Article type : Article

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The Abundance and Distribution of Hatchery and Naturally Produced Chinook Salmon in Columbia River Estuary Nearshore Habitat

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Running Head: Hatchery Chinook Salmon in the Columbia River Estuary

[A] Abstract

Columbia River hatcheries release millions of juvenile Chinook Salmon *Oncorhynchus tshawytscha*, to support fisheries and mitigate for historical productivity declines from dam construction and upriver habitat losses. Recent Federal Biological Opinions for Federal Columbia River Power System operations also have mandated restoration of estuarine rearing habitats to mitigate for mortality of naturally produced (NP) salmon at mainstem dams. We compared genetic stock compositions, distributions, abundances, and length frequencies of hatchery-marked and unmarked Chinook Salmon to evaluate the extent of hatchery influence on nearshore estuarine habitat use and the potential for interactions between hatchery origin (HO) and NP salmon. Cumulative Chinook Salmon releases from all hatcheries in 2010 and 2011 dictated estuarine stock composition, abundance, and life histories during the spring-summer migration peak. NP salmon from the Lower Columbia River populations, including many fry migrants, were most abundant but larger HO juveniles also frequented nearshore sites, where their total biomass often equaled or exceeded that of NP fish. Large releases of a single stock of origin (Spring Creek Group fall run) in April and May created sharp pulses in the estuary entry timing of fingerling-sized salmon and dominated abundance and stock composition at some sites. Successive releases from scores of hatcheries during spring and early summer ensured a consistent presence of HO salmon near shore, regardless of the habitat-specific residence times of individual fish. Overall, the historical replacement of diverse wild populations with fewer hatchery stocks of a narrow size range and migration timing has intensified nearshore habitat use during the spring-summer migration peak and reduced life history variation of Columbia River Chinook Salmon. Such changes could undermine the fish conservation goals of both hatchery mitigation and estuary restoration programs.

[A] Introduction

Tidal wetlands and other shallow nearshore habitats provide juvenile rearing areas (Healey 1980, 1982; Levy and Northcote 1981,1982) that contribute directly to the life-history diversity and productivity of Chinook Salmon *Oncorhynchus tshawytscha*, (Reimers 1973; Bottom et al. 2005a, 2005b) and Coho Salmon *O. kisutch* (Craig et al. 2014; Jones et al. 2014). In the Columbia River Basin, juvenile Chinook Salmon occupy both saline and tidal fresh regions of the estuary year-round (Roegner et al. 2012), and many fry ($\leq$60 mm FL) and fingerlings (subyearlings > 60 mm FL) utilize wetland channels and other shallow, nearshore habitats to feed and grow before entering the ocean (Lott 2004; Roegner et al. 2010; McNatt et al. 2016). Historically, the Columbia River accounted for the world's largest runs of Chinook Salmon (Van Hyning 1973) but current annual production of this and other salmon species has declined to a fraction of the estimated 11 to 16 million adults that once returned to the basin (Northwest Power Planning Council 1986). The extirpation of hundreds of Columbia River salmon populations and the listing of 13 salmon evolutionarily significant units (ESUs; Myers et al. 1998)—conservation units recognized under the Endangered Species Act—has focused increasing attention on the estuary's role in salmon decline and recovery (Kareiva et al. 2000; Bottom et al. 2005b; Fresh et al. 2005; Gustafson et al. 2007). Salmon access to two-thirds or more of the estuary's historical shallow-water rearing habitat has been lost to diking, shoreline development, and flow regulation (Thomas 1983; Kukulka and Jay 2003; Marcoe and Pilson 2017). Recent Federal Biological Opinions have mandated estuary restoration to help mitigate for continued mortality of salmon associated with Federal Columbia River Power System operations (NMFS 2008, 2010, 2020). This mandate takes for granted that estuary restoration will improve survival of naturally produced salmon despite continued releases of millions of hatchery-reared salmon that also congregate in the estuary. To better understand the potential interactions between hatchery origin (HO) and naturally produced (NP) juvenile Chinook Salmon, we compared the abundance, distribution, life history, and genetic stock composition of hatchery-marked and unmarked juveniles in nearshore habitats of the Columbia River estuary, defined as the entire area of tidal influence from the river mouth (rkm 0) to Bonneville Dam (rkm 234) (Simenstad et al. 2011).

Columbia River hatcheries have produced salmon for recreational and commercial harvest and to offset basin-wide productivity declines for more than a century (Hilborn 1992;

Bottom 1997; Lichatowich 1999; Lichatowich et al. 2006). In 1938 the Mitchell Act established a salmon restoration program that included hatchery facilities to replace freshwater habitat and salmon production losses caused by a series of mainstem dams that eliminated or impeded fish passage to much of the basin, including large areas of the Snake River and upper Columbia River (NRC 1996; Bottom et al. 2005b). When efforts largely failed to relocate interior Chinook Salmon stocks (primarily spring run fish) to tributaries below Grande Coulee Dam (rkm 960), the Lower Columbia Fishery Development Program expanded the production of Lower Columbia River stocks (primarily fall run fish) “to build up the existing runs in the lower basin to the maximum capacity of the streams utilized (Laythe 1950).” Since 1946 as many as 20 Mitchell Act hatcheries have released salmon (*Oncorhynchus* spp.) and steelhead (*O. mykiss*) continuously (NMFS 2014). Throughout the Columbia River 82 federal, state, and tribal hatcheries now support salmon mitigation and other fish conservation goals and release on average ~140 million salmon and steelhead annually or approximately 3,700 to 6,000 metric tons (Flagg 2015). Average annual hatchery releases have declined only ~5% in the years since these totals were reported (Pacific States Marine Fisheries Commission 2019).

The majority of the adult salmon now returning to the Columbia River basin are hatchery produced, with estimates ranging from 80% (Lichatowich et al. 2006) to between 65% and 75% (NMFS 2011). Chinook Salmon, considered among the most estuarine dependent of salmon species (Healey 1982) and a major target of habitat restoration under the Columbia Estuary Ecosystem Restoration Program (Thom et al. 2013), currently accounts for nearly 75% of all the hatchery fish released in the Columbia River basin (NMFS 2014; Pacific States Marine Fisheries Commission 2020). Hatchery programs often are cited as a factor contributing to natural salmon population declines (NRC 1996; Lichatowich 1999; Williams 2006; Paquet et al. 2011), but the estuarine distributions of HO salmon or their effects on the habitat use or performance of NP juveniles in the estuary have not been investigated.

Ecological or behavioral interactions with hatchery juveniles could have negative consequences for NP salmon populations that are the focus of the estuary restoration program. Density-related limitations to salmon foraging or growth have been reported in Pacific coastal estuaries (Reimers 1973; Neilson et al. 1985; Korman et al. 1997) and in Puget Sound (Simenstad et al. 1982; Simenstad and Salo 1982; Beamer and Larsen 2004; Greene and Beamer 2012) and often have raised concerns about the salmon carrying capacities of estuaries in basins with large hatchery programs (Reimers 1978; Levings et al. 1986; Korman et al. 1997; ISAB 2015). Attraction of avian and marine mammal predators to large concentrations of hatchery smolts in estuaries has been blamed for density-related declines in the productivity of Oregon coastal Coho Salmon populations (Nickelson 2003) and may account for a negative relationship between the survival of wild Snake River Chinook Salmon (from the upper Columbia River basin) and releases of hatchery steelhead (Levin and Williams 2002). Large densities of hatchery fish could undermine salmon recovery efforts in basins already constrained by considerable reductions in estuarine rearing habitat (Marcoe and Pilson 2017; Brophy et al. 2019; Greene et al. 2021). In Oregon's Salmon River estuary, for example, wetland restoration contributed few adult survivors to the local Coho Salmon population until a decades-old hatchery program also was discontinued (Jones et al. 2014; Jones et al. 2018).

Genetic or phenotypic differences between HO and NP salmon could have negative consequences beyond the effects of fish density (Flagg et al. 2000; Einum and Fleming 2001; Jonsson and Jonsson 2006). Compared to their wild counterparts, for example, propagated salmonids often exhibit increased levels of aggression (Swain and Riddell 1990; Rhodes and Quinn 1998; Peery and Bjornn 2004; Jonsson and Jonsson 2006), and displacement or mortality of wild salmon following the releases of large sized hatchery-reared juveniles has been reported in some stream environments (Nickelson et al. 1986; Peery and Bjornn 2004). To minimize negative interactions with wild fish, hatcheries often are encouraged to restrict the numbers of fish released and to insure spatial and temporal segregation from wild fish (Flagg et al. 2000; Nickelson 2003; Chilcote et al. 2011; Kostow 2012). To the extent such interactions are unavoidable, however, others recommend that hatcheries release fish at size ranges similar to those of NP salmon to minimize a potential competitive advantage for HO juveniles (Einum and Fleming 2001; Kostow 2009).

Hatchery practices in the Columbia River basin vary by program, species, and stock but most juveniles are released at relatively large sizes to encourage rapid seaward migration and to enhance survival (Kostow 2009). Because estuarine residence times and habitat selection by Chinook Salmon tend to vary with fish size (Levings et al. 1986; Healey 1991; Bottom et al. 2005b; Campbell 2010), hatchery production of a narrow range of large phenotypes could concentrate estuarine habitat use in time and space, artificially constraining the estuary's salmon-rearing capacity (Bottom et al. 2005b). Tagging studies using acoustic transmitters have measured rapid seaward migration by large subyearling and yearling salmon (including many hatchery-reared juveniles), primarily through deep estuary channels (e.g., McMichael et al. 2011; Harnish et al. 2012; McNatt et al. 2016). Lower-estuary purse seine surveys found that >90% of the Chinook and Coho Salmon and steelhead in deep-water habitats are of hatchery origin (Weitkamp et al. 2012). Hatchery fish also have been collected in some shallow, nearshore areas (Roegner et al. 2012, 2016; Sather et al. 2016) but the combined effects of all hatchery releases on the densities, stock composition, and size characteristics of salmon in these habitats is poorly understood. It is unknown, for example, whether large size at release effectively segregates HO smolts from NP juveniles in most shallow estuarine nursery areas or whether significant overlap enables hatchery fish to dominate or displace smaller NP juveniles (e.g., Nickelson et al. 1986). The potential benefits of restoring estuarine habitat for at-risk Columbia River salmon thus could depend both on the total densities of hatchery salmon that occupy the estuary at any one time as well as the relative phenotypic similarities between HO and NP juveniles where their distributions overlap.

Until 2008 it was impossible to distinguish most HO and NP juveniles in estuary surveys because only modest proportions of hatchery salmon were marked. More recently, however, >90% of all hatchery juveniles have received an identifying fin clip or tag prior to release, allowing identification of most (though not all) hatchery fish in estuary collections. This paper takes advantage of these increased hatchery marking rates to compare nearshore habitat use by HO and NP juvenile Chinook Salmon and evaluate the implications for estuary restoration and hatchery production programs. Our analyses address the following objectives:

(1) Summarize the estuary-wide contributions of HO and NP juveniles to the genetic stock composition and size distribution of juvenile Chinook Salmon.

(2) Compare the spatial and temporal distribution of HO and NP Chinook Salmon overall and among the major genetic stock groups throughout the estuary.

(3) Estimate the nearshore densities and biomass of HO and NP salmon across the estuary.

(4) Determine whether large hatchery releases modify habitat-specific salmon abundance, genetic stock composition, and life history composition (size variation, timing, and estuarine residency).

[A] Methods

We analyzed hatchery release records, Bonneville Dam fish passage data, and recent field survey results to determine the extent of hatchery influence on Chinook Salmon stock composition and nearshore habitat use in the Columbia River estuary. The data sources, field methods, and analyses are briefly described below.

[C] *Hatchery releases and fish passage at Bonneville Dam.—*We compiled hatchery-release records to compare the cumulative proportions of all Chinook Salmon hatchery stocks released in the Columbia River basin with the stock proportions observed during field surveys of shallow estuarine habitats (Objective 1). For the hatchery records we accessed the Regional Mark Information System database (Pacific States Marine Fisheries Commission 2019) to compile data on releases of juvenile Chinook Salmon from Columbia River Basin hatcheries in 2010 and 2011 (Table 1). Release data were summarized by Chinook Salmon ESU. Hatcheries apply several types of marks to fish that can be used to differentiate HO and NP fish. However, not all released fish are marked. For our study, we considered fish with a clipped fin or implanted with a coded wire tag (CWT) to be marked.

We tracked the timing of estuary entry for upriver releases of hatchery subyearlings and yearlings based on the Bonneville Dam Fish Passage Index reported by the Columbia River DART for 2009 - 2011 (Data Access in Real Time) (University of Washington 2015). The passage index is measured at a single fish bypass that automatically diverts fish every 10 minutes to a sampling facility below the dam. The index is an indicator of run timing and relative magnitude but not the actual number of fish passing the dam since passage efficiency is not equal across the entire face of the dam and varies with river flow. Because Bonneville Dam marks the upper limit of tidal influence on the Columbia River, the Fish Passage Index depicts the estuary entry of all mainstem migrants (HO and NP). The index does not distinguish HO from NP fish. However, we used the index to determine the estuary arrival of Spring Creek stock hatchery fish, which are released weeks or months earlier than subyearling Chinook Salmon from other hatcheries above Bonneville Dam and has an unambiguous signal in the estuary (Pacific States Marine Fisheries Commission 2019). The passage of Spring Creek fish over the dam provided an indicator of hatchery influence on the life history composition of Chinook Salmon as defined by estuary entry timing and size characteristics (e.g., the initial arrival of fingerling sized salmon) (Objectives 1 and 4).

[C] *Field surveys.*—We synthesized data from three beach seining surveys to determine the relative use of nearshore, shallow-water (0 – 6m) habitats by HO and NP juvenile Chinook Salmon (Figure 1). The most spatially extensive dataset was a bimonthly salmon Genetics Survey from March 2010 to March 2012 (Teel et al. 2014). The Genetics Survey quantified the distributions of 11 regional Chinook Salmon stock groups across 6 of the 8 hydrogeomorphic reaches (C – H; Simenstad et al. 2011) of the estuary (Figure 1). The analyses here report collections from two habitat types representing a range of nearshore environments to characterize genetic stock diversity within and among estuary reaches: a relatively exposed, nearshore site along the estuary's main channel ("mainstem" habitat) and a low-velocity, off-channel site on the backside of an island ("backside" habitat). Across the full length of the estuary, nearshore sites encompassed a range of topographies, slopes, sediment types, and depths (< 6m). Columbia River flows in 2010 and 2011 followed a typical seasonal pattern—increasing in the spring to a May or June peak and declining in July to the fall minimum—but the larger spring freshet in 2011 started a month earlier (in mid-May) and continued until mid-July. To summarize the estuary-wide genetic stock composition (Objective 1) and distribution of HO and NP Chinook Salmon (Objectives 2 and 3) we supplemented the bimonthly Genetics Survey results with the relevant 2010-11 catch data from a single long-term monitoring site (mainstem habitat only) at Point Adams Beach (rkm 20) in Reach A (Figure 1) (Roegner et al. 2012; Teel et al. 2014). Hereafter, "estuary-wide" results refer to the combined bimonthly data from the Genetics Survey and the Point Adams Beach monitoring (described below).

For most sites and time periods during the Genetics Survey we collected fish with a 3- x 38-m variable-mesh bag seine (10.0- and 6.3-mm wings, 4.8-mm bag). However, when seasonally high current velocities and water elevations intermittently prevented deployment of the bag seine at upper-estuary sites (i.e., Reaches G and H), we walked a 1.8-m x 4.6-m pole seine (4.6-mm mesh) alongshore to collect juvenile salmon. We collected no March samples in reaches G and H in 2010, but added an additional survey date during the first week of April 2012 to provide comparable seasonal data for the six estuary reaches for two full years. We substituted the April 2012 survey data for the missing Reach G and H data to approximate salmon abundance and stock composition for March 2010.

Besides extending the spatial coverage of the bimonthly genetics dataset to include Reach A, the more frequent Point Adams Beach monitoring also provided an index of the outmigration timing, stock composition, and size characteristics of HO and NP Chinook Salmon near the estuary mouth (Figure 1; Roegner et al. 2016). We used the biweekly collections at Point Adams Beach to estimate the effects of large hatchery release groups on habitat-specific abundance trends near the river mouth and to estimate the ocean entry timing and overall estuary residency of the selected hatchery release groups (Objective 4). We sampled Point Adams Beach biweekly (March (2011) or April (2010) – July) to monthly (all other months) with a tapered, 3- × 50-m variable-mesh beach seine (19.0, 12.7, 9.5 mm) and a 1.0-cm knotless-mesh bunt. After anchoring one end and stacking the net on the beach, the other end was towed downstream with a small boat, forming a semi-circle along the shoreline. Both ends of the net were then pulled to shore by hand, and all fish were guided into the bunt to sort, identify, and enumerate.

A 2009 fish survey at Hayden and Sauvie Island near the Willamette and Columbia River confluence (rkm 163) sampled juvenile Chinook Salmon monthly from February to June (Figure 1) (Sol et al. 2009). Sample collections from the six Hayden Island Survey sites approximated the estuary arrival timing for marked and unmarked Chinook Salmon from the mid- and upper basin (i.e., fish above Bonneville Dam) and from the Willamette River. We analyzed these data to evaluate the effects of large upriver releases of HO Chinook Salmon on the habitat-specific abundance, genetic stock composition, and life history composition of juveniles in the upper estuary (Objective 4). Fish samples from the Hayden Island Survey were collected with a 37m Puget Sound bag seine (tapered 0.91 to 2.4 m in the center) with a knotless mesh, stretched measurement of 12.7mm for the wings and 0.31mm for the bag. The net was deployed by extending one tow line from shore and setting the net from the bow of the boat parallel to shore. The opposite tow line was returned to shore and retrieved in a U-shape, funneling all fish toward the center bag.

During all three field surveys, we measured, weighed, identified hatchery marks (i.e., fin clips) and scanned for CWTs each of the first 30 juvenile Chinook Salmon; measured and checked the mark status for the next 31-100 individuals; and counted all remaining individuals in each sample. We collected fin tissue samples from up to 30 juvenile Chinook Salmon for genetic stock identification. Additional replicate beach seine hauls were made at sampling sites as needed to approximate the 30 fish target for genetic analysis. When total catches exceeded 30 fish, all fish were counted and individuals were selected randomly for genetic sampling regardless of size or presence of a hatchery mark. In sample collections with more than 100 individuals, we estimated the total number of marked and unmarked fish based on the proportions counted in each subsample. We assigned age categories (subyearling vs. yearling) for juvenile Chinook Salmon according to standard biweekly to monthly length frequency cutoffs validated from previous beach seining surveys in the lower estuary (Bottom et al. 2011). The cutoff size for yearlings ranged from > 80 mm on March 1 to >145 mm after July 15.

For the 2010-11 estuary-wide survey data we used a linear regression model comparing measured lengths to weights of fish to assign weights to individuals that were measured only. We first transformed measured weights and lengths by $\log_{10} + 1$ to linearize the observed relationship and then modeled weight as a function of length, year and mark status, and their two-way interactions. We compared models with all combinations of these variables using AICc (Burnham and Anderson 2002) to find the best and most parsimonious model. Based on this analysis, we pooled years but used separate regression models to estimate the weights of marked and unmarked individuals because their length:weight relationships were significantly different (Table S.1). For individuals that were neither measured nor weighed, we assigned the mean lengths and weights from other marked or unmarked fish captured during the same sampling event. After each fish was assigned a length and weight, the biomass of marked and unmarked groups was estimated by dividing the mean weight by the fish density.

Approximately 9% of all the Chinook Salmon were released from hatcheries without an identifiable mark in 2009-11. Since the actual marking rates varied by hatchery, and stock of origin was unknown for all but a random subsample of fish chosen for genetic analysis, we made no attempt to adjust the unmarked fish totals to estimate the true number of hatchery salmon in our sample collections. The results for marked fish reported in this analysis thus should be viewed as minimum estimates of the number, biomass, or proportion of HO Chinook Salmon.

[C] *Genetic stock of origin.*—We used the genetic stock identification (GSI) methods described by Teel et al. (2014) to estimate the origins of juvenile Chinook Salmon collected during each of the estuary surveys. Accordingly, the terms “stock” or “genetic stock” herein refer to the groups delineated directly from the genetic data. Briefly, GSI was conducted using 13 Chinook Salmon microsatellite DNA loci and genotypic data that were standardized by several west coast salmon genetics laboratories (Seeb et al. 2007; Moran et al. 2013). Genomic DNA was extracted from fin tissues taken during field surveys using Wizard Genomic DNA Purification Kits (Promega Corp.). Polymerase chain reactions (PCRs) were then used to amplify the microsatellite loci and genotyping was conducted using the Applied Biosystems Inc. (ABI) 3700 genotyping platform. The GSI computer program ONCOR (Kalinowski et al. 2007) and a "baseline" representing 45 populations were used to assign individual fish to one of 11 regional genetic stocks (Teel et al. 2014). Fish with relative assignment probabilities < 0.90 were excluded from the analysis to improve the overall stock assignment accuracy in the data (Teel et al. 2014). The analysis included a coastal region genetic stock to identify estuary-caught juveniles originating from outside the Columbia River basin (Roegner et al. 2012). The 10 Columbia River basin genetic stocks were generally congruent with the basin's Chinook Salmon ESUs (Table 1). However, because of the lack of genetic differentiation between the Mid- and Upper Columbia River Spring ESUs, genetic assignments were made to a single Mid-Upper Columbia River spring stock. In addition, three genetic stocks were distinct within the Lower Columbia River ESU and assignments were made to West Cascade fall, West Cascade spring, and Spring Creek Group fall stocks. The West Cascade fall and spring stocks include natural spawning and hatchery fish originating in several tributaries to the lower Columbia River and estuary. The Spring Creek Group fall stock is native to tributaries to the Columbia River gorge, upstream of Bonneville Dam. However, the stock's current distribution also includes populations further downstream resulting from the use since 1901 of fish and eggs from Spring Creek National Fish Hatchery to stock hatcheries and rivers throughout the lower Columbia River (Myers et al. 2006). Assignments to the three stocks were also summed to represent the entire ESU, consistent with the broad conservation and recovery framework applied under the Endangered Species Act. Chinook Salmon from the non-native Rogue River fall genetic stock do not belong to one of the basin's ESUs, because those fish are descended from stock transfers to Columbia River hatcheries from southern Oregon coastal hatcheries beginning in the 1980's (North 2006). We included Rogue River fall genetic stock in our GSI analyses, because descendants from those transfers reproduce in some lower river tributaries (Roegner et al. 2010) and are also released from Columbia River hatcheries to support fisheries in off-channel areas of the lower river (North et al. 2006; Table 1).

[C] *Numerical and biomass densities.*—We estimated the numerical and biomass densities of marked and unmarked salmon for each site and date sampled during the Genetics Survey and the Point Adams Beach monitoring. Gear efficiencies and sample areas were not measured during each sampling event and were undoubtedly influenced by changing hydrological conditions. We estimated salmon densities by the areas swept for each gear type and survey method. A standard set for the survey at Point Adams Beach was a semi-circular haul of the 50-m beach seine, for an estimated area of ~397 m^2. The Genetics Survey employed both "standard" and "towed" beach seine hauls to collect individuals for genetic analysis. A standard semi-circular haul of the 38-m seine enclosed ~218 m^2. When Chinook Salmon abundance was too low to adequately meet the genetics sample goal (n = 30) using the standard-haul method, we towed the beach seine for ~38 m before pulling it to shore, sampling an area approximating twice the area of a standard set (total ~436 m^2). Finally, in the upper-estuary reaches G and H water velocities and elevations were sometimes too high to effectively sample nearshore habitats with the beach seine. For those periods and locations, we walked a 7.6-m pole seine in an arc sampling approximately ~39 m^2 for each pole-seine effort. We recorded the total sampling effort for each sampling event.

For the Genetics Survey we investigated the impact of factors Year (2010/2011), Month (May/July), Location (Mainstem/Backside), and Rearing Type (marked/unmarked) on the mean Chinook Salmon density and biomass for the six estuary reaches (C – H) where both mainstem and backside habitats were sampled. We considered reach as a random rather than a fixed factor. The reaches were a collection of random units of all possible units within the estuary, and each unit contained responses for all levels of all the fixed factors. Therefore, the variability among the reaches could be accounted for in the model and removed for fixed factor comparisons. We compared linear regression models of all combinations of these factors and all two-way interactions between them with reach included in all models. We used AIC to rank the models and chose the model with minimum AICc as the best model unless other models were within 2 of the best. If multiple models were included from this process, we then applied weighted averaged estimates across those models (Burnham and Anderson 2002).

[A] Results

[B] Estuarine stock composition and size characteristics of marked and unmarked salmon

Annual releases of hatchery Chinook Salmon in 2010-11 averaged 105 million fish—approximately 75% of all hatchery salmonids released in the basin—with ~2/3 released as subyearlings and ~1/3 as yearlings (Table 1). More than 87% of all hatchery subyearling Chinook Salmon and 98.5% of all yearlings received an identifiable mark or tag prior to release. Two stocks with large subyearling releases were marked at relatively low rates—Upper Columbia River summer/fall (UCR-su/f; 73.9%) and Snake River fall (SR-f; 64.5%). In total, Lower Columbia River (LCR) stocks accounted for nearly 40% of all salmon released from Columbia River hatcheries (Table 1, Figure 2a-b). With the exception of UCR-su/f (25%) and Snake River spring (SR-sp; 13%), each of the remaining Mid and Upper Columbia River stocks constituted less than 10% of the total hatchery production in 2010-11.

Together the three LCR stocks—West Cascade fall (WC-f) (44%), West Cascade spring (WC-sp) (4%), and Spring Creek Group fall (SCG-f) (18%)—were responsible for two-thirds of the juvenile Chinook Salmon sampled estuary-wide in 2010-11 (Figure 2c). Most of the remaining fish were identified as UCR-su/f (22%) and Upper Willamette River spring (UWR-sp) (7%). Combining all survey dates and sites, unmarked fish were most prevalent but the three most abundant stocks included significant proportions of marked hatchery fish: LCR (34%), UCR-su/f (23%), and UWR-sp (42%) (Figure 2d). Deschutes River hatcheries released no fall Chinook Salmon in 2010-2011 (Figure 2c). The 19% (n=8) of DR-f stock in estuary collections listed as hatchery marked (Figure 2d) are most likely a mis-assignment of UCR su/f stock, which is genetically similar to DR-f (Teel et al. 2015).

Marked and unmarked juvenile Chinook Salmon exhibited distinct estuary-wide size distributions (all months and at all stations combined). The pattern for marked juveniles was bimodal, with a near-normal size distribution centered around ~80 mm FL and a smaller peak at ~120mm FL (Figure S.1). Few marked fish <60mm FL were captured in the estuary. Unmarked fish exhibited a much wider size range and a highly skewed size distribution with a peak at ~40 mm FL, ranging upwards to 120 mm FL.

The bimonthly size distributions and temporal trends also differed among marked and unmarked juveniles (Figure 3). Groups of marked yearling fish (~120mm FL) were collected in March (Figure 3), and marked subyearlings were prevalent in the May and July collections. The size distributions for marked salmon were nearly identical during the May and July surveys (median ~ 80mm FL). Few marked fish were captured in the fall. Unmarked subyearling salmon were present in the estuary March through November and exhibited a wider range of sizes (30 – 120 mm FL) than marked fish (60 – 120 mm FL) (Figure 3 and S.1). In contrast to marked juveniles, the size distributions for unmarked subyearlings increased with each bimonthly sampling period from March through November and may reflect continued growth of estuary-resident juveniles, size-selective mortality of smaller juveniles, the arrival of larger fish from upriver, and a combination of these factors.

[B] Spatial and temporal distribution of marked and unmarked salmon

Use of shallow-water habitats by marked hatchery fish varied considerably among sampling sites in 2010-11 (Figure 4). More than half of the fish sampled in Reach A and more than 30% of the fish collected in reaches E and F (near the Willamette River confluence or immediately downstream) were hatchery marked. The proportions of marked fish were slightly higher along the mainstem channel, averaging 28% compared with 20% for fish collections in the backside habitats. The peak proportions of marked fish in spring and summer reflected hatchery rearing schedules with most hatchery releases occurring from May to July. In both years the relative abundance of marked fish in shallow habitats peaked between 30% and 40% in July (Figure 4).

The relative contributions of Chinook Salmon genetic stock groups in the estuary varied geographically in 2010-11 (Figure 5). Lower Columbia River stocks (SCG-f and WC-f) were widely distributed, but the proportional abundance of WC-f was greatest in the mid-estuary (Reaches C – E) (Figure 5). The proportion of UWR-sp stock in nearshore habitats peaked near the Willamette R. confluence (Reach F) and declined downstream while the percentage of UCR-su/f was greatest in upper estuary reaches, G and H. The influence of hatchery programs on estuary-wide distributions and proportional abundances also varied by salmon stock. Hatchery programs increased the proportional abundances of SCG-f and UWR-sp in nearshore habitats but made relatively smaller contributions to the total abundance of WC-f and UCR-su/f stocks. The low proportion of marked UCR-su/f subyearlings at estuary survey sites may reflect the relatively low hatchery marking rate (74%) for this stock, the abundance of UCR-su/f fish naturally produced in an undammed stretch of the mainstem Columbia River (i.e., Hanford Reach), or a combination of factors.

[B] Nearshore density and biomass

Estuary-wide peak densities and biomass of Chinook Salmon ranged from .07 to 1.35 fish/ m^{-2} and 0.15 – 5.18 g m^{-2} in May 2010 and from .05 to 1.81 fish/ m^{-2} and 0.16 to 3.89 g/ m^{-2} in May 2011 (Table S.2). Other salmon species, including yearling Coho and subyearling Chum Salmon, also were present near shore in the spring but occurred less frequently at most sites and were not included in our density estimates. Throughout the peak of the May-July salmon migration in 2010 and 2011, marked hatchery fish averaged more than a third of the numerical density and about half of the biomass density of Chinook Salmon sampled near shore (Table S.2).

Marked individuals accounted for 39% to 72% of the Chinook Salmon biomass sampled at all nearshore locations and more than half of the total biomass in 8 of the 13 habitats surveyed (Figure 4). Large yearlings dominated the catches of marked fish in March (Figure 3) and contributed almost 90% of the Chinook Salmon biomass in the estuary in March 2010 and almost 60% of the biomass in March 2011 (Figure 4). The biomass contributions from subyearling hatchery Chinook Salmon peaked in May at >50% and declined with each bimonthly sampling event thereafter.

The biomass densities of marked and unmarked Chinook Salmon followed similar temporal patterns in most estuary reaches, with bimonthly peaks in May or July and low values during the subsequent September and November surveys (Figure 6). Total biomass density estimates for marked and unmarked salmon in reach H were relatively low during spring and summer months in 2010 and 2011 and in reaches E, F, and G in 2011.

The linear regression modeling of factors affecting density showed that the model with month, mark status and their interaction better explained variation in mean density than all other model combinations (i.e., change in AIC >> 2.0, except for the same model with location added for which AIC = 1.8; we chose the smaller model, Table S.3). Density of unmarked individuals in May was much higher (0.39 fish/m^2, 95% CI (0.35,0.54)) than marked (0.14 fish/m^2, 95% CI (0.04, 0.23) (Table S.4). Densities decreased in July and were similar for marked and unmarked fish (0.06 and 0.11 fish/m^2, respectively). Bimonthly densities of subyearlng Chinook Salmon broadly paralleled the seasonal hydrograph, which increased through the spring to a May or June peak in both years and declined in July to a fall minimum. Differences among years and habitat locations were small.

Similar linear regression modeling of factors affecting biomass showed that the model with only month best explained the variation in mean biomass (i.e., change in AIC > 2.0 for all combinations except the two models adding habitat location and year individually (AIC = 1.48 and 1.49, respectively), so we chose the parsimonious month-only model, Table S.5). Biomass in May was higher than in July (0.75g m^{-2}, 95% CI (0.55, 0.96) versus 0.36g m^{-2}, 95% CI (0.15,0.56) (Table S.6). Differences between years, habitat locations, and marked/unmarked group were not significant.

[B] Habitat-specific abundance, size characteristics, and residency of large hatchery release groups

Large releases of hatchery fish above Bonneville Dam directly influenced the estuary entry timing and relative abundance of fingerling Chinook Salmon in the upper estuary. In three successive years (2009-2011) two sharp peaks in the Fish Passage Index for subyearling Chinook Salmon occurred within days of large April and May releases of SCG-f stock hatchery fish, indicating an immediate post-release migration to the upper estuary (Figure 7, Table 2). A more protracted distribution of the Fish Passage Index during June and July coincided with the estuary arrival of all other HO and NP stocks (yearlings and subyearlings) from the mainstem Columbia River (Figure 7) (University of Washington 2015).

Early spring releases of SCG-f fish above Bonneville Dam in 2009 directly influenced the composition, abundance, and size distribution of Chinook Salmon stocks sampled at upper estuary sites during the Hayden Island survey (Figure 8). Successive large releases of SCG-f fish occurred on April 13 (8.4 million fish from Spring Creek National Fish Hatchery (NFH) and Little White Salmon NFH above the dam), May 1 (4.8 million from Spring Creek NFH), and May 15 (2.5 million from Bonneville Hatchery) (Table 2, Pacific States Marine Fisheries Commission 2019). The number of marked SCG-f fish at Hayden Island increased substantially in April soon after the first hatchery releases and remained abundant through May. Stock richness and abundance of unmarked salmon declined at Hayden Island in April and May but increased again in June after most HO stocks had vacated the site.

Marked Chinook Salmon at Hayden Island averaged >80 mm FL, ~40 to 50 mm FL greater than their unmarked cohorts on most survey dates (Figure 8). A considerably greater size disparity occurred during the March survey, when a large group of Willamette River hatchery yearlings averaging ~140 mm FL were present. The mean size of all unmarked subyearlings at this time was ~50 mm FL.

Multiple large releases of SCG-f hatchery fish also directly affected the composition, abundance, and size classes of juveniles in the lower estuary as indicated by the 2011 biweekly sample collections at Point Adams Beach (Figure 8). A total of 20.6 million SCG-f salmon were released above and below Bonneville Dam between March 12 and May 25 (Table 2), and marked SCG-f fish dominated abundance and stock composition at Point Adams Beach from mid-April through mid-June. Marked SCG-f fish averaged >80 mm FL in May and June, considerably larger than the co-occurring unmarked fry (<60 mm FL) (Figure 8). Abundance of unmarked and other marked fish (mostly WC-f stock) declined with the arrival of SCG-f hatchery fish in April and gradually increased as SCG-f fish steadily declined in June. We collected marked SCG-f fish at Point Adams Beach through the end of June, >30 days after the last release group from Bonneville Hatchery entered the upper end of the estuary (Table 2). We also observed large numbers of SCG-f fish at PAB during the May 2010 surveys with a corresponding decline in the abundance of unmarked juveniles. However, the narrow (one-month) release period for all SCG-f hatchery fish (Table 2) and our biweekly sampling frequency limited the temporal resolution of the SCG-f hatchery contributions at PAB in 2010.

[A] Discussion

Increased hatchery marking rates since 2008 have enabled the first comprehensive comparisons of HO and NP Chinook Salmon habitat use in the Columbia River estuary. Our study expanded on site-specific (Roegner et al. 2012; 2016) and reach-scale surveys (Sather et al. 2016) to examine the estuary-wide effects of hatchery releases on genetic stock composition, abundance, life histories, and co-occurrence with NP juveniles in shallow, nearshore habitats. We found the life history variations of HO juveniles were simplified, as measured by migration timing and size range, while their distributions overlapped considerably with NP fish in time and space. The combined annual releases of >100 million hatchery Chinook Salmon from all Columbia River hatchery programs dictated the frequency and abundance of fingerling Chinook

Salmon in nearshore habitats throughout the peak spring-summer migration period. Large releases from mitigation hatcheries magnified the estuary contributions of LCR fall Chinook Salmon and dominated salmon abundance in some habitats for weeks or months. Marked and unmarked juvenile salmon co-occurred near shore throughout the estuary but the size of marked hatchery subyearlings was much larger than their unmarked (presumably NP) cohorts. Average spring-summer biomass density of marked salmon therefore was about the same as that of unmarked juveniles despite their lower average abundance. Hatchery releases of a narrow range of large subyearling and yearling phenotypes (e.g., freshwater riverine smolts) at the peak of the salmon migration concentrated estuarine habitat use in time and space and increased the likelihood of adverse interactions with smaller NP fry in prime shallow-water rearing areas (Nickelson et al. 1986; Flagg et al. 2000; Einum and Fleming 2001; Jonsson and Jonsson 2006). Hatchery production levels have remained relatively stable in the last decade (Pacific States Marine Fisheries Commission 2019), and therefore, these patterns have not likely changed in the years since our surveys were conducted.

Chinook Salmon of the same genetic stock groups exhibited distinct seasonal and spatial distribution patterns, and marked and unmarked fish of the same genetic stock co-occurred in many of the same nearshore habitats (Teel et al. 2014). The temporal overlap between HO and NP juveniles reflected the release timing of each hatchery stock. For example, early spring abundance of both marked and unmarked yearling UWR-sp Chinook Salmon peaked simultaneously at mainstem and backside habitats in reaches C/D and E/F following peak hatchery releases of this stock in February and March; July abundance of marked and unmarked fingerling UCR-su/f stock peaked concurrently at the same sites in reaches E/F and G/H, corresponding to the large hatchery releases in June-July (Teel et al. 2014). Although we captured the greatest number of unmarked WC-f fry and fingerlings in May, overlap between NP and HO fish was maximum in reaches A and C-D during the July survey, consistent with the timing of peak hatchery releases of this stock in June-July (Teel et al. 2014).

Despite considerable overlap in their nearshore distributions, hatchery salmon exhibited a much narrower range of life history traits than unmarked fish of the same stock. Hatchery release dates often coincided with the peak abundance period for NP cohorts and dictated the estuary arrival times for marked subyearlings (Teel et al. 2014; Figure 7). In contrast, unmarked fry from LCR fall stocks (e.g., WC-f and SCG-f) appeared in the estuary months earlier and continued to arrive later than their hatchery-reared cohorts, presumably the result of natural variations in the emergence and migration times of individuals from diverse lower-river populations (e.g., Reimers and Loeffel 1967). Hatchery selection for large size at release also accounted for a much larger size-at-age and narrower length distribution for marked than for unmarked salmon of the same or different stocks. Near identical length frequencies for marked subyearlings captured in May and July suggest a uniform release size for hatchery fingerlings during the spring and early summer. Intensive hatchery selection for release size and timing reduced variation in the life history composition of salmon in the estuary (e.g., estuary arrival size and timing, estuary residency, ocean entry size and timing) and may explain the narrower range in estuarine growth variability for marked than for unmarked juveniles in 2010-11 (Goertler et al. 2016).

The nearshore dominance of fall-run juveniles, including many fish produced in hatcheries may reflect the historical decline of interior salmon production (Gustafson et al. 2007; NRC 1996) and its replacement with hatchery fish from the lower basin (Laythe 1950). As recently as 1991, 92.7 million hatchery Chinook Salmon were released in the lower river basin below The Dalles Dam (~rkm 309) compared with just 3.3 million in the entire mid- and upper basin regions above (Cone and Riddlington 1996). The vast majority (92%) of these releases consisted of fall-run stocks. By 2017 and 2018 ~44% of all Chinook Salmon stocks were released above the Dalles Dam, and total hatchery production had declined by ~7 million (Pacific States Marine Fisheries Commission 2020). Nonetheless, ~40 million fall Chinook Salmon fingerlings released each year below the dam still account for the single largest hatchery component and continue to magnify LCR stock contributions to the total estuary abundance and nearshore densities of juvenile salmon.

Intensive propagation of relatively few Columbia River salmon stocks and phenotypes has been identified as a likely factor in the apparent simplification of Chinook Salmon life history variation during the last century (Burke 2004; Bottom et al. 2005b; Campbell 2010). Most notable is the paucity of late-season migrants that were once abundant in the estuary during late summer and fall (Rich 1920). Hatchery practices that dictate the timing of estuary entry by subyearling migrants also dictate the timing of ocean entry (Weitkamp et al. 2015) and therefore, the range of growth and survival conditions juveniles will experience during the critical first weeks or months at sea (e.g., Beamish and Mahnken 2001; Beamish et al. 2004; Duffy and Beauchamp 2011; Sharma et al. 2012). Hatchery programs that synchronize the migration timing and size characteristics of Chinook Salmon stock complexes may weaken their overall capacities to buffer the effects of environmental variability through portfolio effects (Hilborn et al. 2003; Moore et al. 2010; Huber and Carlson 2015; Satterthwaite and Carlson 2015). The loss of diverse wild populations and their replacement with few hatchery stocks have been identified as important factors in the collapse of Oregon coastal Coho Salmon (Nickelson et al. 1986; Lichatowich 1999; Bottom et al. 2006) and Sacramento River Fall Chinook Salmon (Lindley et al. 2009; Huber and Carlson 2015).

Our results contradict the general characterization that hatchery selection for large "ocean-ready" smolts eliminates the risk of significant interaction with NP juveniles in shallow nursery habitats (Bottom et al. 2005b; NMFS 2014; Flagg 2015). Estuarine habitat use by juvenile Chinook Salmon generally varies with fish length, with the smallest size classes frequenting shallow-water habitats near shore (Healey 1991; Bottom et al. 2005b). However, the relatively large size at estuary entry for most hatchery salmon did not prevent considerable overlap in the nearshore distributions of marked and unmarked subyearlings. Large size could limit the total interaction times between hatchery individuals and other NP juveniles because estuary residence times generally decrease with fish length (Campbell 2010). Nonetheless, successive releases from > 80 Columbia River hatchery programs insured large numbers of hatchery juveniles remained near shore through the spring and summer when NP juvenile Chinook Salmon also were most numerous. Mean nearshore densities of HO salmon were less than for NP fish but marked subyearlings dominated fish abundance at some mainstem sampling sites (Table S.2), similar to the patterns we observed at Hayden Island and Point Adams Beach. With a mean biomass density equal to that of NP juveniles, the nearshore energy demand of HO juveniles could be significant during the spring-summer peak. The relative proportion of HO fish was greater in mainstem than in backside habitats in most reaches but we found no significant difference in the numerical or biomass densities of HO and NP salmon between the two habitat types.

The potential for interactions between HO and NP juveniles is illustrated by the abundance and residency of marked SCG-f fish near shore following a series of large hatchery releases. SCG-f fingerlings occupied nearshore habitats soon after the first April release and dominated genetic stock composition and fish abundance at upper and lower estuary sites until June. Repeated inverse trends in the abundances of marked SCG-f fingerlings and unmarked fry at Hayden Island in 2009 and at Point Adams Beach in 2011 may be evidence of competitive exclusion or displacement of smaller NP juveniles from productive shallow-water habitats, although we cannot discount seasonal variations or other factors that might otherwise account for these patterns. Similar displacement of juvenile Chinook Salmon fry from prime rearing habitats has been reported in the Skagit River Delta in years when the abundance of large wild smolts leaving the basin exceeds a threshold level (Beamer and Larsen 2004; Beamer et al. 2005).

Juvenile densities in estuarine habitats dramatically increase when large groups of similarly sized individuals of a single hatchery stock are released, synchronizing their estuary arrival and migration timing (e.g., Figures 7 and 8). The combined releases from all hatcheries during the spring-early summer period likely contribute to the pronounced seasonal peak in estuary abundance (Burke 2004; Campbell 2010; Bottom et al. 2011) in contrast to the much more protracted abundance period (i.e., through the fall) reported during the first Columbia River Chinook Salmon study in 1914-16 (Rich 1920; Burke 2004). Loss of wetland rearing habitats combined with large pulsed releases of hatchery fish could concentrate salmon abundance near shore. During the spring peak in 2010 and 2011, estimated nearshore densities for HO and NP Chinook Salmon combined averaged 0.53 fish/m^2 and >1 fish/m^2 at some sites, similar to values reported for the Skagit River delta (0.32-1.20 fish /m^2; Beamer et al. 2005) but greater than the peak values for tidal marsh channels in the Salmon River, Oregon (0.04 - 0.09 fish/m^2; Gray et al. 2002; Hering 2009); a marsh complex in the Fraser River estuary (0.35 fish/m^2; Levy and Northcote 1982); and for selected Columbia River sites in the lower estuary (rkm 35-101; 0.05 - 0.20 fish/m^2; Bottom et al. 2011) and Sandy River Delta (rkm 188-202; 0.12 fish/m^2; Sather et al. 2016). The average spring-summer Chinook densities in our study were ~ 20 times greater than the capacity estimate (252.7 fish/ha or .0257 fish/m^2) for four large tidal river deltas (Nooksack, Skagit, Snohomish, Nisqually) in Puget Sound (Greene et al. 2021).

Studies in the Columbia River have provided evidence of density dependence during the juvenile stage but little is known about density effects in tributaries below Bonneville Dam and in the estuary (ISAB 2015). Studies in other estuaries have reported salmon densities during peak periods can limit foraging success and growth (Reimers 1973, 1978; Simenstad and Salo 1982; Neilsen et al. 1985; Korman et al. 1997), increase outmigration, or reduce the mean size of rearing juveniles (Beamer and Larsen 2004; Beamer et al. 2005; Rice et al. 2011). Chinook Salmon densities in Puget Sound tidal deltas surpass the estimated rearing capacity in most years, and in some systems, density limitations imposed by historical wetland loss are compounded by large hatchery releases (Greene et al. 2021). The large number of salmon populations and hatchery programs and variability in the temporal and spatial distributions of different genetic stock groups complicate efforts to estimate rearing capacity for the Columbia River estuary or to demonstrate density-dependent relationships.

[B] Management Implications

Salmon habitat use in the Columbia River estuary occurs at the intersection of two mitigation programs with different management objectives targeting different subareas of the basin and life stages of salmon: a hatchery program to replace lost habitat and fishery production potential from the interior basin caused by the construction of mainstem dams (NMFS 2014), and a habitat restoration program in the estuary to offset the mortality of naturally produced juveniles caused by the operations of the Federal Columbia River Power System (Thom et al. 2013; Johnson et al. 2018). The considerable spatial and temporal overlap among HO and NP salmon imply a strong potential for interactions in the estuary that are not anticipated by either program but could undermine the salmon conservation goals of both. Recent efforts to reform the hatchery program elevate the conservation of NP populations to a priority equal to the production of fish for harvest (Paquet et al. 2011; Flagg 2015; NMFS 2014, 2017). These reforms focus on reducing impacts on NP fish at the watershed level by adjusting practices at individual hatcheries to minimize the risks from interbreeding and competition in natal streams (e.g., Levin et al. 2001; Zaporozhets and Zaporozhets 2005; Araki 2007a,b; Bailey et al. 2010; Chilcote et al. 2011). Our results suggest the balancing of fish production and conservation goals may also require adjustment of hatchery operations to account for ecological interactions in the estuary (Williams 2006), where all HO and NP salmon pass en route to the ocean.

The appropriate actions to realign Columbia River fish production and estuary restoration objectives may vary among hatcheries but could include adjustments to the timing, size characteristics, and numbers of fish released. Releasing fish in smaller groups over longer periods rather than in large pulses at an “optimum” time could reduce estuary densities by dispersing migrants in time and space. Strategies for rearing fish in semi-natural environments and allowing volitional rather than forced releases have been used successfully by some hatcheries to produce juveniles more similar in size and behavior to NP fish (Fuss and Byrne

2002; Maynard et al. 2004; Fast et al. 2008) and without compromising adult returns (e.g., Brouwer et al. 2014). Hatchery studies frequently have reported better survival for large smolts compared to smolts released at small sizes (Martin and Wertheimer 1989; Morley et al. 1996; Claiborne et al. 2011) but the definitions for “large” and “small” vary among studies, and a larger size at release does not necessarily increase adult survival or production efficiency (Feldhaus et al. 2016). Releasing hatchery salmon that more nearly resemble the size ranges for NP juveniles at estuary entry would likely reduce the risks of competitive dominance by larger individuals (Einum and Fleming 2001; Kostow 2009) but tradeoffs may exist between fish size and densities in prime estuarine rearing areas. Smaller hatchery fish are more likely to reside in the estuary for longer periods (Campbell 2010; Healey 1991), increasing the proportion of HO juveniles in shallow areas and the relative frequency of interactions with NP juveniles.

Controlled studies of the estuarine interactions between HO and NP juveniles could help to establish ecological criteria for managing hatchery practices. Our results reinforce previous recommendations (Rand et al. 2012) for designing hatchery releases as experiments to (1) test for density-related effects on salmon consumption, growth, or residence times in estuarine rearing habitats; and (2) identify any significant behavioral interactions and effects, including whether large pulses of hatchery fish displace NP juveniles from prime rearing areas (e.g., Spilseth and Simenstad 2011). A better understanding of density effects is needed to determine whether hatchery practices are working in concert with estuary restoration to achieve the survival and life history diversity objectives for NP populations (Bottom et al. 2005b; Fresh et al. 2005; Krueger et al. 2017). Otolith chemical methods for reconstructing the estuarine life histories of juveniles and their contributions to adult returns (Campbell 2010; Miller et al. 2010; Volk et al. 2010; Jones et al. 2014; Rose 2015) could provide useful population-level indicators of the effectiveness of estuary restoration actions under particular hatchery production levels.

Ultimately, reductions in the aggregate number of hatchery fish may be necessary to decrease the concentrations of hatchery subyearlings in the estuary during spring and summer. The disproportionate contribution of large SCG-f hatchery fingerlings to the abundance and biomass density of Chinook Salmon in shallow-water rearing habitats is a particular concern. A formal system-wide review of this and other large mitigation hatchery programs may be necessary to better align management actions among multiple legal mandates for Columbia River fishery production (NMFS 2014, 2017), salmon conservation (e.g., Myers et al. 1998; NMFS 2005), and estuary restoration (NMFS 2008, 2010, 2020).

[A] Acknowledgements

The U.S. Army Corps of Engineers, the City of Portland, and NOAA National Marine Fisheries Service provided support for the field surveys. We are grateful to Pascale Goertler, Brian Kelly, George McCabe, Regan McNatt, and Rick Nelson for their assistance in the field. Thanks also to Mary Ramirez and Gordon Rose for their assistance with several graphics. Drafts of this manuscript benefitted from the constructive reviews of Kim Jones, Regan McNatt, Jim Myers, and Rich Zabel. We are particularly grateful for the detailed comments and suggestions for improving this manuscript from Dr. David Geist and three anonymous reviewers.

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Table 1. Annual mean number (millions) and percentage of marked (with a clipped fin and/or cwt) Chinook salmon released from hatcheries. Hatchery release data are for January 2010 through December 2011. Evolutionarily significant unit (ESU) status: E = endangered; T = threatened; NW = listing not warranted. Abbreviations for the 10 Columbia River basin genetic stocks are as follows: LCR= Lower Columbia River; WC-f = West Cascade, fall; WC-sp = West Cascade, spring; SCG-f = Spring Creek Group, fall; UWR-sp = Upper Willamette River, spring; MCR-sp = Mid-Columbia River, spring; UCR sp = Upper Columbia River, spring; DR-f = Deschutes River, fall; UCR-su/f = Upper Columbia River, summer/fall; SR-f = Snake River, fall; SR-sp = Snake River, spring, RR-f = Rogue River, fall.

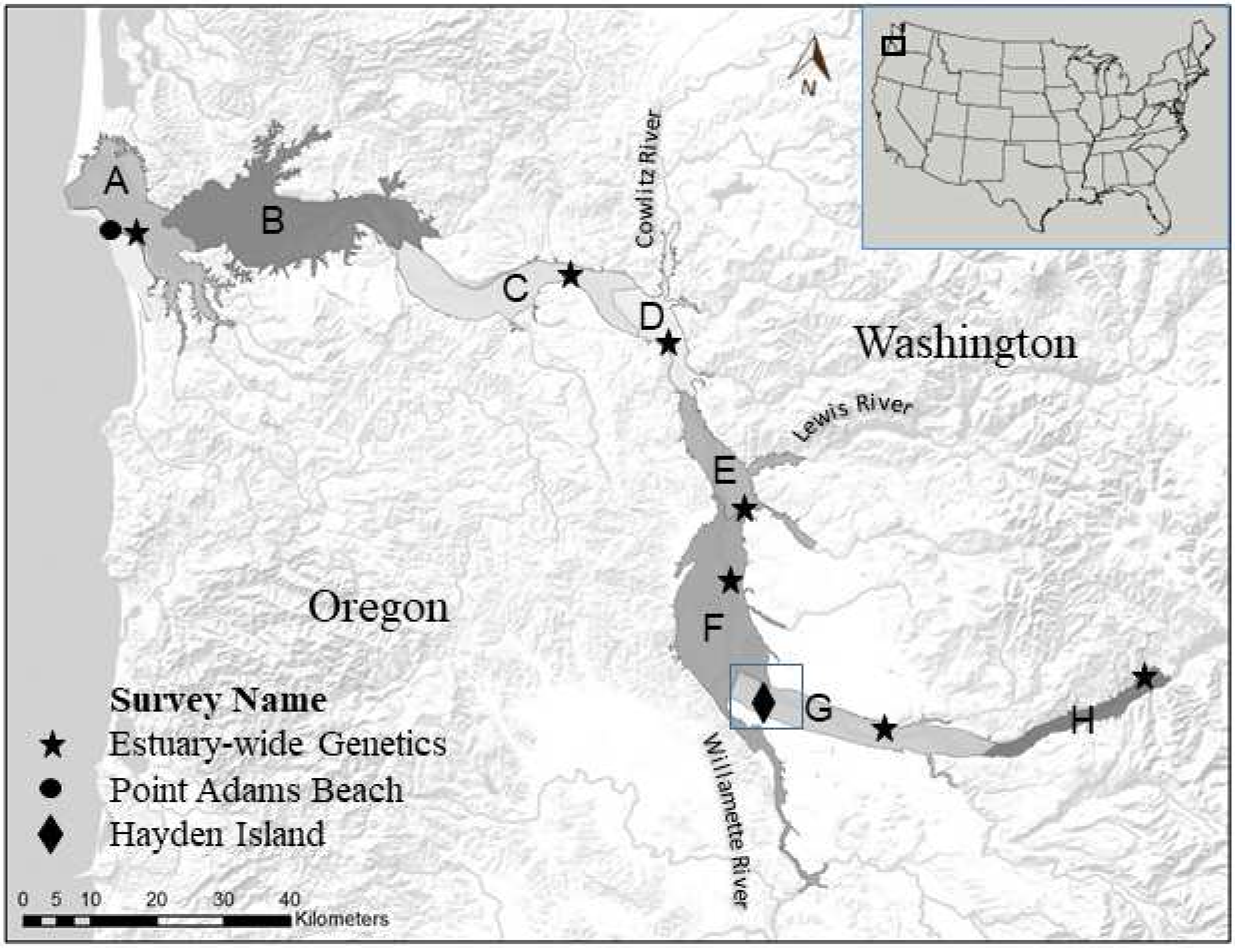
		Subyearling		Yearling		Total	
ESU (status)	Genetic stock	Number (millions)	Percent marked	Number (millions)	Percent marked	Number (millions)	Percent marked
Lower Columbia River (T)	LCR:WC-f	17.0	99.3	0.0	----	17.0	99.3
	LCR:WC-sp	0.3	0.0	2.9	99.3	3.2	89.7
	LCR:SCG-f	21.1	99.6	0.0	----	21.1	99.6
Lower Columbia River ESU total		38.4	98.7	2.9	99.3	41.3	98.7
Upper Willamette River (T)	UWR-sp	2.1	99.9	5.8	99.8	7.9	99.8
Mid-Columbia River Spring (NW)	MCR-sp	0.5	0.0	5.5	93.9	6.0	86.5
Upper Columbia River spring (E)	UCR-sp	0.1	96.4	2.7	98.7	2.8	98.7
Deschutes River Summer-Fall (NW)	DR-f	0.0	----	0.0	----	0.0	----
Upper Columbia River Summer-Fall (NW)	UCR-su/f	23.3	73.9	3.4	100.0	26.7	77.2
Snake River Fall (T)	SR-f	4.8	64.5	0.9	99.4	5.7	70.2
Snake River Spring-Summer (T)	SR-sp	0.9	62.9	12.9	99.1	13.8	96.7

None	RR-f	1.1	100.0	0.0	----	1.1	100.0
All releases		71.3	87.2	34.0	98.5	105.3	90.9

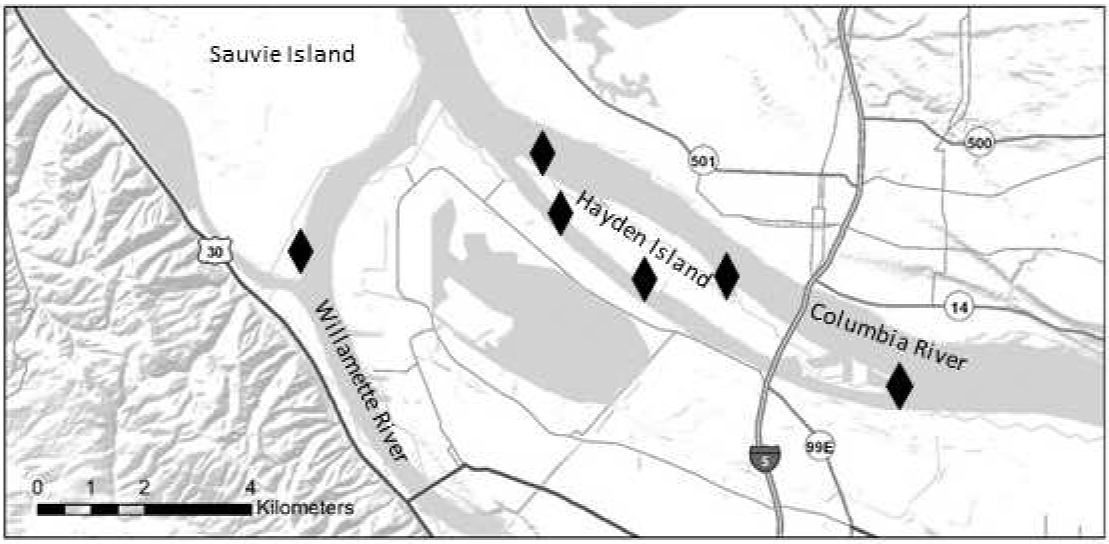
Table 2. Release date, location, and numbers of hatchery origin SCG-f Chinook Salmon released in the Columbia River basin, 2009-11 (Pacific States Marine Fisheries Commission 2019). Fish released into the NF Klaskanine R. (Reach A), Big Cr. (Reach B), and Tanner Cr. (Reach H) enter the estuary below Bonneville Dam. All other releases occurred above the dam.

Year	Release Date	Release Location	Hatchery	Millions Released	Percent Marked
2009	13-Apr	Spring Cr and Little White Salmon R	Spring Cr NFH and Little White Salmon NFH	8.3	100.0
	1-May	Spring Cr	Spring Cr NFH	4.8	100.0
	11-May	Big Cr	Big Cr Hatchery	5.7	98.0
	15-May	Tanner Cr	Bonneville Hatchery	2.5	96.4
Total				21.3	99.4
2010	12-Apr, 13-Apr	Spring Cr and Little White Salmon R	Spring Cr NFH and Little White Salmon NFH	8.0	100.0
	30-Apr, 3-May, 4-May	Big Cr and NF Klaskanine R	Big Cr Hatchery and Klaskanine Hatchery	6.0	98.8
	3-May	Tanner Cr	Bonneville Hatchery	2.9	98.0
	10-May	Spring Cr	Spring Cr NFH	4.6	100.0
Total				21.5	99.4
2011	12-Mar	Little White Salmon R	Little White Salmon NFH	1.8	100.0
	12-Apr	Spring Cr	Spring Cr NFH	6.2	100.0
	15-Apr	NF Klaskanine R	Big Cr Hatchery	1.9	99.9
	4-May	Spring Cr	Spring Cr NFH	4.6	100.0
	16-May	Big Cr	Big Cr Hatchery	3.3	99.8
	25-May	Tanner Cr	Bonneville Hatchery	2.8	99.8
Total				20.6	99.8

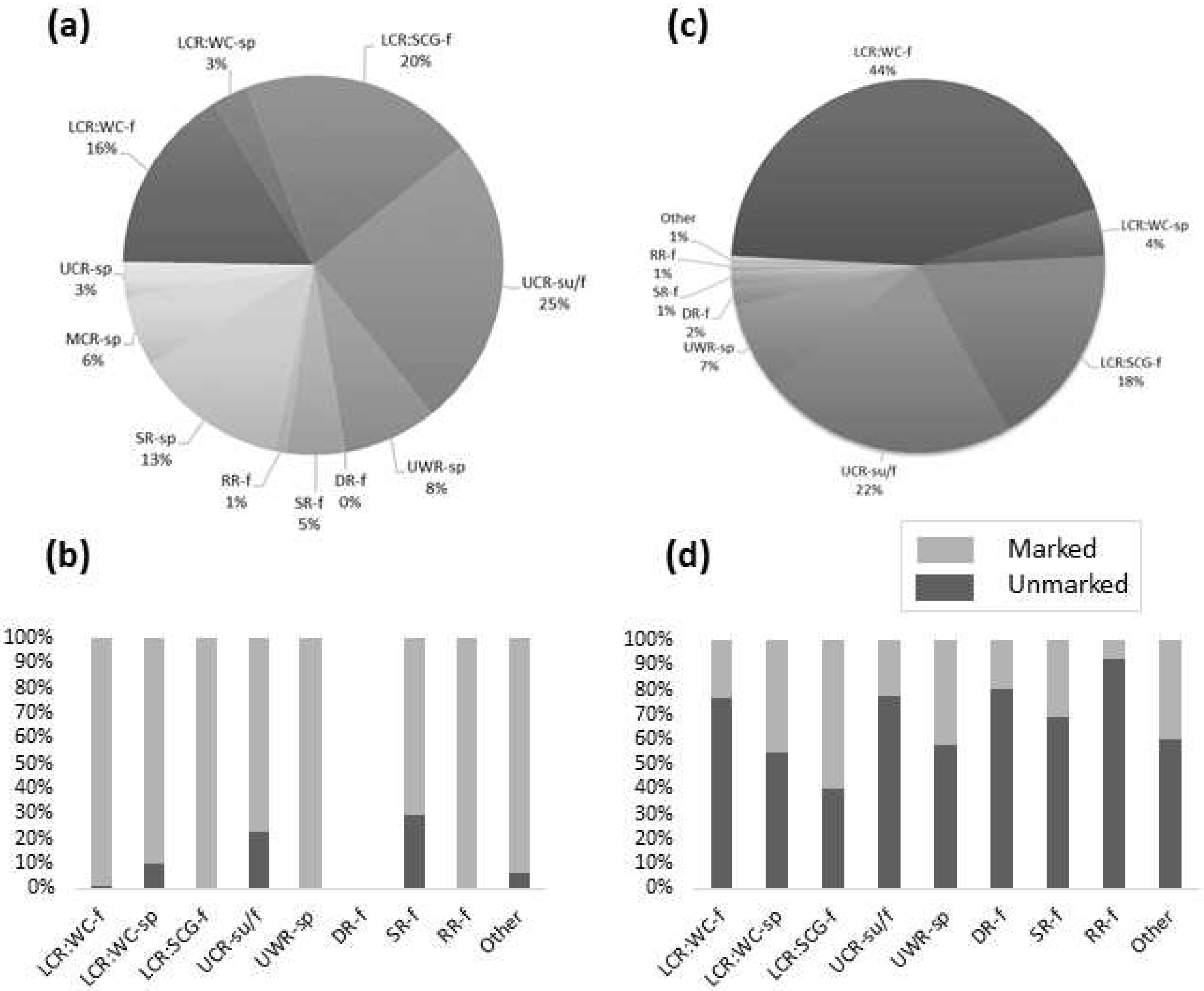
(a)



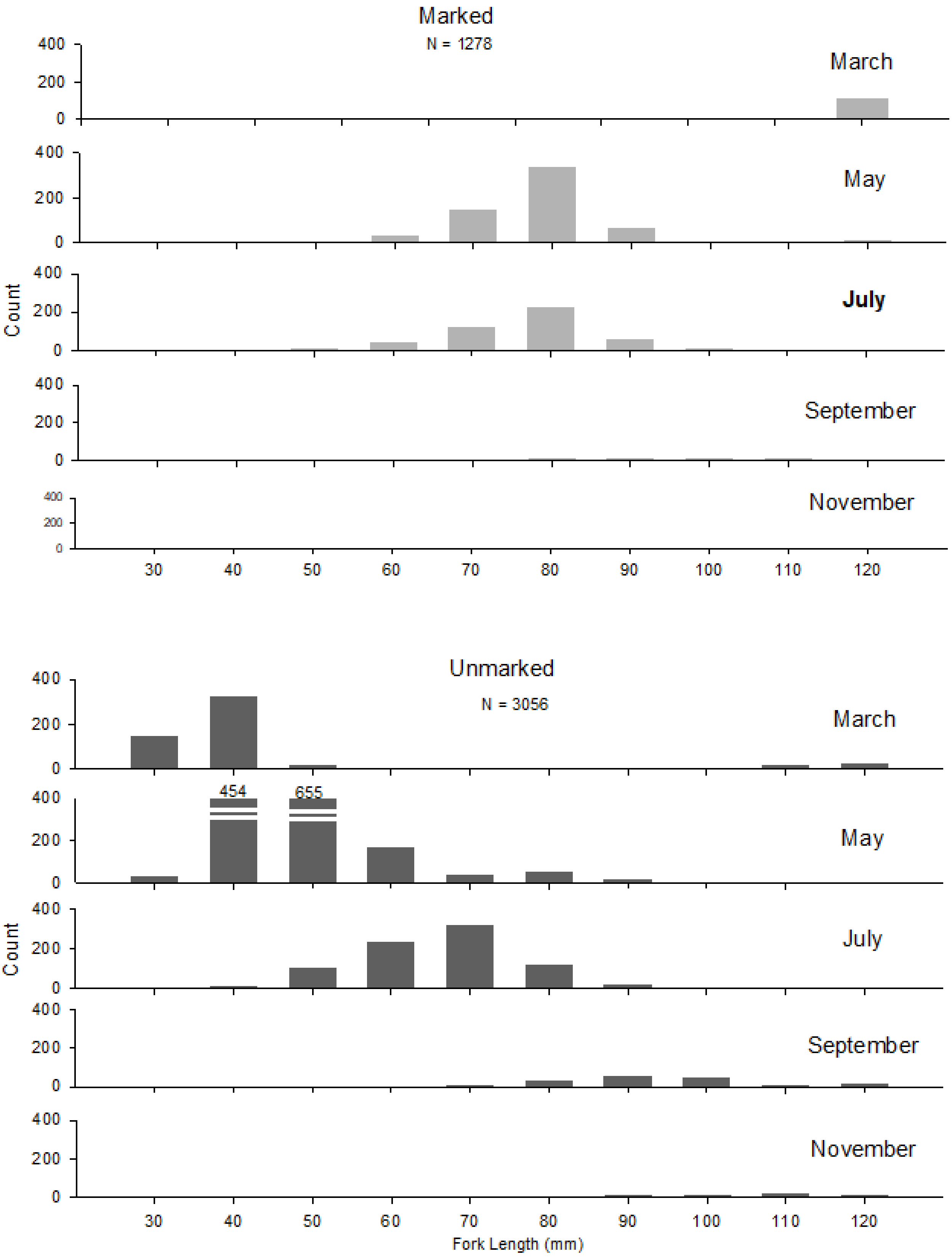
(b)

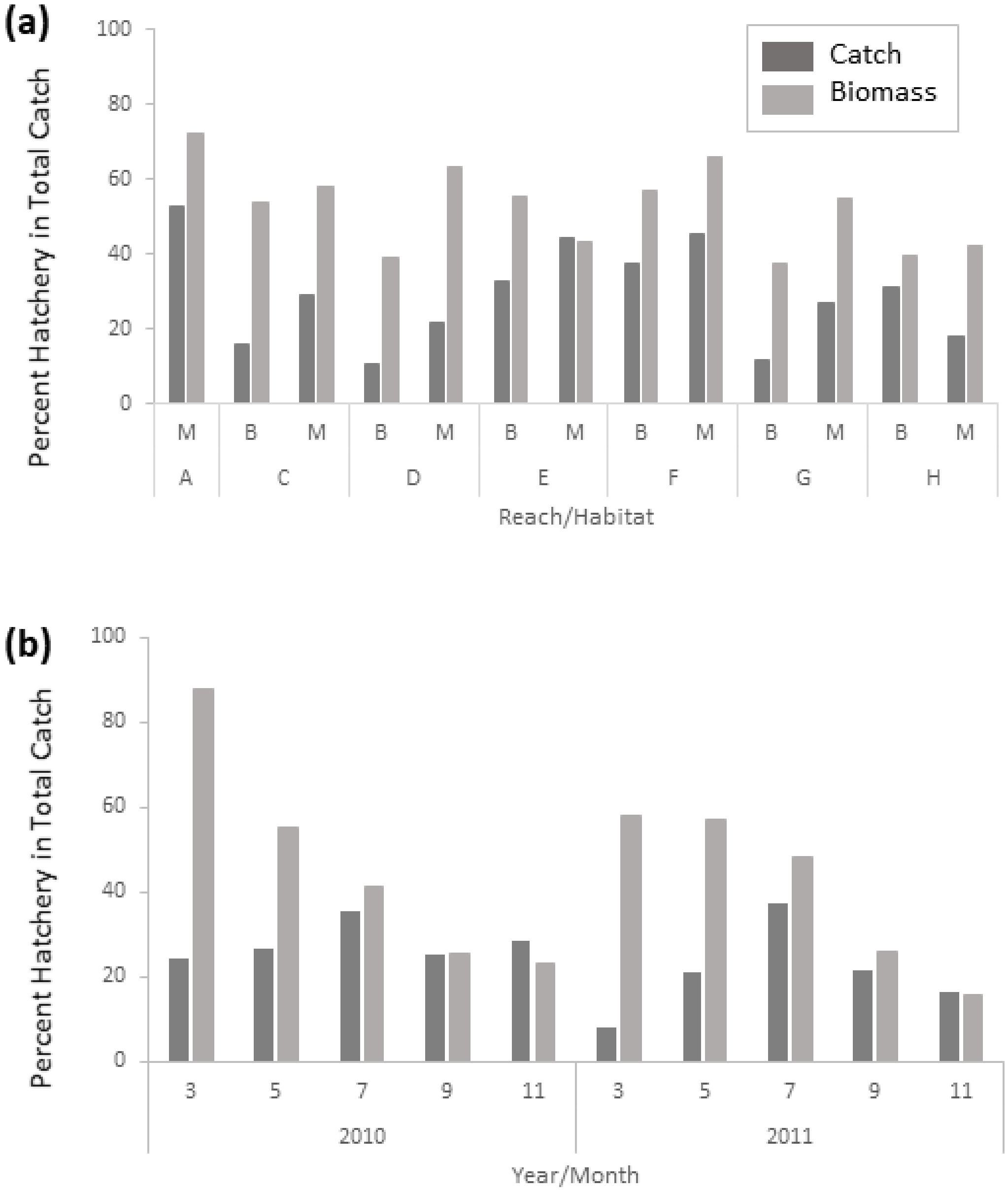


nafm_10671_f1 gray.tif

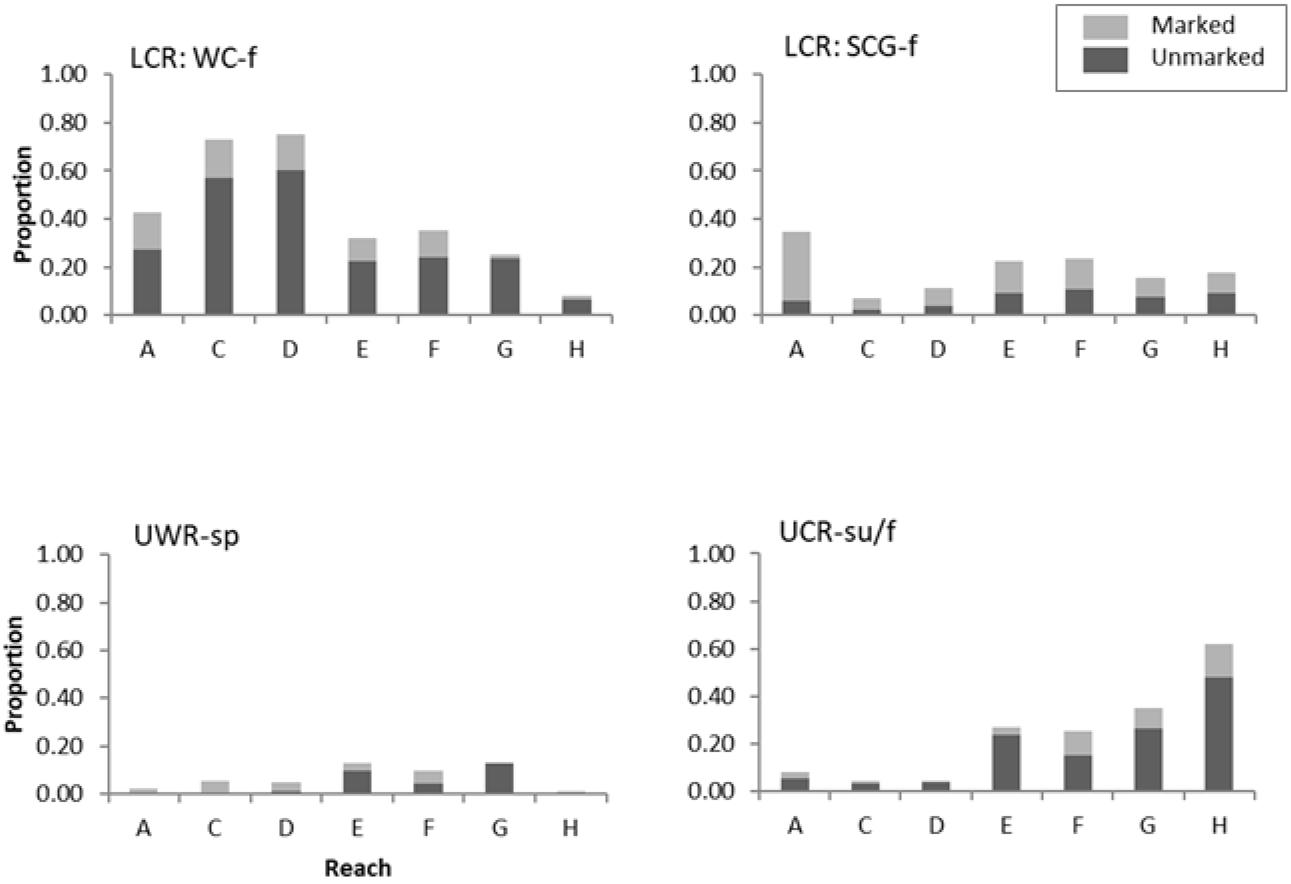


nafm_10671_f2 gray.tif

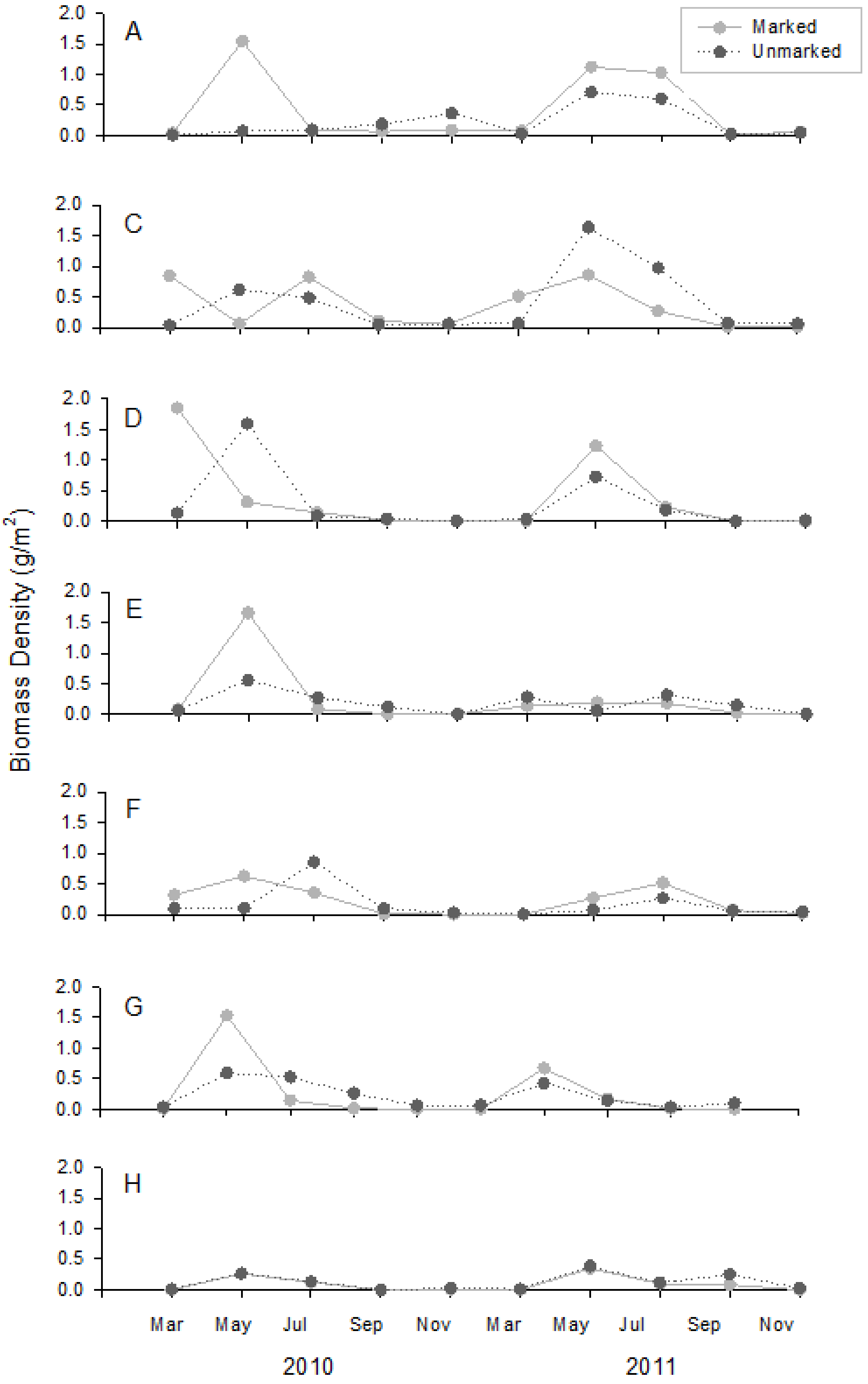

Marked
N = 1278
March
May
July
September
November
Count
400
200
0
30
40
50
60
70
80
90
100
110
120
Unmarked
N = 3056
454
655
Fork Length (mm)



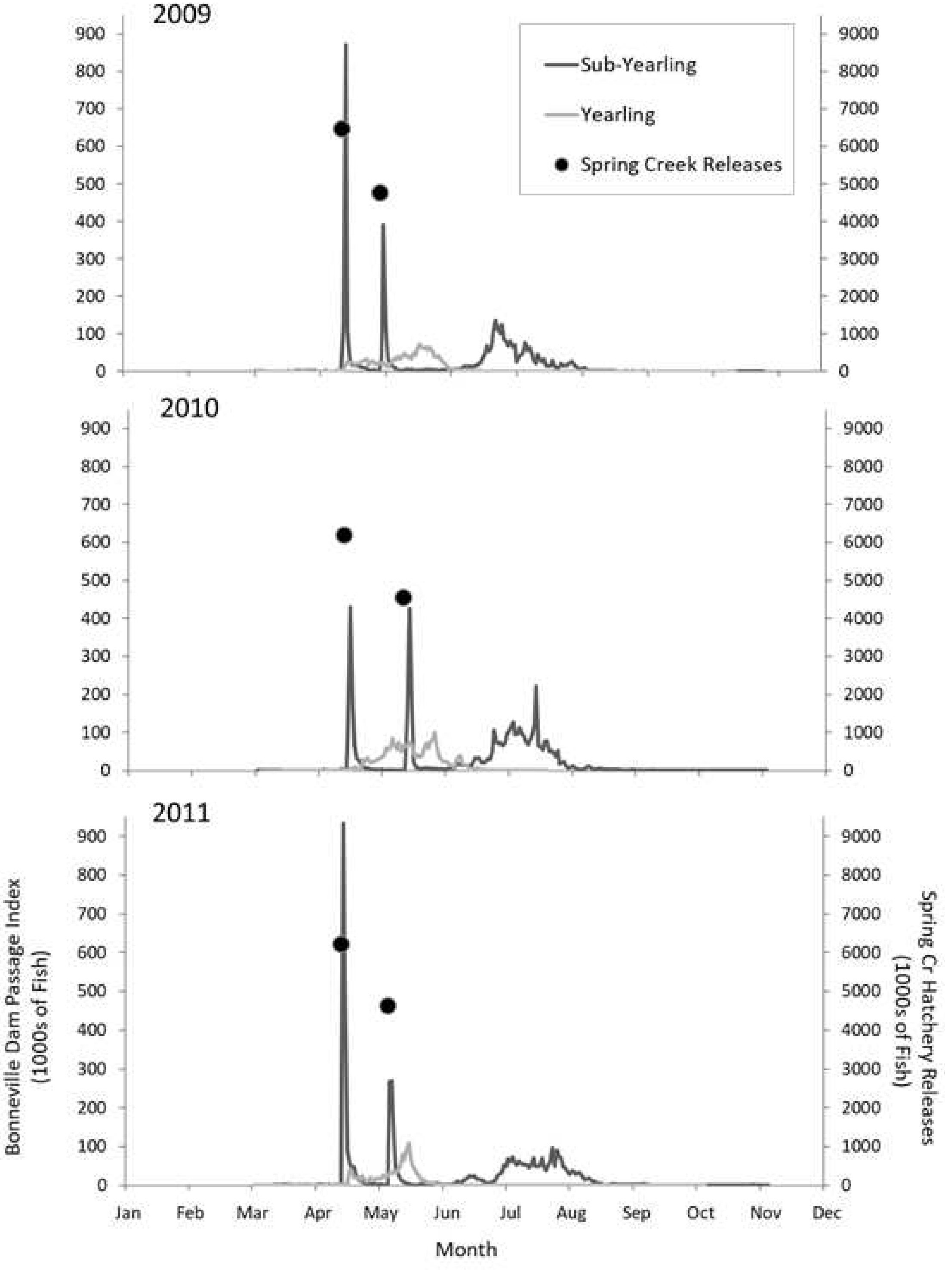
nafm_10671_f4 gray.tif



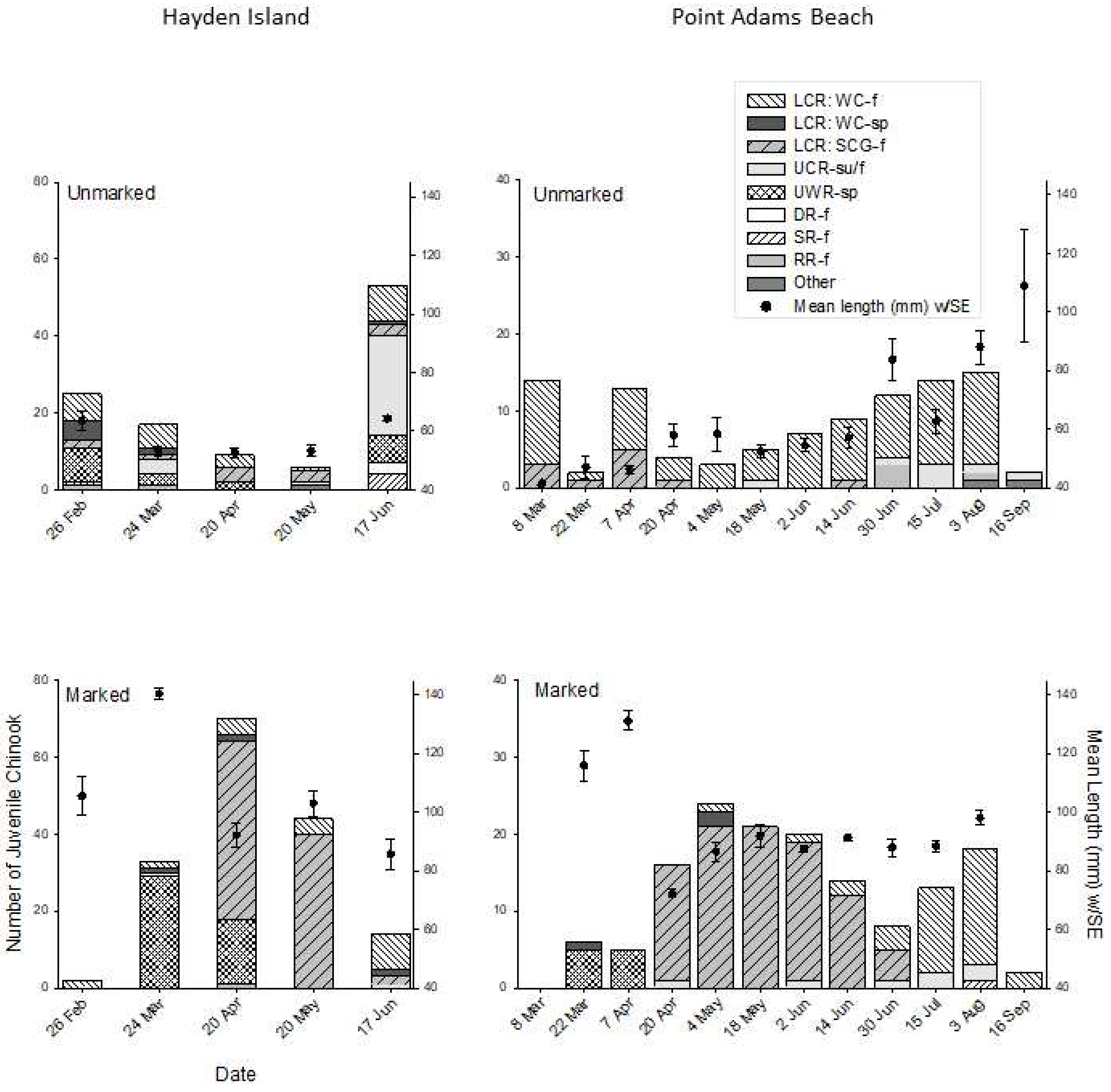
nafm_10671_f5 gray.tif



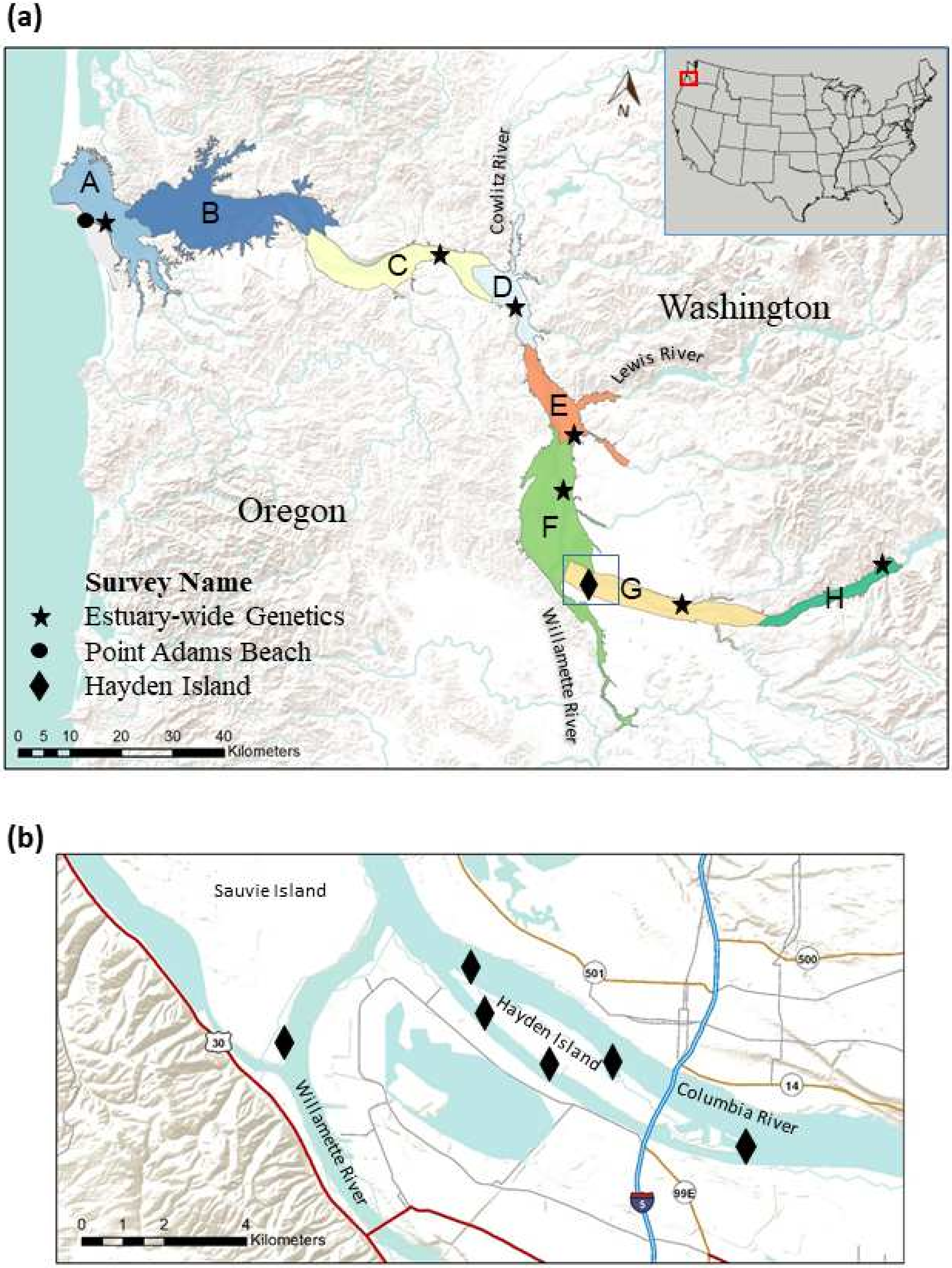
nafm_10671_f6 gray.tif



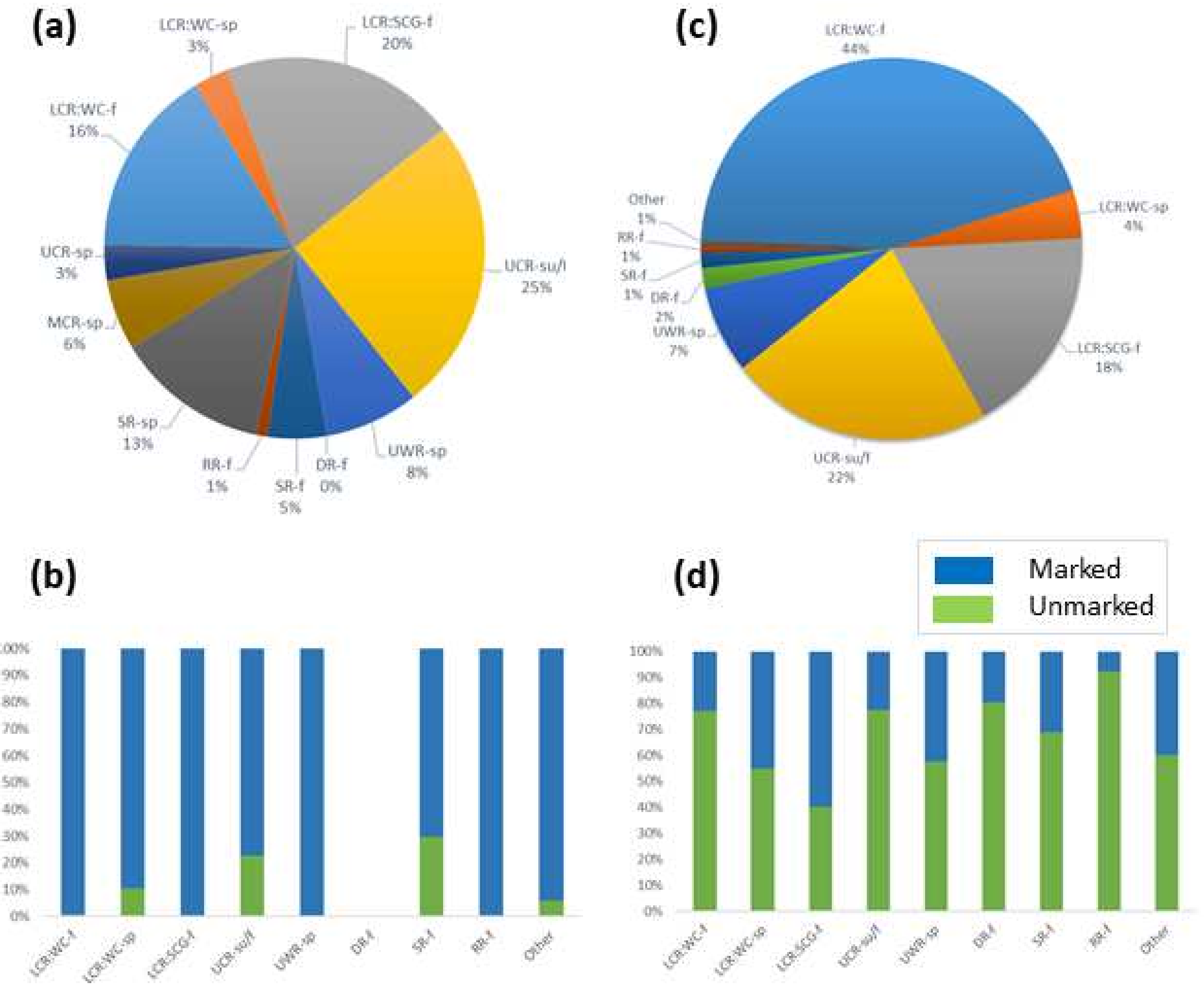
nafm_10671_f7 gray.tif



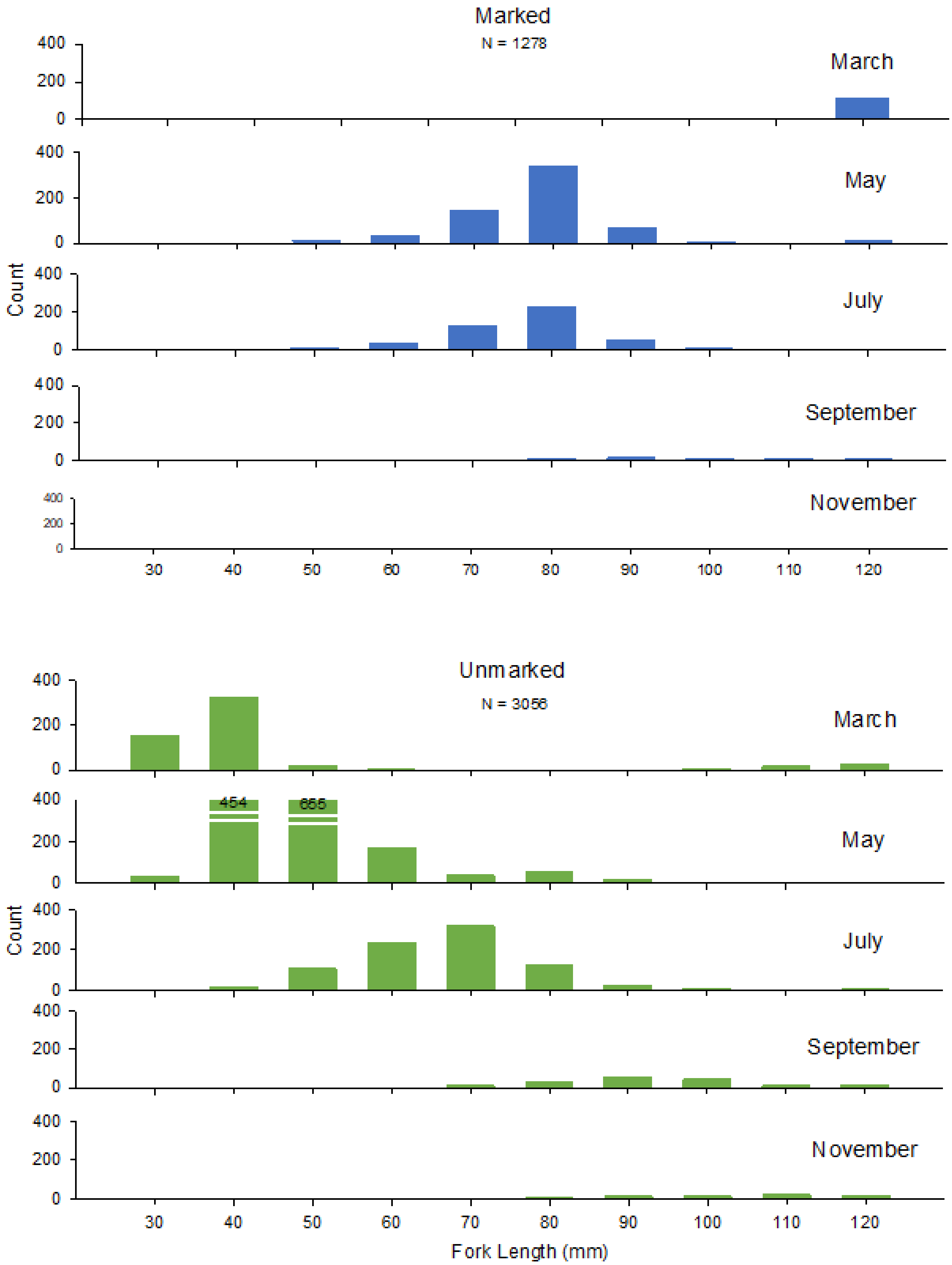
nafm_10671_f8 gray.tif



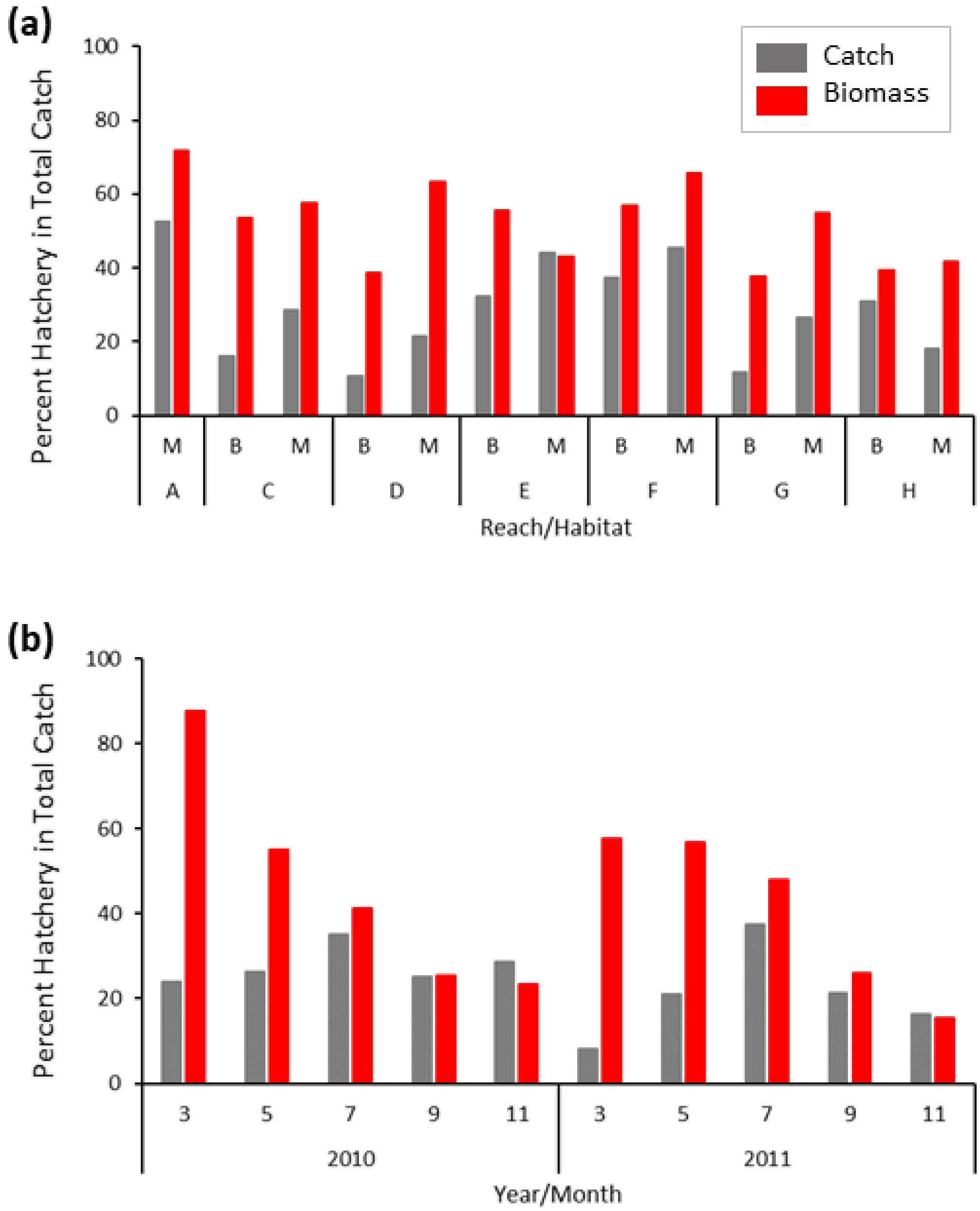
nafm_10671_f1.tif



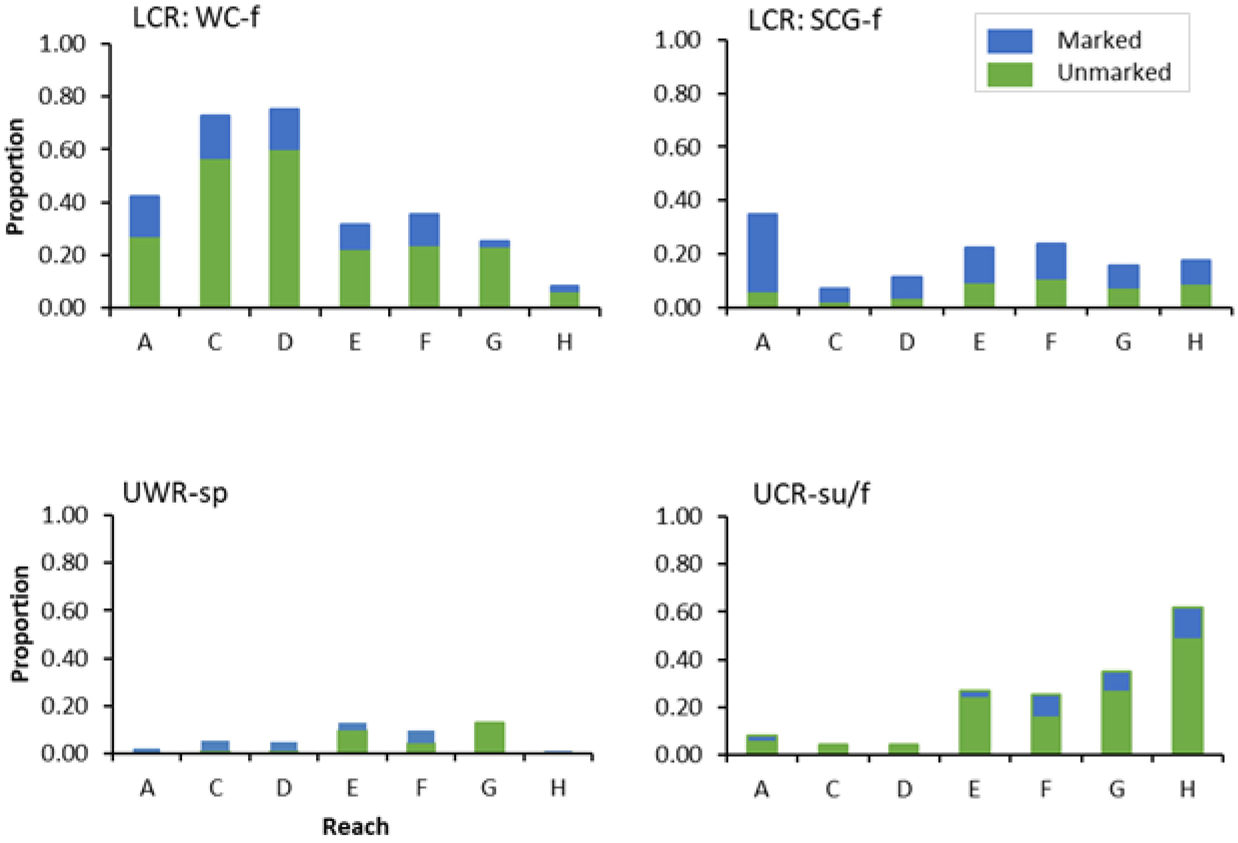
nafm_10671_f2.tif



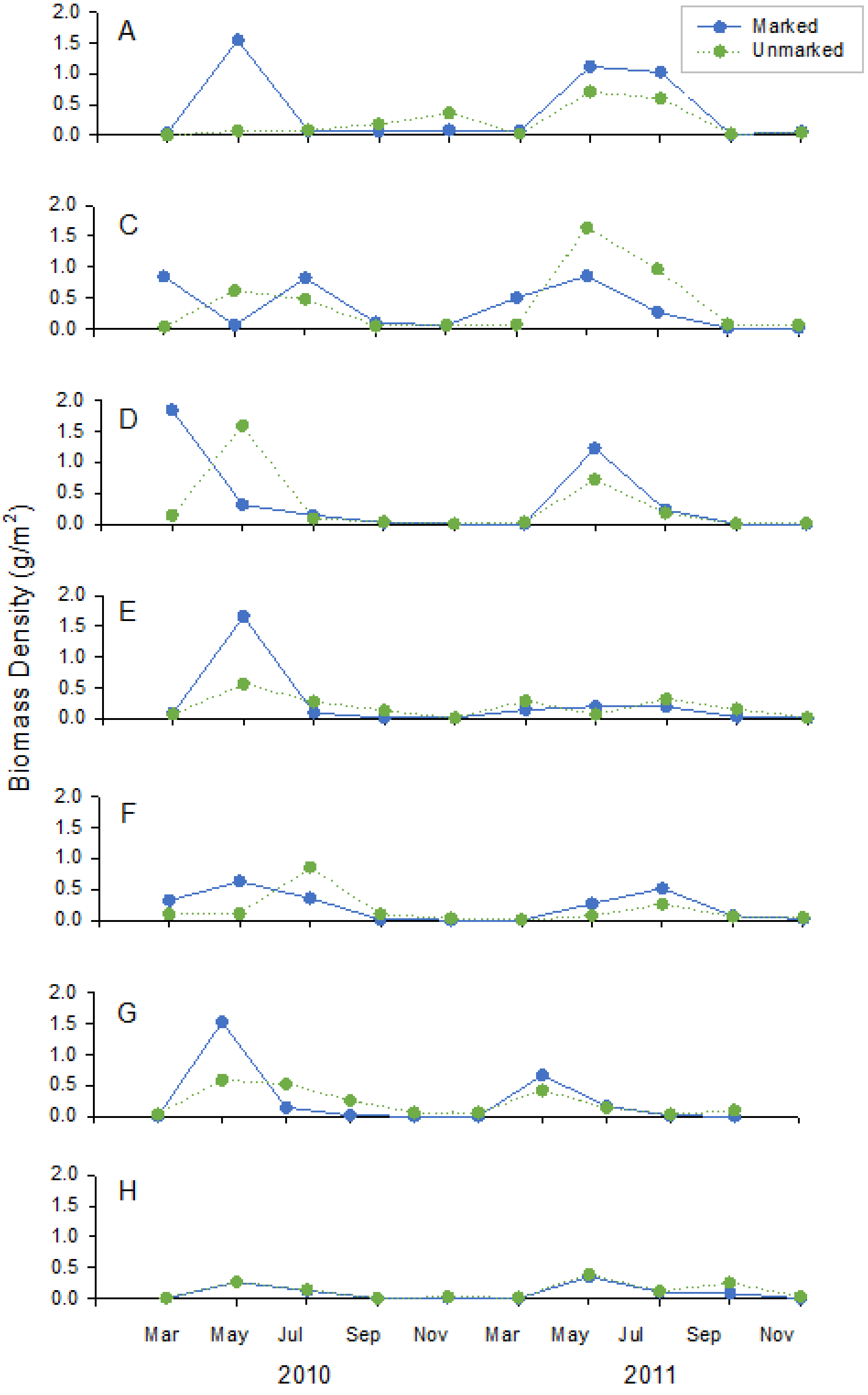
nafm_10671_f3.tif



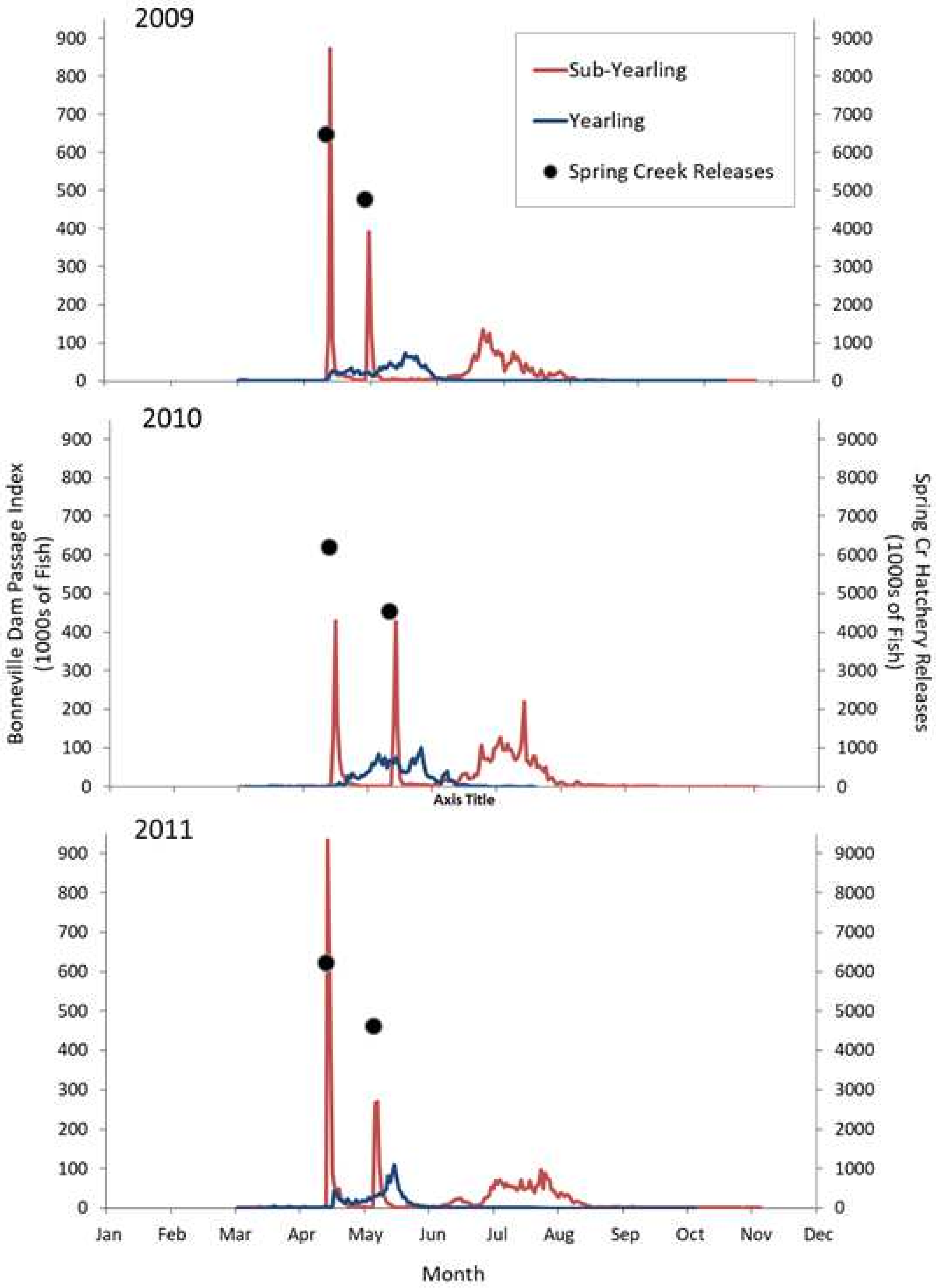
nafm_10671_f4.tif



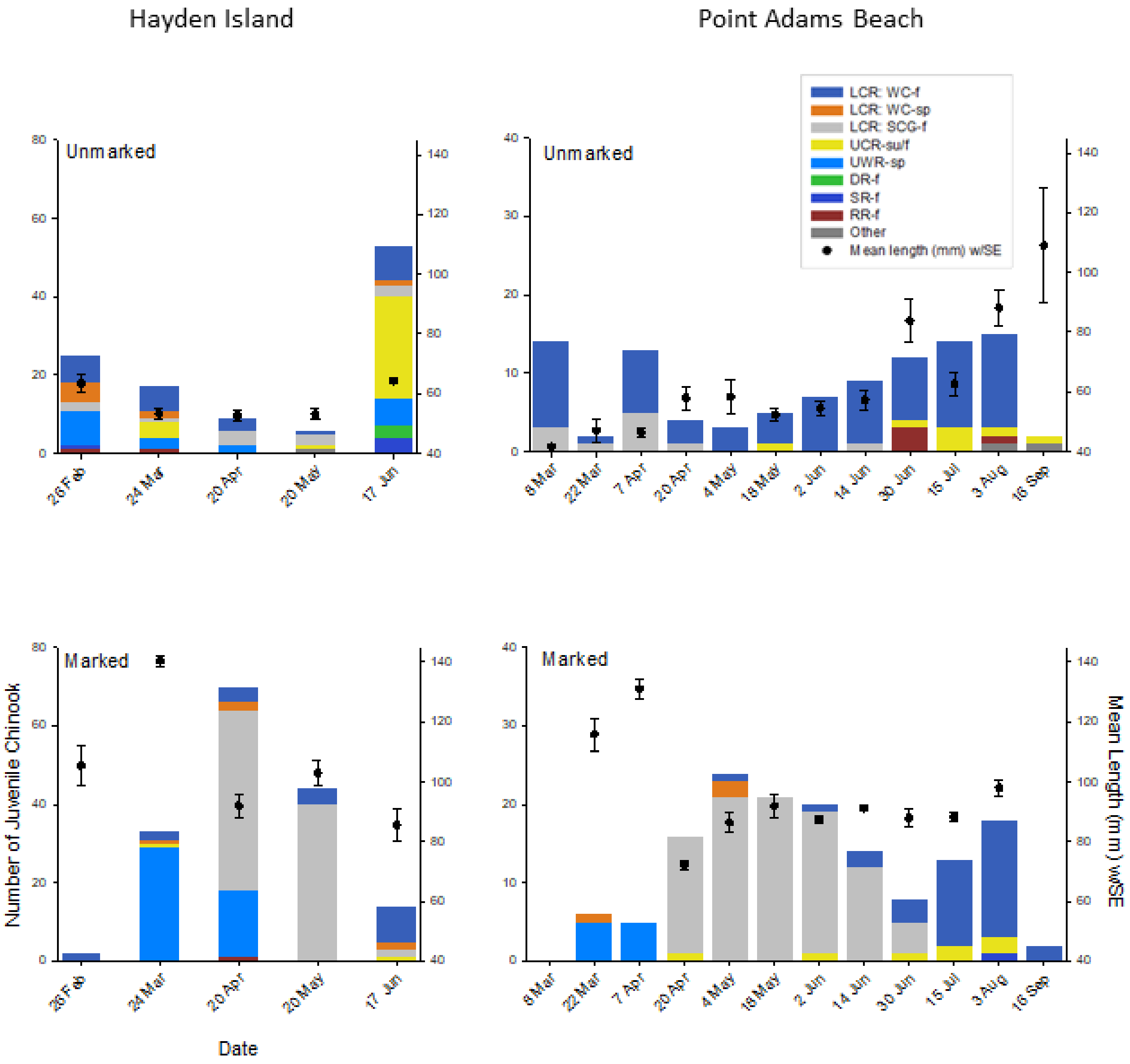
nafm_10671_f5.tif



nafm_10671_f6.tif



nafm_10671_f7.tif



nafm_10671_f8.tif