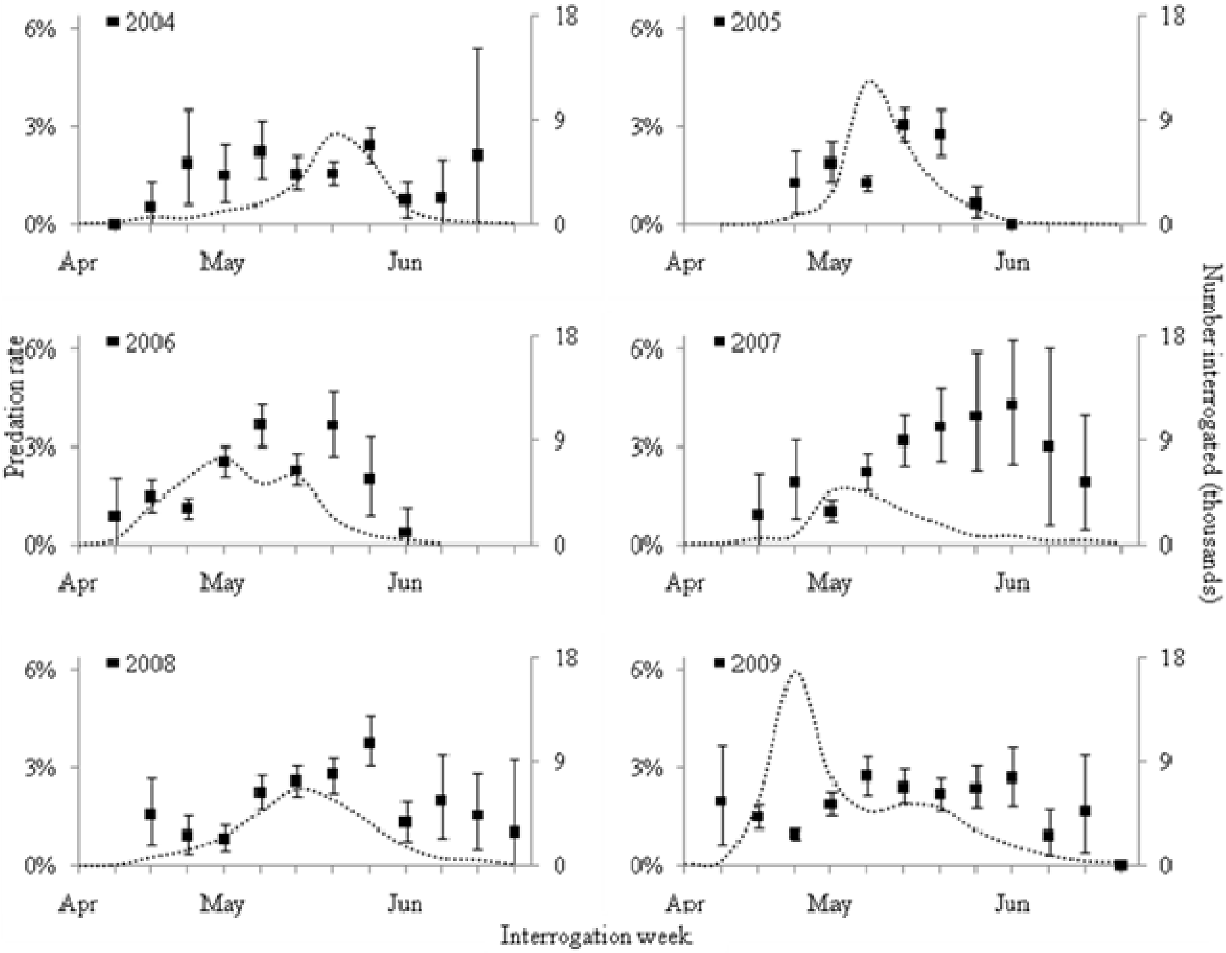
Appendix C.22. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Willamette summer steelhead smolts by Caspian terns nesting on East Sand Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Sullivan Dam were included. Dashed lines represent the number of PIT-tagged Willamette summer steelhead smolts detected at Sullivan Dam.



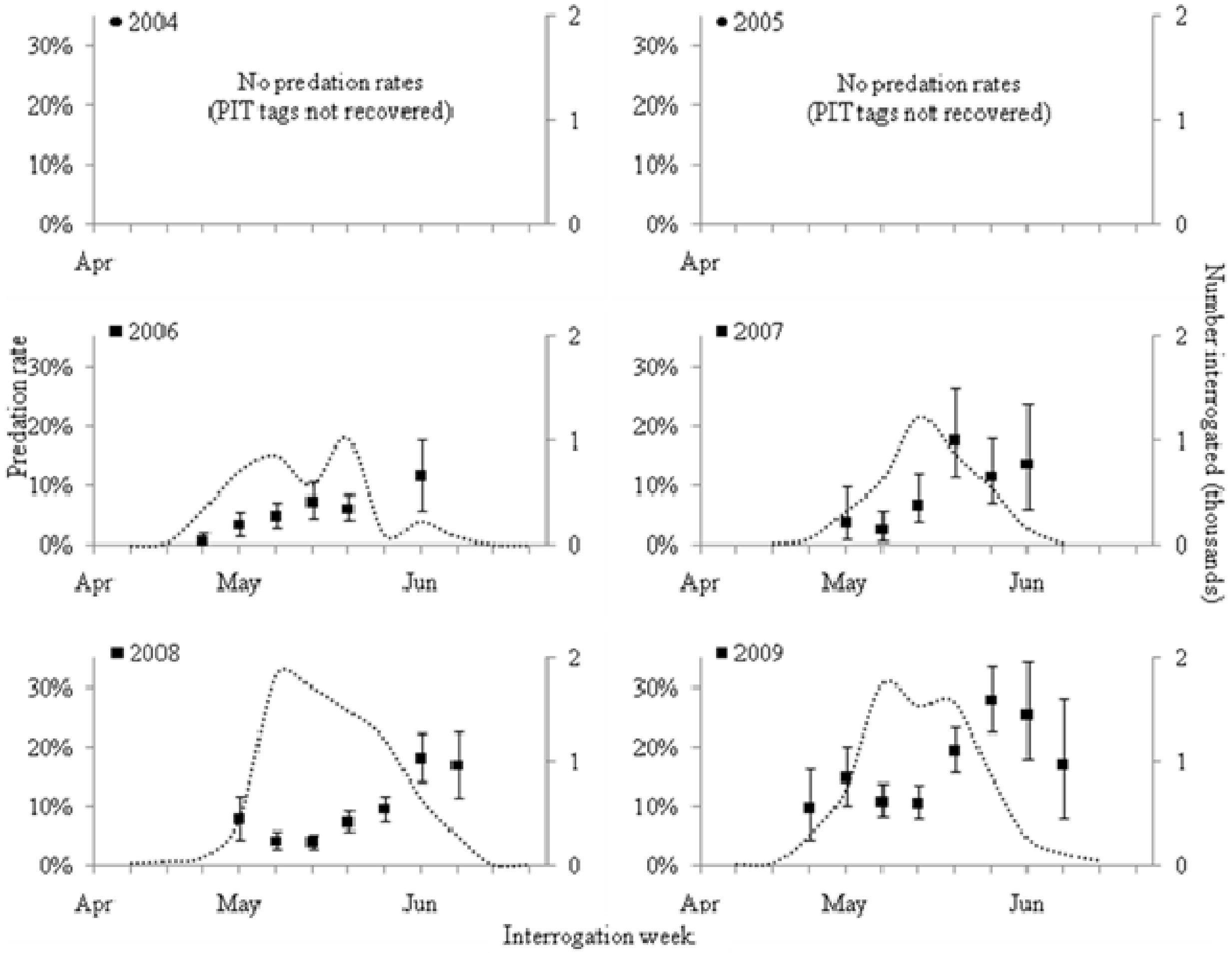
Appendix C.23. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River summer steelhead smolts by Caspian terns nesting on Crescent Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Lower Monumental Dam were included. Dashed lines represent the number of PIT-tagged Snake River summer steelhead smolts detected at Lower Monumental Dam.



Appendix C.24. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia summer steelhead smolts by Caspian terns nesting on Crescent Island during 2004-2009. Only weeks when >100 PIT-tagged summer steelhead were interrogated at Rock Island Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia summer steelhead smolts detected at Rock Island Dam.



Appendix C.25. Weekly predation rates (with 95% confidence intervals) of PIT-tagged Snake River summer steelhead smolts by double-crested cormorants nesting on Foundation Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Lower Monumental Dam were included. Dashed lines represent the number of PIT-tagged Snake River summer steelhead smolts detected at Lower Monumental Dam.



Appendix C.26. Weekly predation rates (with 95% confidence intervals) of PIT-tagged upper Columbia summer steelhead smolts by Caspian terns nesting on Goose Island during 2004-2009. Only weeks when ≥100 PIT-tagged summer steelhead were interrogated at Rock Island Dam were included. Dashed lines represent the number of PIT-tagged upper Columbia summer steelhead smolts detected at Rock Island Dam.

CHAPTER 4

BIOTIC AND ABIOTIC FACTORS INFLUENCING SUSCEPTIBILITY OF JUVENILE STEELHEAD TO PREDATION BY CASPIAN TERNS AND DOUBLE-CRESTED CORMORANTS NESTING ON THE COLUMBIA PLATEAU

This chapter has been prepared for the U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This chapter summarizes data from the Snake River steelhead PIT-tagging project during 2007-2009.

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SUMMARY

Identifying factors that influence susceptibility to predation can aid in developing management strategies to recover fish populations of conservation concern. Predator-prey relationships can be influenced by numerous biotic and abiotic factors, including prey condition, prey size, and environmental conditions. We investigated these factors using juvenile steelhead (*Oncorhynchus mykiss*) from the Snake River (Pacific Northwest, USA), an evolutionarily significant unit (ESU) that is listed as threatened under the U.S. Endangered Species Act (ESA). Steelhead smolts were captured at Lower Monumental Dam and Ice Harbor Dam on the lower Snake River, examined for external indicators of health status, marked with passive integrated transponder (PIT) tags, and released to continue out-migration during 2007-2009 (n = 25,909 fish). Recoveries of fish PIT tags on the downstream Crescent Island Caspian tern (*Hydroprogne caspia*) colony (n = 913 tags) indicated that steelhead susceptibility to tern predation increased significantly with declining steelhead external condition, decreased water discharge, decreased water clarity, and increased steelhead length, although susceptibility of steelhead peaked at about 202 mm fork length and decreased for longer and shorter steelhead. Recoveries of PIT tags on the downstream Foundation Island double-crested cormorant (*Phalacrocorax auritus*) colony (n = 493 tags) indicated that steelhead susceptibility to cormorant predation also increased significantly with declining steelhead external condition, plus steelhead of hatchery origin were more susceptible as compared to their wild counterparts. Results indicate that steelhead susceptibility to avian predation is condition-dependent, length-dependent, and influenced by river conditions and rearing environment. These results suggest that increases in smolt survival associated with reductions in avian predation may be compensated for by other mortality factors as avian predators disproportionately consume steelhead in degraded condition. However, the low prevalence of externally degraded steelhead smolts observed in this study, plus the recovery of PIT tags from steelhead that were in good condition on bird colonies, suggests that some proportion, perhaps substantial, of smolt mortality due to avian predation is additive and reductions in mortality from avian predation may benefit smolt survival. This information is relevant to those management efforts to restore ESA-listed Columbia Basin salmonids that focus on reducing predation rates on out-migrating smolts.

INTRODUCTION

Predation is a key ecological process influencing the size of fish populations and the composition of fish communities (Sih et al. 1985; Kerfoot and Sih 1987). Predator-prey relationships are often influenced by numerous factors, including: (1) prey condition (see review by Mesa et al. 1994), (2) prey size (see review by Sogard 1997), and (3) environmental conditions (Gregory 1993; Mesa 1994; Mesa and Warren 1997; Gregory and Levings 1998; De Robertis et al. 2003). Understanding how these biotic and abiotic factors influence predator-prey relationships has important implications for fitness at both the individual and population levels, plus provides valuable information regarding the efficacy of top-down ecosystem management (i.e., predator management).

The theory that predators disproportionately prey on individuals in substandard condition (e.g., weak, sick, stressed, or inexperienced; sensu Temple 1987) is widely accepted and has been well supported in fish predation studies (see review by Mesa et al. 1994). The occurrence and magnitude of condition-dependent predation may vary as a function of predator foraging strategy. In theory, predators that chase their prey should be more likely to disproportionately take individuals in poorer condition compared to predators that ambush their prey (Estes and Goddard 1967, Schaller 1968). Studies evaluating predator-prey interactions and the efficacy of predator management, however, rarely consider the influence of prey condition and predator foraging strategies (Mesa et al. 1994). For instance, the success of predator management efforts to increase prey populations would be diminished if prey would have died from other causes (e.g., disease, competition, or other predators) regardless of the predation event (Errington 1956; Temple 1987). Thus, the degree to which the mortality caused by predation is compensatory is a primary consideration for programs that seek to restore prey populations through predator management.

Within Pacific salmonid (*Oncorhynchus* spp.) populations, increased susceptibility to predation has been attributed to differences in fish behavior, condition, size, rearing, and environmental conditions (Gregory and Levings 1998; Mesa et al. 1998; Collis et al. 2002; Schreck et al. 2006; Kennedy et al. 2007). Certain environmental conditions experienced by juvenile salmonids during out-migration are known to increase stress, reduce fish performance, and increase susceptibility to predation (Raymond 1979; Budy et al. 2002; Schreck et al. 2006). External condition of out-migrating juvenile steelhead (*O. mykiss*) has been linked to both internal fish condition and survival during out-migration (Hostetter et al. i*n press*). Although some studies have suggested that smolt condition influences susceptibility to avian predation (Schreck et al. 2006; Kennedy et al. 2007), no direct link between external fish condition and susceptibility to avian predation has yet been documented in the wild.

Avian predation on salmonid smolts has been identified as one of several factors limiting recovery of some evolutionarily significant units (ESUs; Waples 1991) of salmonids from the Columbia River basin that are listed as threatened or endangered under the U.S. Endangered Species Act (ESA) (Roby et al. 2003; Lyons 2010). Smolt mortality due to avian predation is primarily attributable to two species of colonial waterbirds, Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus*; Collis et al. 2002). These predators employ very different foraging behaviors, as double-crested cormorants are pursuit-divers that actively hunt their prey underwater (Hatch 1999), whereas Caspian terns are plunge-divers that capture (i.e., ambush) their prey at or near the water surface (Cuthbert and Wires 1999). In addition to foraging behavior, these predators also differ in size, gape width, and foraging range from the breeding colony (Cuthbert and Wires 1999; Hatch 1999), suggesting that susceptibility of juvenile salmonids to each of these two primary avian predators may be uniquely associated with individual fish characteristics and environmental variables. Currently, management efforts to reduce the impact of avian predation on survival of juvenile salmonids within the Columbia River estuary are being implemented (Roby et al. 2002; USFWS 2006). However, knowledge of how individual fish characteristics and environmental factors influence the susceptibility of juvenile salmonids to avian predation is extremely limited.

This study tested several hypotheses regarding the predator-prey relationship between fish-eating birds (e.g., Caspian terns and double-crested cormorants) and anadromous salmonids (e.g., steelhead) in the Columbia River basin. Our hypotheses included: (1) the probability that a smolt will be consumed by an avian predator is influenced by both individual fish characteristics (e.g., size, rearing type [hatchery vs. wild], external condition) and environmental conditions (e.g., turbidity, water discharge rate); (2) avian predation on salmonid smolts is condition dependent, such that the probability of a smolt being consumed by an avian predator increases with declining external condition of the smolt; and (3) factors influencing smolt susceptibility to avian predation will vary with the species of avian predator. The prey used in this experiment were juvenile steelhead released at the lower-most dams on the Snake River , while the predators were Caspian terns and double-crested cormorants nesting on islands in the mid-Columbia River, downstream of the release site and within the documented foraging range of these bird species (Hatch 1999; Anderson et al. 2004; Lyons et al. 2007; Maranto et al. 2010). Snake River steelhead were selected for this study because prior research suggested that steelhead smolts were the most susceptible salmonids to avian predation in the mid-Columbia River (Antolos et al. 2005). In addition, data to evaluate the impact of avian predation are needed to evaluate recovery options for Snake River steelhead, which are listed under the U.S. Endangered Species Act as threatened (Good et al. 2005).

METHODS

Study Area

Our research efforts focused on out-migrating juvenile steelhead from the Snake River basin, which were guided into juvenile collection facilities at either Lower Monumental (LMN) Dam (river kilometer [Rkm] 589) or Ice Harbor (ICH) Dam (Rkm 538) on the lower Snake River, Washington, USA (Figure 4.1). We investigated two piscivorous waterbird breeding colonies on two different islands located on the mid-Columbia River in an impoundment formed by McNary Dam (hereafter referred to as McNary Pool), just downstream of the confluence of the Snake and Columbia rivers and downstream of the smolt capture and release sites: (1) a Caspian tern colony on Crescent Island (Rkm 510) and (2) a double-crested cormorant colony on Foundation Island (Rkm 518; Figure 4.1). Smolt capture and release locations were specifically selected because they were directly upstream and near the tern and cormorant colonies (< 60 km between the furthest bird colony and release site; Figure 4.1). The maximum foraging radius of a Caspian tern has been estimated at 82 km from its breeding colony (Maranto et al. 2010), while double-crested cormorants have been estimated to forage up to 62 km from their breeding colonies (Hatch 1999). Distances suggest that steelhead tagged and released as part of this study were within the general foraging radius of Crescent Island terns and Foundation Island cormorants immediately following release from LMN or ICH dams.

Fish Capture, Tagging, and External Examination

In general, juvenile steelhead were sampled six days per week at LMN Dam and two days per week at ICH Dam during the 2007, 2008, and 2009 steelhead out-migration periods. All sampling for this work was conducted as part of the juvenile fish sampling at each of these sites. Sampling corresponded with the run at-large, starting in early April and ending in early July, or until capture numbers dropped below 100 steelhead per week. Steelhead were collected daily and held for up to 24 h in the juvenile collection facility's holding tanks with flow-through river water prior to sampling. Steelhead were held in a 7,500 liter holding tank with flow-through river water for up to 24 hours at LMN Dam, while at ICH Dam steelhead were held for < 6 hours in a 5,500 liter hold tank. Daily samples of captured juvenile steelhead were separated into small batches (10-50 fish) via a slide gate, anesthetized with tricaine methanesulfonate (MS-222), and tagged with a 12 mm (length) x 2 mm (width) passive integrated transponder (PIT) tag (134.2 kHz) via a modified hypodermic syringe with a 12-gauge needle (Prentice et al. 1990a; Prentice et al. 1990b; Nielson 1992). To reduce disease transmission, needles were soaked for a minimum of 10 min in 70% ethyl alcohol prior to PIT tag loading. Previous studies have found no significant effects of PIT-tagging on subsequent survival of juvenile salmonids (Prentice et al. 1990a).

Steelhead were sampled and PIT-tagged for this study regardless of their condition, rearing type, or length in an effort to represent the overall population of out-migrating juvenile steelhead. Methods for non-invasive examination of steelhead smolts followed those of Hostetter et al. (*in press*), and are briefly summarized here. After a steelhead was PIT-tagged, it was placed in a sample-tray, measured (fork length, ± 1 mm), weighed (± 1 g), assigned as originating from a natural (presence of an adipose fin) or hatchery (absence of a adipose fin or erosion of pectoral, pelvic, caudal, or dorsal fins) rearing environment, and digitally photographed (Canon EOS Rebel XTi camera; Canon EF 50mm f/2.5 Compact Macro lens; Bencher Copymate II copy stand with fluorescent producer light source). Digital photographs were taken of both sides of the steelhead to allow for detailed classification of external symptoms by type and severity after the study fish was released, thereby reducing the total handling time for each fish (< 30 seconds). Detailed information on external condition (e.g., body injuries, descaling, external signs of disease, fin damage, and ectoparasite infestations; classifications described in detail in Table 4.1) of each steelhead was collected by analyzing digital photographs. Finally, each steelhead smolt was scored for overall external condition (good, fair, or poor; modified from Evans et al. 2004) based on scores for each of these five categories of external condition (see Table 4.1).

Following the examination process, daily groups of PIT-tagged steelhead were placed in a holding tank with flow-through river water. At LMN Dam, steelhead were held in a 1,100 liter holding tank with flow-through river water for 4 to 18 hours, and then released into LMN Dam tailrace via the juvenile bypass facility out-flow pipe. At ICH Dam, steelhead were held in a 900 liter holding tank with flow-through river water for 1 to 10 hours, and then released into ICH Dam tailrace via the juvenile bypass facility out-flow pipe. Release times alternated between mornings and evenings to reduce possible bias in steelhead susceptibility to predation associated with release time. Release times at LMN Dam alternated between 18:00 - 23:00 Pacific Daylight Time (PDT) the day of processing and 07:00 - 11:00 PDT the day following processing. Release times at ICH Dam alternated between 09:00 -13:00 PDT and 18:00 - 22:00 PDT the day of processing. All mortalities and ejected PIT tags were removed from temporary holding tanks prior to release and excluded from further analyses.

PIT Tag Recovery

PIT-tagged steelhead smolts were considered to have been consumed by a Caspian tern or a double-crested cormorant if the unique PIT tag associated with a steelhead was detected on the Crescent Island tern colony or the Foundation Island cormorant colony, respectively. PIT tags were recovered from these bird colonies after nesting birds dispersed following the breeding season each year (July-August). A detailed description of the methods used to recover PIT tags from bird colonies can be found in Ryan et al. (2003) and additional methods specific to this study are briefly summarized here. On the tern colony, PIT tags were recovered by systematically scanning the area occupied by terns during the nesting season with a flat-plate PIT tag detector mounted on a 4-wheel-drive vehicle. Pole-mounted, hand-held transceivers were then used to detect PIT tags in areas inaccessible to the flat-plate detector. The thick woodland on Foundation Island did not allow for use of a vehicle; therefore, the entire cormorant colony was scanned using pole-mounted, hand-held transceivers.

Detection Efficiency

Recoveries of PIT tags on bird colonies provide a minimal estimate of predation on PIT-tagged smolts because (1) an unknown proportion of consumed PIT tags are deposited off-colony, (2) PIT tags deposited on the colony may be lost due to wind and water erosion, (3) some PIT tags are damaged either before or after deposition to the point of being unreadable , and (4) detection efficiency of functioning PIT tags on the colony is less than 100% due to signal collision and other factors (Ryan et al. 2003; Evans et al. In press). To better account for possible inter-annual differences in tag loss and damage, we measured detection efficiency each year of the study by systematically sowing known PIT tags, of identical dimensions and design to those implanted in steelhead, on both the tern and cormorant colonies. To investigate possible intra-seasonal variation in PIT tag detection efficiency, known PIT tags were sown on each colony (1) prior to the birds arrival at the colony (March), (2) during the egg incubation period (May), (3) near the time of chick fledging (June), and (4) once the birds had left the colony following the nesting season (late July to early August). Recoveries of these sown PIT tags during PIT tag recovery allowed us to use logistic regression to estimate weekly detection efficiencies by interpolation. Weekly estimates of detection efficiency were then included as fixed effects in all models to account for seasonal variation in colony-specific detection efficiency.

Environmental Factors

Previous studies have suggested that salmonid susceptibility to avian predation in the Columbia River basin may be influenced by abiotic factors, including environmental conditions experienced during salmonid out-migration (Collis et al. 2001, Ryan et al. 2003, Antolos et al. 2005). Environmental variables evaluated as part of this study included: (1) water discharge rate (kcfs) from LMN Dam (hereafter referred to as discharge), (2) water clarity (Secchi depth in meters) in the Snake River, (3) steelhead release location (LMN or ICH dam), (4) prey abundance (estimated number of in-river steelhead in McNary Pool), (5) predator abundance (estimated number of adult and juvenile Caspian terns or double-crested cormorants at the colonies on Crescent Island or Foundation Island, respectively), and (6) migration year.

Steelhead were PIT-tagged and released at two locations upstream of the bird colonies (ICH or LMN dams) to evaluate whether either bird species disproportionately consumed steelhead released at one of these two locations. Measurements of the number of in-river juvenile steelhead passing McNary Dam and water discharge at LMN Dam were obtained from the Fish Passage Center website (www.fpc.org) and the Data Access in Real Time (DART) website (www.cbr.washington.edu/dart). In-river juvenile steelhead abundance was calculated by averaging estimates of the number of juvenile steelhead passing McNary Dam each day into a weekly (Sunday through Saturday) average. Although the in-river steelhead index at McNary Dam is not an exact measurement of the overall abundance of juvenile steelhead in McNary Pool, it does provide a quantitative estimate of relative numbers of in-river juvenile steelhead within and between migration years. The McNary Dam passage index was selected (relative to passage indices at other dams) because it includes steelhead from both the Snake and Columbia rivers and is located just downstream and within the presumed foraging radius of Caspian terns and double-crested cormorants nesting on Crescent Island and Foundation Island, respectively (Figure 4.1). Also, changes in in-river steelhead abundance at MCN Dam were highly correlated with changes in in-river steelhead abundance at LMN Dam (Pearson correlation coefficient, $R > 0.80$). Data on weekly predator abundance were obtained from the website of Bird Research Northwest (www.birdresearchnw.org). These data were estimated by averaging three to eight weekly counts of the numbers of adults and juveniles present on the Crescent Island tern colony and the Foundation Island cormorant colony from observation blinds located at the periphery of each colony. Water clarity was estimated using averages of two to four weekly Secchi measurements taken from a boat in the main channel of the Snake River just above the confluence of the Snake and Columbia rivers. Discharge, water clarity, predator abundance, and prey abundance variables were not highly correlated with one another (Pearson correlation coefficient, $R < 0.60$).

Susceptibility to Avian Predation

A combination of logistic regression models was used to evaluate the influence of environmental factors and individual fish characteristics on the probability of recovering a PIT-tagged steelhead on the Crescent Island tern colony or on the Foundation Island cormorant colony. We considered a model including colony-specific detection efficiency, number of birds on-colony, in-river steelhead abundance, and migration year as our null model (hereafter referred to as the base model) due to the biological importance of these variables in predicting steelhead susceptibility to avian predation (Ryan et al. 2003; Antolos et al. 2005). Variables included in the base model were also included in all candidate models to account for variation in steelhead susceptibility to avian predation associated with these variables prior to investigating relationships between susceptibility to avian predation and explanatory variables of interest.

We fit logistic regression functions:

$$\text{logit}\ (p_i) = \beta_0 + \beta_1 D_i + \beta_2 P_i + \beta_3 S_i + \beta_4 Y_i \ldots + \beta_p X_i$$

where p_i is the probability of being detected on a specific bird colony for steelhead i, β_0 is the regression intercept, $\beta_1 D_i$ is the regression coefficient for the weekly colony-specific detection efficiency for steelhead i, $\beta_2 P_i$ is the regression coefficient for the weekly colony-specific predator abundance for steelhead i, $\beta_3 S_i$ is the regression coefficient for the weekly in-river steelhead abundance for steelhead i, $\beta_4 Y_i$ is the regression coefficient for the migration year of steelhead i, and $\beta_p X_i$ is the regression coefficient for the independent explanatory variable X associated with steelhead i. Independent explanatory variables evaluated by this study included the river conditions and the individual fish characteristics described above.

The influence of individual fish characteristics and environmental factors on the probability of recovering a PIT-tagged steelhead on a bird colony was evaluated by four general models: (1) a base model that only included variables previous literature indicated were biologically important, (2) a global additive model that included all variables of interest, (3) a best fit model from a backwards stepwise selection process that began with the global additive model, and (4) individual models that evaluated each explanatory variable irrespective of other individual fish characteristics and environmental factors of interest. Only additive models were investigated due to the small proportions of PIT tags recovered on bird colonies (~ 2 - 4% per colony, per year) and the high number of explanatory variables evaluated. All models were ranked and compared using Akaike's Information Criteria corrected for small sample size (AIC_c) and AIC_c differences (ΔAIC_c) (Burnham and Anderson 2002). Relative differences in the probability of recovering a PIT-tagged steelhead on a bird colony were further investigated through probabilities, odds, and odds ratios of specific explanatory variables. Evaluation of explanatory variables by AIC_c values, stepwise selection, and odds ratios allowed us to address our hypotheses by identifying the most influential explanatory variables, while also assessing the direction and strength of these variables for explaining susceptibility to each avian predator species. All analyses were conducted in SAS, version 9.2 (SAS Institute, Inc.) with statistical significance set at $\alpha = 0.05$.

RESULTS

Steelhead Capture and Condition

A total of 25,909 juvenile steelhead were PIT-tagged and released from either LMN Dam (n = 22,401) or ICH Dam (n = 3,508) as part of this study. Sampling effort was relatively consistent across the three-year study, with annual releases of 7,065, 9,143, and 9,701 PIT-tagged steelhead in 2007, 2008, and 2009, respectively. Of these PIT-tagged steelhead, 59% were classified as being in good condition, 28% in fair condition, and 13% in poor condition (Table 4.1). The most prevalent external fish condition was fin damage (78%), followed by > 5% de-scaling (34%), body injuries (19%), external symptoms of disease (4%), and ectoparasite infestations (3%; Table 4.1). Released PIT-tagged steelhead consisted of more hatchery-reared smolts (n = 22,135) than wild smolts (n = 3,774), which corresponded with the relative abundance of these two rearing types among run-of-the-river steelhead captured at LMN and ICH dams (Table 4.2). Average fork length of hatchery-reared steelhead ($\bar{\chi} = 225$ mm) was 21.2% greater than that of wild steelhead ($\bar{\chi} = 185$ mm; 95% CI of difference: 20.7% to 21.7%, two-tailed t-test $P < 0.001$). Despite this difference, fork lengths of hatchery-reared and wild steelhead overlapped considerably (hatchery-reared fork lengths = 132 - 375 mm, wild fork lengths = 131 - 354 mm; Table 4.2). External condition of hatchery-reared steelhead also showed some differences from their wild counterparts. For instance, wild steelhead displayed a lower percentage of body injuries, de-scaling, and fin damage (Table 4.3). However, only 1% of hatchery-reared steelhead had ectoparasites compared to 12% of wild steelhead observed with ectoparasites (Table 4.3). Integrated condition ranks reflected these differences among rearing types, with 57% of hatchery-reared steelhead classified in good condition as compared to 70% of wild reared steelhead classified in good condition (Table 4.3).

PIT Tag Recoveries

Of the 25,909 PIT-tagged steelhead released as part of this study, we recovered 913 PIT tags (3.5%) on the Crescent Island Caspian tern colony and 493 PIT tags (1.9%) on the Foundation Island double-crested cormorant colony (Table 4.4). There was a positive association between detection efficiency of PIT tags sown on the Crescent Island tern colony and date when sown, indicating that the probability of recovering a smolt PIT tag was higher for tags deposited later in the nesting season. This trend was significant in 2007 ($\chi^2 = 140.9$, df = 1, $P < 0.001$), 2008 ($\chi^2 = 153.1$, df = 1, $P < 0.001$), and 2009 ($\chi^2 = 92.1$, df = 1, $P < 0.001$). Results from Foundation Island cormorant colony indicated that the probability of recovering a PIT-tag on this colony was inversely related to date when sown. However, this relationship was not significant in 2007 ($\chi^2 = 1.8$, df = 1, $P = 0.174$), but suggestive in both 2008 ($\chi^2 = 3.3$, df = 1, $P = 0.071$) and 2009 ($\chi^2 = 5.0$, df = 1, $P = 0.026$).

Susceptibility to Tern Predation

Both individual fish characteristics and environmental factors were associated with susceptibility of steelhead to predation by Caspian terns (Table 4.5). Fork length, discharge, water clarity, and integrated condition rank (good, fair, or poor) were the most influential explanatory variables in predicting the susceptibility of steelhead to predation by Caspian terns (Table 4.5). A quadratic function of steelhead fork length, was the most important individual fish characteristic in predicting susceptibility to tern predation. Steelhead fork length was highly significant in the top model (Table 4.6) and had the lowest AIC_c value of any model based on a single explanatory variable (Table 4.5). Results indicated that the relationship between fork length and susceptibility to tern predation was convex, with the greatest susceptibility of steelhead at fork lengths of around 202 mm, but lower susceptibility at greater and lesser fork lengths (Figure 4.6).

Similar trends were observed when investigating discrete associations between steelhead susceptibility to tern predation and individual fish characteristics or environmental factors. Once again, the quadratic function of steelhead fork length was the most important variable in predicting the relative susceptibility of steelhead smolts to predation by terns (Table 4.5, Appendix D.1). Several individual fish characteristics, including severe body injuries, > 20% de-scaling, and the integrated condition rank, were associated with higher susceptibility to tern predation, supporting the hypothesis that steelhead susceptibility to avian predation is condition-dependent (Appendix D.1). Specifically, the odds of recovering a steelhead in poor condition on the Caspian tern colony were 1.3 times greater than steelhead in good condition (95% CI: 1.1 to 1.6, $P = 0.006$). Similarly, the odds of recovering steelhead with > 20% de-scaling and severe body injuries were 1.4 times (95% CI: 1.0 to 1.9, $P = 0.035$) and 1.2 times (95% CI: 1.0 to 1.6, $P = 0.057$), respectively, greater than steelhead without descaling or body injuries. Several individual fish characteristics were not included in the top model and were also not significantly associated with susceptibility of steelhead to predation by Caspian terns in the discrete models (Table 4.6; Appendix D.1). For instance, rearing type was not included in the top model (Table 4.6) and only a small and suggestive relationship was detected in the discrete model indicating that the odds of recovering a hatchery-reared steelhead were 0.8 times as great as wild steelhead (95% CI: 0.7 to 1.0; $P = 0.053$; Appendix D.1).

In all models, results indicated that steelhead susceptibility to predation by terns increased when the number of in-river steelhead decreased, water clarity decreased, and discharge decreased, (Table 4.6, Appendix D.1). For instance, results from the top model indicated that steelhead susceptibility to tern predation increased when in-river steelhead abundance decreased ($P < 0.001$), water clarity decreased ($P < 0.001$), and discharge decreased ($P < 0.001$; Table 4.6). Thus high river flows, high water clarity, and large numbers of out-migrating steelhead smolts all favor a reduction in risk of tern predation for individual steelhead smolts. There was no evidence, however, of a relationship between the number of terns on Crescent Island and susceptibility of steelhead to tern predation, after accounting for other variables in the model ($P = 0.380$; Table 4.6). Although not included in the top model, there was a significant relationship between release location and steelhead susceptibility to tern predation in the discrete model (Appendix D.1). This result indicated that the odds of recovering a steelhead released at LMN Dam, farther upriver from the tern colony, were 1.3 times greater than steelhead released at ICH Dam, closer to the colony (95% CI: 1.0 to 1.7; $P = 0.028$).

Susceptibility to Cormorant Predation

Similar to results for predation by Caspian terns, steelhead susceptibility to predation by double-crested cormorants was associated with both individual fish characteristics and environmental factors. External symptoms of disease appeared to be the most important individual fish characteristic for predicting steelhead susceptibility to cormorant predation. Variables for both moderate and severe external symptoms of disease were highly significant ($P < 0.001$) in the top model (Table 4.7) and had the lowest AIC_c value of any individual characteristic model (Table 4.5). Results from the top model indicated that the odds of recovering steelhead with moderate or severe external symptoms of disease on the cormorant colony were 2.8 times (95% CI: 1.7 to 4.7, $P < 0.001$) and 2.9 times (95% CI: 2.1 to 4.2, $P < 0.001$), respectively, greater than steelhead without external symptoms of disease symptoms (Table 4.7). Results from the top model also indicated that the odds of recovering hatchery-raised steelhead on the cormorant colony were 1.5 times greater than wild steelhead (95% CI: 1.1 to 2.1, $P = 0.005$; Table 4.7).

Results from models based on individual explanatory variables provided additional support for the hypothesis that steelhead susceptibility to cormorant predation was condition-dependent. Several external conditions, including body injuries ($P = 0.050$), external symptoms of disease ($P < 0.001$), fin damage ($P = 0.012$), and the integrated condition rank ($P < 0.001$), indicated that as the condition of steelhead declined susceptibility to cormorant predation increased (Appendix E.1). No relationship was detected between other individual fish characteristics and susceptibility to cormorant predation (Appendix E.1). For example, unlike steelhead susceptibility to tern predation, susceptibility to predation by cormorants was not associated with steelhead fork length (linear or quadratic).

Environmental factors included in the top model indicated that steelhead susceptibility to predation by cormorants increased when cormorant abundance increased ($P < 0.001$) and steelhead were released closer to the cormorant nesting colony (ICH Dam; $P < 0.001$). Unlike results for terns, there was no evidence of predator swamping associated with cormorant predation on steelhead, as in-river steelhead abundance was not related to steelhead susceptibility to predation by cormorants ($P = 0.946$). Although not included in the top model, water clarity was associated with steelhead susceptibility to predation by cormorants when modeled as a separate explanatory variable. Results from the model including water clarity as the single explanatory variable indicated that as water clarity increased, steelhead susceptibility to cormorant predation increased ($P = 0.047$). Discharge was not associated with steelhead susceptibility to cormorant predation as it was not included in the top model nor was it significant when modeled as the single explanatory variable (*P* = 0.959).

DISCUSSION

This study tested hypotheses regarding the influence of individual fish characteristics and environmental variables on the relative susceptibility of steelhead smolts to avian predation. We found that the size and condition of juvenile steelhead, as well as river conditions at the time of release, were related to a steelhead's probability of being eaten by an avian predator, but the importance and strength of these factors differed between avian predator species. One consistent trend across both avian predators in this study was the disproportionate consumption of steelhead in degraded condition. These results corroborate previous work that suggested susceptibility of Chinook salmon (*O. tshawytscha*) smolts to predation by Caspian terns in the Columbia River estuary was influenced by the incidence of disease in out-migrating smolts (Schreck et al. 2006). Kennedy et al. (2007) found similar relationships between decreased saltwater preparedness and increased susceptibility of juvenile steelhead to avian predation in the Columbia River estuary; further supporting the hypothesis that salmonid susceptibility to avian predation is associated with individual fish characteristics and condition.

The results presented here support the theory that predator foraging strategies play an important role in the incidence and magnitude of condition-dependent predation (Estes and Goddard 1967; Schaller 1968; Temple 1987). Condition-dependent susceptibility was more evident for cormorant predation (pursuit-diving foraging strategy) compared to tern predation (plunge-diving foraging strategy); four different external indices of health status (body injuries, external symptoms of disease, fin damage, and integrated condition rank) were significantly associated with susceptibility of steelhead to cormorant predation (Appendix E.1), but only one external health index (integrated condition rank) was associated with susceptibility to tern predation (Appendix D.1). Also, the magnitude of condition-dependent susceptibility was greater for cormorant predation compared to tern predation. For instance, the odds of recovering steelhead in poor condition were 1.7 times greater than steelhead in good condition on the cormorant colony (Appendix E.1), but only 1.3 times greater than steelhead in good condition on the tern colony (Appendix D.1). Taken together, these results indicated that both avian predator species disproportionately consumed steelhead in degraded condition; however, the magnitude of condition-dependent susceptibility varied by avian predator species.

Selective predation can result from several conditional events, including differences in predator-prey encounter rates, attack rates, and/or capture rates (Temple 1987). For instance, the behavior of potential prey in poorer health could make them more conspicuous to predators, thus increasing encounter rates. Likewise, predators could selectively attack prey in poorer condition to enhance foraging efficiency (rate of successful attacks) and expend less energy to capture substandard prey relative to healthy prey (Stephens and Krebs 1986). Finally, increased predation rates on prey in substandard condition could result from predators attacking all encountered individuals of the prey population, but having a higher capture rate on fish in poorer health. (Temple 1987). Although this study could not address the mechanisms influencing selective predation, Caspian terns and double-crested cormorants both disproportionately consumed steelhead in degraded condition.

The external indices of steelhead health status used in this study are associated with other metrics of overall fish condition, including increased pathogen infections and reduced survival (Hostetter et al. *in press*). Hostetter et al. (*In press*) demonstrated that juvenile steelhead with external symptoms of degraded condition were significantly less likely to survive compared to relatively undamaged smolts. These previous results, coupled with the condition-dependent avian predation demonstrated in the present study, support the hypothesis that avian predators are disproportionately consuming smolts that were less likely to survive to adulthood, indicating that smolt mortality from avian predation is partly compensatory. Further studies will be required to evaluate if reductions in smolt mortality due to avian predation will be compensated for by other mortality factors. The level of compensation associated with each predator species may vary as disproportionate consumption of steelhead in degraded condition was greater by double-crested cormorants than Caspian terns.

Individual fish characteristics, including fork length and rearing type (hatchery vs. wild), were also related to differences in steelhead susceptibility to avian predation. The hypothesis that susceptibility to predation is influenced by prey body size is not new to ecological theory (Sogard 1997). The size of individual prey and the relationship between the size of an individual relative to the prey population at large can have a major influence on survival probability (Rice 1993). One of the most prevalent theories regarding the influence of juvenile fish size on susceptibility to predation is that bigger-is-better (Sogard 1997), which predicts that larger prey will have a survival advantage over smaller prey. Our results, however, supported an alternative prediction from optimal foraging theory, that prey size selection by a predator should be dome-shaped, with the largest and smallest prey individuals having a survival advantage due to predators optimizing energy intake by selecting intermediate-sized prey (MacArthur and Pianka 1966; Rice et al. 1997). Steelhead susceptibility to Caspian tern predation supported this intermediate size selection hypothesis; susceptibility was highest for steelhead with fork lengths around 202 mm, and lower for both larger and smaller steelhead. Evidence that salmonid susceptibility to predation by Caspian terns may be positively related to salmonid smolt fork length was previously presented by Collis et al. (2001) and Ryan et al. (2003). These studies noted that the relative susceptibility of various salmonid species to Caspian tern predation in the Columbia River estuary was correlated with size, as juvenile Chinook and coho (*O. kisutch*) salmon were less susceptible compared juvenile steelhead, which are generally larger. Caspian terns nesting on Crescent Island in McNary Pool were also found to disproportionately consume steelhead compared to the relatively smaller Chinook salmon (Antolos et al. 2005). Overall, our results imply that fork length is an important factor influencing salmonid susceptibility to predation by Caspian terns. Size-dependent predation on steelhead by Caspian terns provides strong empirical evidence in support of the hypothesis that relative size differences among salmonid ESUs are responsible, at least in part, for differences in susceptibility to Caspian tern predation (Collis et al. 2001; Ryan et al. 2003).

A growing body of evidence suggests that behavioral and physical traits associated with hatchery-raised salmonids enhance susceptibility to predation (Olla and Davis 1989; Johnsson and Abrahams 1991; Alvarez and Nicieza 2003; Fritts et al. 2007). Several studies in the Columbia River estuary have noted that hatchery-reared salmonids were more susceptible to avian predation compared to their wild counterparts (Collis et al. 2001; Ryan et al. 2003; Kennedy et al. 2007). Our results indicate that hatchery-reared steelhead were more susceptible to some avian predators in freshwater systems; cormorants disproportionately consumed hatchery-reared steelhead compared to wild-origin steelhead, however this was not the case for

Caspian terns. Although the mechanisms associated with increased susceptibility of hatchery-reared salmonids to avian predation are not completely elucidated, numerous traits including a lack of innate and learned predator avoidance behaviors (Olla and Davis 1989; Berejikian 1995), greater surface orientation (Mason et al. 1967), or increased stress levels associated with handling (Schreck 1981; Olla and Davis 1989) could play a role in the higher susceptibility of hatchery-reared salmonids to cormorant predation. However, similar to results from this study, increased predation rates on hatchery-reared salmonids are not observed at all avian colonies or in all years (Collis et al. 2001; Ryan et al. 2003; Chapter 3). Additionally, several confounding variables complicated comparisons among hatchery-reared and wild steelhead as hatchery steelhead were on average longer and displayed more external conditions compared to their wild counterparts.

Environmental factors have been shown to alter salmonid susceptibility to predation in both field and laboratory settings (Raymond 1979; Gregory 1993; Gregory and Levings 1998; Korstrom and Birtwell 2006). For instance, Antolos et al. (2005) suggested that low river flows and reduced in-river salmonid abundance were associated with increased predation rates on salmonids by Caspian terns nesting on Crescent Island. In this study, decreased water discharge was strongly associated with increased steelhead susceptibility to tern predation. Water discharge, which is correlated with water velocity, is a key factor determining how quickly juvenile salmonids migrate through reservoirs (Berggren and Filardo 1993) and beyond the foraging range of central place foraging predators such as colonial piscivorous waterbirds. However, this relationship was not consistent across predator species, as steelhead susceptibility to cormorant predation was not significantly related to discharge.

Decreased water clarity (i.e., increased turbidity) can decrease susceptibility of fish prey to predation by piscivorous fishes due to a potential reduction in predator-prey encounter rates (Gregory 1993; Gregory and Levings 1998; De Robertis et al. 2003). Strod et al. (2008) found that increased turbidity reduced detection and predation of fish by the great cormorant (*Phalacrocorax carbo sinensis*), a pursuit-diver similar to double-crested cormorants. Similarly, this study found that steelhead susceptibility to double-crested cormorant predation decreased with increased turbidity; indicating that as turbidity increased, the probability that a steelhead would be consumed by a cormorant decreased. Unlike susceptibility to cormorant predation, however, steelhead susceptibility to Caspian tern predation was positively related to turbidity; indicating that as turbidity increased, the probability that a steelhead would be consumed by a tern also increased. Differences in the influence of turbidity on steelhead susceptibility to avian predation are likely due to variation in foraging behavior between terns and cormorants. Increased turbidity reduced steelhead susceptibility to cormorant predation, possibly due to a decrease in encounter rates that would affect a pursuit-diving predator (Strod et al. 2008). However, increased turbidity may have had the opposite effect on steelhead susceptibility to an ambush-style predator, like Caspian terns; as decreased reaction times and reduced use of cover by salmonids in more turbid water (Gregory 1993; Gregory and Levings 1998; Korstrom and Birtwell 2006) may increase steelhead susceptibility to plunge-diving predators like terns.

Large numbers of prey can swamp the short-term capacity of predators to attack, handle, and consume them, which in turn can improve an individual prey's chances of survival (Ims 1990). Consistent with the predator swamping hypothesis, increased abundance of in-river steelhead decreased the susceptibility of individual steelhead to predation by terns. Ryan et al. (2003) attributed a similar relationship between increased salmonid abundance and reduced salmonid susceptibility to avian predation in the Columbia River estuary to a greater potential for predator satiation and an improved ability of schooling fish to avoid predation. The association between in-river steelhead abundance and susceptibility to avian predation was, however, specific to terns. A lack of a relationship between in-river steelhead abundance and susceptibility to cormorant predation was likely due to the strong correlation between peak steelhead abundance and peak cormorant abundance (Figure 4.2c and e). The increased food demands associated with more breeding pairs at a larger cormorant colony, along with metabolic requirements of more growing chicks (Roby et al. 2003), presumably superseded any influence that steelhead abundance would have on the susceptibility of steelhead to predation by cormorants.

Prey often live in communities that include several predator species. The majority of studies, however, only examine predation impacts associated with one predator species (Sih et al. 1998). In the Columbia River basin, two avian species are responsible for the majority of smolt losses due to avian predation, Caspian terns and double-crested cormorants (Collis et al. 2002). Predator-specific differences in foraging behavior, size, and gape width (Cuthbert and Wires 1999; Hatch 1999) suggest that the functional roles of these two predator species may be vastly different, and thus their impacts on prey populations (e.g., salmonids) may also differ. The influences of individual fish characteristics and river conditions on steelhead susceptibility to avian predation were often predator specific in this study; further demonstrating the need to evaluate predator-specific impacts. Information regarding predator-specific impacts from multiple predators will improve top-down ecosystem management (i.e., predator management) to recover fish populations of conservation concern where warranted and applicable.

The efficacy of predator control in an effort to restore prey populations of conservation concern depends on whether reductions in mortality due to predation are compensated by other mortality factors. Separating ultimate causes of mortality (e.g., degraded fish condition) from proximate causes (e.g., avian predation of fish in degraded condition) can provide valuable insight into complex predator-prey interactions. For instance, if most juvenile salmonids consumed by avian predators would have died from other causes regardless, then reductions in avian predation will not result in commensurate increases in the number of returning adult salmonids (i.e., smolt-to-adult survival; Schreck et al. 2006). Our results suggest that the efficacy of management actions to reduce avian predation may be somewhat discounted by the disproportionate predation of steelhead in degraded condition. However, the low prevalence of externally degraded steelhead smolts observed in this study (see Table 4.1), plus the recovery of PIT tags from steelhead that were apparently in good condition on bird colonies, suggests that some proportion, perhaps substantial, of smolt mortality due to avian predation is additive. At this time studies to quantify the level of compensatory mortality associated with avian predation in the Columbia River basin have yet to be published. Further, it appears that hydrosystem operations and river conditions (e.g., discharge and water clarity) can influence the susceptibility of steelhead to avian predation. Identification of individual smolt characteristics and hydrosystem practices that affect smolt survival and susceptibility to predation will aid in development of management strategies that contribute to the recovery of ESA-listed stocks of salmonids from the Columbia River basin.

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LITERATURE CITED

Alvarez, D., and Nicieza, A.G. 2003. Predator avoidance behaviour in wild and hatchery-reared brown trout: the role of experience and domestication. J. Fish Biol. **63**(6): 1565-1577.

Anderson, C.D., D.D. Roby, and K. Collis. 2004. Foraging patterns of male and female double-crested cormorants nesting in the Columbia River estuary. Can. J. Zool. **82**(4): 541-554.

Antolos, M., Roby, D.D., Lyons, D.E., Collis, K., Evans, A.F., Hawbecker, M., and Ryan, B.A. 2005. Caspian tern predation on juvenile salmonids in the mid-Columbia River. Trans. Am. Fish. Soc. **134**(2): 466-480.

Berejikian, B.A. 1995. The effects of hatchery and wild ancestry and experience on the relative ability of steelhead trout fry (*Oncorhynchus mykiss*) to avoid a benthic predator. Can. J. Fish. Aquat. Sci. **52**(11): 2476-2482.

Berggren, T.J., and Filardo, M.J. 1993. An analysis of variables influencing the migration of juvenile salmonids in the Columbia River basin. N. Am. J. Fish. Manage. **13**(1): 48-63.

Budy, P., Thiede, G.P., Bouwes, N., Petrosky, C.E., and Schaller, H. 2002. Evidence linking delayed mortality of Snake River salmon to their earlier hydrosystem experience. N. Am. J. Fish. Manage. **22**(1): 35-51.

Burnham, K.P., and Anderson, D.R. 2002. Model selection and inference - a practical information-theoretic approach, second edition. Springer-Verlag, New York, New York.

Collis, K., Roby, D.D., Craig, D.P., Ryan, B.A., and Ledgerwood, R.D. 2001. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary: vulnerability of different salmonid species, stocks, and rearing types. Trans. Am. Fish. Soc. **130**: 385-396.

Collis, K., Roby, D.D., Craig, D.P., Adamany, S., Adkins, J.Y., and Lyons, D.E. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: implications for loses of juvenile salmonids to avian predation. Trans. Am. Fish. Soc. **131**: 537-550.

Cuthbert, F.J., and Wires, L.R. 1999. Caspian tern (*Sterna caspia*). *In* The birds of North America, number 403. *Edited by* A. Poole and F. Gill. The Birds of North America Inc., Philadelphia, Pennsylvania.

De Robertis, A., Ryer, C.H., Veloza, A., and Brodeur, R.D. 2003. Differential effects of turbidity on prey consumption of piscivorous and planktivorous fish. Can. J. Fish. Aquat. Sci. **60**(12): 1517-1526.

Errington, P.L. 1956. Factors limiting higher vertebrate populations. Science **124**(3216): 304-307.

Estes, R.D., and Goddard, J. 1967. Prey Selection and hunting behavior of the African wild dog. J. Wildl. Manage. **31**(1): 52-70.

Evans, A.F., Beaty, R.E., Fitzpatrick, M.S., and Collis, K. 2004. Identification and enumeration of steelhead kelts at a Snake River hydroelectric dam. Trans. Am. Fish. Soc. **133**(5): 1089-1099.

Evans, A.F., Roby, D.D., Collis, K., Cramer, B.M., Sheggeby, J.A., Adrean, L.J., Battaglia, D.S., and Lyons, D.E. *In press*. Recovery of coded wire tags on a Caspian tern colony in San Francisco Bay: a technique to evaluate impacts of avian predation on juvenile salmonids. N. Am. J. Fish. Manage.

Fritts, A.L., Scott, J.L., and Pearsons, T.N. 2007. The effects of domestication on the relative vulnerability of hatchery and wild origin spring Chinook salmon (*Oncorhynchus tshawytscha*) to predation. Can. J. Fish. Aquat. Sci. **64**(5): 813-818.

Good, T.P., Waples, R.S., and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commerce NOAA Tech. Memo NMFS-NWFSC-66: 598.

Gregory, R.S. 1993. Effect of turbidity on the predator avoidance-behavior of juvenile Chinook salmon (*Oncorhynchus tshawytscha*). Can. J. Fish. Aquat. Sci. **50**(2): 241-246.

Gregory, R.S., and Levings, C.D. 1998. Turbidity reduces predation on migrating juvenile Pacific salmon. Trans. Am. Fish. Soc. **127**(2): 275-285.

Hatch, J.J., and Weseloh, D.V. 1999. Double-crested cormorant (*Phalacrocorax auritus*). *In* The birds of North America, number 441. *Edited by* A. Poole and F. Gill. The Birds of North America Inc., Philadelphia, Pennsylvania.

Hosmer, D.W., and Lemeshow, S. 2000. Applied logistic regression. John Wiley & Sons, Inc., Hoboken, N.J.

Hostetter, N.J., Evans, A.F., Roby, D.D., Collis, K., Hawbecker, M., Sandford, B.P., Loge, F.J., and Thompson, D.E. *In press*. Relationship of external fish condition to pathogen prevalence and out-migration survival in juvenile steelhead. Trans. Am. Fish. Soc.

Ims, R.A. 1990. On the adaptive value of reproductive synchrony as a predator-swamping strategy. The American Naturalist **136**(4): 485-498.

Johnsson, J.I., and Abrahams, M.V. 1991. Interbreeding with domestic strain increases foraging under threat of predation in juvenile steelhead trout (*Oncorhynchus mykiss*): an experimental study. Can. J. Fish. Aquat. Sci. **48**(2): 243-247.

Kennedy, B.M., Gale, W.L., and Ostrand, K.G. 2007. Relationship between smolt gill Na+, K+ ATPase activity and migration timing to avian predation risk of steelhead trout (*Oncorhynchus mykiss*) in a large estuary. Can. J. Fish. Aquat. Sci. **64**(11): 1506-1516.

Kerfoot, W. C. and Sih A.. 1987. Predation: direct and indirect impacts on aquatic communities. Hanover, New Hampshire, University Press of New England

Korstrom, J.S., and Birtwell, I.K. 2006. Effects of suspended sediment on the escape behavior and cover-seeking response of juvenile Chinook salmon in freshwater. Trans. Am. Fish. Soc. **135**(4): 1006-1016.

Lyons , D.E., D.D. Roby, and K. Collis. 2007. Foraging patterns of Caspian terns and double-crested cormorants in the Columbia River estuary. Northwest Sci. **81**(2): 91-103.

Lyons, D.E. 2010. Bioenergetics-based predator-prey relationships between piscivorous birds and juvenile salmonids in the Columbia River estuary. Ph.D. dissertation, Department of Fisheries and Wildlife, Oregon State University, Corvallis, Oregon.

MacArthur, R.H., and Pianka, E.R. 1966. On optimal use of a patchy environment. The American Naturalist **100**(916): 603.

Mason, J.W., Brynildson, O.M., and Degurse, P.E. 1967. Comparative survival of wild and domestic strains of brook trout in streams. Trans. Am. Fish. Soc. **96**(3): 313-319.

Mesa, M.G. 1994. Effects of multiple acute stressors on the predator avoidance ability and physiology of juvenile Chinook salmon. Trans. Am. Fish. Soc. **123**(5): 786-793.

Mesa, M.G., Poe, T.P., Maule, A.G., and Schreck, C.B. 1998. Vulnerability to predation and physiological stress response in juvenile Chinook salmon (*Oncorhynchus tshawytscha*) experimentally infected with *Renibacterium salmoninarum*. Can. J. Fish. Aquat. Sci. **55**(7): 1599-1606.

Mesa, M.G., Thomas, P.P., Dena, M.G., and James, H.P. 1994. Are all prey created equal? A review and synthesis of differential predation on prey in substandard condition. J. Fish Biol. **45**(Suppl. A): 81-96.

Mesa, M.G., and Warren, J.J. 1997. Predator avoidance ability of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) subjected to sublethal exposures of gas-supersaturated water. Can. J. Fish. Aquat. Sci. **54**(4): 757-764.

Nielsen, L.A. 1992. Methods for marking fish and shellfish. *In* American Fisheries Society Special Publication. Bethesda, Maryland. pp. 104-109.

NOAA (National Oceanic and Atmospheric Administration) Fisheries. 2007. Draft biological opinion for consultation on remand for operation of the Columbia River power system and 11 bureau of reclamation projects in the Columbia Basin and ESA section 10(a)(1)(A) permit for juvenile fish transportation program, Portland, Oregon.

Olla, B.L., and Davis, M.W. 1989. The role of learning and stress in predator avoidance of hatchery-reared coho salmon (*Oncorhynchus kisutch*) juveniles. Aquaculture **76**(3-4): 209-214.

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

Prentice, E.F., Flagg, T.A., McCutcheon, C.S., Brastow, D.F., and Cross, D.C. 1990b. Equipment, methods and an automated data-entry station for PIT tagging. In Fish-marking techniques. Edited by N.C. Parker, A.E. Giorgi, R.C. Heidinger, D. Jester Jr, E.D. Prince and G.A. Winans. American Fisheries Society, Symposium 7, Bethesda, Maryland.

Raymond, H.L. 1979. Effects of dams and impoundments on migrations of juvenile Chinook salmon and steelhead from the Snake River, 1966 to 1975. Trans. Am. Fish. Soc. **108**(6): 505-529.

Rice, J.A., Crowder, L.B., and Marschall, E.A. 1997. Predation on juvenile fishes: dynamic interactions between size structured predators and prey. *In* Early life history and recruitment in fish populations. *Edited by* R.C. Chambers and E.A. Trippel. Chapman and Hall, London. pp. 333-356.

Rice, J.A., Crowder, L.B., and Rose, K.A. 1993. Interactions between size-structured predator and prey populations: experimental test and model comparison. Trans. Am. Fish. Soc. **122**(3): 481-491.

Roby, D.D., Collis, K., Lyons, D.E., Craig, D.P., Adkins, J.Y., Myers, A.M., and Suryan, R.M. 2002. Effects of colony relocation on diet and productivity of Caspian terns. J. Wildl. Manage. **66**(3): 662-673.

Roby, D.D., Lyons, D.E., Craig, D.P., Collis, K., and Visser, G.H. 2003. Quantifying the effect of predators on endangered species using a bioenergetics approach: Caspian terns and juvenile salmonids in the Columbia River estuary. Can. J. Zool. **81**(2): 250-265.

Ryan, B.A., Smith, S.G., Butzerin, J.M., and Ferguson, J.W. 2003. Relative vulnerability to avian predation of juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary, 1998-2000. Trans. Am. Fish. Soc. **132**(2): 275-288.

Schreck, C.B. 1981. Stress and compensation in teleostean fishes: response to social and physical factors. *In* Stress and fish. *Edited by* A.D. Pickering. Academic Press, New York. pp. 295-321.

Schreck, C.B., Stahl, T.P., Davis, L.E., Roby, D.D., and Clemens, B.J. 2006. Mortality estimates of juvenile spring-summer Chinook salmon in the lower Columbia River and estuary, 1992-1998: evidence for delayed mortality? Trans. Am. Fish. Soc. **135**(2): 457-475.

Schaller, G.B. 1968. Hunting behaviour of the cheetah in the Serengeti National Park, Tanzania. Afr. J. Ecol. **6**(1): 95-100.

Sih, A. 1987. Predators and prey lifestyles: an evolutionary and ecological overview. *In* Predation: direct and indirect impacts on aquatic communities. *Edited by* W.C. Kerfoot and A. Sih. University Press of New England Hanover, New Hampshire, USA. pp. 203-224.

Sih, A., Englund, G., and Wooster, D. 1998. Emergent impacts of multiple predators on prey. Trends Ecol. Evol. **13**(9): 350-355.

Sogard, S.M. 1997. Size-selective mortality in the juvenile stage of teleost fishes: a review. Bull. Mar. Sci. **60**(3): 1129-1157.

Stephens, D.W., and Krebs, J.R. 1986. Foraging theory. Princeton University Press, Princeton, New Jersey.

Strod, T., Izhaki, I., Arad, Z., and Katzir, G. 2008. Prey detection by great cormorant (*Phalacrocorax carbo sinensis*) in clear and in turbid water. J. Exp. Biol. **211**(6): 866-872.

Temple, S.A. 1987. Do predators always capture substandard individuals disproportionately from prey populations? Ecology **68**(3): 669-674.

USFWS (U.S. Fish and Wildlife Service) 2006. Record of decision: Caspian tern management to reduce predation of juvenile salmonids in the Columbia River estuary: final environmental impact statement. Portland, Oregon.

Waples, R.S. 1991. Definition of ‘‘species’’ under the Endangered Species Act: application to Pacific salmon, U.S. Department of Commerce, NOAA Technical Memorandum NMFS-NWFSC-194.

Table 4.1. External condition description and prevalence (%) for steelhead captured and marked with a passive integrated transponder (PIT) tag at Lower Monumental or Ice Harbor dams during the 2007- 2009 (n = 25,909). External condition descriptions from Hostetter et al. (*in press*).

External Condition		Tagged	%	Description
Body injury	Absent	20,963	81%	No visible hemorrhaging, scarring, or other damage to the head, trunk, operculum, or eyes
	Moderate	3,025	12%	Closed or healed scars to the head, trunk, operculum, or eyes
	Severe	1,921	7%	Deformities, open wounds, or large surface area scarring on the head, trunk, operculum, or eyes
De-scaling	< 5%	16,975	65%	Scale loss on < 5% of body
	5-20%	7,916	31%	Scale loss on 5-20% of body
	>20%	1,018	4%	Scale loss on >20% of body
Disease	Absent	24,917	96%	No external symptoms of bacterial, fungal, or viral infections
	Moderate	349	1%	Visible infection limited to one external area
	Severe	643	3%	Visible infection in multiple areas or symptoms that suggest a systemic infection
Ectoparasites	Absent	25,217	97%	No visible ectoparasites
	Moderate	523	2%	Visible ectoparasites found in 1 area
	Severe	169	1%	Visible ectoparasites in > 1 area or on gills
Fin Damage	Absent	5,762	22%	Fin wear and damage < 50% on any fin
	Moderate	14,553	56%	Fin wear and damage > 50% on 1-2 fins
	Severe	5,594	22%	Fin wear and damage > 50% on ≥ 3 fins
Integrated Condition[a]	Good	15,228	59%	No noticeable external injury or symptoms of disease; de-scaling ≤ 10% of body surface
	Fair	7,197	28%	Minor scars or other closed external damage; de-scaling > 10% but ≤ 50% of body surface
	Poor	3,484	13%	Any steelhead with externally apparent fungal, parasitic, or bacterial infections, or de-scaling > 50% of body, or open external body lesions

[a] Modified version of the procedure developed by Evans et al. (2004) for scoring external condition in adult steelhead

Table 4.2. Rearing-type (hatchery or wild) and length of steelhead captured and marked with a passive integrated transponder (PIT) tag at Lower Monumental or Ice Harbor dams during the 2007- 2009 (n = 25,909).

			Length		
Year	Rearing-type	Number Sampled	Minimum	Mean	Maximum
2007	Hatchery	6,101	144	223	305
	Wild	964	140	193	285
2008	Hatchery	7,786	132	224	375
	Wild	1,357	134	179	354
2009	Hatchery	8,248	144	227	341
	Wild	1,453	131	186	316
Combined	Hatchery	22,135	132	225	375
	Wild	3,774	131	185	354

Table 4.3. External condition prevalence (%) for hatchery and wild steelhead captured and marked with a passive integrated transponder (PIT) tag at Lower Monumental or Ice Harbor dams during the 2007- 2009 (n = 25,909).

		Hatchery		Wild	
External Condition		Tagged	%	Tagged	%
Body injury	Absent	17,528	79%	3,435	91%
	Moderate	2,852	13%	173	5%
	Severe	1,755	8%	166	4%
De-scaling	< 5%	13,790	62%	3,185	84%
	5-20%	7,373	33%	543	14%
	>20%	972	4%	46	1%
Disease	Absent	21,235	96%	3,682	98%
	Moderate	319	1%	30	1%
	Severe	581	3%	62	2%
Ectoparasites	Absent	21,886	99%	3,331	88%
	Moderate	215	1%	308	8%
	Severe	34	0%	135	4%
Fin Damage	Absent	3,140	14%	2,622	69%
	Moderate	13,519	61%	1,034	27%
	Severe	5,476	25%	118	3%
Integrated Condition[a]	Good	12,569	57%	2,659	70%
	Fair	6,754	31%	443	12%
	Poor	2,812	13%	672	18%

[a] Modified version of the procedure developed by Evans et al. (2004) for scoring external condition in adult steelhead

Table 4.4. Steelhead smolt PIT tags detected on the Crescent Island Caspian tern (tern) and the Foundation Island double-crested cormorant (cormorant) colonies during 2007-2009.

		Recovered on tern colony		Recovered on cormorant colony	
Year	Number released	Number	%	Number	%
2007	7,065	212	3.0%	139	2.0%
2008	9,143	279	3.1%	178	1.9%
2009	9,701	422	4.4%	176	1.8%
Combined	25,909	913	3.5%	493	1.9%

Table 4.5. Model selection results used to evaluate the influence of individual fish characteristics and environmental variables on the susceptibility of juvenile steelhead to predation by Caspian terns (Tern) or double-crested cormorants (Cormorant).

		Tern		Cormorant	
Model[a]	df	AIC_c	ΔAIC_c	AIC_c	ΔAIC_c
Individual characteristic					
Body injury	8	6881.7	262.8	4861.7	68.3
De-scaling	8	6881.3	262.4	4865.4	72.0
Disease	8	6883.3	264.4	4829.2	35.8
Ectoparasites	8	6881.9	263.0	4861.1	67.7
Fin damage	8	6882.7	263.8	4858.7	65.3
Integrated condition	8	6878.6	259.7	4848.9	55.5
Rearing type	7	6880.3	261.4	4854.7	61.3
Length	7	6769.9	151.0	4864.9	71.5
Length + length^2	8	6635.8	16.9	4863.5	70.1
Environmental variable					
Discharge	7	6881.1	262.2	4865.4	72.0
Water clarity	7	6875.0	256.1	4861.5	68.1
Release location	7	6878.8	259.9	4834.1	40.7
A priori					
Base model[a]	6	6881.9	263.0	4863.3	70.0
Global model[b]	23	6627.0	8.1	4804.0	10.6
Backward stepwise					
Length + length^2 + integrated condition + discharge + water clarity	12	6618.9	0.0	-	-
Disease + release location + rearing type	10	-	-	4793.4	0.0

[a] All models controlled for detection efficiency, predator abundance, prey abundance, and migration year ("Base Model").

[b] Additive model that included all individual characteristics and environmental variables.

Table 4.6. Results from the top model used to evaluate susceptibility of juvenile steelhead to predation by Caspian terns.

Variable[a]	Effect	df	χ^2	Odds Ratio	95% CI	P
Length	1-cm increase	1	83.1	3.95	2.94 - 5.31	< 0.001
Length^2	1-cm increase	1	96.4	0.97	0.96 - 0.97	< 0.001
Steelhead abundance	1,000-increase	1	61.6	0.97	0.96 - 0.98	< 0.001
Water clarity	1-m increase in clarity	1	13.2	0.56	0.41 - 0.77	< 0.001
Discharge	10-kcfs increase	1	12.3	0.95	0.92 - 0.98	< 0.001
Migration year	2008 vs. 2007	1	44.2	2.60	1.96 - 3.44	< 0.001
	2009 vs. 2007	1	3.5	1.34	0.99 - 1.83	0.060
Integrated condition	Fair vs. Good	1	0.5	1.06	0.90 - 1.25	0.494
	Poor vs. Good	1	6.7	1.29	1.06 - 1.57	0.010
Tern abundance	100-bird increase	1	0.8	1.03	0.97 - 1.09	0.380

[a] See Methods for variable descriptions

Table 4.7. Results from the top model used to evaluate susceptibility of juvenile steelhead to predation by double-crested cormorants.

Variable[a]	Effect	df	χ^2	Odds Ratio	95% CI	P
Disease	Moderate vs. Absent	1	15.3	2.78	1.67-4.65	< 0.001
	Severe vs. Absent	1	33.8	2.94	2.05-4.24	< 0.001
Release Location	LMN vs. ICH	1	33.7	0.53	0.43-0.66	< 0.001
Cormorant Abundance	100-bird increase	1	12.0	1.15	1.06-1.25	< 0.001
Rearing	Hatchery vs. Wild	1	7.8	1.54	1.14-2.09	0.005
Release Year	2008 vs. 2007	1	5.9	1.43	1.07-1.90	0.015
	2009 vs. 2007	1	3.2	1.45	0.97-2.16	0.072
Steelhead Abundance	1,000-increase	1	0.0	1.00	0.99-1.01	0.946

[a] See Methods for variable descriptions

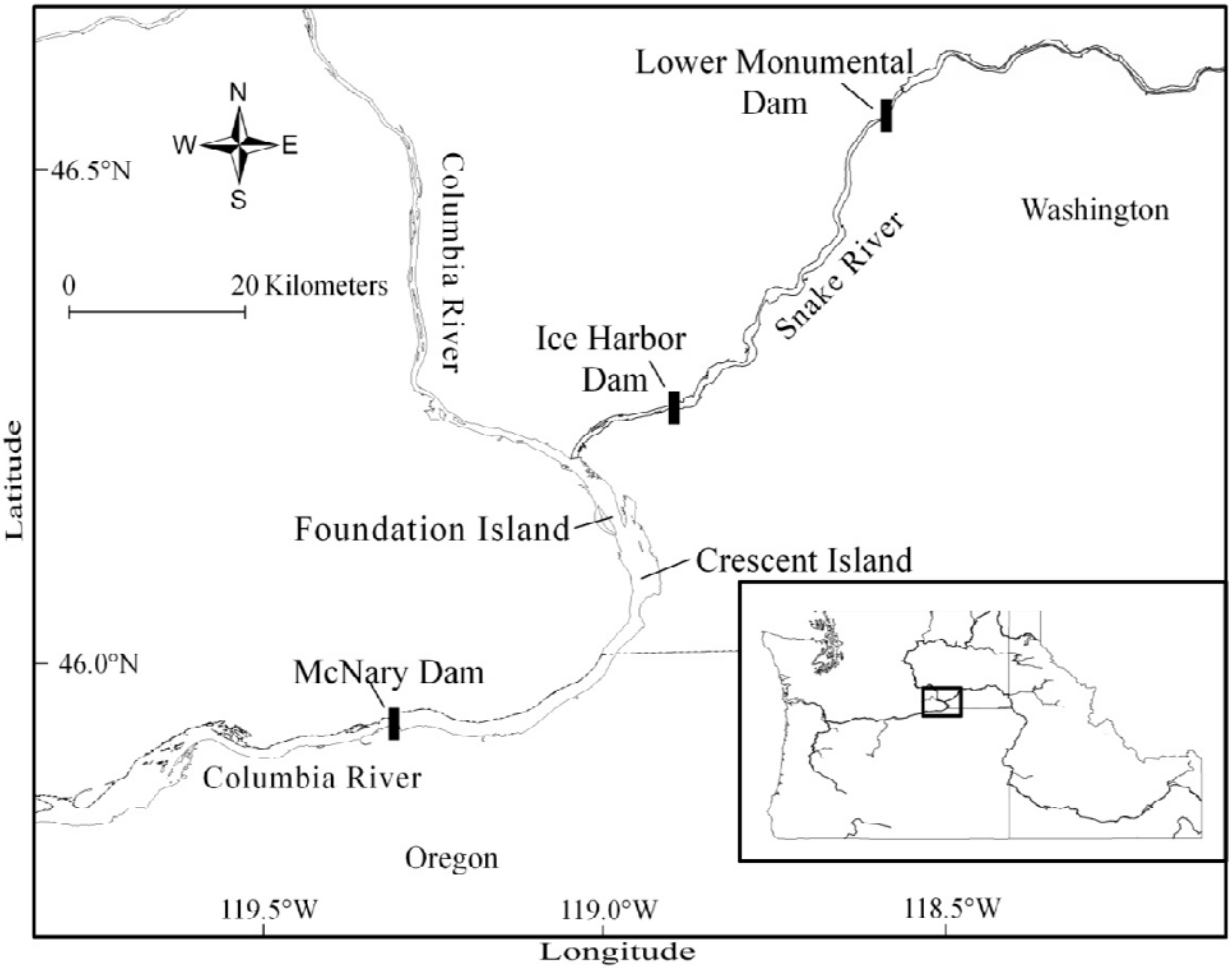

Figure 4.1. Map of the mainstem Columbia and Snake rivers showing hydroelectric dams (bars) where smolts were PIT-tagged and released, the Caspian tern colony at Crescent Island, and the double-crested cormorant colony at Foundation Island.

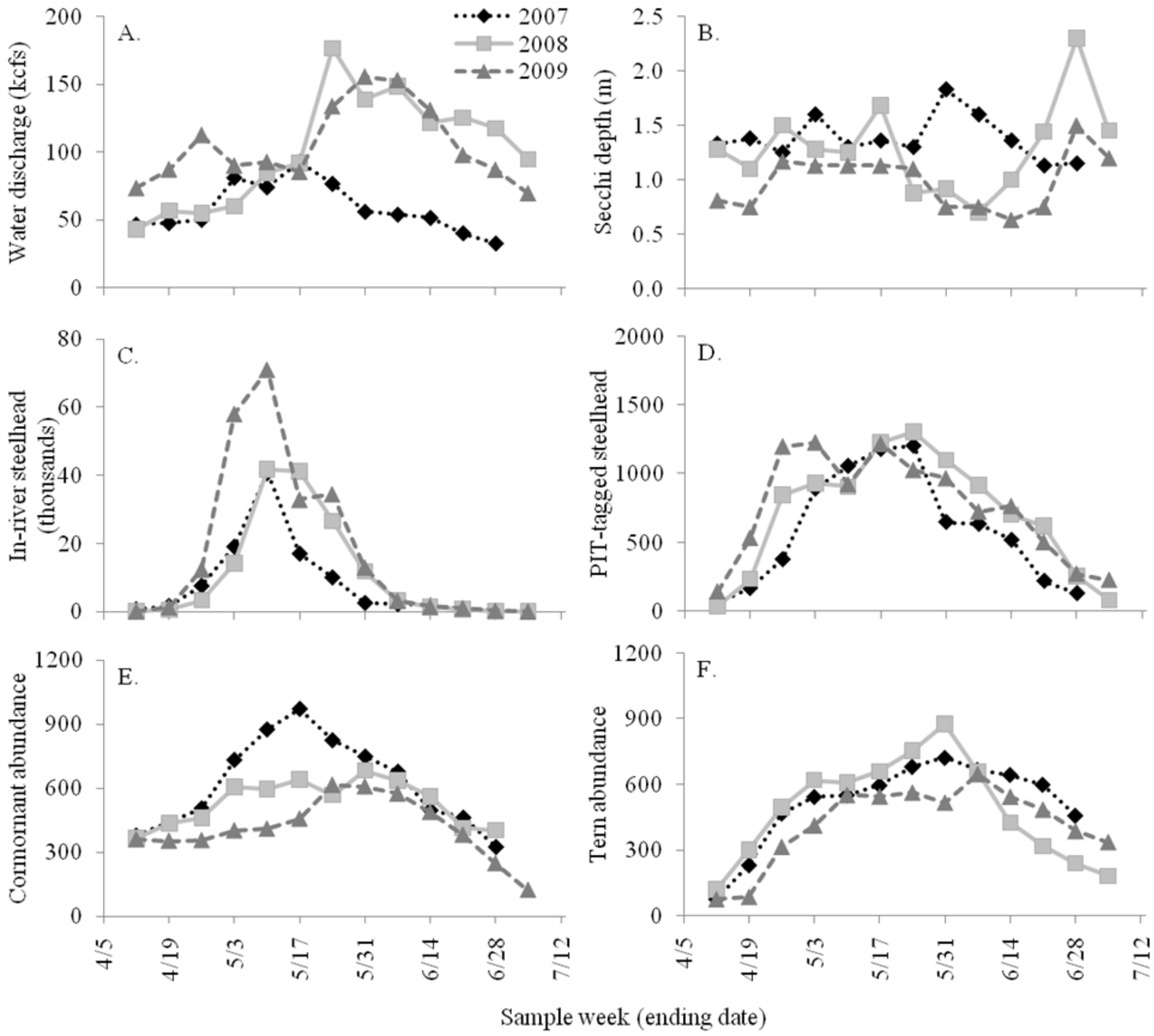

Figure 4.2. Weekly averages of (A) water discharged (kcfs) from Lower Monumental Dam, (B) water clarity (Secchi depth in meters), (C) number of in-river steelhead smolts, (D) number of steelhead smolts tagged (passive integrated transponder [PIT] tags) and released as part of this study, (E) number of double-crested cormorants on Foundation Island, and (F) number of Caspian terns on Crescent Island. Data points are plotted on the last day of each sample week. See Methods for complete variable descriptions.

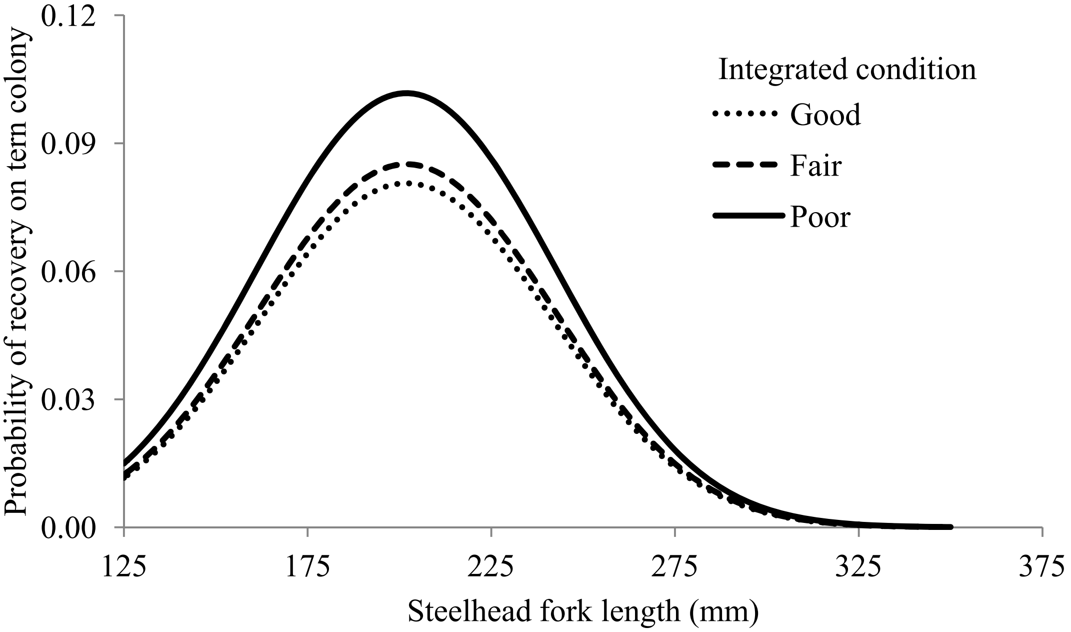

Figure 4.3. Regression lines for the relationship between steelhead fork length and juvenile steelhead susceptibility to predation by Caspian terns. Regression lines were calculated when other variables in the top model (colony-specific PIT tag detection efficiency, number of steelhead smolts in-river, number of Caspian terns on the Crescent Island colony, water clarity, and discharge) were held constant at their respective median values.

Appendix D.1. Individual variable results from logistic regression models used to evaluate the susceptibility of juvenile steelhead to predation by Caspian terns.

Variable[a]	df	χ^2	P	Effect	Odds Ratio	95% CI	P
Individual characteristic							
Body injury	2	4.4	0.112	Moderate vs. Absent	0.94	0.76 - 1.16	0.569
				Severe vs. Absent	1.24	0.99 - 1.55	0.057
De-scaling	2	4.9	0.086	5-20% vs. < 5%	1.09	0.93 - 1.27	0.298
				>20% vs. < 5%	1.40	1.02 - 1.92	0.035
Disease	2	2.9	0.232	Moderate vs. Absent	1.55	0.86 - 2.77	0.145
				Severe vs. Absent	1.25	0.78 - 2.00	0.353
Ectoparasites	2	4.5	0.105	Moderate vs. Absent	1.56	1.02 - 2.37	0.039
				Severe vs. Absent	0.79	0.28 - 2.22	0.660
Fin damage	2	3.2	0.198	Moderate vs. Absent	0.87	0.74 - 1.03	0.115
				Severe vs. Absent	0.98	0.80 - 1.21	0.855
Integrated condition	2	7.7	0.022	Fair vs. Good	1.05	0.89 - 1.23	0.584
				Poor vs. Good	1.31	1.08 - 1.58	0.006
Rearing type	1	3.7	0.053	Hatchery vs. Wild	0.83	0.68 - 1.00	0.053
Length	1	111.8	< 0.001	1-cm increase	0.90	0.88 - 0.91	< 0.001
Length + length^2	2	97.8	< 0.001	1-cm increase	-	- -	< 0.001
Environmental variable							
Discharge	1	2.8	0.093	10-kcfs increase	0.98	0.95 - 1.00	0.093
Water clarity	1	8.7	0.003	1-m increase in clarity	0.66	0.50 - 0.87	0.003
Release location	1	4.8	0.028	LMN vs. ICH	1.32	1.03 - 1.69	0.028

[a] Associations determined after controlling for detection efficiency, predator abundance, prey abundance, and migration year.

Appendix E.1. Individual variable results from logistic regression models used to evaluate the susceptibility of juvenile steelhead to predation by double-crested cormorants.

Variable[a]	df	χ^2	P	Effect	Odds Ratio	95% CI	P
Individual characteristic							
Body injury	2	6.0	0.050	Moderate vs. Absent	1.19	0.91-1.57	0.200
				Severe vs. Absent	1.42	1.05-1.93	0.025
De-scaling	2	2.0	0.376	5-20% vs. < 5%	1.15	0.95-1.39	0.164
				>20% vs. < 5%	1.09	0.69-1.70	0.721
Disease	2	48.9	< 0.001	Moderate vs. Absent	2.69	1.61-4.50	< 0.001
				Severe vs. Absent	3.08	2.15-4.43	< 0.001
Ectoparasites	2	0.2	0.917	Moderate vs. Absent	0.86	0.43-1.75	0.680
				Severe vs. Absent	NA[b]	NA[b]	0.958
Fin Damage	2	8.8	0.012	Moderate vs. Absent	1.17	0.91-1.49	0.218
				Severe vs. Absent	1.49	1.13-1.96	0.005
Integrated Condition	2	19.9	< 0.001	Fair vs. Good	1.20	0.97-1.48	0.092
				Poor vs. Good	1.72	1.36-2.19	< 0.001
Rearing type	1	9.4	0.002	Hatchery vs. Wild	1.61	1.19-2.18	0.002
Length	1	0.5	0.482	1-cm increase	1.01	0.98-1.04	0.482
Length + Length^2	2	3.1	0.078	NA	-	-	0.078
Environmental variable							
Discharge	1	0.0	0.959	10-kcfs increase	1.00	0.96-1.04	0.959
Water Clarity	1	3.9	0.047	1-m increase in clarity	1.44	1.00-2.07	0.047
Release Location	1	34.9	< 0.001	LMN vs. ICH	0.52	0.42-0.65	< 0.001

[a] Associations determined after controlling for detection efficiency, predator abundance, prey abundance, and migration year.

[b] No steelhead with severe ectoparasites were recovered on the cormorant colony.

CHAPTER 5

ABUNDANCE, DISTRIBUTION, AND DIET COMPOSITION OF DOUBLE-CRESTED CORMORANTS OVER-WINTERING ON THE LOWER SNAKE RIVER

This chapter has been prepared for the U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This chapter summarizes data from double-crested cormorants that over-wintered on the Snake River during 2007-2009.

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SUMMARY

The abundance, distribution, and diet of double-crested cormorants (*Phalacrocorax auritus*) over-wintering on the lower Snake River in eastern Washington were assessed from October 2007 to February 2010. We conducted monthly boat-based surveys of the lower 224 river kilometers (Rkm) of the Snake River during the winters of 2008-09 and 2009-10 and found cormorants throughout the study area. A monthly average of 256 cormorants was observed on this reach of the Snake River, or an average of about 1.2 cormorants per Rkm. Roughly 22% of these cormorants were recorded within 0.25 Rkm of the four federal hydroelectric dams in the study area. During the winters of 2007, 2008, and 2009we lethally collected 130 cormorants with identifiable prey remains in their foreguts to determine whether over-wintering cormorants were consuming U.S. Endangered Species Act (ESA) listed fall Chinook salmon (*Oncorhynchus tshawytscha*) that also over-winter in the lower Snake River. Salmonids comprised an average of 11.7% (by mass) of the cormorant diet, with both juvenile and adult (precocious males or jacks) salmonids confirmed in the diet. ESA-listed fall Chinook salmon comprised an average of just 3.4% of the diet by mass but were the most numerous salmonid found by number (individual fish). The most prevalent non-salmonid prey in the foregut contents samples were centrarchids (bass and sunfishes), which accounted for 34.3% by mass of the diet during the study. Juvenile shad (15.0%), cyprinids (minnows and carp; 11.7%), catostomids (suckers; 7.0%), ictalurids (catfish; 6.3%), and percids (perch; 3.4%) were the next most prevalent non-salmonid prey types in the cormorant diet samples.

Overall diet composition of cormorants over-wintering in this stretch of the lower Snake River was highly variable and changed throughout the season, but juvenile fall Chinook salmon appeared to be a minor component of the overall diet. The impact this small component has on the fall Chinook population, however, depends on the relative availability of juvenile fall Chinook and, ultimately, the probability that these fish will survive to return as adults. Estimates of total salmonid biomass consumed by cormorants over-wintering on the lower Snake River were between 3,100 – 11,000 kg annually, significantly less than that of cormorants nesting on nearby Foundation Island during the breeding season.

INTRODUCTION

The double-crested cormorant (*Phalacrocorax auritus;* hereafter referred to as cormorant) is a wide-spread, fish-eating, colonial nesting bird species found throughout North America (Hatch and Weseloh 1999). Since the 1970s, the cormorant population, as a whole, has grown significantly throughout the species range (Wires et al. 2001). Cormorant populations east of the Continental Divide have increased at both breeding and over-wintering locations (Hatch 1995, Scherr 2010). The Western North America Population of the species also increased over the past three decades, but at a lower rate than the population east of the Continental Divide and with most of the growth occurring at a single cormorant colony in the Columbia River estuary (Chapter 1; Adkins and Roby 2010; Lyons 2010). While most cormorants east of the Continental Divide have clear migratory patterns in fall and spring, most cormorants in the Western Population remain as year-round residents, migrating only short distances (< 200 km) seasonally (Adkins and Roby 2010). However, little has been published on the distribution and abundance of cormorants from the Western North America Population that over-winter at inland sites (Adkins and Roby 2010).

Fisheries managers have been concerned that cormorant predation negatively affects fish populations (Blackwell et al. 1997, Derby and Lovvorn 1997, Jones et al. 2010). Avian predation has been identified as one of several factors limiting recovery of some evolutionarily significant units (ESUs; Good et al. 2005) of salmonids (*Oncorhynchus* spp.) from the Columbia River basin that are listed as threatened or endangered under the U.S. Endangered Species Act (ESA) (Roby et al. 2003; Lyons 2010). Smolt mortality due to avian predation in the Columbia River basin is most significant from two species of piscivorous colonial waterbirds, Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (Collis et al. 2002), with comparatively lower impacts from other fish-eating waterbirds (e.g., California gulls [*Larus californicus*], ring-billed gulls [*Larus delawarensis*], common mergansers [*Mergus merganser*]; Collis et al 2001, Wiese et al. 2008, Chapter 3).

Most juvenile salmonids from the Columbia River basin migrate to the ocean during April – July, making them susceptible to predation by piscivorous waterbirds that breed in the basin (Collis et al. 2001, Chapter 3). However, a proportion of juvenile fall Chinook salmon (*Oncorhynchus tshawytscha*) that originates from the Snake River over-winters in the mainstem of the Snake and Columbia rivers (reservoir-type; Connor et al. 2005). Connor et al. (2002) concluded that the creation of dams on the lower Snake River changed the river environment, specifically rearing temperatures, and influenced the life history of fall Chinook salmon. The study found that the reservoir-type life history was most prevalent in fall Chinook salmon originating from the Clearwater River, where ca. 53% of wild smolts over-wintered in Snake River reservoirs. Survival of reservoir-type fall Chinook salmon is of particular interest to managers because this life history strategy is believed to have a higher survival rate to adulthood compared to fall Chinook salmon that out-migrated to the ocean in their first year (ocean-type; Connor et al. 2005).

Because reports of cormorants over-wintering on the lower Snake River increased from a few individuals to several hundred over a five-year period (S. Dunmire, USACE, pers. comm.), a pilot study was initiated from September-December 2007 (referred to hereafter as 2007), in which opportunistic counts of cormorants present in the forebays and tailraces at Lower Granite and Little Goose dams occurred several times a day, on up to three days per week. Counts from the pilot study confirmed that cormorants over-wintered on the lower Snake River and utilized the habitat immediately surrounding the dams. Thus, the pilot study data suggested possible interactions between reservoir-type fall Chinook salmon and piscivorous waterbirds over-wintering in the Columbia River basin that had yet to be addressed in previous studies, even though substantial cormorant predation has been shown to occur during the spring and summer months (Collis et al. 2002, Lyons 2010).

The objectives of this study were to (1) document the abundance and distribution of piscivorous waterbirds, particularly double-crested cormorants, over-wintering on the lower Snake River, (2) locate areas of high double-crested cormorant abundance or density, and (3) determine the diet composition of over-wintering cormorants, with an emphasis on evaluating consumption of ESA-listed fall Chinook salmon.

METHODS

Study Area

The Snake River is the largest tributary of the Columbia River. The lower Snake River is characterized by slow-moving water that passes through large reservoirs separated by four hydroelectric dams (Figure 5.1). Our study area encompassed the lower Snake River from the confluence with the Columbia River to the confluence with the Clearwater River, a total length of 224 Rkm. We separated our study area into five river reaches separated by four mainstem hydroelectric dams: Ice Harbor Dam (Rkm 16), Lower Monumental Dam (Rkm 67), Little Goose Dam (Rkm 113), and Lower Granite Dam (Rkm 173; Figure 5.1). The coinciding reaches were defined as (1) confluence of the Columbia River to Ice Harbor Dam (16 Rkms), (2) Ice Harbor Dam to Lower Monumental Dam (51 Rkms), (3) Lower Monumental Dam to Little Goose Dam (46 Rkms), (4) Little Goose Dam to Lower Granite Dam (60 Rkms) and (4) Lower Granite Dam to the confluence of the Clearwater River (51 Rkms; Figure 5.1).

Distribution and Abundance

To estimate the abundance and distribution of piscivorous waterbirds on the lower Snake River, surveys were conducted from a boat travelling through each reach at 10-15 knots, with two trained individuals (a boat operator and an observer) onboard. To survey each river reach, two counts were made, one as the boat travelled up-river, and one as the boat travelled down-river, with at least 30 min separating the two directional surveys. The observer was stationed in the bow of the boat and image stabilizing binoculars (10x42) were used to observe and enumerate all piscivorous birds. We were able to positively identify birds on on the river and both shorelines from the middle of the river. Surveys were limited to days and time periods when surface waves, rain, and fog did not prohibit observations. Surveys of each river reach required 1-2 days to complete and each entire lower Snake River study area survey required 5-7 days to complete. Entire study area surveys were completed monthly during the winters of 2008 (October 2008-February 2009) and 2009 (November 2009-February 2010).

The presence and number of all piscivorous waterbirds were recorded during the survey of each river reach. In addition to cormorants, data was also collected on four groups of piscivorous birds that were frequently observed; ring-billed and California gulls (*Larus delawarensis* and *L. californicus*), Clark's and western grebes (*Aechmophorus clarkii* and *A. occidentalis*), American white pelicans (*Pelecanus erythrorhynchos*), and common mergansers (*Mergus merganser*). Because of difficulties in identify each species, gull spp. and grebe spp. were only identified to genus, while pelican and mergansers were identified to species. Monthly bird abundance for each genus or species in each reach was determined by averaging the up-river and down-river counts in a reach during a survey. Average bird abundance per reach was summed across reaches to estimate total abundance of piscivorous waterbirds on the lower Snake River for each month. The presence or absence of birds within 0.25 Rkm of mainstem dams was recorded for all piscivorous waterbirds. For cormorants flocks of more than five birds, Rkm and GPS location were also recorded. Cormorant densities were calculated for each of the nine months when surveys were conducted by dividing the number of individuals observed in a reach by the length (Rkm) of the reach. Each dam was given a fixed length of 0.5 Rkm to calculate density of birds observed in the fore-bay and tailrace of each dam. Although the possibility of double-counting birds in different reaches existed, there was no evidence to suggest that movement rates varied among or between reaches. We assumed equal probability of birds entering or leaving a reach at any given time.

Roost locations for cormorants were identified in four of the five river reaches. In reach 1 an island in the tailrace of Ice Harbor dam was identified as a known nighttime roost. In addition, the Foundation Island breeding colony, 4 Rkm downstream of the confluence with the Snake River, was also confirmed as a nighttime roost for cormorants. In reach 3 the Lyons Ferry Bridge was identified as a roost location, as was the Central Ferry Bridge in reach 4. In reach 5 Swallows Park on the Snake River, five Rkm upriver of the confluence with the Clearwater River, was identified as a roost in 2009 and surveyed monthly (Figure 5.1).

Diet Collections

Diet composition was determined by lethally collecting cormorants on the lower Snake River during the winters of 2007, 2008, and 2009. Cormorants were opportunistically shot using shotguns between Lower Monumental and Lower Granite Dams, where cormorant densities were observed to be high during surveys (e.g., at dams, Lyons Ferry Bridge, Central Ferry Bridge; Figure 5.1). A minimum of 10 cormorants were collected each month (October to February); however, 10 specimens with identifiable prey remains in the foregut were not always collected. Dissections were performed promptly in the field to remove the foreguts of each collected cormorant. The complete foregut contents of each cormorant that contained identifiable prey remains was scraped into a sample bag, weighed, labeled, and frozen for later laboratory analysis. During laboratory analysis at Oregon State University, we identified prey to the lowest possible taxonomic classification using identification keys (Wydoski and Whitney 2003) and previously developed protocols (Collis et al. 2002). Tissue samples were collected from all presumed salmonid prey items for genetic analyses conducted by D. Kuligowski with NOAA Fisheries. Species identifications were carried out by amplifying (PCR) the mitochondrial DNA fragment COIII/ND3 as outlined by Purcell et al. (2004). Samples identified as Chinook salmon were genotyped with 13 standardized microsatellite DNA markers (Seeb et al. 2007). Stock origins of individual Chinook salmon were estimated using standard genetic assignment methods (Van Doornik et al. 2007).

Unidentifiable fish soft tissue samples were artificially digested according to the methods of Peterson et al. (1990; 1991). Once digested, diagnostic bones (i.e., otoliths, cleithra, dentaries, and pharyngeal arches) were removed from the sample and identified to species using a dissecting microscope (Hansel et al. 1988). Unidentified fish soft tissue samples that did not contain diagnostic bones and samples comprised of bones only (i.e., no soft tissue) were excluded from diet composition analysis. Taxonomic composition of double-crested cormorant diets was expressed as percent of identifiable prey biomass. Prey composition was calculated for each month, with monthly collections averaged across all three years. Due to the opportunistic nature of our collections and the uneven distribution of samples across years, diet composition data were pooled across all years.

All diet samples collected from cormorant foreguts were scanned for the presence of passive integrated transponder (PIT) tags from salmonid smolts. When present, PIT tags were recovered and information associated with the tag code, including species, run, and rear-type (hatchery or wild) of the fish was retrieved from the PIT Tag Information System (PTAGIS 2010) maintained by the Pacific States Marine Fisheries Commission (PTAGIS 2010). Detailed information from PIT tag codes was then used to verify visual identification of prey species and to identify any ESA-listed salmonid species in the cormorant diet.

We used the information collected on cormorant diet and abundance to estimate the over-all mass of salmonids consumed by cormorants wintering on the lower Snake River. Bioenergetics calculations were performed as in Chapter 2 for cormorants nesting at Foundation Island, but modified by several simplifying assumptions. First, daily energy requirements for wintering double-crested cormorants have not been measured in the region. We assumed two plausible possibilities and performed the calculations for both scenarios: (1) daily energy expenditure (DEE) was similar to that estimated for double-crested cormorants post-breeding (August – October) in the Columbia River estuary (Lyons 2010) and (2) DEE was equivalent to that measured in great cormorants (*Phalacrocorax carbo*) wintering in Germany (Keller and Visser 1999). Second, we assumed that the average number of cormorants seen across the season in surveys (256 individuals; see results) was a reasonable approximation of the size of the cormorant population that was present to prey upon salmonids across the winter. Third, we assumed a 5-month (154-day) period of possible consumption (October – February). Summary diet composition data obtained as described above were used for calculations. All other variables, and the bioenergetics model structure, followed that used to estimate consumption by cormorant breeding colonies (see Lyons 2010). Finally, due to small sample sizes and a lack of information on the life stage (juvenile, adult) of consumed salmonids in the diet of cormorants, we were unable to convert total salmonid biomass into an estimate of total fish consumption.

RESULTS

Distribution and Abundance

Cormorants were observed in all five river reaches surveyed, from the Columbia River to the Clearwater River, in every month of the study. A geospatial database was constructed to map congregations of five or more cormorants (Figure 5.1). Congregations were observed throughout the study area, including at each of the four dams. Cormorants observed throughout the river and not associated with dam structures were often found on or near other man-made structures, such as channel markers, pilings, log booms, suspended cables, and bridges. To a lesser extent natural habitat was utilized by cormorants, typically islands and exposed snags; very rarely were cormorants observed loafing on the river's shoreline. Large flocks of cormorants were frequently observed at or near the roost locations identified during the surveys (Figure 5.1).

The number of over-wintering cormorants on the lower Snake River was smaller in comparison to that of cormorants nesting on nearby Foundation Island (confluence of the Snake and Columbia rivers) during the spring breeding season (ca. 300 – 350 breeding pairs) (Chapter 1). A monthly average of 256 individual cormorants (calculated by averaging all boat based survey totals) was observed in the study area from October 2008 – February 2010. However, abundance varied by month and decreased through the winter in both 2008 and 2009 (Figure 5.2). High counts of cormorants were observed in November of each year, with a maximum count of 395 in November 2008. The fewest cormorants were observed in January and February, with a minimum of 159 counted in January 2010. Overall fewer cormorants were observed during the winter 2009 winter surveys, a monthly average of 224, compared to the 2008 winter surveys, when an average of 281 cormorants was counted.

Gull spp. were the most abundant piscivorous waterbirds encountered during the surveys (Figure 5.3), peaking in November 2008 with 686 individuals counted. Grebe spp. were also commonly observed throughout the study area, with a maximum of 497 individuals counted in November 2008. Both gulls and grebes followed similar seasonal trends as cormorants, peaking in November and declining as the winter progressed. Fewer American white pelicans and common mergansers were observed; these were almost exclusively in reach 1 below Ice Harbor Dam. Maximum counts of 115 pelicans and 107 mergansers were recorded in January 2010 and December 2008, respectively. These results indicate that, compared to double-crested cormorants, other piscivorous waterbirds over-winter on the lower Snake River in greater numbers, but with similar seasonal trends in abundance.

Cormorants were widespread throughout the study area during the winters of 2008 and 2009. Although we observed cormorants in varying abundance at each of the four lower Snake River dams during each winter, most cormorants were observed away from dams (74% in 2008 and 80% in 2009, Figure 5.2). In addition, cumulative monthly counts of cormorants observed within 0.25 Rkm from dams were always less than the cumulative counts of cormorants along the river reaches (Figure 5.2). Counts of cormorants at all dams peaked in November or December. At dams, cormorants were commonly observed using structures for loafing, such as lock walls, lights, log booms, and various signs and markers.

The numbers of cormorants per river kilometer was calculated for the vicinity of dams and for each of the five river reaches in order to compare cormorant densities across the study area. While cormorants were more abundant away from dams, densities of cormorants were higher at dams than in the river reaches (Figure 5.4). The density of cormorants varied both between reaches and between months, but remained low, ranging from 0.1 – 5.0 cormorants/Rkm, with the highest densities in reach 1, below Ice Harbor Dam. In comparison, densities at dams were much higher, but also were more variable, both between months and between dams, ranging from 0 to 162 cormorants/Rkm. Seasonal cormorant density was most variable at Ice Harbor Dam (20 – 142 cormorants in 2009) and Lower Monumental Dam (14-162 cormorants in 2008). No cormorants were counted at Lower Granite in two months (January and February) and at Little Goose Dam in one month (February).

Diet Collections

From October 2007 to February 2010 a total of 160 cormorants were collected, of which 130 or 81.3% containing identifiable prey remains in the foregut to determine diet composition. Analysis of foregut samples indicated that a small percentage of the diet of over-wintering cormorants on the lower Snake River consisted of salmonids, specifically juvenile fall Chinook salmon. Salmonids comprised 11.7% of identifiable biomass from all samples combined. The proportion of salmonids was highest in November (24.2%) and lowest in February, when no salmonids were found (Table 5.1). The salmonid percentage was influenced by the presence of precocious adult male (jack) salmon in the samples. Jack salmon were significantly larger than the juvenile salmon of interest in this study. Precocious salmon exhibit a life history in which they partially out-migrate and then quickly return to spawn as sexually mature individuals (Connor et al. 2005). Juvenile fall Chinook salmon were also found in several cormorants collected in November, December, and January (Figure 5.5). Taken together (jacks and juveniles) fall Chinook comprised 3.4% of identifiable biomass from samples. Juvenile steelhead and possibly resident rainbow trout (*O. mykiss)* were also identified in samples from several months and comprised 3.4% of the diet. By mass, the most prevalent salmonid was actually coho salmon (*O. kisutch*), which comprised 3.6% of the diet. Coho, however, were only identified in one year (2007) and were from a limited number of foregut samples (n=8). Partially digested coho found in foregut samples were also significantly larger (> 300 mm) than typical-sized juveniles, suggesting coho were jacks or residualized, not smolts. Fall Chinook salmon were most abundant in terms of number of identified prey items (n = 19), followed by steelhead/rainbow trout (n = 9), and coho salmon (n = 2; Figure 5.5).

The most prevalent non-salmonid prey types identified in foregut samples were centrarchids (sunfish and basses), clupeids (juvenile shad), cyprinids (minnows), and catostomids (suckers) which comprised 34.3%, 15.0%, 11.7%, and 7.0% of prey biomass, respectively. Other fish families identified in cormorant foregut contents samples included ictalurids (catfishes; 6.3%) and percids (perch; 3.4%). Variation in proportions of prey types were observed across months for most fish families (Table 5.1). The most notable changes were observed in the proportions of juvenile shad consumed, which ranged from 0 - 59.1%, and of suckers consumed, which ranged from 0 - 20.1%. The seasonal increase in shad consumption by cormorants was apparently related to the out-migration of juvenile shad, which move through the lower Snake River in November and December; however, there is no apparent explanation for the large increase in sucker consumption in February.

Further evidence of fall Chinook salmon in the diet of over-wintering double-crested cormorant came from PIT tags found in several cormorant foregut samples. Recovered PIT tags provided another opportunity - in concert with genetic analysis - to confirm that ESA-listed fall Chinook salmon were present in the diet, and provided some life history information on the fish based on PIT-tag detections at dams on Snake and Columbia rivers. In total, 30 PIT tags were recovered from cormorant stomachs, 17 of which were confirmed to be Snake River fall Chinook salmon by querying records from PTAGIS. Of the 17 fall Chinook PIT tags recovered, two were positively identified as jack salmon (due to detections of the fish at adult fishways on the mainstem Columbia River) and 10 were confirmed as juveniles (due to detection at juvenile bypass facilities on the lower Snake River just days before the tag was recovered in a cormorant stomach).

Estimates of salmonid consumption (both juveniles and jacks) by cormorants wintering on the lower Snake River ranged from 5,900 kg (95% CI: 3,100 – 8,700 kg), assuming the energy expenditure rate of wintering great cormorants, to 7,500 kg (95% CI: 4,100 – 11,000 kg), assuming the estimated energy expenditure rate of post-breeding double-crested cormorants in the Columbia River estuary.

DISCUSSION

While extensive research has documented cormorant abundance and diet in the Columbia River basin during the breeding season, this study is the first attempt to characterize a local population in the basin during winter. Our study confirmed reports that cormorants over-winter on the lower Snake River and demonstrated that cormorants were present throughout the study area in all years and months of the study. An average of 256 cormorants reach^{-1} month^{-1} was observed throughout the study period, 1.2 cormorants/Rkm.

The distribution of cormorants along the lower Snake River varied among surveys, with cormorants frequently observed in aggregations at dams and other major structures (e.g., log booms, bridges, and channel markers). Other studies have shown that over-wintering cormorant distribution and abundance can be related to various extrinsic factors, such as human activity, potential roost locations, and, particularly, forage fish availability (King et al. 1995, Simmonds et al. 1997). During the early part of the winter period, use of dams could be associated with the abundance of high-energy juvenile shad found throughout the Columbia River basin during the fall (Roby et al. 1998; Petersen et al. 2003). Other studies have concluded that increased prey availability, and potentially injured or stunned prey just below dams, results in an increase in predator abundance, including piscivorous waterbirds such as cormorants (Blackwell et al. 1997, Wiese et al. 2008).

The reported increase of cormorants over-wintering on the lower Snake River does not appear to be the result of growth at breeding colonies on the mid-Columbia River or in the larger Columbia Plateau region. At Foundation Island, just 4 Rkm down-river from the study area, the cormorant breeding colony has remained fairly stable since annual monitoring was initiated in 2002 (250 – 350 breeding pairs; Chapter 1). Also, the larger cormorant breeding colony in North Potholes Reserve near Moses Lake, Washington (ca. 800 pairs), roughly 80 km northeast of the lower Snake River, has declined somewhat in the last few years (Chapter 1). Nonetheless, it seems likely that some portion of cormorants nesting at or fledged from either the Foundation Island or the North Potholes Reserve colony now over-winter along the mid-Columbia and lower Snake rivers. Additionally, observations throughout the study period confirmed that cormorants were present on Foundation Island throughout the non-breeding part of the year.

There are currently no studies underway to confirm where over-wintering cormorants on the Snake River came from and what proportion of these birds were breeders at either coastal or inland colonies. Two recent studies (Clark et al. 2006; Adkins and Roby 2010) have documented that few individuals marked with leg bands or satellite tags in the Columbia River estuary have migrated east of the Cascade Mountains; just 3.1% (5/161) and 1.0% (1/101), respectively, of cormorants in each study were tracked east of the crest of the Cascade Range. These findings suggest that it is unlikely that cormorants over-wintering on the lower Snake River are seasonal migrants from coastal colonies. Further research is necessary to determine where double-crested cormorants over-wintering on the lower Snake River are originating from and whether over-wintering at inland sites east of the Cascade/Sierra Nevada mountain ranges is a widespread and recent phenomenon for the Western North America Population of double-crested cormorants.

The diet composition of double-crested cormorants over-wintering on the lower Snake River was variable over the study period. Throughout the range of the species, diets of cormorants are known to be highly variable both spatially and temporally, reflecting changes in prey distribution, abundance, and availability (Neuman et al. 1997; Wires et al. 2001). In the Columbia River estuary, Collis et al. (2002) found that diet differed significantly between two local breeding cormorant colonies, and varied temporally with proportions of prey types consumed changing through the breeding season. As in similar studies, it is important to interpret our results cautiously, as our sample sizes and collection locations may not be representative of all over-wintering cormorants on the lower Snake River. Cormorants were collected between Lower Monumental Dam and Lower Granite Dam in part due to the identification of relatively large numbers of cormorants roosting under two bridges at night. We could not determine whether these nighttime roosts attracted cormorants from throughout the lower Snake River or from a larger or smaller geographic area. Opportunistic counts during our study suggested that greater numbers of cormorants roosted at these sites than were observed during surveys of the surrounding reservoir. Similar fish prey types are found throughout the study area (Wydoski and Whitney 2003); however, potential differences in relative prey abundance among the four reservoirs on the lower Snake River are unknown.

Diet studies conducted at the double-crested cormorant breeding colony on Foundation Island during 2005-2010 concluded that 22% of the cormorant diet from early April to early July consisted of juvenile salmonids (Chapter 2). While this proportion is higher than that of over-wintering cormorants on the lower Snake River (11.7%), it is evidence that cormorants in this region rely on salmonids as a food source. Similar to results presented here, the proportion of salmonids in the diet of Foundation Island cormorants is also highly variable across the nesting season, and peaks in May when out-migrating salmonid smolts are most abundant in the lower Snake and mid-Columbia rivers. Unlike the diet of Foundation Island cormorants, however, the diet of over-wintering cormorants included jack (adult) salmon and the presence of these larger-sized fish influenced the over-all biomass calculations.

Estimates of salmonid biomass (juveniles and adults combined) consumed by cormorants over-wintering on the lower Snake River (3,100 – 11,000 kg) were approximately one third of estimates for the Foundation Island breeding colony of cormorants (22,000 kg/breeding season; Chapter 2) and approximately one half of estimates for the Crescent Island Caspian tern colony (13,000 kg/breeding season; Chapter 2). Estimates of salmonid biomass consumption by over-wintering cormorants could not be converted into numbers of fish consumed because the mass distribution of fall Chinook, both adults and juveniles, taken by cormorants was unknown.

The impact of predation by cormorants over-wintering on the lower Snake River on the survival of over-wintering juvenile fall Chinook salmon remains unclear. While our study confirms that a portion of the diet of over-wintering cormorants consists of juvenile salmonids, and specifically fall Chinook salmon, the proportion of fall Chinook in the diet is small (3.4%) and this number is influenced by the presence of jack salmon to an unknown degree. Additional research would be needed to quantify predation rates on juvenile fall Chinook by cormorants over-wintering on the lower Snake River and to determine the impact of this level of predation on the population growth rate (lambda) of this ESA-listed species. Future research could seek to characterize the distribution and abundance of fall Chinook salmon and other prey types throughout the lower Snake River to relate changing fish availability to seasonal changes in cormorant abundance, distribution, and diet composition. Additional monitoring may also be warranted to determine whether the numbers of over-wintering double-crested cormorants on the lower Snake River are changing, whether there is inter-annual variation in the distribution of over-wintering cormorants on the lower Snake River (particularly at dams), and, if so, what factors are influencing these changes. Finally, a large proportion of the diet of cormorants over-wintering on the lower Snake River consisted of bass, shad, catfish, and perch, all non-native, introduced taxa that may compete with and/or depredate juvenile fall Chinook salmon that over-winter in the lower Snake River; further complicating the net impact of cormorant predation on this ESA-listed stock.

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LITERATURE CITED

Adkins, J. Y., and D. D. Roby. 2010. A status assessment of the double-crested cormorant (*Phalacrocorax auritus*) in Western North America: 1998-2009. A Report by USGS - Oregon Cooperative Fish and Wildlife Research Unit for U.S. Army Corps of Engineers, Portland, Oregon.

Blackwell, B. F., and W. B. Krohn. 1997. Spring foraging distribution and habitat selection by Double-crested Cormorants on the Penobscot River, Maine USA. Colonial Waterbirds 20:66-76.

Blackwell, B. F., W. B. Krohn, N. R. Dube, and A. J. Godin. 1997. Spring prey use by Double-crested Cormorants on the Penobscot River, Maine, USA. Colonial Waterbirds 20:77-86.

Clark, A. C., T. M. Kollasch, and A. W. Don. 2006. Movements of double-crested cormorants fledged on the Columbia River estuary. Northwestern Naturalist 87:150-152.

Collis, K., D. D. Roby, D. P. Craig, B. A. Ryan, and R. D. Ledgerwood. 2001. Colonial waterbird predation on juvenile salmonids tagged with passive integrated transponders in the Columbia River estuary: Vulnerability of different salmonid species, stocks, and rearing types. Transactions of the American Fisheries Society 130:385-396.

Collis, K., D. D. Roby, D. P. Craig, S. Adamany, J. Y. Adkins, and D. E. Lyons. 2002. Colony size and diet composition of piscivorous waterbirds on the lower Columbia River: implications for losses of juvenile salmonids to avian predation. Transactions of the American Fisheries Society 131:537-550.

Connor, W. P., H. L. Burge, R. Waitt, and T. C. Bjornn. 2002. Juvenile life history of wild fall Chinook salmon in the Snake and Clearwater rivers. North American Journal of Fisheries Management 22:703-712.

Connor, W. P., J. G. Sneva, K. F. Tiffan, R. K. Steinhorst, and D. Ross. 2005. Two alternative juvenile life history types for fall Chinook salmon in the Snake River basin. Transactions of the American Fisheries Society 134:291-304.

Glahn, J. F., D. S. Reinhold, and C. A. Sloan. 2000. Recent population trends of Double-crested Cormorants wintering in the delta region of Mississippi: responses to roost dispersal and removal under a recent depredation order. Waterbirds 23:38-44.

Good, T. P., R. S. Waples, and P. Adams. 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. NOAA Tech. Memo. NMFS-NWFSC-66 *In* U.S. Department of Commerce, Seattle, Washington.

Hansel, H. C., S. D. Duke, P. T. Lofy, and G. A. Gray. 1988. Use of diagnostic bones to identify and estimate original lengths of ingested prey fishes. Transactions of the American Fisheries Society 117:55-62.

Hatch, J. J. 1995. Changing populations of Double-crested Cormorants. Colonial Waterbirds 18:8-24.

Hatch, J. J., and D. V. Weseloh. 1999. Double-crested Cormorant (*Phalacrocorax auritus*), The Birds of North America, Philadelphia, Pennsylvania.

Jones, A. W., C. M. Dalton, E. S. Stowe, and D. M. Post. 2010. Contribution of declining anadromous fishes to the reproductive investment of a common piscivorous seabird, the Double-crested Cormorant (*Phalacrocorax auritus*). Auk 127:696-703.

Keller, T. M., and G. H. Visser. 1999. Daily energy expenditure of great cormorants wintering at Lake Chiemsee, southern Germany. Ardea 87:61-69.

King, D. T., J. F. Glahn, and K. J. Andrews. 1995. Daily activity budgets and movements of winter roosting Double-crested Cormorants determined by biotelemetry in the delta region of Mississippi. Colonial Waterbirds 18:152-157.

Lyons, D. E. 2010. Bioenergetics-based predator-prey relationships between piscivorous birds and juvenile salmonids in the Columbia River estuary. Unpubl. Ph.D. dissertation, Oregon State University, Corvallis, OR.

Neuman, J., D. L. Pearl, P. J. Edwins, R. Black, D. V. Weseloh, M. Pike, and K. Karwowski. 1997. Spatial and temporal variation in the diet of double-crested cormorants (*Phalacrocorax auritus*) breeding on the lower Great Lakes in the early 1990s. Canadian Journal of Fisheries and Aquatic Sciences 54:1569-1584.

Petersen, J. H., R. A. Hinrichsen, D. M. Gadomski, D. H. Feli, and D. W. Rondorf. 2003. Biodiversity, status, and conservation of the world's shads. American Fisheries Society Symposium 35.

Petersen, J. H., D. B. Jepsen, R. D. Nelle, R. S. Shivey, R. A. Tabor, and T. P. Poe. 1991. System-wide significance of predation of juvenile salmonids in Columbia and Snake River reservoirs, Annual Report of Research. Bonneville Power Administration, Portland, Oregon.

Petersen, J. H., M. G. Mesa, J. Hall-Griswold, W. C. Schrader, G. W. Short, and T. P. Poe. 1990. Magnitude and dynamics of predators on juvenile salmonids in Columbia and Snake river reservoirs. Bonneville Power Administration, Portland, Oregon.

PTAGIS 2010. The Columbia Basin PIT Tag Information System. <http://www.ptagis.org> (12/20/2010 2010).

Purcell, M., G. Mackey, E. LaHood, H. Huber, and L. Park. 2004. Molecular methods for the genetic identification of salmonid prey from Pacific harbor seal (*Phoca vitulina richardsi*) scat. Fishery Bulletin 102:213-220.

Roby, D. D., D. P. Craig, K. Collis, and S. L. Adamany. 1998. Avian predation on juvenile salmonids in the lower Columbia River. Report by Oregon State University for U. S. Army Corps of Engineers, Portland, Oregon.

Roby, D. D., D. E. Lyons, D. P. Craig, K. Collis, and G. H. Visser. 2003. Quantifying the effect of predators on endangered species using a bioenergetics approach: Caspian terns and juvenile salmonids in the Columbia River estuary. Canadian Journal of Zoology 81:250-26.

Scherr, H., J. Bowman, and K. F. Abraham. 2010. Migration and winter movements of Double-crested Cormorants breeding in Georgian Bay, Ontario. Waterbirds 33:451-460.

Simmonds, Jr., R. L., A. V. Zale, and D. M. Leslie Jr. 1997. Physical and biological factors affecting seasonal Double-crested Cormorant densities on Oklahoma reservoirs. Colonial Waterbirds 20:31-40.

Seeb, L. W., A. Antonovich, M. A. Banks, T. D. Beacham, M. R. Bellinger, S. M. Blankenship, M. R. Campbell, N. A. Decovich, J. C. Garza, C.M. Guthrie III, T. A. Lundrigan, P. Moran, S. R. Narum, J. J. Stephenson, K. J. Supernault, D. J. Teel, W. D. Templin, J. K. Wenburg, S. F. Young, and C. T. Smith. 2007. Development of a standardized DNA database for Chinook salmon. Fisheries 32:540-552.

Van Doornik, D. M., D. J. Teel, D. R. Kuligowski, C. A. Morgan, and E. Casillas. 2007. Genetic analyses provide insight into the early ocean stock distribution and survival of juvenile coho salmon off the coasts of Washington and Oregon. North American Journal of Fisheries Management 27:220-237.

Wiese, F. K., J. K. Parrish, C. W. Thompson, and C. Maranto. 2008. Ecosystem-based management of predator–prey relationships: piscivorous birds and salmonids. Ecological Applications 18:681-700.

Wires, L. R., F. J. Cuthbert, D. R. Trexel, and A. R. Joshi. 2001. Status of the double-crested cormorant (*Phalacrocorax auritus*) in North America. Final report to USFWS, Fort Snelling, Minnesota.

Wydoski, R. S., and R. R. Whitney. 2003. Inland fishes of Washington. American Fisheries Society and University of Washington Press, Bethesda, Maryland and Seattle, Washington.

Table 5.1. Average monthly diet composition (percent identifiable biomass) of double-crested cormorants over-wintering on the lower Snake River during the winters of 2007-08, 2008-09, and 2009-10. The numbers of cormorants collected each month with prey remains are shown in parentheses.

		Monthly Average % Biomass				
		October	November	December	January	February
	Seasonal Average	(6)	(33)	(42)	(23)	(26)
Centrarchidae (bass and sunfish)	**34.3**	66.7	26.0	9.2	43.8	25.6
Clupeidae (shad)	**15.0**	0.0	16.0	59.1	0.0	0.0
Cyprinidae (minnows and carp)	**11.7**	16.7	12.1	4.8	2.8	22.1
Salmonidae (salmon and steelhead)	**11.7**	16.7	24.2	9.1	8.4	0.0
Unidentified non-salmonids	**7.2**	0.0	6.3	4.5	16.6	8.7
Catostomidae (suckers)	**7.0**	0.0	3.0	4.8	20.1	7.1
Ictaluridae (catfish)	**6.3**	0.0	3.0	2.4	0.0	26.0
Percidae (perch)	**3.4**	0.0	0.0	0.6	6.4	10.2
Other prey	**2.7**	0.0	6.1	5.6	1.9	0.0

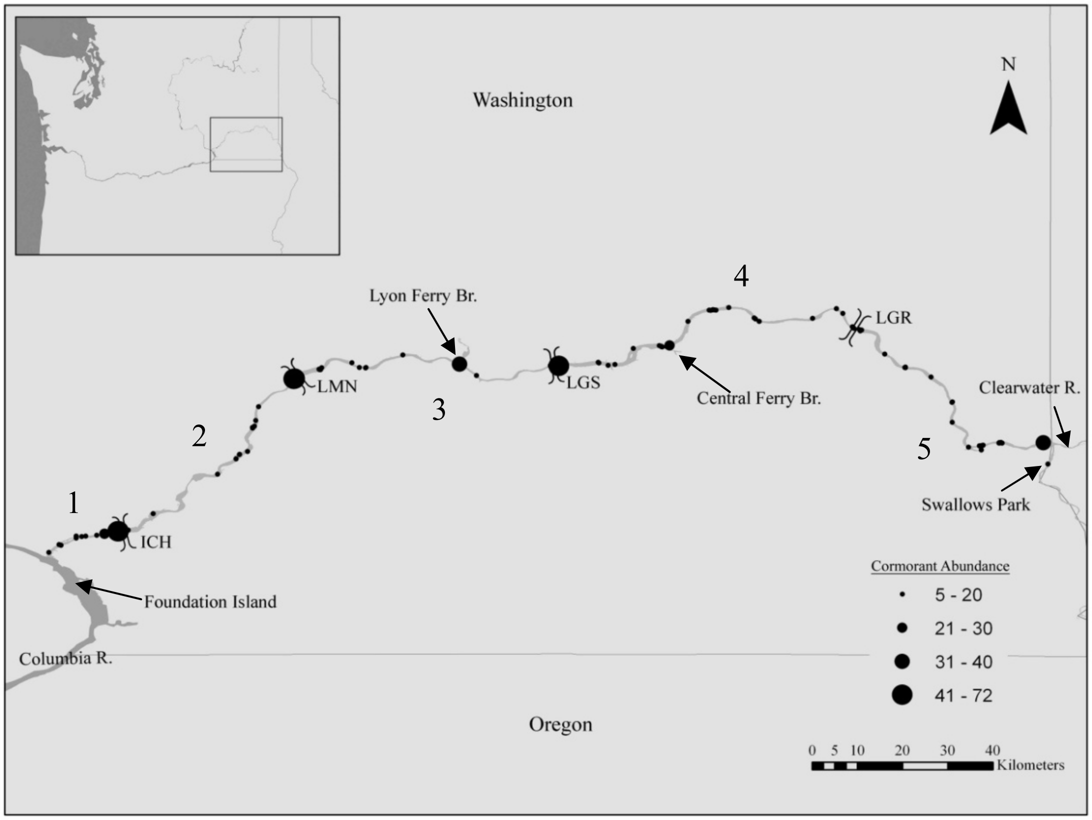

Figure 5.1. Map of the lower Snake River (Rkm 0 - 224) showing the distribution of over-wintering double-crested cormorant flocks (≥ 5 cormorants) observed during the winters of 2008 and 2009. The study area was separated into 5 survey reaches; 1 - Columbia River to Ice Harbor Dam (ICH), 2 – ICH to Lower Monumental Dam (LMN), 3 – LMN to Little Goose Dam (LGS), 4 – LGS to Lower Granite Dam (LGR), and LGR – Clearwater River.

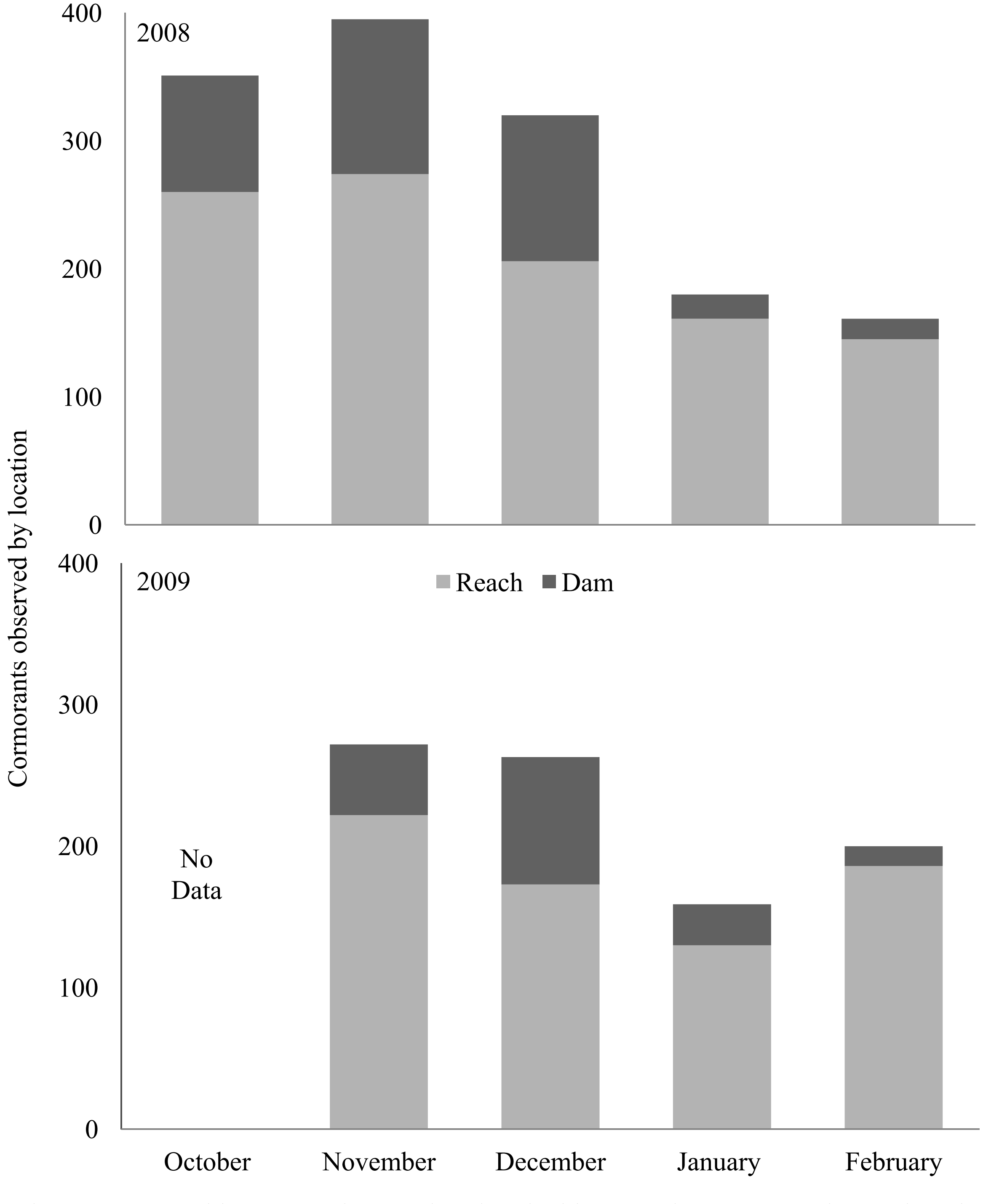

Figure 5.2. Monthly counts of over-wintering double-crested cormorants observed on the lower Snake River during the winters of 2008 and 2009. Bar colors indicate numbers of cormorants counted ≤ 0.25 km from a dam (Black) or > 0.25 km from a dam (Gray).

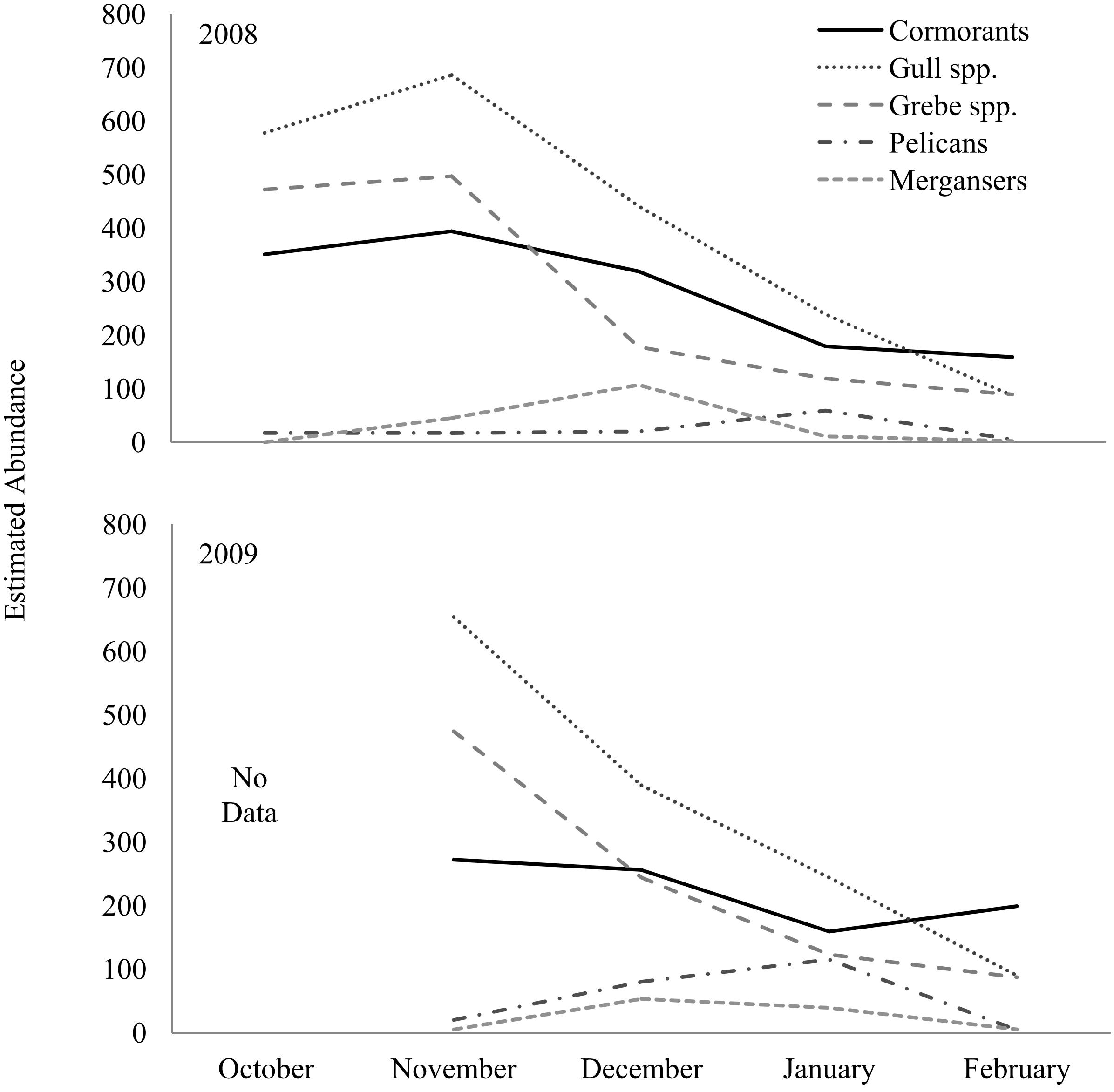

Figure 5.3. Monthly abundance of over-wintering piscivorous waterbirds observed on the lower Snake River during the winters of 2008 and 2009.

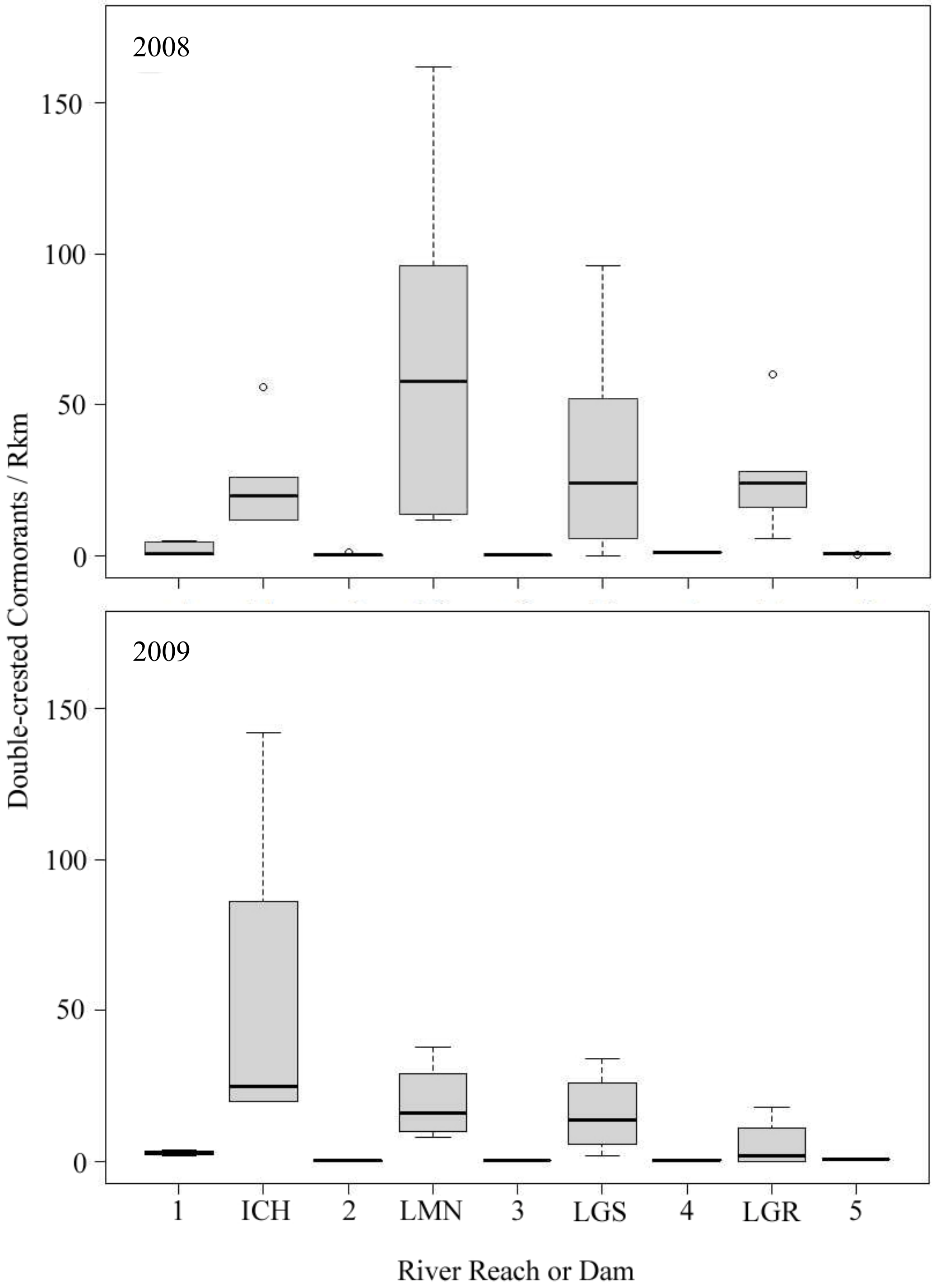

Figure 5.4. Density of double-crested cormorants observed on the lower Snake River during the winters of 2008 and 2009. Locations of cormorants were separated by those counted ≤ 0.25 km from a dam (Dams) or > 0.25 km from a dam (River Reaches).

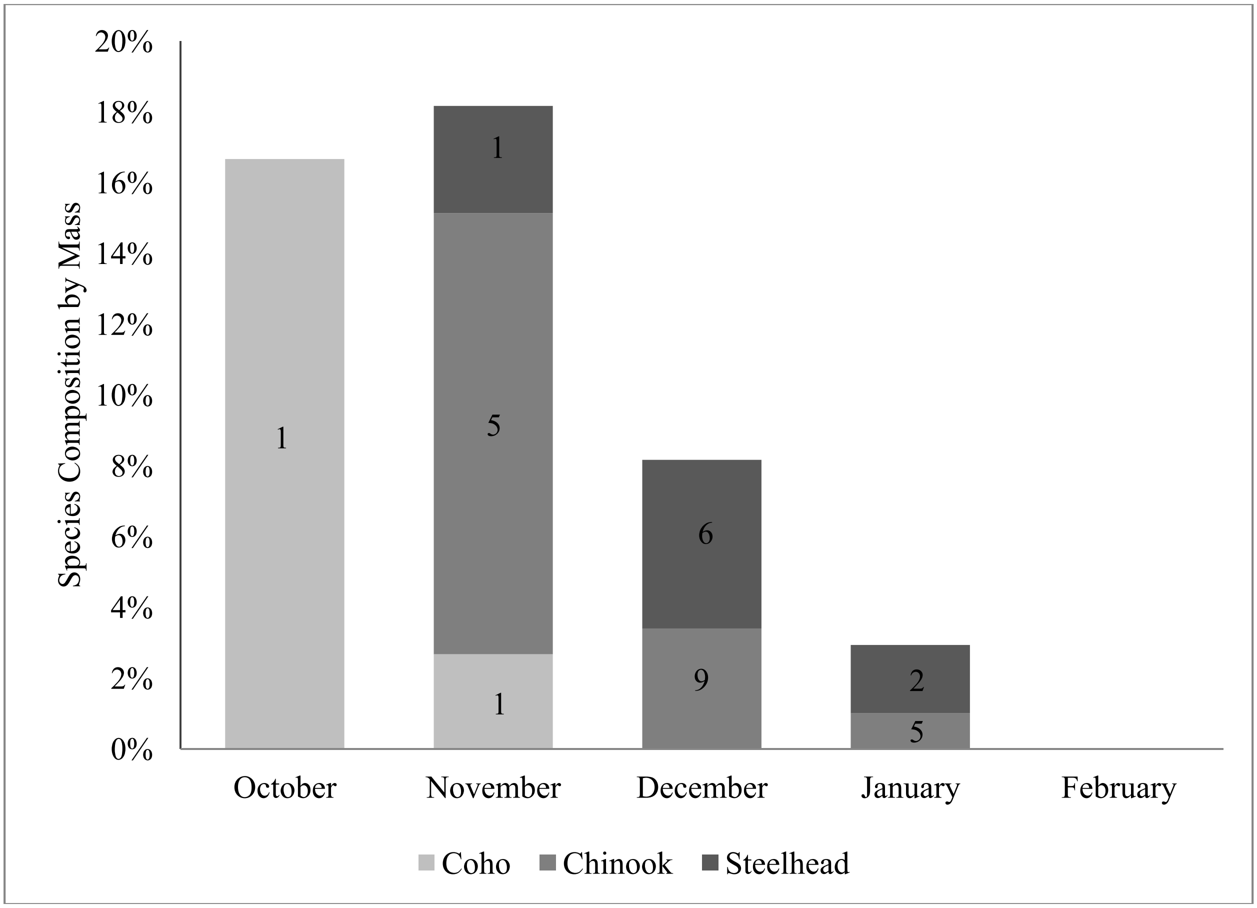

Figure 5.5. Species composition (percent of all identifiable prey biomass) of salmonid species identified in foregut contents of double-crested cormorants over-wintering on the lower Snake River during the winters of 2007, 2008, and 2009. Data labels represent the number of fish identified for each species.

CHAPTER 6

SYNOPSIS AND CONCLUSIONS:

IMPACTS OF AVIAN PREDATION ON SALMONID SMOLTS FROM THE COLUMBIA AND SNAKE RIVERS

This chapter has been prepared for the U.S. Army Corps of Engineers – Walla Walla District for the purpose of assessing project accomplishments. This chapter synthesizes and summarizes the results and conclusions from Chapters 1-5 of this report.

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Our objective was to determine the magnitude of avian predation on juvenile salmonids (*Oncorhynchus* spp.) from the Snake and Columbia rivers, and to identify which piscivorous waterbird colonies in the Columbia Plateau region are responsible for the greatest losses of salmonid smolts. We conducted a 6-year study during 2004-2009, which allowed our research group to realize this objective and to determine trends in piscivorous waterbird populations, as well as trends in their impacts on survival of salmonid smolts. A key finding is that the two species of piscivorous colonial waterbirds that are responsible for most losses of juvenile salmonids in the Columbia River estuary, Caspian terns (*Hydroprogne caspia*) and double-crested cormorants (*Phalacrocorax auritus*), are also responsible for most smolt losses to avian predators in the Columbia Plateau region. This despite the much higher numbers of other species of piscivorous colonial waterbirds (i.e., California and ring-billed gulls [*Larus californicus* and *L. delawarensis*], American white pelicans [*Pelecanus erythrorhynchos*]) nesting in the Columbia Plateau region (see Chapter 1).

Caspian Terns

The total breeding population of Caspian terns in the Columbia Plateau region (ca. 850 pairs) is about an order of magnitude less than the Caspian tern breeding colony in the Columbia River estuary (ca. 9,500 pairs). Furthermore, about half of the Caspian terns nesting in the Columbia Plateau region are using colonies a considerable distance from the Columbia or Snake rivers (see Chapter 1). Consequently, we expected that Caspian terns nesting at inland colonies would have a minor, perhaps even negligible, impact on survival of juvenile salmonids compared to the large colony at East Sand Island in the Columbia River estuary. Over the last 10 years, East Sand Island terns have consumed an average of about 5.4 million juvenile salmonids annually. An average of only about 425 pairs of Caspian terns nested at the Crescent Island colony, the largest tern colony on the mid-Columbia River during the study period, but less than one 20th the size of the East Sand Island colony. Nevertheless, Crescent Island Caspian terns annually consumed an average of about 415,000 juvenile salmonids (see Chapter 2). This unexpectedly high annual smolt consumption is a consequence of the diet composition of Caspian terns nesting at Crescent Island; about two-thirds of the fish they consume are salmonids, whereas only about one-third of the diet of East Sand Island terns consists of salmonids. The Crescent Island Caspian tern colony declined by about 34% during the study period, continuing a trend that started in 2001. While the estimated total annual consumption of juvenile salmonids by Caspian terns nesting at this colony also declined during the study period, the estimated annual consumption of steelhead smolts (*O. mykiss*) did not.

Our estimates of salmon and steelhead smolt consumption by Caspian terns nesting at Crescent Island are derived from demand-based bioenergetics calculations (see Chapter 2). Predation rates on particular ESUs or stocks of salmonids by Crescent Island terns are derived from smolt PIT tags (passive integrated transponders) that are ingested by terns and regurgitated on the tern colony, where we recover them at the end of each nesting season (see Chapter 3). Using PIT tag recoveries on the Crescent Island tern colony to estimate predation rates on in-river migrants belonging to particular salmonid stocks can be subject to several sources of bias, especially due to low detection efficiency of smolt PIT tags deposited on bird colonies. Our group has devised a method to correct or adjust for this potential source of significant bias. Estimates of smolt predation rates based on PIT tag recoveries, adjusted for detection efficiency and expressed on a per capita basis, reveal that the Crescent Island Caspian tern colony has consistently had the highest per capita smolt predation rates of all the piscivorous waterbird colonies studied in the Columbia Basin, including the tern and cormorant colonies in the Columbia River estuary.

Smolt PIT tag recoveries on the Crescent Island tern colony indicate that steelhead smolts from the Snake River ESU are consistently more susceptible to predation by Caspian terns from this colony than other salmonid stocks. Minimum predation rates by Crescent Island terns on Snake River summer steelhead averaged 7.7%; this estimated predation rate increases to about 12% once the proportion of ingested PIT tags that were regurgitated off-colony is taken into consideration. This predation rate is similar to the predation rate on the same salmonid stock by Caspian terns nesting at the colony on East Sand Island in the Columbia River estuary, a colony that is 20 times larger. Susceptibility of steelhead smolts from the threatened Snake River ESU to predation by Crescent Island terns is not only higher than for any other salmonid ESU, it also increases significantly with declining steelhead external condition and decreasing water discharge (see Chapter 4). Thus steelhead smolts in poorer condition and out-migrating during low flows are more likely to be preyed upon by Caspian terns. These results are definitive proof that Caspian tern predation on steelhead smolts is not completely additive; in other words, smolts consumed by terns do not have the same chance of returning as adults as smolts not consumed by terns. It remains unclear, however, what proportion of the smolt losses to Caspian terns are compensatory (fish with little or no chance of surviving to adulthood).

A second Caspian tern colony of roughly the same size as the Crescent Island colony is located on Goose Island in Potholes Reservoir. This colony is situated at least 35 km from Columbia River, and seems an unlikely threat to the survival of juvenile salmonids out-migrating from the upper Columbia River. Smolt PIT tag recoveries on this tern colony, however, indicated that on average terns from the Goose Island colony consumed at least 10% of Upper Columbia summer steelhead smolts, a threatened ESU. The remarkable susceptibility of steelhead from this ESU to Caspian terns nesting at a considerable distance from the Columbia River confirms the ability of Caspian terns to commute long distances and forage efficiently on vulnerable salmonid stocks. Predation rates on other salmonid ESUs by terns nesting at this colony are much lower (see Chapter 3). Our estimates of predation rates on salmonid stocks, particularly steelhead stocks, by Caspian terns nesting on Goose Island in Potholes Reservoir differ considerably from those previously published by Maranto et al. (2010). Maranto et al. (2010) estimated an average predation rate of just 0.6% on Upper Columbia summer steelhead by terns nesting at Potholes Reservoir during 2003-2006. Conversely, we measured a predation rate on Upper Columbia steelhead by terns nesting at Potholes Reservoir ranging from 5.5% to 15.6% per year during 2006-2009. This increase in estimated predation rate is the result of both temporal changes in the Caspian tern colony at Potholes Reservoir and methodological improvements in calculating predation rates based on PIT tag recoveries on-colony.

A third Caspian tern colony, located on the mid-Columbia River, first formed in 2005 in the Blalock Islands of John Day Pool. Although this Caspian tern colony has exceeded 100 breeding pairs in some years since then, it has produced few, if any, young. Despite small size and poor nesting success, the Blalock Islands Caspian tern colony has persisted, and during the period 2007-2009 an average of over 1,200 smolt PIT tags were recovered from the colony. This indicates that the Blalock Islands tern colony, like the Crescent Island tern colony, preys mostly on juvenile salmonids, especially steelhead smolts (see Chapter 3).

Re-sightings of banded Caspian terns on the Crescent Island and Goose Island colonies indicate that there is considerable inter-colony movement of adult terns, especially immigration from the declining Crescent Island tern colony to the expanding Goose Island (Potholes

Reservoir) tern colony (see Chapter 1). Although there are very few re-sightings of banded Caspian terns in the Blalock Islands, it is safe to assume that considerable exchange of breeding adult terns occurs between this colony and others in the region. Colony switching in Caspian terns is a well-known phenomenon, and an important behavioral trait to consider when designing potential management options for the species. If management of Caspian terns to reduce predation rates on both Snake River summer steelhead and Upper Columbia summer steelhead is deemed warranted, then the tern colonies at Crescent Island, Goose Island (Potholes Reservoir), and the Blalock Islands should all be considered a single management unit, rather than separate management units.

In 2008, the U.S. Army Corps of Engineers – Portland District began implementing the management actions outlined in the Final EIS (FEIS) and the Records of Decision (RODs) for Caspian tern management in the Columbia River estuary, a plan to redistribute about two-thirds of the East Sand Island Caspian tern colony (ca. 6,000 breeding pairs) to alternative colony sites in Oregon and California by 2015 (USFWS 2005, 2006). The reduction in the size of the Caspian tern in the Columbia River estuary is expected to result in significant increases in salmonid smolt survival (especially steelhead survival) in the Columbia River estuary. Potential immigration to colonies in the Columbia Plateau region of large numbers of Caspian terns that formerly nested on East Sand Island is a source of concern for salmonid managers. Concern is heightened by the much higher per capita smolt predation rates by Caspian terns nesting at the two largest colonies in the Columbia Plateau region, and the unexpectedly high predation rates on Upper Columbia and Snake River summer steelhead ESUs.

Movement of Caspian terns from the estuary, where management to reduce the size of the East Sand Island tern colony is on-going, to locations in the Columbia Plateau region is plausible given the ephemeral nature of Caspian tern nesting habitats and the inter-colony movements documented for Caspian terns (Cuthbert 1988; Quinn and Sirdevan 1998; Wires et al. 2001). Re-sightings of Caspian terns banded at East Sand Island have documented recruitment to colonies as far from the Columbia River estuary as the Copper River Delta in Alaska (Y. Suzuki, unpubl. data). Antolos et al. (2005) concluded, however, that suitable nesting habitat for Caspian terns on Crescent Island was limited and, consequently, large increases in the size of the Crescent Island Caspian tern colony were unlikely.

Immigration to the Columbia Plateau region of a significant fraction of the Caspian terns that will disperse from the Columbia River estuary is highly unlikely due to the paucity of suitable nesting habitat for terns in the Columbia Plateau region (Antolos 2004). In addition to limits on suitable nesting habitat, the numbers of Caspian terns currently nesting in the Columbia Plateau region are limited by mammalian nest predation, inter-specific competition for limited nesting habitat, food availability, and human disturbance. Data from re-sighting of banded terns support this conclusion, as natal colony philopatry and colony site fidelity are low at Crescent Island compared to East Sand Island, and few terns banded at East Sand Island have immigrated to Crescent Island (this study; S. Collar, unpubl. data). Nevertheless, even if a small percentage of Caspian terns that formerly nested at East Sand Island were to immigrate to colonies in the Columbia Plateau region, this could result in a disproportionately large increase in regional Caspian tern numbers and a significant increase in predation rates on two listed salmonid ESUs: Snake River summer steelhead and Upper Columbia summer steelhead.

Double-crested Cormorants

The only double-crested cormorant breeding colony on the mid-Columbia River that persisted throughout the study period and was of sufficient size to warrant concern over potential impacts to salmonid smolt survival was on Foundation Island, just below the confluence of the Snake and Columbia rivers. This colony averaged about 330 breeding pairs during the study period, and there was no apparent trend in colony size during this period. By comparison, the double-crested cormorant colony on East Sand Island in the Columbia River estuary averaged ca. 12,500 breeding pairs, about 38 times larger than the Foundation Island cormorant colony.

By far the largest breeding colony of double-crested cormorants in the Columbia Plateau region is located at the north end of Potholes Reservoir, where over 1,100 breeding pairs nested as recently as 2006. The North Potholes cormorant colony has declined significantly since 2006, and with it the overall number of cormorants nesting in the region. Efforts to recover smolt PIT tags from the North Potholes cormorant colony have revealed very small numbers (< 50 per year), indicating that this cormorant colony currently poses no significant threat to survival of juvenile salmonids from upper Columbia River stocks. This conclusion is in stark contrast to our conclusion regarding the current impact of predation from Caspian terns nesting at Potholes Reservoir.

We examined the possibility that the Foundation Island colony of double-crested cormorants in the mid-Columbia River is a significant mortality factor for juvenile salmonids using bioenergetics methods to estimate prey consumption. Estimated annual consumption of salmonids by Foundation Island cormorants ranged from 470,000 to 880,000 smolts (see Chapter 2). Despite a somewhat smaller colony and less specialization on salmonids, cormorants consumed more salmonid biomass than Caspian terns nesting at the nearby Crescent Island colony, due primarily to the larger body size of cormorants and consequent greater individual energy requirements. There was no apparent trend in salmonid smolt consumption by Foundation Island cormorants during the study period.

PIT tag recoveries on the Foundation Island cormorant colony indicated that overall predation rates on PIT-tagged salmonid stocks were much lower compared to the colony on East Sand Island in the Columbia River estuary. Once predation rates on PIT-tagged smolts were adjusted for colony size (i.e., smolt consumption per bird), however, per capita predation rates were substantially higher for cormorants nesting at the Foundation Island colony compared to the colony in the Columbia River estuary (see Chapter 3). While the Foundation Island cormorant colony is much smaller than its counterpart in the estuary, Foundation Island cormorants are more reliant on salmonids as a food source. This greater reliance on salmonids, coupled with lower diversity of available salmonid stocks compared to the estuary, is responsible for the unexpectedly high impact of the small Foundation Island cormorant colony on specific stocks of salmonids, particularly Snake River summer steelhead and Snake River sockeye. Predation rates on Snake River summer steelhead (> 2.0%) and Snake River sockeye (> 1.7%) by Foundation Island cormorants were high compared to those of other bird colonies in the Columbia Plateau region. PIT tag recoveries on the Foundation Island cormorant colony also indicated that steelhead susceptibility to cormorant predation increased significantly with declining steelhead external condition, plus steelhead of hatchery origin were more susceptible to predation from Foundation Island cormorants compared to their wild counterparts (see Chapter 4). These results indicated that steelhead susceptibility to cormorant predation is condition-dependent and is influenced by rearing environment (hatchery vs. wild). As with results from PIT tags recovered on the Crescent Island Caspian tern colony, these findings unequivocally demonstrate that at least a portion of the smolt mortality caused by cormorant predation in the mid-Columbia River is compensatory.

Several management agencies have proposed reducing the amount of nesting habitat for double-crested cormorants on East Sand Island as a means to reduce the impact of cormorant predation on survival of juvenile salmonids in the Columbia River estuary. Such potential management has raised concerns that large numbers of double-crested cormorants might immigrate to cormorant colonies in the Columbia Plateau region, resulting in higher predation rates on Snake River and upper and mid-Columbia salmonid stocks. Reductions in the size of the cormorant colony on East Sand Island would be unlikely to result in a large and sudden influx of nesting double-crested cormorants in the Columbia Plateau region, based on current demographic connectivity. Banding and satellite tracking studies have suggested that movement of double-crested cormorants between the large colony in the Columbia River estuary and those in the Columbia Plateau region is limited (see Chapter 1). Nevertheless, even if a small percentage of double-crested cormorants that currently nest at East Sand Island were to immigrate to the Foundation Island colony, could result in a disproportionately large increase in colony size and a significant increase in predation rates on salmonid smolts, in particular Snake River summer steelhead and Snake River sockeye ESUs.

We also assessed the abundance, distribution, and diet of double-crested cormorants over-wintering on the lower Snake River in eastern Washington to investigate the potential for significant impacts from cormorant predation on survival of ESA-listed fall Chinook salmon that over-wintering in the lower Snake River (see Chapter 5). We conducted monthly boat-based surveys of the lower 224 river kilometers (Rkm) of the Snake River during the winters of 2008-09 and 2009-10 and found cormorants throughout the study area. A monthly average of 256 cormorants was observed on this reach of the lower Snake River, or an average of about 1.2 cormorants per Rkm. Roughly 22% of the cormorants recorded were within 0.25 Rkm of the four federal hydroelectric dams in the study area. During the winters of 2007-08, 2008-09, and 2009-10 we lethally collected a total of 130 cormorants with identifiable prey remains in their foreguts to determine whether over-wintering cormorants were consuming over-wintering juvenile fall Chinook salmon. Overall diet composition of cormorants was highly variable and changed as winter progressed. The most prevalent prey types in the foregut samples were centrarchids (sunfishes and bass; 34.3% by mass), followed by shad (15.0%), cyprinids (11.7%), and salmonids (11.7%). Fall Chinook salmon comprised an average of 3.4% by mass of the cormorant diet. Although juvenile fall Chinook salmon were a minor component of the cormorant diet relative to other fishes, biomass consumption of all salmonids by overwintering cormorants was estimated at 3,100 to 11,000 kg, or about one third of the estimated salmonid biomass consumption by cormorants nesting at Foundation Island (see Chapter 5). The bulk of the diet of over-wintering cormorants, however, consisted of non-native fishes that compete with or depredate juvenile salmonids. Due to limited information on the abundance and distribution of fall Chinook in the lower Snake River, it is unknown what proportion of over-wintering fall Chinook are annually consumed by cormorants. This information is needed to more fully evaluate the potential impacts of over-wintering cormorants on survival of this ESA-listed salmon stock.

Gulls and Pelicans

Numbers of breeding American white pelicans at the Badger Island colony on the mid-Columbia River, the sole breeding colony for the species in the State of Washington, increased substantially during the study period; numbers of pelicans counted on-colony increased from about 900 to 1,800 individuals. Overall numbers of breeding gulls, the most numerous piscivorous colonial waterbirds in the region, declined during the study period, mostly because of the abandonment of a large gull colony on Island 18 in the mid-Columbia River and the decline of a large gull colony on Threemile Canyon Island (see Chapter 1).

Predation rates on smolts by gulls nesting at colonies on Miller Rocks and Crescent Island, and by pelicans nesting at the colony on Badger Island, were minor (generally < 0.5% of available smolts) compared to smolt losses from Caspian terns nesting at Crescent Island or Goose Island (Potholes Reservoir), or double-crested cormorants nesting at Foundation Island. Of the gull and pelican colonies examined, smolt predation rates were highest for the gull colony on Miller Rocks (see Chapter 3).

Conclusions

We have used recoveries of smolt PIT tags from nine colonies of piscivorous colonial waterbirds in the Columbia River basin to estimate predation rates on specific stocks of salmonids from the Snake River and the upper and mid-Columbia River. We have also provided bioenergetics-derived estimates of the numbers of juvenile salmonids consumed by avian predators nesting at the two colonies in the Columbia Plateau region where the highest numbers of smolt PIT tags have been detected. Based on these results, the identified priorities for management consideration in the Columbia Plateau region are Caspian tern and double-crested cormorant colonies. Gulls and white pelicans nesting at colonies in the Columbia Plateau region are a relatively minor source of salmonid smolt mortality, and predation on juvenile salmonids by these species is best managed using deterrence methods focused at sites where smolts are especially vulnerable to these avian predators (e.g., dams).

While our results demonstrate that Caspian terns and double-crested cormorants nesting on the mid-Columbia River consume large numbers of juvenile salmonids, very few juvenile lamprey were consumed. Taken together, we estimated that the birds at the Crescent Island tern colony and the Foundation Island cormorant colony consume less than 10,000 lamprey macropthalmia per year. A greater diet sampling frequency would be necessary to more precisely estimate lamprey consumption, as it appears that use of lamprey as a prey resource is limited to occasional episodes during the birds' breeding season. Our diet sampling frequency for cormorants was too low to detect any cormorant consumption of adult lamprey, although it may occur.

Based on the results of this study, the greatest potential for increasing survival of smolts from ESA-listed salmonid stocks by managing piscivorous colonial waterbirds in the Columbia Plateau region would be realized by focusing management efforts on Caspian terns nesting at colonies on Crescent Island, Goose Island, and the Blalock Islands. Reductions in the size of these tern colonies would enhance survival of Upper Columbia and Snake River summer steelhead stocks in particular. More limited enhancement of smolt survival for Snake River steelhead and Snake River sockeye could be achieved by managing the double-crested cormorant colony at Foundation Island. Based on smolt predation rates as inferred from PIT tags recovered on-colony, management of other piscivorous waterbird colonies in the Columbia Plateau region would provide relatively small and perhaps undetectable increases in stock-specific smolt survival.

Further work is necessary, however, to translate estimates of smolt consumption and smolt predation rates into assessments of the potential benefits for threatened and endangered salmonid populations of reducing avian predation in the Columbia Plateau region. The analysis of potential benefits from management of piscivorous waterbirds for restoring ESA-listed stocks of salmonids is key to informed decision-making, as resource managers consider whether management of specific waterbird colonies in the Columbia Plateau region is warranted and, if so, what the management objectives should be.