Measuring estuary avian predation on juvenile salmon by electronic recovery of passive integrated transponder tags from nesting colonies, 2013

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Executive Summary

Avian predation is one factor limiting the recovery of threatened and endangered populations of Pacific salmon *Oncorhynchus* spp. in the Columbia River Basin. To measure, monitor, and manage the effects of avian predation, estimated predation rates are needed for individual Evolutionarily Significant Units (ESUs) and Distinct Population Segments (DPSs) of Pacific salmon. One method to estimate predation rates is to retrieve codes from passive integrated transponder (PIT) tags deposited on avian nesting colonies after birds have consumed a tagged juvenile fish. Proportions of fish consumed (number consumed/number available) can then be estimated for all PIT-tagged salmonids in that group. If the tagged group is representative of a specific ESU or DPS, avian predation impacts can then be extrapolated to non-tagged fish in that ESU or DPS.

This report presents results from recovery of salmonid PIT tag codes after the breeding season on three seabird colonies on East Sand Island in the Columbia River Estuary. Tag-code recoveries were used in collaboration with Bird Research Northwest to derive estimates of estuary predation on juvenile salmonids by Caspian terns *Hydroprogne caspia*, double-crested cormorants *Phalacrocorax auritus,* and Brandt's cormorants *P. penicillatus*.

We present results from the following two primary study components:

1. Recovery of PIT-tag codes from nesting colonies on East Sand Island
2. Estimation of ESU/DPS-specific estuary predation rates, including
    - i. Adjustments for tag-code detection efficiency and on-colony deposition rates where available
    - ii. Estimates of estuary predation rate for 10 ESU/DPS groups originating above Bonneville Dam (Columbia River) or above Sullivan Dam (Willamette River)
    - iii. Reporting of tag releases and recoveries of PIT tag codes from Lower Columbia River Chinook salmon found on East Sand Island
    - iv. Minimum estimates of estuary predation rate for barge-transported and in-river migrant Snake River salmon and steelhead originating at or above Lower Granite Dam

On the East Sand Island Caspian tern colony, we recovered 11,860 unique PIT tag codes from juvenile fish that migrated downstream in 2013. Tag codes recovered included fish from 10 of the 13 Pacific salmon groups tagged in 2013 that were listed as threatened or endangered under the U.S. Endangered Species Act. On the double-crested cormorant colony, we recovered 11,020 unique tag codes, also representing all 10 listed groups.

Daily detection efficiencies of PIT-tag codes on East Sand Island varied through the nesting season, ranging from 41 to 72% on the Caspian tern colony and 45 to 72% on the double-crested cormorant colony. These efficiency ranges were comparable to those measured in prior years. However, we did detect a downward trend in detection efficiencies on the tern colony beginning in 2009. This suggests PIT tags are accumulating in surface layers and causing tag-code collisions, which reduce efficiency when scanning for tags.

Bird Research Northwest used our tag-code recovery data to estimate on-colony tag deposition rates by double-crested cormorants. They estimated that 60% (95% CI = 47-73%) of tags consumed by double-crested cormorants were deposited on the colony, implying that up to 40% of the tags consumed by double-crested cormorants were deposited off-colony. Detection efficiency and on-colony deposition adjustments were used to estimate 2013 predation rates for PIT-tagged groups of salmonids originating above Bonneville Dam and Sullivan Dam.

We estimated estuary predation rates for in-river-migrant PIT-tagged fish from 10 ESUs/DPSs with geographical origins above Bonneville Dam (Columbia River) or Sullivan Dam (Willamette River). Deposition-adjusted estimates showed Caspian terns had the greatest impact on steelhead DPSs (8.6-12.5%, depending on DPS), with a lesser impact on salmon ESUs (0.6-1.4%, depending on ESU). By ESU/DPS, double-crested cormorants had the greatest impact on Snake River spring/summer Chinook salmon (2.9%), with impacts ranging 0.7-2.6% on other groups. Double-crested cormorant predation on salmonids declined compared to 2012, even though the number of breeding pairs increased and reproductive success was higher on East Sand Island in 2013. In general, Upper Willamette River spring Chinook salmon experienced the least avian predation impact (< 1.0% for both Caspian terns and double-crested cormorants), and Brandt's cormorants appeared to have minimal impacts (<0.3%) on all groups evaluated in 2013.

Fish included in the Lower Columbia River Chinook salmon ESU exhibit complex life history types, and while some components of this ESU are PIT-tagged, there is no comprehensive, representative PIT-tagging program for the ESU as a whole. Thirteen different sources contributed a total of 97,255 PIT-tagged juveniles from the Lower Columbia River Chinook salmon ESU during migration year 2013. However, just three hatcheries above Bonneville Dam accounted for 72% of these fish. Due to this extremely skewed representation of the ESU in the PIT-tagged groups, we elected not to estimate predation rates on this ESU. Many groups from this ESU, including wild fish above and below Bonneville Dam, were clearly not represented by tagging in 2013.

However, we did evaluate tag recoveries from avian colonies on East Sand Island for Lower Columbia River Chinook salmon, and those recoveries indicated the following patterns: (1) the majority of tag codes (83.6%) were detected on the double-crested cormorant colony, (2) the majority of tagged fish detected on a colony were released from a hatchery on or before 1 May (75.3%), and (3) 78.7% of all tag codes recovered on East Sand Island originated from releases that occurred on or before 2 May 2013. These results suggest that existing tagged hatchery releases from this ESU are particularly susceptible to double-crested cormorant predation.

During 2013, 123,595 PIT-tagged sockeye, steelhead, and Chinook salmon from the Snake River were transported through the hydropower system by barge and released directly below Bonneville Dam; of these, 4,035 were recovered on East Sand Island bird colonies. At Bonneville Dam, 26,469 in-river migrants from the Snake River were detected; of these, 719 were subsequently recovered on East Sand Island bird colonies. We compared avian predation impacts between barge-transported fish vs. in-river migrants in two ways. First, we estimated minimum annual predation using all available data from 2013. Second, we compared estimated minimum weekly predation between barge release groups and in-river migrants detected at Bonneville Dam for weeks during which at least 100 tagged fish were present in both groups.

When examining patterns in estimates of minimum annual predation rates for transported vs. in-river migrant Snake River fish, qualitative comparisons suggest barged fish experience lower tern predation but higher cormorant predation. Caspian terns had the largest impacts on steelhead (4.7-9.0%), as did double-crested cormorants (1.0-2.3%). All groups experienced low predation from Brandt's cormorants ($\leq$ 0.2%). When comparing estimated mean minimum weekly predation rates of barged fish and in-river migrants, unpaired *t*-tests did not show statistically significant differences in 12 possible comparisons. Results imply that barging in 2013 was not linked to large changes in susceptibility of Snake River salmon or steelhead to estuary avian predation. However, sample sizes for statistical comparison were small ($n \leq 8$), and therefore the power of the tests to detect anything but large differences between mean minimum predation rates was relatively weak. In addition, test results would be confounded if there is any as-yet-unidentified within-season temporal variation in the vulnerability of barged fish or in-river migrants.

To improve understanding of estuary avian predation on Columbia River salmon, including the potential effects of barge transport on Snake River groups, we recommend that future work include support to determine the mechanisms driving variation in seasonal and annual predation rates. Specifically, we recommend

1. Measurement of temporal and spatial abundance of dominant non-salmonid prey species in the estuary, to test whether or not annual and weekly/seasonal changes in predation rates are correlated with changes in the availability of non-salmonid prey species, and
2. A synthesis of multiple years of data for barged fish vs. in-river migrant comparisons, to improve the statistical power necessary to resolve what, if any, biologically meaningful differences exist in estimated minimum predation rates

With specific regard to basin-wide PIT tag programs, improvement in our ability to measure and resolve avian predation estimates would benefit from

1. A comprehensive tagging and interrogation program for Lower Columbia River salmonids, to more accurately characterize overall estuary predation for this ESU as a whole
2. Improvements to PIT-tag detection capability at Bonneville Dam, or as close to estuary entry as possible, so as to provide sufficient detection numbers to measure estuary entry timing of in-river migrant groups.

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Introduction

In 1987, research biologists began tagging juvenile Pacific salmon *Onchorhynchus* spp. with passive integrated transponder (PIT) tags to measure survival through the Federal Columbia River Power System (Prentice et al. 1990; Marvin 2012). During the ensuing years, all Columbia River dams and fishways managed by the U.S. Army Corps of Engineers were instrumented with PIT-tag detection systems. These systems allow scientists and mangers to track survival of juvenile salmon and steelhead as they migrate seaward through dams within the basin, as well as allowing them to track adult fish passing these dams during the spawning migration.

Annual releases of PIT-tagged juvenile salmon have grown from an initial number less than 50,000 to over 2 million (Marvin 2012). When fish are tagged, data specifying the species, origin, release site, and release date for each tagged individual are recorded in a basin-wide regional database known as the Columbia Basin PIT Tag Information System (PTAGIS 2013). Subsequent detections or recoveries of individual PIT tags are also recorded in this database.

Survival estimates derived from PIT-tag data have been used to identify times, places, and agents of salmon mortality. These data in turn inform development of action plans to identify the factors limiting population recovery of threatened or endangered salmon and to implement recovery actions for ESA-listed populations (NMFS 2008, 2010).

One potential limiting factor affecting salmon recovery was identified in 1998, when fisheries biologists discovered thousands of PIT tags from juvenile salmon being deposited on seabird colonies (Collis et al. 2001). These included colonies of Caspian terns *Hydroprogne caspia* and double-crested cormorants *Phalacrocorax auritus* on Rice Island (rkm 34) in the lower Columbia River. In 1998, NMFS developed land-based PIT-tag detectors to allow large-scale recovery of PIT-tag codes from avian colonies (Ryan et al. 2001). Results from this work demonstrated that birds were consuming millions of salmon annually (Collis et al. 2001; Ryan et al. 2003).

Resource managers responded to this finding by initiating a program to relocate the birds from Rice Island (rkm 34), where the primary prey fish available were juvenile salmon, to East Sand Island (rkm 8), where alternative prey, such as marine forage fish, were known to be available (Bottom and Jones 1990). The expectation was that moving bird colonies closer to a non-salmonid food source would reduce avian predation impacts on salmon. Subsequent data from PIT-tag recovery on bird colonies showed that estuary avian predation impacts were reduced by moving the nesting areas to East Sand Island.

However, basin-wide assessments continued to indicate that estuary avian predation is a factor limiting recovery of ESA-listed species of Pacific salmon (Good et al. 2005; Evans et al. 2012; Sebring et al. 2013). Resource managers support recovery of PIT-tag codes from estuary bird colonies in compliance with mandates of the National Marine Fisheries Service Biological Opinion (NMFS 2008), supplemental Biological Opinion (NMFS 2010), and the Adaptive Management Implementation Plan (NMFS 2009). Management agencies also support the data processing and analysis necessary to estimate avian predation rates on Columbia River Basin salmonids. Specifically, estimation of Caspian tern and double-crested cormorant predation rates and ongoing monitoring of Caspian tern population impacts on juvenile salmon in the estuary (NMFS 2009, reasonable & prudent alternative actions 45-46 and 65).

Finally, there is ongoing management interest in the potential survival/mortality effects of programs to transport Snake River juvenile salmonids past the hydropower system and release them below the last mainstem dam (Bonneville Dam, rkm 235) or directly into the estuary (Ryan et al. 2006; Marsh et al. 2008, 2010; Anderson et al. 2013). While transportation avoids several sources of direct juvenile mortality occurring in the hydropower system, barged fish do become vulnerable to avian predation after they are released and during the remainder of their juvenile migration to sea through the Columbia River plume.

The tasks necessary to address freshwater and estuary avian predation issues have historically been shared among NMFS biologists and Bird Research Northwest (BRNW, formerly Columbia Bird Research). Each research group reports on tasks partitioned to them under annual research contracts with the U.S. Army Corps of Engineers (USACE) or the Bonneville Power Administration (BPA). During 2013, we assumed all responsibility for PIT-tag code recoveries on East Sand Island bird colonies and BRNW assumed responsibility for recoveries on all other colonies in the Columbia River Basin, in addition to their avian ecology research during the nesting season. In this report we summarize results from our PIT-tag recovery efforts on East Sand Island. Select summary results from BRNW tasks are also presented here. A more detailed description of BRNW methods and results can be found in Roby et al. (2014).

This report presents estimated predation rates by ESU or DPS to meet management needs and to provide comparability with the 2013 BRNW companion report. Prior to 2012, predation estimates were reported by species/run/rear types, not by ESU/DPS group (Sebring et al. 2009, 2010a,b, 2012).

Study Area

Piscivorous waterbirds have established numerous nesting colonies throughout the Columbia River Basin (Figure 1). Most of these colonies are active (e.g. East Sand Island, Crescent Island), although some are no longer in use (e.g. Rice Island). Nearly all individual colonies have been studied at one time or another by three collaborative groups: Bird Research Northwest, a research partnership between Oregon State University, the U.S. Geological Service, and Real Time Research; the National Marine Fisheries Service, and the University of Washington. Of all avian nesting colonies, the largest are those presently located in the lower Columbia River estuary on East Sand Island, OR. We recovered PIT-tag codes from this location in 2013.

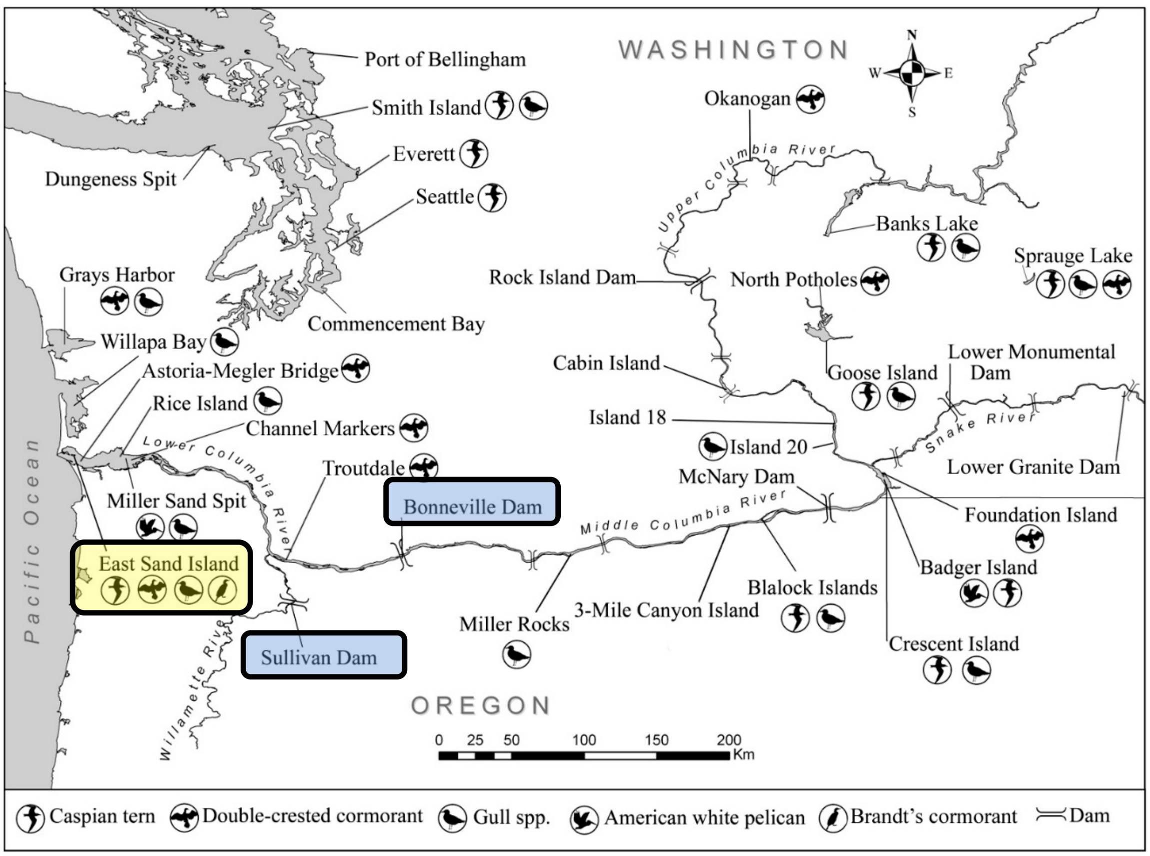


Figure 1. Map of Columbia River Basin and coastal Washington showing the location of active and former breeding colonies of piscivorous waterbirds. The study site for this report is East Sand Island (indicated in yellow) at the mouth of the Columbia River. Dams nearest to East Sand Island were Bonneville and Sullivan (indicated in blue). Map provided courtesy of BRNW.

During 2013, all of our PIT-tag code recovery efforts took place on East Sand Island, Oregon (rkm 8). The area of this island is 2.02 by 10^6 m^{-2} (~ 50 acres), and its terrain consists primarily of coarse sand with some topsoil and vegetation. The west end of the island is armored with a stone jetty, and the south end is characterized by a shoreline of stone and rip-rap. Access to this federally owned island required permission from the U.S. Army Corps of Engineers.

To transport crew and gear onto the island, we used a 28-foot research vessel and an inflatable landing skiff. These vessels were moored at the Hammond Mooring Basin in Hammond, OR, approximately 7.5 km SE of East Sand Island. When working on the island, the crew transport vessel was secured to one of three NOAA seasonal moorings placed at the northeast tip (46°16.026'N 123°57.951'W, primary tern colony access), southeast corner (46°15.560' N 123°58.292' W, secondary tern colony access), and northwest corner of the island (46°15.957' N 123°59.252' W, cormorant colony access).

The Caspian tern nesting colony is located on the eastern end of East Sand Island and during 2013 included 6,394 m^{-2} (1.6 acres) of vegetation-free bare sand (Figure 2). This area represents a reduction in colony size of nearly 50% from the 12,545 m^{-2} (3.1 acres) available in 2010 (Figure 3). Prior to initiation of nesting activity, the area is actively managed by the U.S. Army Corps of Engineers to control colony size, maintain bare sand habitat, and eliminate vegetation. The middle section of the island contains dense vegetation with small trees, shrubs, and grass. No waterbirds nest in this section of the island.

Figure 2. Bare sand habitat on the East Sand Island Caspian tern colony. This habitat is actively groomed to maintain a relatively flat surface without vegetation. Black fencing marks the edge of the colony. This picture was taken facing northwest, towards the town of Chinook, WA.

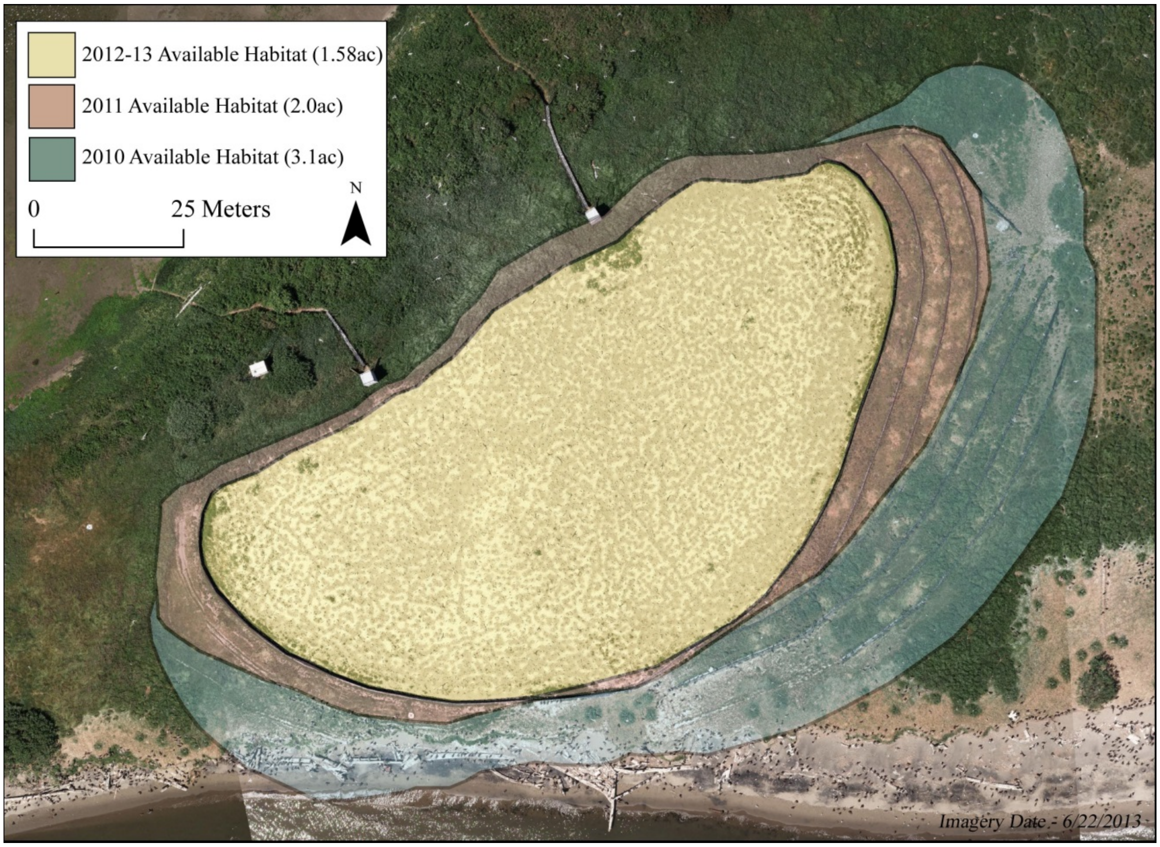


Figure 3. Aerial photo of Caspian tern colony size and shape during 2013. The colored shading in this figure shows the sequential 50% reduction of colony size from 2010 to 2012. Figure provided courtesy of Bird Research Northwest.

Double-crested and Brandt's cormorants nest on the west end of East Sand Island in bare sand (Figure 4) or on stone rip-rap (Figure 5). Originally, cormorants had access to 15,782 m^{-2} (3.9 acres) of nesting habitat, but in 2013 a combination of fencing and hazing was used to restrict active nesting to an area of 12,545 m^{-2} (3.1 acres; Figure 6). In this report, we refer to the area of active nesting habitat as the cormorant colony and the area where cormorants were discouraged from nesting as the dissuasion area.

Figure 4. Bare sand habitat on the East Sand Island cormorant colony. Unlike habitat on the tern colony, this bare sand is not actively managed to maintain a flat surface without vegetation or rocks. Upright sticks marked with arrows indicate the edges of an area surveyed with a hand-held PIT-tag detector.

Figure 5. Rip-rap habitat on the East Sand Island cormorant colony. This habitat is found primarily on the southwestern portion of the island, where the shoreline is armored with stone and where driftwood and other flotsam accumulate in significant quantities.

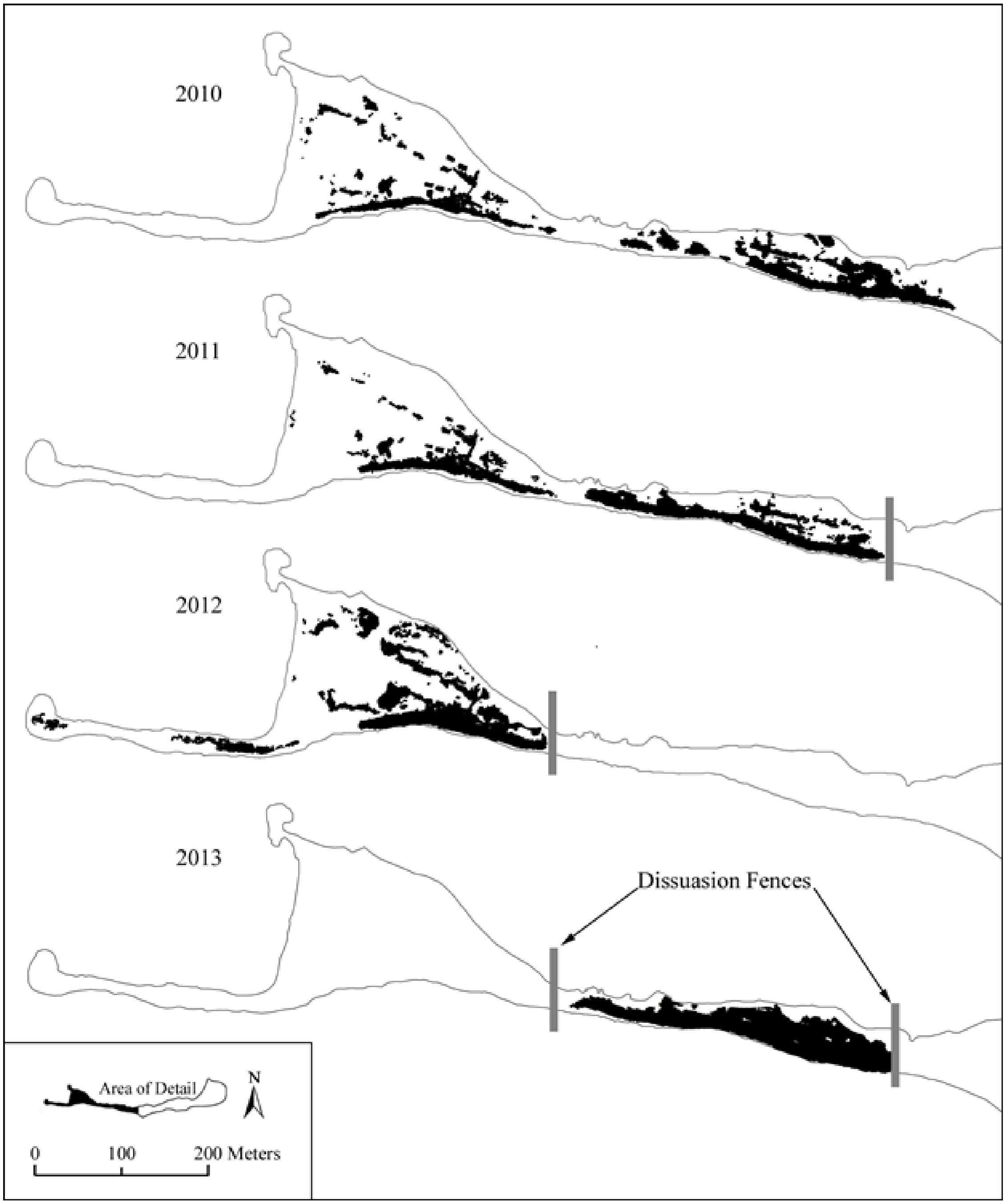


Figure 6. Distributions of the double-crested and Brandt's cormorants nest areas on the west end of East Sand Island. A sequential reduction in colony area was achieved through the use of dissuasion fencing and hazing during the breeding season. Maps provided courtesy of Bird Research Northwest.

Recovery of Tags from East Sand Island

All detection efforts on East Sand Island took place between 31 October and 13 December 2013. A two-person crew surveyed the tern colony, and depending on personnel availability, a two-, four-, or five-person crew surveyed the cormorant colony. After gear transport, set-up, and testing, one complete survey of the tern colony required 2 d, and one complete survey of the cormorant colony required 2-3.5 d. Each complete survey of a colony was referred to as a "pass." Three passes were completed on the tern colony and two on the cormorant colony. We did not scan for tags in the dissuasion area.

Because no genetic material was available, we designated fish as belonging to an ESU or DPS based on species, run type, and watershed of origin (NMFS 2014) and assigned all recovered PIT tag codes to an ESU or DPS. Appropriate combinations of 12-digit hydrological unit code (HUC; PSMFC 2009), species, and run type as reported in the PTAGIS database were assigned to each ESU or DPS (Table 1). To minimize assignment errors due to fish with biological origins in a watershed other than the one into which they were released, we compared tag site information to release site information. Tag- vs. release-site mismatches, and any other tag groups with known origins outside of the ESU/DPS, were excluded from estimates.

Rain events have the potential to wash away PIT tags deposited on the colony, and it is important to note that rainfall in 2013 was significantly different than climatological averages for the lower Columbia River Estuary. April, May, and September of 2013 were unusually wet, while June, July, August, and October were unusually dry. A total of 102 cm (40.1 in) of rainfall accumulated between April and October, compared to the climatological average of 50 cm (19.6 in) for that period (NCDC 2014). There were also record-setting rain events during September 2013 (monthly total of 37.9 cm, or 14.9 in). Much of that rain fell during 27-28 September 2013, when the remains of typhoon Pabuk hit the lower Columbia River estuary and over 12.7 cm (5 in) of rain fell in 24 h. This occurred one month before our crews had access to East Sand Island. Thus, rain and storm surges may have washed away an unknown number of tags from exposed rip-rap habitat before cormorant colony surveys began.

Several non-weather events affected field data collection during 2013 and are also worth noting. First, both terns and cormorants remained on the nesting colonies until the first week of October (Roby et al. 2014). As a consequence, PIT tag surveys could not begin until 10 October. Second, due to a shutdown of the U.S. Federal Government, all project employees were on furlough and operations suspended for 16 d from 1 through 16 October. Finally, between 5 and 22 November, there was a temporary stand-down of all NOAA activities associated with tag-code recovery on the cormorant colony, and this further delayed data collection work.

Table 1. Criteria for assignment of PIT tag codes to an ESU or DPS group in 2013, based on reported hydrological unit code (HUC), species, and run type. Asterisks (*) indicate that the last 8 digits of the HUC could be any combination of digits.

		Release data reported in PTAGIS		
ESU or DPS	ESA status	HUC designation	Species code	Run type code
Snake River sockeye	Endangered	1706*	4	All codes
Chinook salmon				
Snake River spring/summer	Threatened	1706*	1	1 or 2
Upper Columbia River spring	Endangered	1702*	1	1
Middle Columbia River spring	Not listed	1703,* 1707*	1	1
Snake River fall	Threatened	1706*	1	3
Upper Columbia River summer/fall	Not listed	1703,* 1702*	1	2 or 3
Upper Willamette River spring	Threatened	1709*	1	1
Lower Columbia River	Threatened	1708,* 17070105, 17090011, 17090012	1	All codes
Steelhead				
Snake River	Threatened	1706*	3	All codes
Upper Columbia River	Threatened	1702*	3	All codes
Middle Columbia River	Threatened	1703,* 1707*	3	All codes
Upper Willamette River	Threatened	1709*	3	4

Caspian Tern Colony

Methods

Caspian terns nest only on relatively flat, bare sand. Therefore, PIT-tag surveys on the tern colony were performed with detection gear designed for this habitat. To detect PIT tags on or up to 10-15 cm below the surface, we used a 6-coil, flat-plate antenna system deployed from a vehicle. This system allowed us to upload detection records to an on-board laptop computer (Ryan et al. 2001). We developed this flat-plate system in 2001, and it has been modified to be towed at speeds up to ~8 m/min (0.3 mph) by a small tractor driving along overlapping, parallel tracks (Figure 7). In this report, a “track” refers to a single, continuous swath covered by the flat-plate detector as the tractor is driven from one edge of the colony to the opposite edge in a straight line.

Because PIT-tag reading range is most sensitive to the orientation of a tag relative to the antenna detection field, we varied the scanning direction of the flat-plate system in the following two ways. First, each individual track within a pass was run both forward and backward over exactly the same terrain. Second, for each of the three passes over the tern colony, a different track orientation was used. In the first pass, tracks ran parallel to the long axis of the colony. In the second pass, tracks ran perpendicular to the long axis of the colony. In the third and final pass, tracks were oriented diagonally from the northwest corner to the southeast corner of the colony.

Before starting the tracks for each pass, we scanned the entire colony perimeter, towing the flat-plate both forwards and backwards along the perimeter track, for any tags that may have been deposited or washed into the grass on the edges of the colony. Where drift fencing was absent, we scanned at least one plate-width (~1.5 m) into the grassy area bordering the colony. In a few cases, obstacles such as old tire fragments or cables made it difficult to determine whether the flat plate was close enough to the substrate (10-12 cm) to detect potentially buried tags. For these small areas, we employed a hand-scanner to ensure proper coverage near the obstacle.

Figure 7. Flat-plate detection system used on East Sand Island Caspian tern colony. The custom aluminum housing on the rear of the tractor is connected to the tractor alternator and contains AC/DC power systems necessary to power the antenna, multiplex transceiver, and laptop computer. Overlapping tracks are visible on the sand surface.

Results

Between 6 November and 13 December 2013, we completed three passes over the Caspian tern colony and recorded a total of 17,760 PIT-tag codes with no prior history of detection. Of these detections, 11,860 (66.9%) were from juvenile salmon that migrated in 2013. A breakdown of raw tag-code recoveries by ESU/DPS group is provided in Table 2 . Tag codes were recovered from 10 of 13 groups of ESA-listed salmonids in the Columbia River Basin. The three ESA-listed groups not represented included Lower Columbia River coho salmon, chum salmon, and steelhead. None of these three groups had individuals with release records in PTAGIS at the time of this report. Naturally produced Lower Columbia River chum salmon enter the estuary as fry migrants (< 50 mm FL), and are therefore generally too small to tag with PIT tags. We recovered PIT-tag codes from 19 cutthroat trout *Oncorhynchus clarkii*, as well as from 60 fish with no species identification listed in PTAGIS.

Table 2. Summary of all PIT-tags detected on East Sand Island bird colonies, migration year 2013. Tags were recovered from fish in all 10 ESA-listed groups reported as PIT-tagged in 2013 and for two additional non-listed ESUs.

				Detected at Bonneville/Sullivan Dam and recovered on East Sand Island		
ESU or DPS	ESA-status	Detected at Bonneville/Sullivan	Recovered on East Sand Island	Caspian tern	Double-crested cormorant	Brandt's cormorant
Snake River sockeye	Endangered	1,463	310	4	15	0
Chinook salmon						
Snake River spring/summer	Threatened	16,177	5,216	71	189	13
Upper Columbia River spring	Endangered	3,113	619	8	30	0
Middle Columbia River spring	Not listed	4,474	720	22	25	2
Snake River fall	Threatened	4,260	1,547	14	28	0
Upper Columbia R summer/fall	Not listed	4,259	1,243	27	37	3
Upper Willamette River spring	Threatened	2,763	83	18	10	0
Lower Columbia River	Threatened	13,139	2,911	77	415	15
Steelhead						
Snake River	Threatened	8,107	11,358	667	129	9
Upper Columbia River	Threatened	4,521	2,236	8	30	0
Middle Columbia River	Threatened	1,782	720	22	25	2
Upper Willamette River	Threatened	5	35	0	0	0
Total		64,063	26,998	938	933	44

Cormorant Colony

Methods

The cormorant colony is fundamentally different from the tern colony in that the latter contains a single species and a single nesting habitat type (bare sand). In contrast, the cormorant colony contains double-crested and Brandt's cormorants, two nesting habitat types (bare sand, rip-rap), and two experimental treatment areas (dissuasion area, tag-deposition experiment areas).

In addition, cormorant nests have a more complex three-dimensional structure, as they are built with sticks, vegetation, fecal matter, and other materials, whereas tern nests are built as a simple scrape in the sand. Because the larger flat-plate detector cannot be used in the complex, three-dimensional structures of the cormorant colony, all scanning for PIT tags on this colony was done with small, hand-held scanners (Figure 8).

To systematically survey the colonies, hand-scanning teams were assigned to one of five possible survey categories in a species-by-habitat matrix (Table 3).

Figure 8. Hand-held PIT detector used to scan East Sand Island rip-rap habitat on cormorant colonies. Battery-powered transceivers were carried in a backpack system.

Table 3. Survey categories for hand-scanning PIT tags on the cormorant colony.

Species	Habitat type or experimental treatment
Double-crested cormorant	Bare sand
	Rip-rap
Brandt's cormorants	Bare sand, 12 plots
	Rip-rap, 2 plots
Double-crested cormorant	Deposition experimental sites, 1-12

In collaboration with BRNW, we used a combination of aerial photographs and on-the-ground reconnaissance to determine which areas fell into each species and habitat designation. Field crews consulted these maps and discussed strategy for scanning habitat each day before data collection began and as necessary throughout the day

In 2013, on-colony observations and high-resolution aerial photographs collected by Birds Research Northwest revealed that Brandt's cormorant nests were dispersed among several plots throughout the larger colony. These plots were not confined to a few locations on the periphery of areas occupied by double-crested cormorants. Therefore, we performed separate PIT-tag surveys of nesting locations for all plots containing 20 or more spatially contiguous Brandt's cormorant nests. Collaborators agreed that tags in these nest areas were most likely to have been deposited primarily by Brandt's rather than double-crested cormorants.

A total of 13 plots had 20 or more Brandt's cormorant nests (Table 4), with 12 containing bare sand habitat and 2 containing rip-rap habitat. On 9 October, immediately after cormorants vacated the colony, Bird Research Northwest staff marked Brandt's cormorant survey plots with 4-ft long bamboo stakes and non-toxic spray paint. We repainted marks as needed and summed all tag recoveries on Brandt's cormorant plots to estimate predation rates by this species.

Table 4. Brandt's cormorant plots hand-scanned for PIT tag recovery.

Plot designation	Habitat type	Brandt's cormorant nests in plot (n)	Double-crested cormorant nests in plot (n)
AB	Bare sand	407	5
Bonus	Rip-rap	99	1
C	Bare sand	43	0
D1	Bare sand	30	0
D2	Bare sand	70	0
E	Bare sand	86	0
F	Bare sand	27	1
H1	Bare sand	57	0
H2	Bare sand	31	0
H3	Bare sand	68	0
I	Bare sand	65	1
O	Bare sand	34	1
U	Bare sand, Rip-rap	246	2

While scanning for tags, hand-scanning teams manually moved detectors back and forth across/against the surface of the substrate in a manner best described as "aggressive vacuum-cleaning." Whenever possible, we walked systematically along parallel, overlapping, tracks within the area being scanned. To ensure large areas were not missed during hand-scanning, we used a combination of maps, visual landmarks, sticks, non-toxic paint, and scuff marks in the sand to delineate sections of habitat that had already surveyed.

At the end of each day, the survey area covered was marked on laminated copies of the aerial colony maps. Exposed rip rap habitat was scanned first because this habitat type is most vulnerable to tag loss due to wind, rain, and waves associated with fall storm events.

Results

Between 31 October and 27 November 2013, we completed two passes over the approximately 12,545 m^{-2} (3.1 acres) of cormorant nesting area, including all double-crested cormorant nest plots, Brandt's cormorant nest plots, and experimental deposition sites. The first pass was completed on 20 November 2013; the second pass was completed on 27 November.

On the double-crested cormorant colony, hand-scanning recovered 14,500 unique tag codes with no prior history of detection, of which 11,020 were from juvenile salmon that migrated in 2013. In areas occupied primarily by Brandt's cormorants, we recovered an additional 780 unique tags with no prior detection history, of which 477 came from fish that migrated downstream in 2013. A breakdown of all tag-code recoveries by ESU/DPS group is provided in Table 2.

Tag codes were recovered from all 10 PIT-tagged groups of ESA-listed salmonids in the Columbia River Basin. The other three ESA-listed groups (Lower Columbia River coho salmon, chum salmon, and steelhead) had no 2013 PIT tag release records in PTAGIS at the time of this report. On the double-crested cormorant colony, we recovered one PIT tag code from a northern pikeminnow, *Ptychocheilus oregonensis*, and one from a Pacific lamprey, *Entosphenus tridentata*. Forty-two codes had no species identification listed in PTAGIS database. From Brandt's cormorant areas, we recovered one PIT-tag code from a cutthroat trout and one from a northern pikeminnow. Only one code was found that had no species identification listed in the PTAGIS database.

Estimated Rates of Predation

To estimate more accurate predation rates (number consumed/number available) from estuary PIT-tag detections, we adjusted for two types of measurement uncertainty. First, detection efficiencies on bird colonies are not 100%; thus, some proportion of PIT tags actually present on a colony will not be detected during a tag survey. For example, if two or more tags are located very close to each other, their transponders may be excited and emit a tag code at exactly the same time, resulting in a “tag-code collision” wherein the transceiver cannot read either code correctly. Tags may also break over the course of the season or become buried so deeply that they are beyond the reading range of the detector. Methods to account for imperfect detection probabilities are described below and referred to as “detection efficiency adjustments.” Results from this method provide minimum estimates of predation impacts.

Second, it is also known that some proportion of PIT-tags from salmon captured or ingested by birds will be deposited off the colony or damaged during the digestion of prey (Osterback et al. 2013; Roby et al. 2013). These tags are therefore cannot be detected during tag surveys of the colony. Methods that account for this phenomenon are referred to as “deposition adjustments.”

Adjustments for detection efficiency have been made to estimates of predation on East Sand Island in most prior years of this study (e.g. Ryan et al. 2001; Sebring et al. 2012; Roby et al. 2013; Zamon et al. 2013). However, methods of adjusting for colony deposition rates are novel and have been made only for double-crested cormorants and Caspian terns nesting on East Sand Island in 2012, and for Caspian terns in a few cases prior to 2012 (Lyons et al. 2012; Roby et al. 2013, Appendix A; Roby et al. 2014, Appendix A). For salmonid groups originating above Bonneville and Sullivan Dam in 2013, we examined predation rates adjusted for both types of measurement uncertainty.

Adjustments to Tag Recovery Data

Detection Efficiency Adjustments

Methods—Detection efficiency was estimated by sowing a known number of "control" tags, or tags with known codes, onto the colony before, during, and after the nesting season. As in previous years, Bird Research Northwest personnel placed groups of 100 control tags on the East Sand Island Caspian tern and double-crested cormorant colonies to quantify detection efficiency. Because there were two habitat types on the cormorant colony, control tags were divided into bare sand (n = 100) and rip-rap (n = 100) habitat subgroups. Tags were sown once in April before egg-laying began, and again in October, after all chick-rearing activity had ended and adult birds had vacated the colonies. In 2013, Bird Research Northwest sowed 100 additional tags during egg incubation or chick-rearing on the Caspian tern colony. They reported being able to do this without disrupting normal nest attendance. However, to avoid disturbance to birds, control tags were not sown during incubation or chick-rearing periods on the cormorant colony.

Detection efficiency is expected to decrease with time elapsed since the initial deposition of a PIT tag. The longer a tag remains on a colony, the higher the probability that it will be buried, damaged, or washed away from the original deposition site. Evans et al. (2012) used a logistic regression equation to model this time-dependent decrease in detection probabilities. Model parameters were estimated from detection efficiencies measured by control-tag recoveries. We adjusted all predation rates using detection efficiencies estimated from this logistic regression method.

Results—Seven hundred control tags were deployed across tern and cormorant nesting colonies during 2013, with equal numbers sown during each discrete time period and habitat type. However, on the bare-sand habitat of the cormorant colony, some pre-season control tags were sown in areas where birds did not subsequently nest, and we did not know the exact location of these tags. As a result, we were unable to determine how many of the original 200 pre-season control tags ended up in the actual nesting area. To account for potential bias resulting from lost control tags, we estimated the number of control tags recovered over the entire colony in two steps:

1. We counted the actual number of pre-season control tags recovered from rip-rap habitat (n = 57)
2. To this count, we added 100 multiplied by the mean value for pre-season bare sand detection efficiency ($\overline{x}$ = 0.7) during prior years when conditions on the colony were similar to 2013 (2010-2012).

This resulted in a mean pre-season estimated detection efficiency value of 63.5% based on 127 (57 + 70) tags found in during 2013. With this adjustment, tag deployments and recoveries on each colony, as well as raw detection efficiencies for each date, are summarized in Table 5.

Table 5. Observed recoveries of control tags sown to estimate daily detection efficiencies on East Sand Island in 2013. Data from this table were used to generate the logistic regression equation for date-specific detection efficiencies.

Colony	Date sown	Tags sown (n)	Detections (n)	Proportion recovered (%)
Caspian tern				
	10 Apr	100	54	54.0
	17 Jul	100	53	53.0
	10 Oct	100	84	84.0
Double-crested cormorant				
	11 Apr	200	127*	63.5
	9 Oct	200	147	73.5

* Estimated value; see text for explanation

Mean adjusted detection efficiencies in 2013 were 63.3% for the Caspian tern and 68.5% for the double-crested cormorant colonies overall. Daily detection efficiencies for East Sand Island colonies were estimated for the nesting, incubation, and chick-rearing periods period between 1 March and 31 August 2013 using the logistic regression model of Evans et al. (2012; Table 6). Estimated daily detection efficiencies ranged 41-72% for the Caspian tern colony and 61-72% for the double-crested cormorant colony. As in prior years, we applied the cormorant colony adjustment to both double-crested and Brandt's cormorants.

Table 6. Colony-specific coefficients of the binomial logistic regression used to estimate daily detection efficiencies. Nomenclature is as per Evans et al. (2012), Equation 2.

Colony	β_0	β_i
Caspian tern	--0.78	0.00711
Double-crested cormorant	0.29	0.00258

Although detection efficiencies were within the range of those measured in recent years (c.f. Appendix Table 4 in Sebring et al. 2012), we noticed a downward trend in detection efficiencies on the tern colony after 2009 (Table 7). This trend indicated a potential problem with tag collision due to increasing tag density on the tern colony.

The causes of this trend are likely twofold. First, pre-season deep harrowing of the colony surface with a large tractor has not been conducted since 2008, although the colony was harrowed to a shallow-depth (~30 cm) using an ATV in pre-season 2013. Deep harrowing is necessary to redistribute prior years' tags into the substrate and out of the detection range of the flat-plate antenna (D. Roby, pers. comm.). Therefore, tag density has likely been increasing near the surface, increasing tag collisions in the vertical dimension. Second, nest density per square meter has increased as the colony area has decreased (Roby et al. 2013, 2014), probably resulting in tags being deposited more closely together in the horizontal dimension. The net result is increasing near-surface tag densities in both the vertical and horizontal dimensions.

Table 7. Trends in raw control tag code recoveries and recovery of current migration year tag codes on the East Sand Island Caspian tern colony.

	Control tag code recoveries		Proportion of newly detected tag codes from current migration year (%)
Year	Mean recovery efficiency (%)	Pre-season recovery efficiency (%)	
2007	89	92	89
2008	92	90	93
2009	90	91	86
2010	84	76	85
2011	77	76	70
2012	77	60	77
2013	64	54	67

On-colony Deposition Adjustments

Methods—In 2012, Roby et al. (2013) introduced a new method to estimate Columbia River predation rates derived from PIT-tag recoveries. This method attempts to account for PIT tags consumed by birds but deposited at sites away from the breeding colonies. Because birds do not spend 100% of their time on the colony, some proportion of tags they consume will be deposited off the colony (Osterback et al. 2013; Roby et al. 2013). For example, PIT tags egested during a foraging trip may fall into the water or onto off-colony terrain. Due to variation among bird species in colony attendance patterns, foraging trip duration, and other behavior and biological factors (e.g., size and egestion processes), adjustments for deposition rate need to be species-specific.

An overview of methodology for calculating species-specific on-colony deposition rates for adjustments to Caspian tern and double-crested cormorant predation rates was presented in Roby et al. (2014, Appendix A, *PIT Tag Deposition Studies*). Briefly, birds were fed fish containing PIT tags with known codes, and the proportion of those codes recovered on colonies was used to estimate the probability of on-colony vs. off-colony deposition. Bootstrapping techniques were used to calculate a 95% confidence interval for the estimated on-colony deposition rate (Roby et al. 2014).

Results—Detailed results describing species-specific adjustments to Caspian tern and double-crested cormorant predation rates from deposition experiments are presented in appendices to Roby et al. (2013, 2014). Briefly, they used PIT-tag data collected during 2005-2006 from Caspian tern colonies on Crescent and East Sand Islands to estimate a mean on-colony tag deposition rate of 71% (95% CI 62-81%). In 2012 and 2013, respective mean on-colony deposition rates of 44 (36-51%) and 60% (47-73%) were estimated based on experiments conducted with East Sand Island double-crested cormorants.

Estimated Annual Predation for Listed Upriver Groups

Methods

The methods and reporting of estimated predation rates for PIT-tagged groups has changed in 2013 compared to our reports prior to 2012. These two changes are described below.

First, we estimated predation rates by Evolutionarily Significant Unit (ESU) or Distinct Population Segment (DPS). To accomplish this, we began by assigning each tagged release group to a specific ESU or DPS listed by the National Marine Fisheries Service (NMFS 1992; NOAA 2014). As described in Table 1, we assigned individual tag groups to an ESU/DPS unit based on the Columbia Basin Hydrologic Unit Code (HUC), species, and run type associated with each ESU or DPS.

This change was part of an effort to align reporting of research with ESU or DPS management units. Thirteen of 19 anadromous salmonid groups from the Columbia River Basin are listed as threatened or endangered. Because all juvenile salmon and steelhead originating in the Columbia River Basin must pass through the estuary to reach the ocean, all ESUs and DPSs are potentially subject to estuary avian predation. However, few of the 13 listed ESUs or DPSs are PIT-tagged in a representative fashion. Thus, for some listed salmonids, data is unavailable or insufficient for unbiased estimates of predation impact.

Second, our estimated predation rates in 2013 were adjusted for on-colony tag deposition rates by BRNW. These adjustments are intended to produce more biologically realistic estimates of predation (e.g. Ryan et al. 2003, Osterback et al. 2013, Roby et al. 2013, 2014). Methods for adjusting our estimates of on-colony tag deposition rates are reported in Roby et al. (2013; sections 1.4 and 2.4).

Predation rates for ESU and DPS groups were estimated using a two-step process:

1) We estimated how many PIT-tagged fish from each ESU/DPS were *available* to birds in the estuary during the nesting season. This was done by compiling tag codes from fish known to have been detected at Bonneville or Sullivan Dam or released into the estuary from a barge or tagging program on any given day, and
2) We estimated how many PIT-tagged fish were *consumed* by birds on East Sand Island during the nesting season. This was done for each ESU/DPS by adjusting PIT-tag code recovery numbers with estimated detection efficiency and on-colony tag deposition probabilities for East Sand Island terns and cormorants, respectively.

For some listed ESU/DPS groups, fish enter the Columbia and Willamette Rivers above the PIT-tag detectors at Bonneville Dam (Columbia River, rkm 235) or Sullivan Dam (Willamette River, rkm 206). For these groups we used the total number of fish detected passing either of these two dams between 1 March and 31 August 2013 to estimate fish availability to birds in the estuary (Evans et al. 2012). Following the methodology of Evans et al. (2012), we estimated predation rates only for groups with at least 500 PIT-tagged individuals available to birds in the estuary during the season to minimize the chance of spurious results that might arise due to small sample sizes.

Results

Adequate sample sizes were available to calculate estimated predation rates for 8 of the 13 listed ESU/DPS groups, as well as for 2 groups that were not listed (Table 8). We present estimated predation rates adjusted for detection efficiency alone (considered to be minimum predation rates) as well as rates adjusted for both detection efficiency and off-colony tag deposition (Table 8). For Caspian terns, estimated predation rates adjusted for on-colony deposition were an average of 38% higher than those adjusted for detection efficiency alone (differences of 20 to 67%). For double-crested cormorants, deposition-adjusted predation rate estimates were an average of 57% higher than predation rates adjusted for detection efficiency alone (differences of 23 to 70%).

For Brandt's cormorants, adjustments for on-colony deposition were negligible because both minimum and deposition-adjusted estimates, were close to zero (≤ 0.2%). During the remainder of this section of the report, we refer to deposition-adjusted predation rates, and note that results for salmonid groups with highest and lowest predation impacts were the same when examining either minimum estimated predation rates or deposition-adjusted predation rates.

Of the ESU and DPS groups we analyzed, the PIT-tagged fish experiencing the highest predation rate by Caspian terns were Snake River steelhead (12.5%; Table 7). The highest predation rate estimates for double-crested cormorants were observed in Snake River spring Chinook salmon (2.9%) and sockeye salmon (2.6%). The lowest predation rates by Caspian terns were observed on Upper Columbia River spring Chinook salmon (0.6%) and Snake River sockeye salmon (0.7%). The lowest predation rate by double-crested cormorants was observed on Upper Willamette River Chinook salmon (0.7%). Overall, Brandt's cormorants appeared to have low predation rates on all groups evaluated, regardless of ESU/DPS examined (<0.2%).

Table 8. Predation rates for Columbia River Basin ESU/DPS groups with population origins above Bonneville or Sullivan Dams, migration year 2013. Estimates are shown for tag detection rates adjusted for detection efficiency alone and for both detection efficiency and on-colony deposition rate. We calculated estimates only where ≥500 fish from an ESU/DPS were detected at a dam. Information on Lower Columbia River Chinook salmon is presented in Table 9.

		Adjusted estimates of predation rate								
		Caspian tern			Double-crested cormorant			Brandt's cormorant		
		Detection	Deposition		Detection	Deposition		Detection	Deposition	
ESU or DPS	ESA-status	efficiency (%)	(%)	95% CI	efficiency (%)	(%)	95% CI	efficiency (%)	(%)	95% CI
Snake River sockeye	Endangered	0.5	0.7	0.2-1.5	1.6	2.6	1.3-4.1	<0.1	<0.1	n/a
Chinook salmon										
Snake River spring/summer	Threatened	0.8	1.1	0.8-1.5	1.8	2.9	2.4-3.7	0.1	0.2	0.1-0.3
Upper Columbia River spring	Endangered	0.5	0.6	0.2-1.2	1.5	2.4	1.6-3.5	0.1	<0.1	n/a
Middle Columbia River spring	Not listed	0.9	1.3	0.7-1.9	0.8	1.3	0.8-1.9	0.1	0.1	<0.1-0.3
Snake River fall	Threatened	0.6	0.8	0.5-1.3	1.0	1.7	1.1-2.4	<0.1	0.1	<0.1-0.2
Upper Columbia River summer/fall	Not listed	1.1	1.4	0.8-2.0	1.3	2.0	1.3-2.8	0.1	<0.1	n/a
Upper Willamette River	Threatened	0.6	0.9	0.4-1.4	0.5	0.7	0.3-1.4	<0.1	0.2	<0.1-0.2
Steelhead										
Snake River	Threatened	7.5	12.5	10.4-15.1	1.2	2.0	1.5-2.7	0.1	0.1	n/a
Upper Columbia River	Threatened	6.3	8.6	7.1-10.6	1.6	2.7	1.9-3.7	0.1	0.1	<0.1-0.3
Middle Columbia River	Threatened	7.5	9.6	7.1-12.5	1.3	1.6	0.7-2.6	<0.1	0.2	<0.1-0.5

Predation Impacts on Lower Columbia River Chinook Salmon

Methods

Chinook salmon populations included in the Lower Columbia River ESU exhibit extremely diverse life history characteristics. Fish in this ESU may have eight possible life history types, depending on the combination of juvenile life history (yearling or subyearling), origin (hatchery or naturally spawned), and natal stream geography (broadly divided as above or below Bonneville Dam; Appendix A in Lyons et al. 2012). A coordinated tagging program for this ESU would tag each life history subgroup in proportion to its contribution to the ESU as a whole. The lack of such a program means that predation rate estimates based on PIT-tag code recoveries are more complicated and less precise than those for other ESU/DPS groups. Depending on how one chooses to address this diversity with respect to currently available tag data, there are different assumptions associated with different calculation methods.

There are three primary reasons why the calculations for Lower Columbia River Chinook salmon do not follow the same method as for ESU/DPS groups originating above Bonneville or Sullivan Dam. First, PIT-tagged fish from other ESU/DPS groups enter the Columbia or Willamette River upstream from a detection facility (i.e., Bonneville or Sullivan Dam) where downstream migration behavior and entry into the foraging range of estuary birds can be documented. In contrast, Lower Columbia River Chinook salmon may enter the mainstem Columbia River below a terminal dam, with no record of movement other than the release date. Often we do not know when these lower-river fish begin migration through the estuary. Therefore, we have no post-release interrogation data with which to estimate the availability of these fish to birds in the estuary (Lyons et al. 2012).

Second, because there is no coordinated effort to PIT tag Lower Columbia River Chinook salmon, many groups with this ESU (most notably naturally produced fish) are not well-represented among PIT-tagged fish. Furthermore, groups that are PIT tagged in any given year are not tagged in proportion to their contribution to the entire ESU. Therefore, predation estimates calculated from PIT-tag recoveries do not necessarily provide an accurate picture of what is happening to the ESU as a whole, but are limited to specific tag groups. In the absence of a coordinated PIT-tagging program for Lower Columbia River Chinook salmon, generalizations as to avian impacts on this ESU need to be made with caution.

Third, the total number of Lower Columbia River Chinook salmon tagged and released on an annual basis is relatively small compared to numbers tagged and released in other ESUs. Therefore, sample sizes for analyses are inherently too small to produce robust and precise estimates of predation. To obtain sample sizes sufficient to compare predation rates among subgroups of Lower Columbia River Chinook salmon in the past it has been necessary to compile data from several years and to use smaller sample sizes than available for other ESU/DPS groups (e.g., 100 rather than 500 PIT-tagged fish available) (c.f. Appendix A in Lyons et al. 2012).

In attempting to estimate predation impacts on Lower Columbia River Chinook salmon, some investigators have measured the availability of these fish using all annual PIT-tag release data from fish in this ESU, regardless of geographic origin (e.g. Lyons et al. 2012). This had the advantage of producing larger sample sizes but the disadvantage of failing to account for mortality between release and entry into the mainstem below Bonneville Dam. Thus estimates of predation using this method are likely biased low because the number of fish available for consumption by birds is probably less (and thus the proportion consumed greater) than the number originally released due to other unmeasured sources of mortality.

In previous years of this study, researchers have compared relative rates of predation or used indices of availability to estimate impacts to PIT-tagged groups of Lower Columbia River fish (Ryan et al. 2003; Sebring et al. 2013). For components of the ESU originating in upriver areas, an index of availability of fish to birds in the estuary was calculated using dates of detection at Bonneville or Sullivan Dam. For components of the ESU originating below the dams, release date was used for these indices. This approach accounted for mortality that occurred before fish reached a terminal dam for all upriver PIT-tagged groups. However, it had the disadvantages of 1) using different measures of estuary availability for upriver vs. lower river groups, and 2) considerably reducing replicate size for upriver groups.

Both methods assume that the date on which a tagged fish was consumed by a bird within one week of the day on which that fish was detected at an interrogation site, or within one week of the date of release for those lower river fish where interrogation sites were not available. At this time, there is no way to test this assumption because it is not yet possible to discern either the date of a predation event or the date of deposition on a colony for a specific PIT tag.

We present only information on tagged release groups, dam detections (where applicable), and subsequent raw or unadjusted tag recoveries on East Sand Island. We do not make any ESU-wide predation rate estimation because we know that doing so would present a biased estimate for the ESU as a whole. Estimating predation rates for the specific subgroups tagged in 2013 was beyond the scope of this report, although that would be a conservative way to estimate avian predation impacts on PIT-tagged subgroups in the absence of a representative tagging program for the entire ESU.

To estimate how many Lower Columbia River Chinook salmon were available to birds in the estuary on any given day, we summed all daily releases of PIT-tagged groups from this ESU within its geographic boundaries between 1 March and 31 August 2013. We summed tag-code recoveries for each day during this period to estimate a minimum number of fish consumed by birds on East Sand Island. These sums were not adjusted for detection efficiency or on-colony deposition rates; they represent only raw counts of tag recoveries.

Results

Between 1 March and 31 August 2013, groups of PIT-tagged Chinook salmon from the Lower Columbia River ESU were released from 13 sources. The earliest date on which tagged fish from this ESU were released was 1 March 2013; records in PTAGIS indicate no PIT-tagged fish in this ESU were released prior to that day. The last tag group was released on 2 July 2013. A total of 97,255 PIT-tagged Chinook salmon from migration year 2013 were released within the geographic boundaries of this ESU.

Of these releases, 72% came from three hatcheries, all of which released fish to the mainstem Columbia River above Bonneville Dam. These “top three” sources of PIT-tagged fish were Carson National Fish Hatchery, Little White Salmon National Fish Hatchery, and Spring Creek National Fish Hatchery (Table 9). Four additional sources contributed at least 1% to the tagged population. Seven sources accounted for 99.4% (n = 96,672) of all tagged fish in the Lower Columbia River Chinook salmon ESU. All other sources combined accounted for 0.6% (n = 583) of tagged fish in this ESU.

A total of 2,911 PIT-tag codes from the Lower Columbia River Chinook salmon ESU were recovered on East Sand Island bird colonies. The majority of all tag detections were from the double-crested cormorant colony (n = 2,435, or 83.6 %); 411 were recovered from the Caspian tern colony (14.1%), and 65 from Brandt’s cormorant plots (2.2%). Of these 2,911 codes, 507 codes were also detected previously at Bonneville Dam. A breakdown of code detections on East Sand Island by release source and bird colony is presented in Table 10.

The timing of releases from this ESU and recoveries of tag codes from those release groups are shown in Figure 9. The cumulative frequency distributions of both tag releases and tag code recoveries are shown in Figure 10. In 2013, 75.3% of tags from this ESU were released by 1 May, and 78.7% of all tags recovered on East Sand Island originated with groups released on or before 2 May.

Table 9. Proportions of PIT-tagged Chinook salmon released from sources within the Lower Columbia River ESU, migration year 2013.

	Release site	Run type	Rear type	Origin above/below Bonneville Dam	Total released (n)	Proportion of all releases (%)	Cumulative releases (%)	Detected at Bonneville Dam (n)
1	Carson National Fish Hatchery	Spring	Hatchery	Above	29,582	30.4	30.4	3,563
2	Little White Salmon National Fish Hatchery	Fall	Hatchery	Above	14,960	15.4	45.8	1,995
3	Spring Creek National Fish Hatchery	Fall	Hatchery	Above	14,941	15.4	61.2	821
4	Moving Falls Acclimation Pond, Hood River	Spring	Hatchery	Above	12,321	12.7	73.9	1,430
5	Little White Salmon National Fish Hatchery	Spring	Hatchery	Above	10,480	10.8	84.7	1,274
6	Parkdale Hatchery	Spring	Hatchery	Above	6,084	6.3	91.0	401
7	Willard National Fish Hatchery	Spring	Hatchery	Above	4,492	4.6	95.6	475
8	Bonneville Dam Juvenile Bypass Flume/Pipe	Fall	Hatchery	Above	3,712	3.8	99.4	3,186
	All other contributors to tagged groups	Mixed	Mixed	Mixed	683	1.2	100.0	14

Table 10. Detections of PIT tag codes from Lower Columbia River Chinook salmon on bird colonies, East Sand Island, migration year 2013.

		Total recoveries (n)			Recoveries previously detected at Bonneville Dam (n)		
	Release group	Caspian tern	Double-crested cormorant	Brandt’s cormorant	Caspian tern	Double-crested cormorant	Brandt’s cormorant
1	Carson National Fish Hatchery	83	243	7	11	35	3
2	Little White Salmon National Fish Hatchery	118	584	10	25	96	6
3	Bonneville Dam Facility Bypass Flume/Pipe	23	203	3	23	198	3
4	Moving Falls Acclimation Pond, Hood River	30	86	3	7	13	1
5	Spring Creek National Fish Hatchery	118	1,202	36	7	55	4
6	Willard National Fish Hatchery	15	57	4	0	9	0
7	Parkdale Hatchery	24	55	2	4	9	0
8	Hood River	0	2	0	0	0	0
	Totals	411	2,435	65	77	415	15

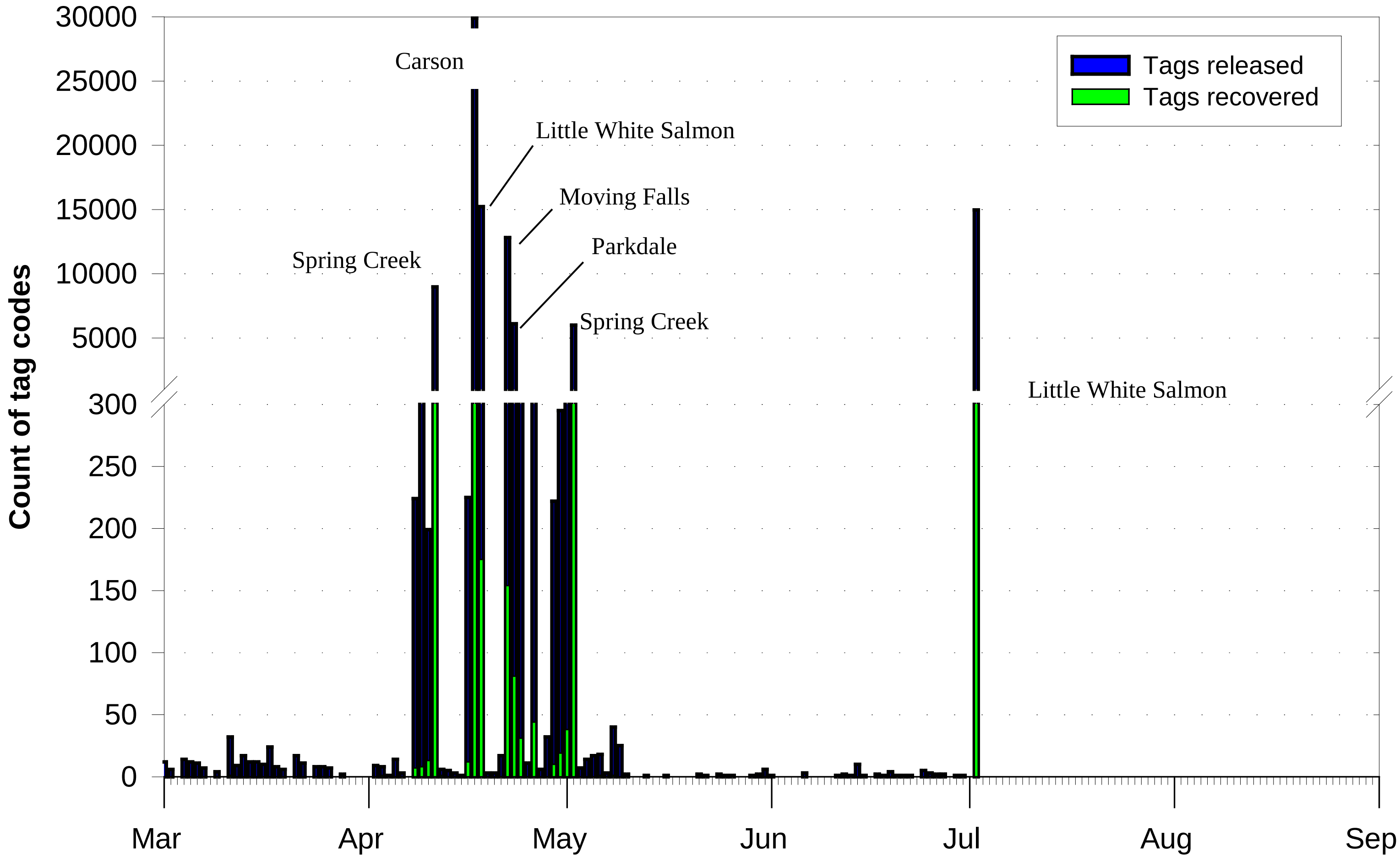


Figure 9. Recoveries of Lower Columbia River Chinook salmon tag codes from East Sand Island, compared to daily releases of PIT-tag groups between 1 March and 31 August 2013. Recoveries are raw counts, and have not been adjusted for detection efficiency or on-colony deposition. Hatcheries responsible for the top seven largest releases are shown on this figure.

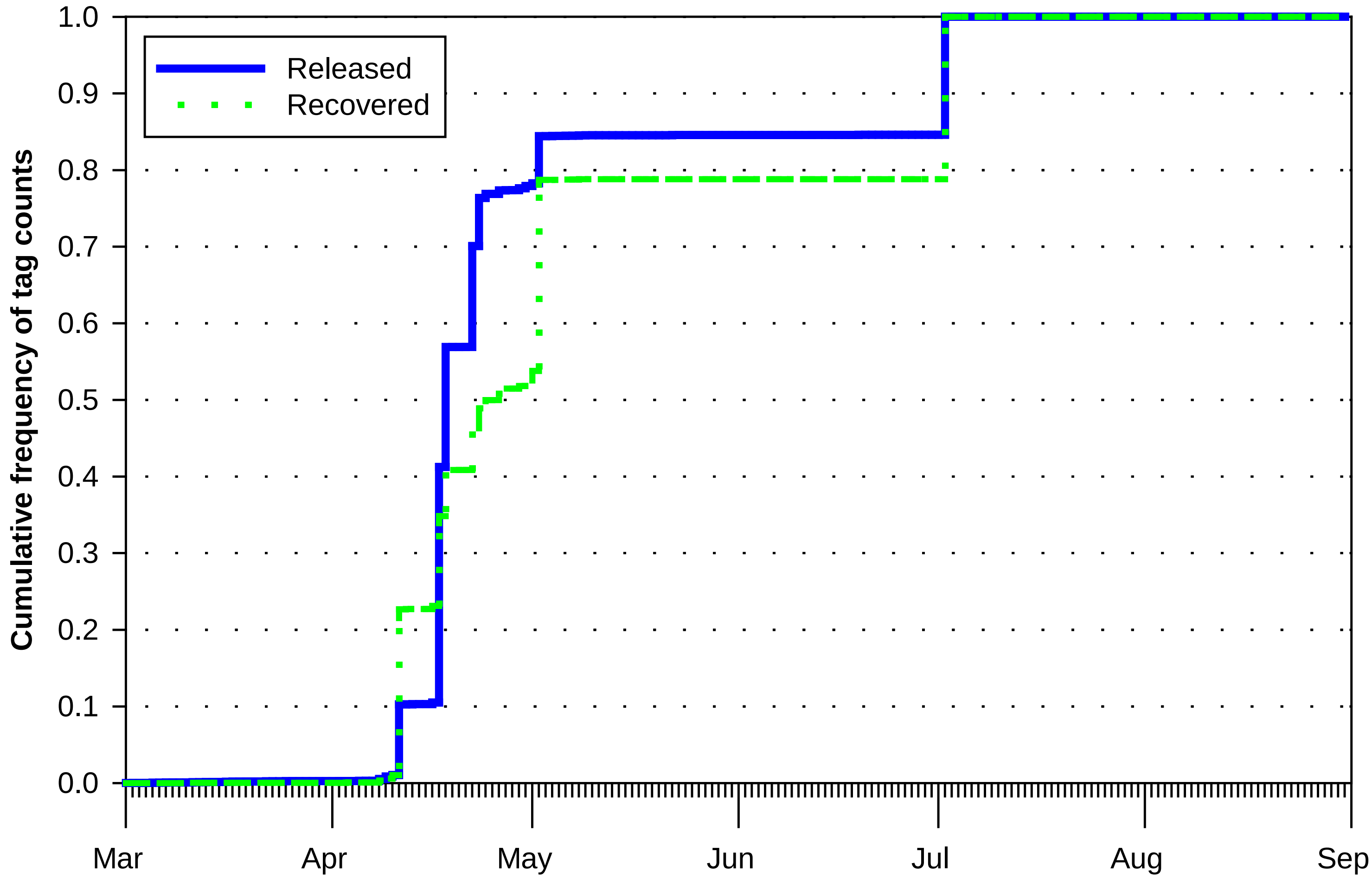


Figure 10. Cumulative frequency distributions for Lower Columbia River Chinook salmon released and recovered between 1 March and 31 August 2013. Recoveries are raw counts of tag detections and have not been adjusted for detection efficiency or on-colony deposition.

Estimated Minimum Predation on Transported vs. In-river Migrant Snake River Salmon and Steelhead

Methods

To measure availability of transported Snake River salmonids to birds in the lower Columbia River estuary, we obtained records of all fish PIT-tagged and released at or above Lower Granite Dam and subsequently routed onto a barge at one of the Snake River dams (Lower Granite Dam, rkm 695; Little Goose Dam; rkm 635; Lower Monumental Dam, rkm 589). Fish were PIT-tagged for experimental comparisons of transportation and a variety of other studies. Some PIT-tagged fish that were intended to remain in the river were placed on barges in error (typically <5% of all bypassed fish, Sandford unpublished data). Transported fish were released at night downstream of Bonneville Dam, near Skamania, WA, (between rkm 222 and 229; median release location rkm 225). Early in the season (April-May), releases occurred every 24 h; later in the season (June-August), releases occurred every 48 h.

To obtain an index of estuary availability for in-river migrant Snake River salmon or steelhead, we used the number of PIT-tag detections at Bonneville Dam between 1 March and 31 August 2013. Because we wanted to compare in-river migrants that experienced similar migration histories until barging took place, we included for analysis only fish originally released at or above Lower Granite Dam.

To measure how many available PIT-tagged fish were consumed by birds, we counted tag-code recoveries from East Sand Island. Tag codes provide no data on when an individual predation or tag deposition event occurred; therefore, we assumed for purposes of estimating daily detection efficiency that an individual fish was consumed on the same day it became available to avian predators. This availability date was the date of barge release for transported fish and date of last detection at Bonneville Dam for in-river migrant fish.

To estimate a minimum predation rate for fish consumed by birds on East Sand Island, we adjusted daily detection numbers for tag codes detected on East Sand Island between 1 March and 31 August 2013 by the daily detection efficiency specified by the logistic regression equation described previously in this report. We then summed daily estimated numbers of PIT-tags from each group of interest consumed by birds. Deposition adjustments were not included for these estimates because deposition estimates for 2013 were not available in time for this report.

We then compared minimum estuary avian predation rates between transported and in-river migrant Snake River salmon and steelhead in two ways. Our comparisons of relative predation rates between transported and in-river fish within a single year do not account for potential variation in deposition rates over the nesting season, and no data are yet available to measure such variation (Roby et al. 2013, 2014; Hostetter et al., BRNW, unpublished manuscript). However, a correction for detection efficiency was applied to account for detection probabilities that changed during the nesting season (Tables 4-5).

First, we estimated the annual minimum predation impact for each group over the entire season by adjusting the number of tag codes recovered on East Sand Island by daily detection efficiencies, and summed the adjusted numbers over the entire season. We then divided each sum by the total number of tags available throughout the season. This calculation was analogous to those described in previous sections of this report except that no adjust for deposition was made. Thus we compared, in a qualitative fashion, cumulative annual predation impacts for transport vs. in-river groups for each species/run/rear type available.

Second, we estimated the weekly minimum predation impacts for each group, and graphically compared these rates with the timing and magnitude of weekly barge releases and in-river migrants passing Bonneville Dam. We also evaluated the null hypothesis of no overall difference in mean predation rates between transported fish and in-river migrants. We tested this hypothesis using an unpaired *t*-test on weekly predation rates estimated for both groups. This method accounted for temporal variability in mean differences over the entire season, although it did not identify within-season temporal trends.

To ensure validation of the test assumption that means were approximately normally distributed, comparisons were restricted to weeks when ≥100 tag codes were available from barge releases or from detections at Bonneville Dam. Week 1 began on 1 March, and we computed weekly predation by summing the adjusted number of tag codes originating from releases/detections in that week and recovered on East Sand Island, and then dividing that sum by the total number of tag codes released/detected for that same week.

Results

Barge releases and recoveries—During migration year 2013, a total of 123,595 PIT-tagged juvenile salmon or juvenile steelhead were transported by barge and released at Skamania, WA, between 27 April and 5 August (Table 11). No barge releases took place before or after those dates. The distribution of barge releases by species, run, and rear type is presented in Table 11. Over three-quarters of transported fish (78.9%) were hatchery-reared. Only 21% were wild fish, and the majority of wild fish were (79.2%) were steelhead. Summary statistics for daily barge releases are presented in Table 12.

Table 11. Summary of barge releases and PIT tag code recoveries for Snake River salmon and steelhead, migration year 2013. Counts include only fish that were originally tagged and released at or above Lower Granite Dam.

Snake River ESU or DPS	Rear type	Released by barge (n)	Recoveries, by bird colony (n): Caspian tern	Double-crested cormorant	Brandt's cormorant	Totals
Sockeye salmon	Hatchery	9,838	20	40	4	64
	Wild	3	0	0	0	0
Steelhead	Hatchery	42,350	1,545	629	32	2,208
	Wild	20,603	542	312	21	875
Spring/summer Chinook salmon	Hatchery	45,019	195	608	21	824
	Wild	5,392	10	48	2	60
Fall Chinook salmon	Hatchery	337	1	2	1	4
	Wild	9	0	0	0	0
Totals		123,595	2,313	1,639	81	4,035

Table 12. Summary statistics for Snake River daily barge releases, migration year 2013.

Snake River ESU or DPS	Rear type	Days with ≥100 fish released by barge	Daily release group size (n): Minimum	Median	Maximum
Sockeye salmon	Hatchery	9	1	8	2,726
Steelhead	Hatchery	37	1	335	6,240
	Wild	30	1	134.5	2,080
Spring/summer Chinook salmon	Hatchery	22	1	47.5	6,094
	Wild	20	1	29	710
Fall Chinook salmon	Hatchery	0	1	6	47

Of these barge releases, 4,035 tag codes were subsequently recovered on East Sand Island, representing 3.3% of original releases. The distribution of recoveries for barged fish is shown in Table 11. Of tag recoveries on East Sand Island, 76.8% (n = 3,100) were from hatchery-reared fish and 23.2% (n = 935) were from wild fish.

In-river migrant detections and recoveries—A total of 26,469 PIT-tagged Snake River salmon or steelhead were released at or above Lower Granite Dam, migrated in-river, and were subsequently detected at Bonneville Dam between 1 March and 31 August 2013 (Table 13). The first of these detections was on 2 March and the last on 25 August. Summary statistics for daily in-river migrant detections are presented in Table 14.

In 2013, 719 PIT-tag codes from Snake River in-river migrant salmon and steelhead were detected at Bonneville Dam and also subsequently recovered on East Sand Island. The distribution of recovered tag codes by species, run, and rear type are shown in Table 12. This sample represented a recovery of 2.7% of the all tag codes detected at Bonneville Dam. The majority of these tag codes (n = 569, 79.1%) were from hatchery-reared fish.

An additional 14,756 tag codes from Snake River fish migrating in 2013 but not detected at Bonneville Dam were also detected on East Sand Island. The distribution of these tag codes by species, run, and rear type is show in Table 14. Of these tags, 84.4% (n = 12,458) were from hatchery-reared fish; the other 15.6% (n = 2,298) were from wild-origin fish. These tags could not be included in estimates of predation because we had no information on the date these in-river migrants passed Bonneville Dam. However, it is clear that these fish survived past Bonneville Dam, entered the estuary, and were then subject to avian predation.

Table 13. Summary of PIT tag detections of in-river migrant Snake River salmon and steelhead from East Sand Island bird colonies, migration year 2013. Counts include only fish that were originally tagged and released at or above Lower Granite Dam; these counts are therefore not necessarily from the same group of fish as presented in Table 2. Recoveries include only tags detected both at Bonneville Dam and on East Sand Island.

			Recoveries detected at Bonneville Dam and on bird colonies (n)			
Snake River ESU or DPS	Rear type	Total detected at Bonneville Dam (n)	Caspian tern	Double-crested cormorant	Brandt's cormorant	Totals
Sockeye salmon	Hatchery	1,420	4	15	0	19
	Wild	43	0	0	0	0
Steelhead	Hatchery	4,968	560	112	8	680
	Wild	2,501	107	17	1	125
Spring/summer Chinook salmon	Hatchery	14,128	65	172	11	248
	Wild	2,040	6	17	2	25
Fall Chinook salmon	Hatchery	1,323	14	6	8	28
	Wild	46	0	0	0	0
Totals		26,469	756	339	30	1,125

Table 14. Summary statistics for daily detections of in-river Snake River salmon and steelhead migrants at Bonneville Dam, migration year 2013.

			Daily release group size (n)		
Snake River ESU or DPS	Rear type	Days with ≥100 fish released by barge	Minimum	Median	Maximum
Sockeye salmon	Hatchery	6	0	0	257
Steelhead	Hatchery	21	0	0	349
	Wild	6	0	0	141
Spring/summer Chinook salmon	Hatchery	20	0	1	1,390
	Wild	4	0	0	110
Fall Chinook salmon	Hatchery	0	0	2	58

Table 15. Summary of PIT tag detections for in-river migrant Snake River salmon and steelhead on East Sand Island, migration year 2013. Counts include only fish originally released at or above Lower Granite Dam with no subsequent detection record at Bonneville Dam. These detections could not be used in the estimation of minimum predation rates.

		Recoveries on bird colonies but not detected at Bonneville Dam (n)			
Snake River ESU/DPS	Rear Type	Caspian tern	Double-crested cormorant	Brandt's cormorant	Totals
Sockeye	Hatchery	85	129	7	221
	Wild	1	3	0	4
Steelhead	Hatchery	5,780	1,438	64	7,282
	Wild	1,455	471	25	1,951
Spring/summer Chinook	Hatchery	1365	2,456	104	3,925
	Wild	83	244	8	335
Fall Chinook	Hatchery	400	604	26	1,030
	Wild	2	6	0	8
Totals		9,171	5,351	234	14,756

Annual minimum predation rates—Estimated annual minimum predation rates on barged vs. in-river migrant fish are presented in Table 16. Because sample sizes were less than 500 tagged fish, it was not possible to estimate predation rates for either transported or in-river migrant groups of wild sockeye (n = 3 transported), hatchery fall Chinook salmon (n = 337 transported), and wild fall Chinook salmon (n = 9 transported). Therefore, these three groups are not included in qualitative comparisons below.

Barged and in-river migrant hatchery sockeye salmon did not appear to be highly susceptible to estuary avian predation, as estimated annual predation rates were less than 1.0% in all cases except double-crested cormorants, which took an estimated 1.6% of in-river migrants in 2013. For hatchery and wild steelhead estimated minimum predation rates from Caspian terns were higher for in-river migrants (9.0 and 7.6%, respectively) than for barged fish (6.6 and 4.7%, respectively). In contrast, higher double-crested cormorant predation was estimated for barged than in-river migrant steelhead (2.3 vs. 1.0%, for both hatchery and wild fish).

Estimated minimum tern predation on hatchery steelhead appeared to be greater than that for wild fish. Estimates for Brandt's cormorant predation on steelhead were very low (0.1-0.2% in all cases); therefore, steelhead of any type did not appear to be particularly vulnerable to predation by Brandt's cormorants, nor did there appear to be any significant differences in Brandt's cormorant impacts on barged fish compared to in-river migrants.

Hatchery spring/summer Chinook salmon exhibited the same, relatively low rates of Caspian tern predation rates for both barged and in-river migrant fish (0.8%). These rates were higher than rates estimated for wild fish. In-river migrant wild fish may have experienced marginally higher predation rates (0.5%) than barged fish (0.3%). In contrast, double-crested cormorant predation on both hatchery and wild spring/summer Chinook was marginally higher for barged fish (2.1 and 1.4%, respectively) than for in-river migrants (1.9 and 1.3%, respectively). Hatchery fish exhibited higher estimated predation rates than wild fish from double-crested cormorants. Estimates of predation by Brandt's cormorants were low for all spring/summer Chinook salmon (0.1-0.2%), regardless of migration history or rear type.

In general, Caspian tern predation seemed to be consistently lower on barged fish compared to in-river migrants (4 of 5 cases), whereas double-crested cormorant predation seemed to be consistently higher on in-river migrants (4 of 5 cases). Brandt's cormorants exhibited very low estimated predation on all groups (0.0-0.2% in 5 of 5 cases).

Table 16. Estimated annual minimum predation rates for barged vs. in-river migrant Snake River salmon and steelhead, migration year 2013. Results are presented by species, run, and rear type for those groups where at least 500 fish were released from barged or detected during migration at Bonneville Dam.

		Estimated annual minimum predation rate (%)					
		Caspian tern		Double-crested cormorant		Brandt's cormorant	
Snake River ESU or DPS	Rear type	Barged	In-river	Barged	In-river	Barged	In-river
Sockeye salmon	Hatchery	0.4	0.5	0.6	1.6	0.1	0.0
Steelhead	Hatchery	6.6	9.0	2.3	1.0	0.1	0.1
	Wild	4.7	7.6	2.3	1.0	0.2	0.1
Spring/summer Chinook salmon	Hatchery	0.8	0.8	2.1	1.9	0.1	0.1
	Wild	0.3	0.5	1.4	1.3	0.1	0.2

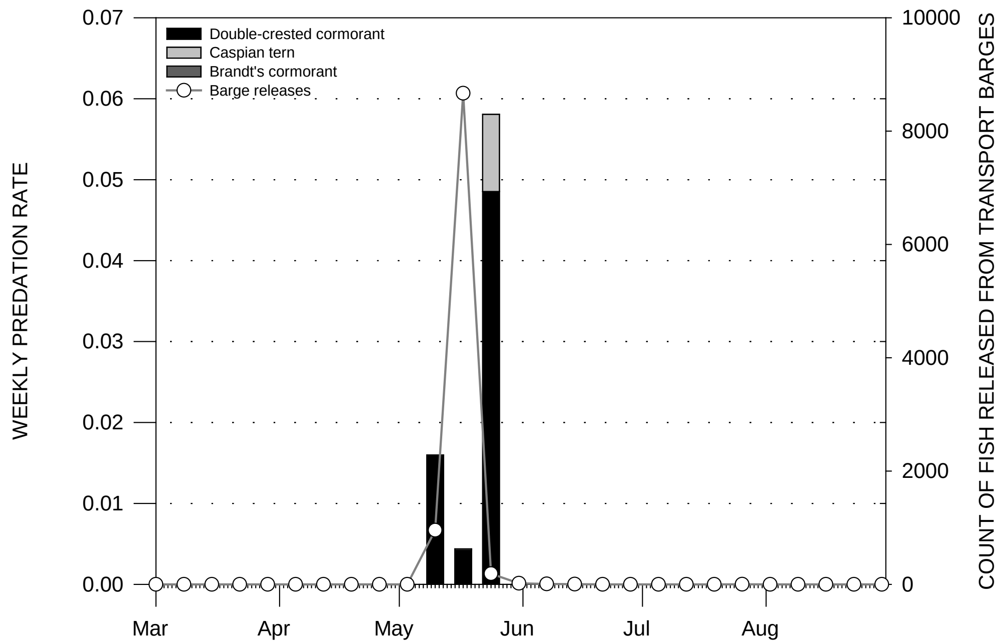
Weekly minimum predation rates—Plots of weekly minimum predation rates, weekly barge release numbers, and weekly detections of in-river migrants at Bonneville Dam are shown for all available fish groups in Figures 11 through 16. These figures present observed temporal trends in barge releases, detections at Bonneville, and estimated minimum weekly predation rates throughout the entire breeding season for avian predators. All raw release or detection numbers are shown, even in cases where there were less than 100 fish per week. Predation estimates were not calculated for weeks when less than 100 fish were released or detected, as these fell below our minimum sample size criterion. Counts of weeks that met the sample size criterion for estimating predation rates in each group are presented in Table 17.

Table 17. Weeks with minimum releases of 100 Snake River PIT-tagged fish, presented by species/run/rear types, migration year 2013. These numbers represent the sample sizes available for unpaired *t*-tests comparing barged vs. in-river fish groups.

		Weeks with ≥ 100 fish (n)	
ESU or DPS		Transported fish released from a barge	In-river migrants detected at Bonneville Dam
Sockeye salmon	Hatchery	3	2
Steelhead	Hatchery	8	7
	Wild	8	8
Spring/summer Chinook salmon	Hatchery	6	7
	Wild	6	7
Fall Chinook salmon	Hatchery	0	7

Sample sizes were adequate to perform unpaired *t*-tests on barged vs. in-river migrant fish for Snake River steelhead and spring/summer Chinook salmon, but not for sockeye salmon or fall Chinook salmon. Mean weekly rates for each group, as well as the *P*-value for each barged vs. in-river comparison, are presented in Table 18. None of the 12 comparisons performed revealed any statistically significant difference between mean estimated rates of predation for barged vs. in-river migrant fish (*P*-values 0.29-0.92, mean value of 0.59).

Barged Snake River Sockeye Salmon - Hatchery



In-river Migrant Snake River Sockeye Salmon - Hatchery

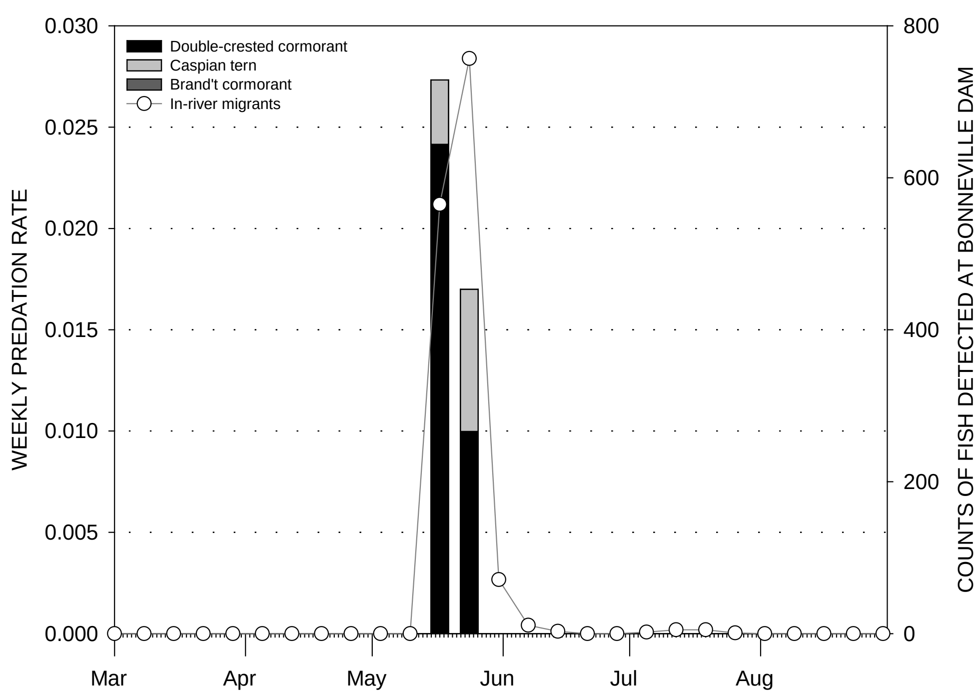
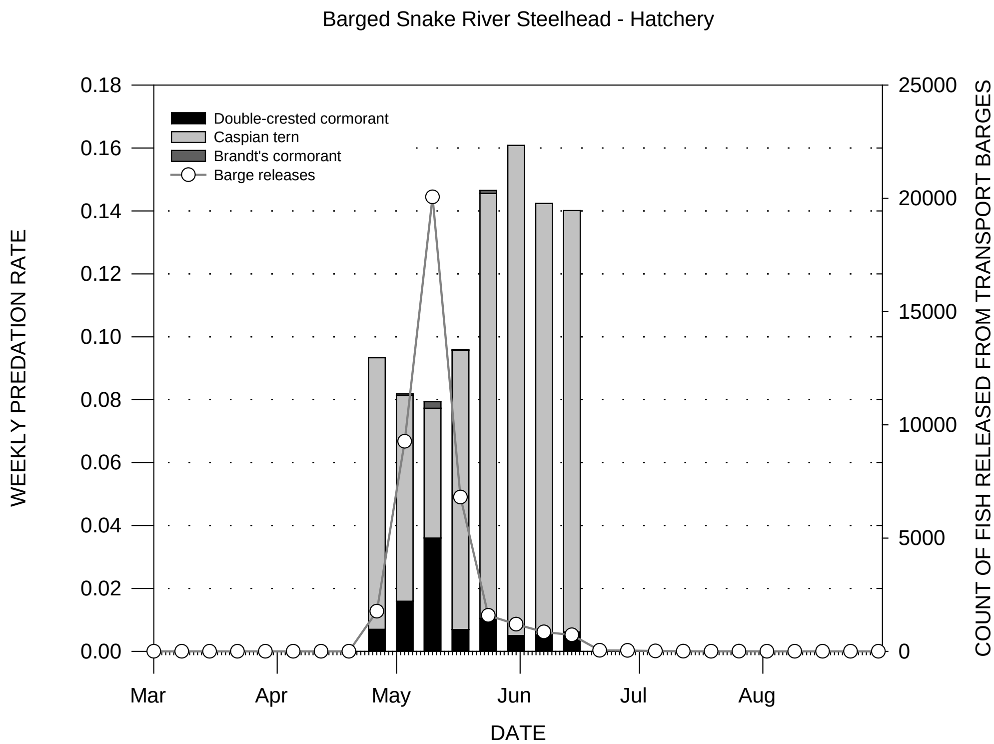


Figure 11. Weekly releases, detections, and estimated minimum predation rates for hatchery-reared Snake River sockeye salmon, migration year 2013.



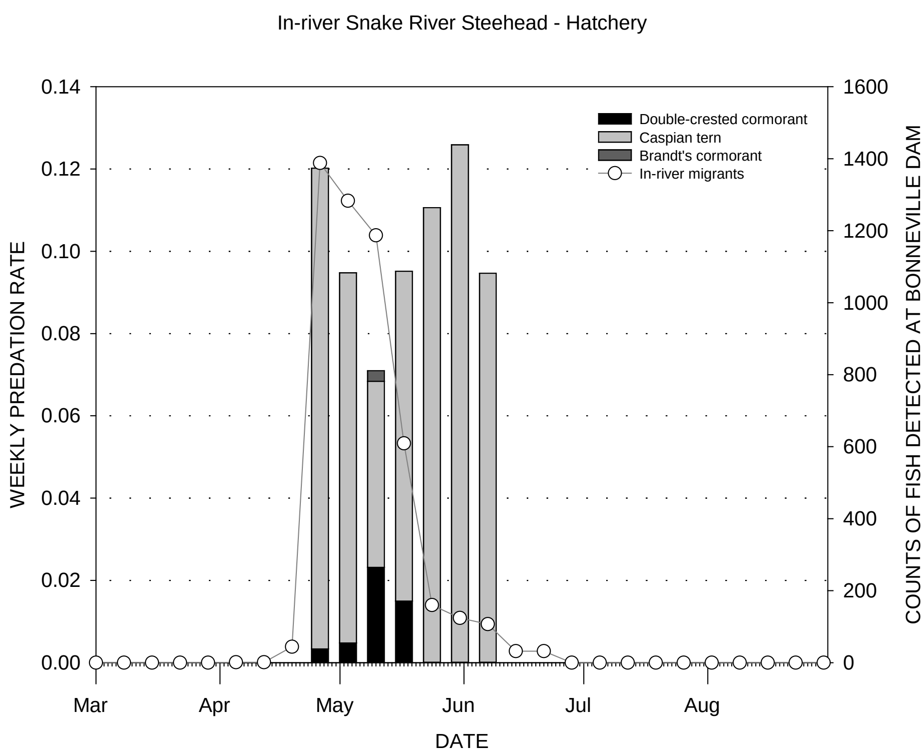
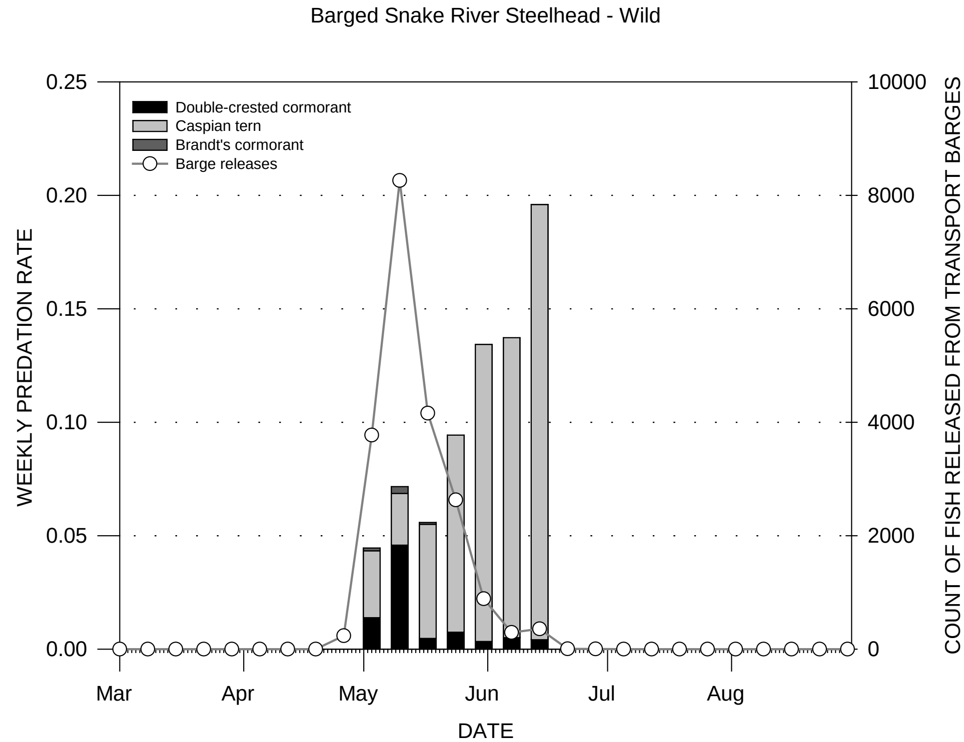


Figure 12. Weekly releases, detections, and estimated minimum predation rates for hatchery-reared Snake River steelhead, migration year 2013.



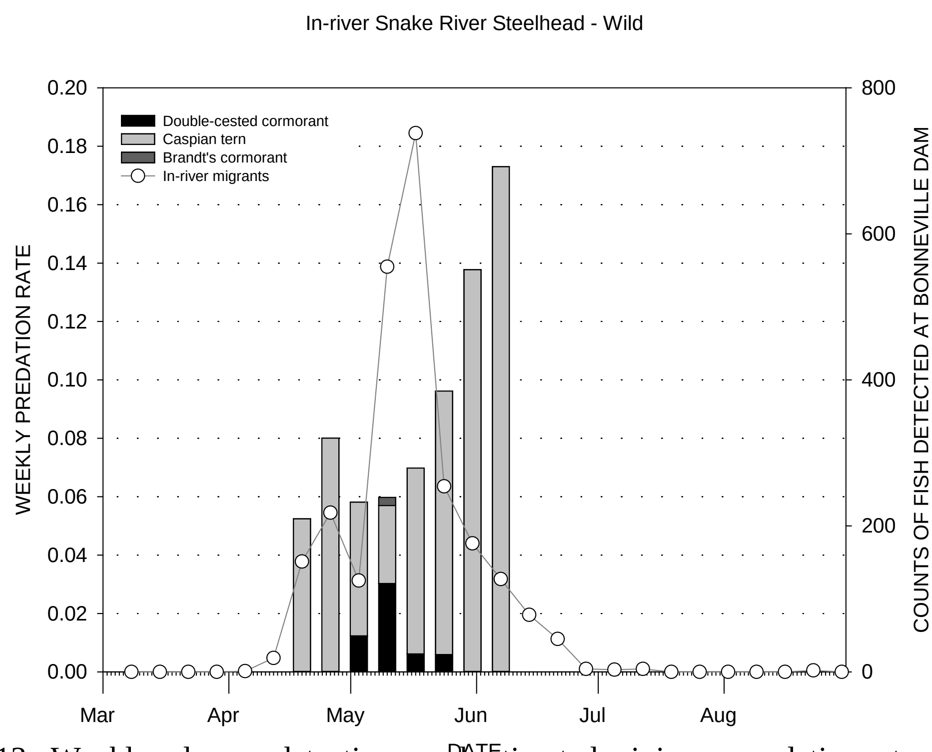


Figure 13. Weekly releases, detections, and estimated minimum predation rates for wild Snake River steelhead, migration year 2013.

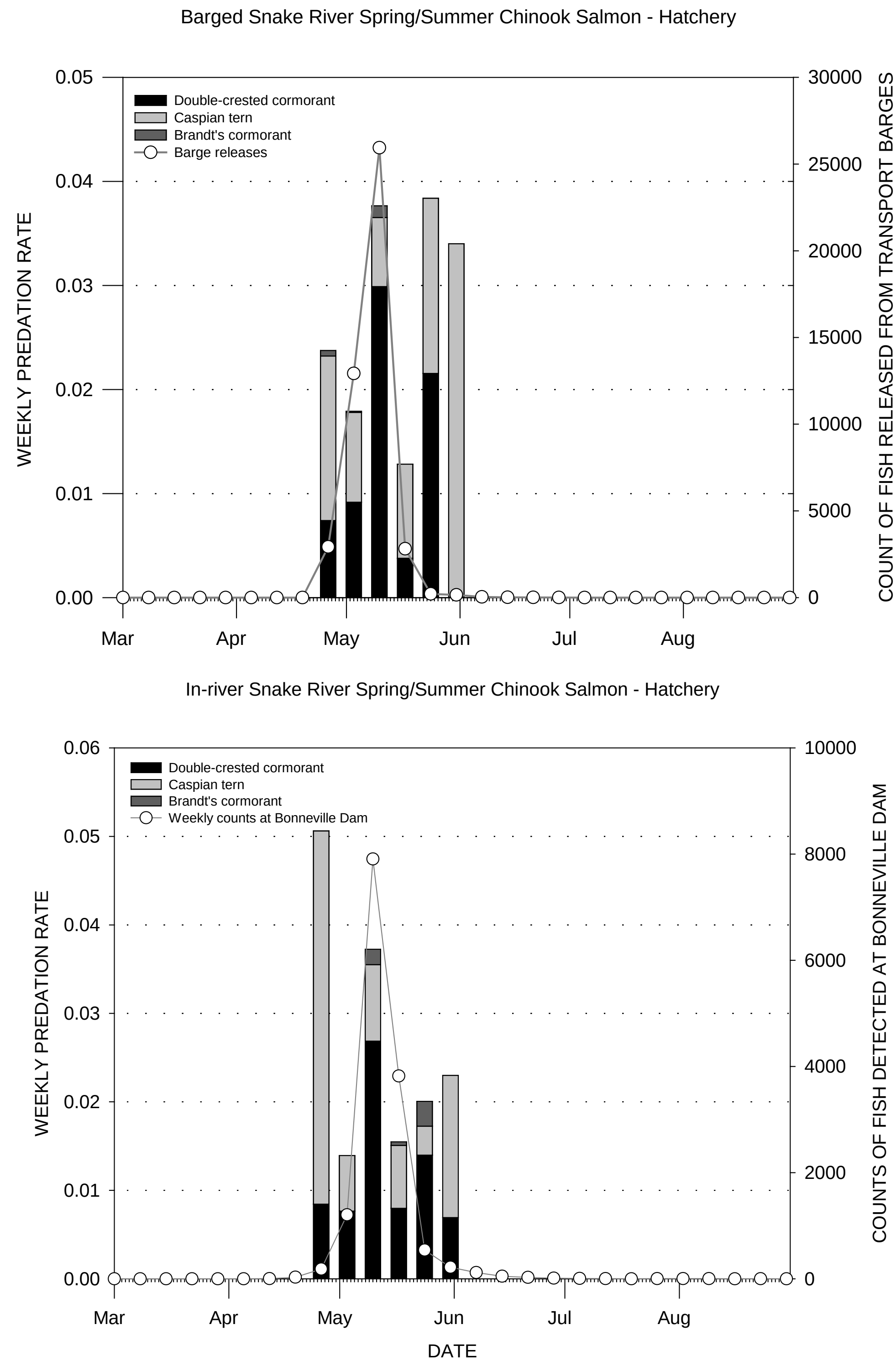


Figure 14. Weekly releases, detections, and estimated minimum predation rates for hatchery Snake River spring/summer Chinook salmon, migration year 2013.

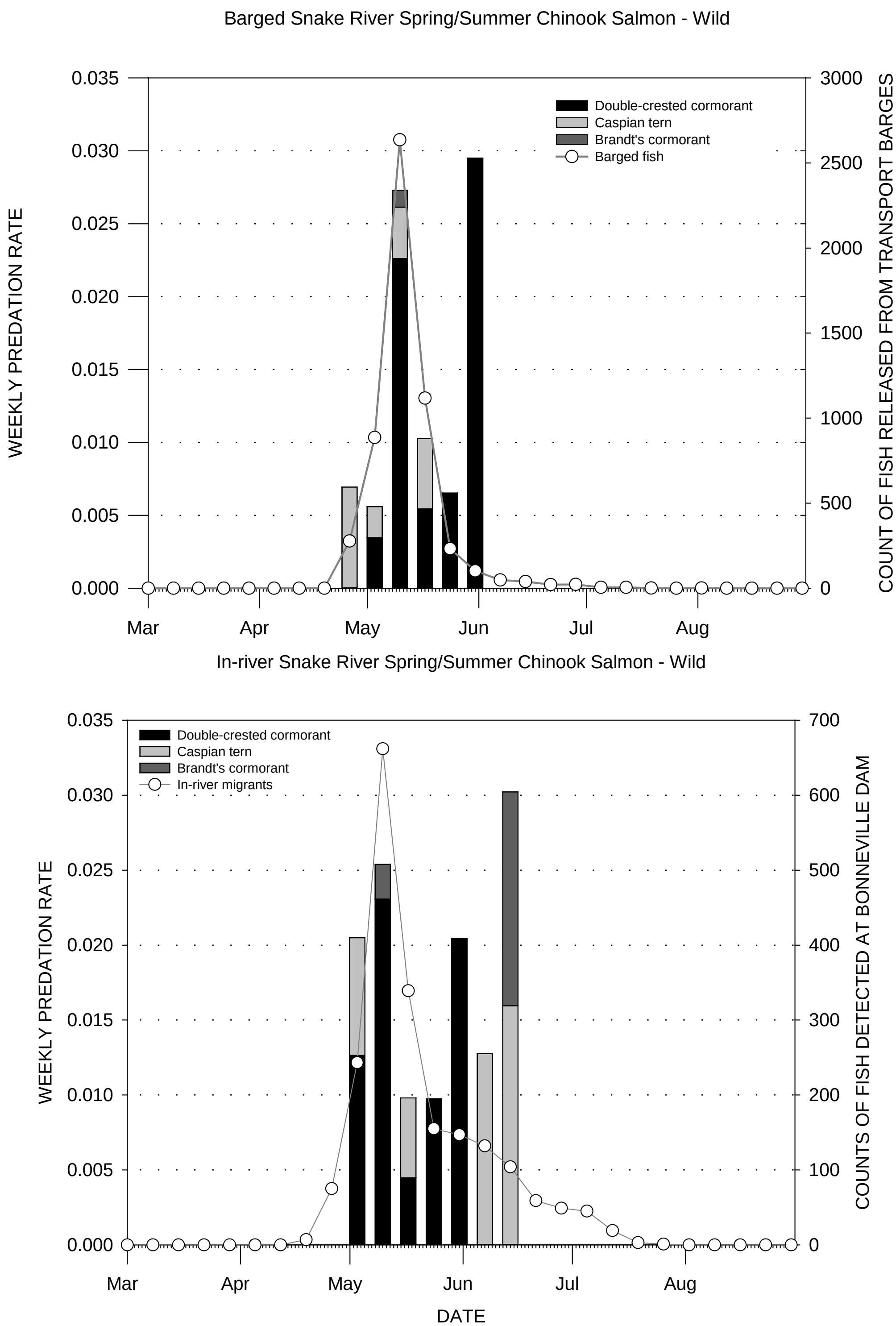


Figure 15. Weekly releases, detections, and estimated minimum predation rates for wild Snake River spring/summer Chinook salmon, migration year 2013.

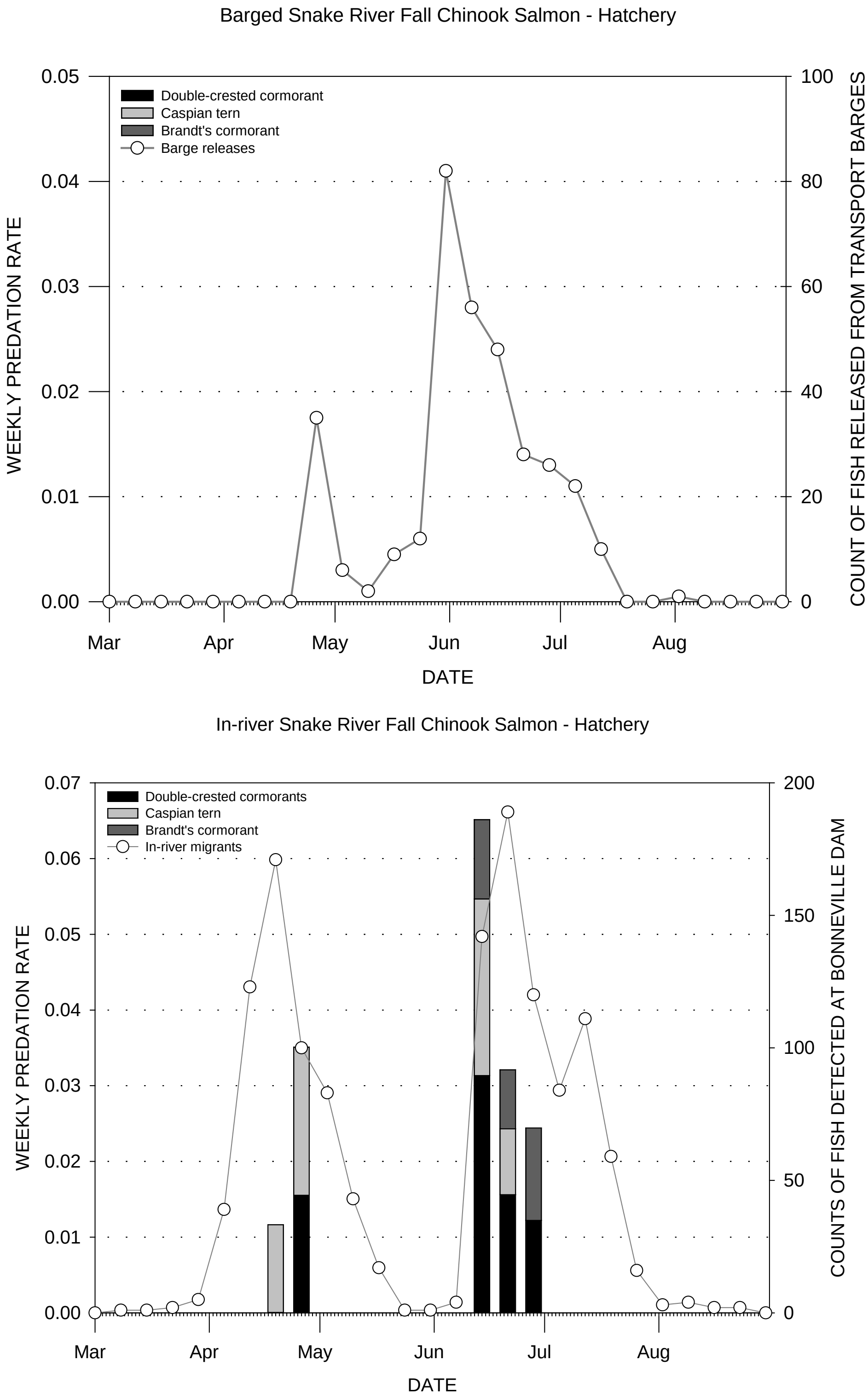


Figure 16. Weekly releases, detections, and estimated minimum predation rates for hatchery Snake River fall Chinook salmon, migration year 2013. Because no week had barge releases of ≥ 100 fish, it was not possible to estimate weekly minimum predation estimates for barged fish.

These results indicated no large overall differences in mean weekly rates between barged fish and in-river migrants. However, all of our tests had relatively low statistical power ($n \leq 8$); therefore, they would have been able to resolve only very large differences in mean predation rates. Additionally, the results of a given *t*-test do not account for any strong patterns of temporal variation in vulnerability (as opposed to availability) of barged fish compared to in-river migrants. The presence or absence of such temporal patterns was not evaluated in the scope of the present analysis.

Other qualitative trends seen in examining the observed mean minimum weekly predation rate estimates (Table 18) included the following. Mean weekly predation was highest on steelhead for Caspian terns (8.1-10.5%). Double-crested cormorant predation was similar but low for both steelhead and spring/summer Chinook salmon (1.0-1.2%). Hatchery steelhead and spring/summer Chinook salmon exhibited higher rates of Caspian tern predation than their wild counterparts. On the other hand, double-crested cormorant predation on these same groups differed by 0.2% between hatchery and wild fish. Brandt's cormorant predation estimates on steelhead and Chinook salmon groups were very low in all cases ($\leq$ 0.2%).

As seen in all other comparisons, tern predation on steelhead showed the highest predation estimates of all groups. These trends mirrored the patterns seen in estimates of annual minimum predation rates for 2013, suggesting our statistical weekly comparisons were a reasonable rough approach for assessing overall rates in the absence of a multi-year analysis.

Table 18. Mean weekly minimum estimated predation rates for barged vs. in-river migrant PIT-tagged Snake River salmon and steelhead, including *P*-values for unpaired *t*-tests, migration year 2013. Means are presented according to bird colonies where tags codes were recovered and all means were calculated from only weeks where ≥ 100 fish were released from barges or detected passing Bonneville Dam.

		Estimated mean minimum weekly predation rate (%)								
		Caspian tern			Double-crested cormorant			Brandt's cormorant		
Snake River ESU or DPS	Rear type	Barged	In-river	*P*	Barged	In-river	*P*	Barged	In-river	*P*
Steelhead	Hatchery	10.5	9.5	0.56	1.2	1.0	0.34	0.1	<0.1	0.83
	Wild	8.1	8.4	0.92	1.1	1.0	0.57	0.1	<0.1	0.60
Spring/summer Chinook salmon	Hatchery	1.5	1.2	0.64	1.2	1.0	0.77	<0.1	0.1	0.40
	Wild	0.3	0.6	0.29	1.1	1.0	0.85	<0.1	0.2	0.32

Conclusions and Recommendations

We successfully accomplished the 2013 study objectives of PIT tag code recovery and predation rate estimation. Recovery of PIT-tag codes from East Sand Island Caspian tern and cormorant colonies was completed on 13 December 2013. On the Caspian tern colony (6,394 m^{-2}, 1.58 acres), we recorded 11,860 unique codes from Pacific salmon tagged for the 2013 migration year. With hand-held PIT tag detectors, we recorded 11,020 unique codes from the 12,545 m^{-2} (3.1 acre) double-crested cormorant colony. These recoveries included tag codes from all 10 of the 13 ESA-listed anadromous salmonid ESUs or DPSs that were PIT-tagged in the Columbia River Basin. Recovery of tag codes from East Sand Island is required for calculations of estimated estuary avian predation rates on all listed groups of PIT-tagged salmon and steelhead in the Columbia River Basin.

To measure detection efficiency of our recovery efforts, Bird Research Northwest sowed 300 control tags across the tern colony, where 3 sets of 100 tags were sown before, during, and after the nesting season. Four hundred control tags were sown across the cormorant colony, with 200 sown immediately before the nesting season and 200 immediately after the nesting season finished. We recovered an estimated 63.3% of control tag codes from the tern colony and 68.5% from the double-crested cormorant colony. These recovery rates were within the range of those estimated in 2002-2012 (Caspian tern colonies: 64-95%; double-crested cormorant colonies: 35-76%; Sebring et al. 2012; Zamon et al. 2014). However, there is a trend of decreasing control tag recovery rates on the tern colony, indicating tag collisions are likely occurring in the surface stratum where recently deposited tags reside.

Maintaining high detection efficiencies is desirable for future work; therefore we recommend that action be taken to reduce the number of old tags in the surface strata. For example, deep-harrowing of the nesting substrate could be resumed to redistribute older tags below the detection range of the flat-plate antenna. It would also be possible to physically remove old tags from the colony. These activities would need to take place after the autumn PIT tag surveys, but before the placement of the next year's pre-season control tags prior to the birds returning to nest in the spring.

Our PIT-tag code recoveries in 2013 supported experiments by staff of Bird Research Northwest, whose goal was to measure on-colony tag deposition rates by double-crested cormorants. Cormorants volitionally consumed 127 PIT-tagged trout on the colony, and BRNW estimated that 60% (95 CI =47-73%) of these tags were deposited on the colony, with 40% presumably deposited elsewhere. We used these deposition data to adjust predation rates for ESA-listed groups originating above Bonneville (on the Columbia River) or Sullivan Dam (on the Willamette River). Extending deposition adjustments to other analyses that currently do not include these adjustments would make direct comparisons of predation rates across studies possible.

Sample sizes of fish detected at terminal dams were sufficient to estimate estuary predation rates for 8 of 13 ESA-listed ESU/DPS groups, as well as for two unlisted Chinook salmon populations: Middle Columbia River spring and Upper Columbia River summer/fall. Of these groups, Upper Willamette River spring Chinook salmon in general were least impacted by estuary avian predation, with predation rates of less than 1% for any bird species. For the steelhead DPSs, Caspian terns had a larger impact (8.6-12.5%) than did double-crested cormorants (1.6-2.7%). Conversely, cormorants had a larger impact on sockeye (2.6%) and on Upper Columbia and Snake River Chinook salmon ESUs (2.0-2.9%) than did terns (0.6-1.4%). Overall, Brandt's cormorants had minimal predation impacts on all ESU/DPS groups we examined (0.2% or less in all cases). These results indicate that in addition to known differences in juvenile migration timing among salmon and steelhead ESU/DPSs, there are likely differences among these fish groups in estuary migration routes and migration behavior, as several of these groups are found in the estuary at the same time.

We noted that estimated rates of predation on steelhead by double-crested cormorants were significantly lower in 2013 than in 2012 (range 3.4-7.2% in 2012 vs. 1.6-2.0% in 2013; Zamon et al. 2014). This is an interesting result because the size of the cormorant colony was larger in 2013 (14,916 breeding pairs) than in 2012 (12,301 breeding pairs). In addition, reproductive productivity of the birds in 2013 was the highest observed in the past 4 years (Roby et al. 2014). If there were a one-to-one, linear relationship between adult bird population size and predation impacts, then there should have been an increase of approximately 17.5% in predation rates on steelhead, not a reduction in predation rates of 50% or more. This result suggests that simply counting breeding pairs does not necessarily provide an accurate way to predict predation impact in any given year. It also suggests that birds must have been successfully consuming prey other than salmon to meet their reproductive needs in 2013.

Diet analysis by Roby et al. (2014), a tag-independent method of assessing predation impact, confirmed that cormorant consumption of salmon was 50% less in 2013 than in 2012. As in past years, northern anchovy *Engraulis mordax* made up a significant component of the diet for double-crested cormorants. We hypothesize that changes in the estuary and nearshore availability of non-salmonid prey, in particular northern anchovy, were likely driving the reduction in salmon predation seen in 2013.

Other studies examining predator-prey relationships and early marine survival in Columbia River salmonids present additional evidence that seasonal changes in the presence of suitable alternative prey in the nearshore ocean and estuary, especially northern anchovy, may be partly responsible variation in juvenile salmon survival in the estuary, plume and nearshore ocean (Kaltenberg et al. 2010; Zamon et al. 2013; Roby et al. 2014). Other biotic and aboitic factors (e.g., river flow or near-shore ocean conditions) may also influence juvenile salmonid susceptibility to avian predation and the proportion of alternative prey available to birds (Bottom and Jones 1990; Weitkamp et al. 2012). Additional research to explore both food-web and ecological factors in the Columbia River Estuary that affect variation in avian predation rates is warranted.

Due to the complex life history and lack of an experimental or representative tagging program for Lower Columbia River Chinook salmon in 2013, we did not estimate predation rates on PIT-tagged groups from this ESU as a whole. However, we were able to report the contribution of PIT-tagged fish groups to the Lower Columbia River Chinook salmon ESU, as well as the tag codes from those groups that were recovered from East Sand Island bird colonies. During 2013, 13 separate sources contributed a total of 97,255 PIT-tagged Lower Columbia River Chinook salmon, although only 7 of these sources accounted for 99.4% of tagged fish. Almost three-quarters (72%) of all tagged fish came from only three national fish hatcheries located above Bonneville Dam: Carson (spring migrants), Little White Salmon (spring and fall migrants), and Spring Creek (spring migrants).

We recovered 2,911 tag codes from Lower Columbia River Chinook salmon on East Sand Island, with 83.6% of these tags found on the double-crested cormorant colony. Most recoveries (78.7%) came from fish released on or before 2 May 2013, suggesting that fish from this ESU are particularly susceptible to predation by double-crested cormorants. The 2013 data are consistent with findings from earlier years of this study (Sebring et al. 2013).

Qualitative comparisons of minimum estimated annual predation for barged and in-river migrants from the Snake River showed interesting trends. For example, estimated annual predation on barged fish was lower for Caspian terns in 4 of 5 cases, higher for double-crested cormorants in 4 of 5 cases, and very low for Brandt's cormorants in all cases. This result suggests that impacts on transported Snake River fish vary by predator species. In general, hatchery fish exhibited either equivalent or higher estimated annual predation rates than wild fish. Caspian terns and double-crested cormorants both showed the largest estimated effects on steelhead.

However, quantitative comparisons of mean estimated weekly predation on barged and in-river fish did not reveal statistically significant differences in any of the

12 possible comparisons made. This means that either there were no differences in predation impacts between transported fish and in-river migrants or the test failed to detect such differences. An unpaired *t*-test is a very simple, general tool for examining differences in means between groups. In our case, the power of the tests was low due to small sample sizes, and we were unlikely to find small differences between groups. In addition, there may be some within-season temporal variation in the vulnerability of barged or in-river fish that would require a more in-depth analytical approach.

Better resolution of potentially small but biologically meaningful differences in predation rates on barged vs. in-river fish groups would require larger sample sizes as well as more in-depth statistical analysis. Increasing the sample sizes to better resolve temporal trends or any other differences between transported fish and in-river migrants would require (1) performing a multi-year analysis on barged and in-river data from previous years or (2) a focused study with tagged release groups specifically designed to measure avian predation impacts of barge transportation compared to in-river migration (c.f. Ryan et al. 2007, Marsh et al. 2008).

For Snake River in-river migrant fish, our estimated annual minimum predation rates for Caspian terns, double-crested cormorants, and Brandt's cormorants were similar but not identical to deposition-adjusted rates reported by Roby et al. (2014). Differences in estimates are due to two factors. First, our comparison of migration histories excluded Snake River fish that entered the river below Lower Granite Dam because we wanted to compare fish groups that shared the same migration history prior to barging. Roby et al. (2014) included those fish in their analysis. Second, Roby et al. (2014) adjusted their estimates for off-colony deposition as well as for daily detection efficiencies, whereas we adjusted only for the latter (deposition adjustments were not available for our migration history comparisons). Of these two factors, the deposition adjustment likely accounted for most of the difference between the two estimates; however, this adjustment would probably not have changed the outcome of our relative comparisons.

As in 2012, we recovered a significant number of tag codes from East Sand Island that had no prior detection record at Bonneville Dam during 2013. These codes were recovered from in-river migrants with origins in the Snake River (n = 18,431; Table 1) and elsewhere in the Columbia Basin (n = 7,788; Table 1). This means that in 2013, only about 7.1% of all tags recovered on East Sand Island were actually used to estimate estuary avian predation. Clearly, the remaining 92.9% of tag codes detected on East Sand Island were from fish that entered the estuary in 2013 and were consumed by birds. Until there are improvements to detection systems at or below Bonneville Dam, we are likely to continue missing important information on estuary entry timing and avian impacts to in-river migrant fish.

Overall, the 2013 data continue to support the present understanding that

1. Caspian terns have the largest impact on steelhead ESU/DPS groups, taking on the order of 8-12% of tagged steelhead from DPSs originating above Bonneville Dam;
2. Double-crested cormorants have the largest impact on salmon stocks overall, taking on the order of 1-4.6% of any given ESU in 2013, with additional impacts on the Lower Columbia River Chinook salmon ESU that are likely to be biologically significant; and
3. Brandt's cormorants do not likely have a biologically significant predation impact on any salmonid ESU/DPS group, regardless of species, run type, rearing history, or migration history.

Data also support an emerging hypothesis that there is not likely to be one overall effect of barging fish that can be applied to all species of avian predators. Avian predation patterns in 2013 show what appear to be predator-specific effects on transported fish.

The specific mechanisms governing variation in seasonal and annual predation impacts of Caspian tern and double-crested cormorants on listed groups are not yet understood. Interestingly, while the double-crested cormorant population was larger in 2013 than in 2012 and its reproductive success was the highest observed in the past 4 years, estimated predation on salmon was lower than in 2012. Mechanistically, this counterintuitive result implies that cormorants were consuming more non-salmonid prey in 2013 than in 2012. In our 2012 report, we suggested avian predation impact on a given ESU/DPS is probably a function several factors, including:

1. Physical estuary conditions during estuary entry/residence, such as temperature, salinity, flow, and turbidity.
2. Physiological and biological condition of individual fish
3. Biological conditions during estuary entry residence, such as:
    a. Distribution/abundance of prey resources for juvenile salmon
    b. Distribution/abundance of other juvenile salmon/steelhead with regard to direct or indirect density effects and as alternative prey for salmon predators
    c. Distribution/abundance of alternative prey for salmon predators, such as marine forage fish
    d. Distribution/abundance of avian predators
4. Timing of estuary residence, including date of arrival, dates of residence, and travel time
5. Migratory pathways/habitat used by fish
6. Time and locations of highest avian foraging activity

This year's decrease in cormorant predation impacts despite increases in cormorant population size and reproductive success strongly suggest that addressing how physical and biological factors affect alternate prey in particular would be a fruitful avenue to improve our understanding of factors driving variation in avian predation impacts.

A variety of established research tools for addressing distribution and abundance of non-salmon prey in the Columbia River estuary are available. Specifically, net sampling of salmon and forage fish (e.g. Weitkamp et al. 2012), combined with calibrated hydroacoustic mapping of estuary fish distributions, would be powerful tools to fill our knowledge gap in this area. Modest resources could support an initial investigation as to whether or not seasonal changes in predation on juvenile salmon were correlated with changes in river flow and relative abundance of non-salmonid prey (especially northern anchovy) in the estuary.

In addition to encouraging the investigation of mechanisms driving variation in avian predation rates, research, monitoring, and management estuary avian predation impacts would benefit from the following specific actions:

1. Researchers should strive to recover PIT-tag codes from the East Sand Island cormorant colony as soon as possible after access to the colony is granted (within 1-2 weeks). Additional qualified field personnel and hand-scanning equipment should be ready no later than mid-September. This would allow completion of two full detection passes on the cormorant colony before heavy weather begins to wash tags off the colony.
2. A consensus should be reached to standardize how and when on-colony deposition adjustments to predation estimates should be made. A recently published study strove to resolve this issue for gull predation on wild steelhead in small California watersheds (Osterback et al. 2013), and recent deposition studies on Caspian terns, double-crested cormorants, and California gulls are now available for the Columbia River Basin (Roby et al. 2014; Hostetter et al., BRNW, unpublished manuscript). Clearly, our results in Table 8 and the California study show there is a significant difference in estimated absolute impact in cases where deposition adjustments are made, with an over-all mean increase in estimated predation impact.
3. We recommend a well-designed PIT-tagging program to provide proportional representation of the diverse populations (above/below dam facilities) and life history types (spring/fall migrants, hatchery/wild rear types) within the Lower Columbia River Chinook salmon ESU.

   In the continuing absence of such a program, estimates of predation impacts should be provided for available source-specific release groups without generalizing to the entire ESU. Lower Columbia River Chinook salmon are especially vulnerable to predation by double-crested cormorants; however, cumulative avian predation impacts on this ESU remain poorly understood (Sebring et al., 2013).

4. Improvements to PIT-tag detection capability prior to estuary entry are needed for accurate assessment of migration timing, estuary avian predation impacts, and robust statistical comparisons of barge transported and in-river migration histories. During 2013, many PIT-tagged fish passed Bonneville Dam undetected, entered the estuary, and were taken by East Sand Island birds. Because we lacked previous detection data, we were able to use less than 8% of all tags recovered from migration year 2013 to estimate estuary predation rates for in-river migrant fish.

5. Resolution of within-season temporal and spatial patterns in predator-prey relationships would provide insight into when and where along the estuary migration corridor juvenile salmon are most vulnerable to predation events and the mechanisms determining why predation impacts vary from year to year and from fish group to fish group. The specific times and locations of most intensive avian foraging activity compared to the availability of salmonids and alternative prey in the estuary are not well-understood. Contemporary spatial and temporal information on both prey distributions and predator foraging activity patterns in the estuary are lacking, although results from NMFS alternate transport studies indicate that changes in temporal and spatial components of salmonid releases do make a measurable difference in salmon mortality due to estuary avian predation (Marsh et al. 2010).

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