NOAA Technical Memorandum NMFS-NWFSC-88

Fish Habitat Use and Chemical Contaminant Exposure

at Restoration Sites in Commencement Bay, Washington

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O. Paul Olson, Lyndal Johnson, Gina Ylitalo,
Casey Rice, Jeff Cordell,[1] Tracy K. Collier,
and Jennifer Steger[2]

Northwest Fisheries Science Center
Environmental Conservation Division
2725 Montlake Boulevard East
Seattle, Washington 98112

[1] University of Washington
School of Aquatic and Fishery Sciences
Seattle, Washington 98195

[2] NOAA Restoration Center
Northwest Region Office
7600 Sand Point Way Northeast
Seattle, Washington 98115

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Executive Summary

Commencement Bay is a heavily industrialized estuary in central Puget Sound that has suffered extensive loss of its historical estuarine wetlands and chemical contamination from both point and nonpoint sources. In October 1991 the Natural Resource Trustees responsible for the area's wildlife (National Oceanic and Atmospheric Administration [NOAA], U.S. Fish and Wildlife Service, U.S. Bureau of Indian Affairs, Puyallup Indian Tribe, Muckleshoot Indian Tribe, Washington Department of Ecology, Washington Department of Fish and Wildlife, and Washington Department of Natural Resources) formally initiated a natural resource damage assessment (NRDA) action involving several Commencement Bay industries. This process eventually resulted in the development and implementation of a contaminated sediment remediation and habitat restoration plan.

In 2002 and 2003, the Environmental Conservation Division of the National Marine Fisheries Service (NOAA Fisheries Service) Northwest Fisheries Science Center and the NOAA Restoration Center conducted an assessment program to evaluate the success of restoration projects at seven restoration sites in Commencement Bay, Washington. The objective of this study was to monitor these restoration sites to determine fish presence, individual fish health, and the degree of toxic chemical contaminants at restored sites. Restoration sites were sampled for fish assemblage composition, salmonid diets, and anthropogenic chemical contamination of fish tissues and sediments.

Monitoring results showed that all sites were being utilized by fish but assemblage composition varied by habitat type and date. The Skookum Wulge and Yowkwala sites were utilized most heavily by juvenile salmon (*Oncorhynchus* spp.*)*, while other sites were dominated by species such as shiner perch (*Cymatogaster aggregata*), surf smelt (*Hypomesus pretiosus*), and Pacific staghorn sculpin (*Leptocottus armatus*). Mean species richness was greatest at Olympic View and Yowkwala, and lowest at Squally Beach and the Middle Waterway sites, consistent with expectations based on habitat type. Species richness, as well as total number of fish captured, increased from spring to early summer, then declined. The mean species density and mean fish density were lower in 2003 than 2002. In both years, marked (i.e., known hatchery origin) juvenile Chinook salmon (*O. tshawytscha*) and coho salmon (*O. kisutch*) outnumbered unmarked juveniles, and were generally larger. Unmarked juvenile Chinook were present in Commencement Bay for a longer period of time than marked fish. Diet analysis of juvenile Chinook stomach contents demonstrated differences in prey consumed based on both site and length of fish.

Anthropogenic chemical contaminants, including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and dichlorodiphenyltrichloroethanes (DDTs), were measured in fish tissue, fish prey, and sediments from these restoration sites. Contaminant concentrations in sediments at most of the restoration sites were comparable to levels observed at Commencement Bay sites used as reference areas for the Hylebos Waterway damage assessment

sediment evaluation studies conducted as part of the NRDA process (EVS 1996), and substantially cleaner than sediments in the heavily industrialized sections of the Hylebos Waterway (EVS 1996). However, there were some exceptions. At Middle Waterway at Simpson, Middle Waterway at City, and Squally Beach, total PAH concentrations in sediments were in the 8,000–15,000 parts per billion range, comparable to concentrations measured at some of the more contaminated sites in the Hylebos Waterway as part of the NRDA study (Collier et al. 1998a, EVS 1996). Concentrations of DDTs in sediments were also elevated at two sites. At Squally Beach, levels (13 ng/g dry wt) were within the higher range of DDT concentrations recorded in the Hylebos Waterway during NRDA (EVS 1996). At Mowitch, DDT concentrations (120 ng/g dry wt) were an order of magnitude higher than concentrations measured at any of the restoration sites, or at any sampling sites in the Hylebos Waterway or Commencement Bay during NRDA (EVS 1996).

Similarly, contaminant concentrations in salmon stomach contents were lower than those measured in salmon from the Hylebos Waterway during NRDA at most of the restoration sites (Stehr et al. 2000), but concentrations of PCBs and DDTs in stomach contents of juvenile salmon from the Middel Water at Simpson site, and PAHs in stomach contents of juvenile salmon from the Olympic View site, were elevated. Body burdens of PCBs in juvenile Chinook and chum salmon from the Yowkwala and Mowitch sites were at threshold concentrations associated with adverse effects in salmon (Meador et al. 2002), but cannot be compared with Hylebos data (Collier et al. 1998a) as no whole body samples were analyzed as part of that study.

The major goals of this study were 1) to assess the effectiveness of restoration efforts in improving fish habitat accessibility and use, especially for juvenile salmon, and 2) to assess the effectiveness of sediment remediation efforts in reducing contaminant levels in Commencement Bay and protecting estuarine biota from damage associated with toxicant exposure. Unfortunately, the lack of prerestoration baseline data limited the evaluation of restoration efforts at the sites in this study. Most importantly, we were unable to determine whether habitat use by salmonids and other fish species had increased or decreased as a result of restoration, as no data were available on their prior utilization of the restoration sites. Moreover, the assessment of temporal changes in contamination and fish exposure at the restoration sites was complicated by the fact that sites sampled as part of the damage assessment process were different from those selected for restoration. True reference sites with which to compare fish habitat usage data would also have been helpful, although they may be impossible to find in highly industrialized embayments such as Commencement Bay. Because of the lack of baseline data, longer term monitoring is needed to better assess the effectiveness of restoration activities at these sites.

Despite these limitations, the study yielded valuable information on the status of restoration sites and aquatic biota in Commencement Bay. In summary, we found that:

- All the Commencement Bay restoration sites studied are providing some level of functional fish habitat.
- There was an increase in the physical quality of the sites (capacity) due to restoration efforts that should improve the system functionality of these sites.

- As yet, no clear trends in habitat use at the restoration sites can be observed. While there were some yearly differences in species abundances, the differences were relatively small considering the natural year-to-year variation that is known to occur.
- Sediment remediation activities at some of the restoration sites have not been fully successful in reducing contaminant levels below trustee site cleanup goals or resource injury guidelines.
- Improvements in the "realized function" of the restoration sites were compromised by the presence of chemical contaminants in the sediments and tissues of fish that may have adverse health effects on their long-term health and survival.

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Introduction

Commencement Bay, a heavily urbanized estuary at the southern end of central Puget Sound, is the harbor for the City of Tacoma, Washington (Shreffler et al. 1990, Shreffler et al. 1992, Simenstad 2000). More than 98% of the historical estuarine wetland has been lost or altered through anthropogenic activities (Bortleson et al. 1980, Collins and Montgomery 2001, Collins et al. 2003) (Figure 1). The intertidal areas leading into the bay have been filled and channelized, so that they now form eight heavily industrialized waterways (Hylebos, Blair, Sitcum, Milwaukee, St. Paul, Middle Waterway, Thea Foss, and Wheeler-Osgood).

Industries along the waterways include pulp and lumber mills, shipbuilding and ship repair facilities, shipping docks, marinas, chlorine and chemical production facilities, concrete production facilities, aluminum smelting facilities, oil refineries, food processing plants, automotive repair shops, railroad operations, and numerous other storage, transportation, and chemical manufacturing companies (national resource damage assessment [NRDA] Web site, http://www.darrp.noaa.gov/pacific/cbay/index.html). More than 700 point and nonpoint discharges into the bay have been identified, a number of which may contribute to sediment and water contamination (Commencement Bay Nearshore/Tideflats Record of Decision, U.S. Environmental Protection Agency [EPA], 1989).

Coastal and estuarine systems play a vital role in the life cycle of many organisms, including rearing habitat for early life stages of numerous fish species (Levy and Northcote 1982, Day et al. 1989, Beck et al. 2001, Beck et al. 2003, Rice et al. 2005). These ecosystems are of particular importance in the recovery of species at risk (Feist et al. 2003), and are heavily impacted by human activities (Shreffler et al. 1990, Cederholm et al. 2000, Beck et al. 2001, Beck et al. 2003, Rice et al. 2005).

Commencement Bay, Washington, was the first Superfund site designated in marine waters of the United States, and the National Oceanic and Atmospheric Admininstration (NOAA) and other natural resource agencies have been concerned for a number of years about the potential impacts of hazardous chemicals on marine and estuarine animals that use Commencement Bay, including salmon listed under the Endangered Species Act. In 1991 the trustees (NOAA, U.S. Fish and Wildlife Service, U.S. Bureau of Indian Affairs, Puyallup Tribe of Indians, Muckleshoot Indian Tribe, Washington Department of Ecology, Washington Department of Fish and Wildlife, and Washington Department of Natural Resources) formally initiated resource damage assessment and restoration planning actions in Commencement Bay under the NRDA process (http://www.darrp.noaa.gov/pacific/cbay/injury.html).

Damage assessment studies conducted by NOAA and the trustees over the last decade have demonstrated the uptake of polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethane (DDTs), hexachlorobenzene (HCB), and various organochlorine pesticides (OCs) by juvenile Chinook (*Oncorhynchus tshawytscha*) and

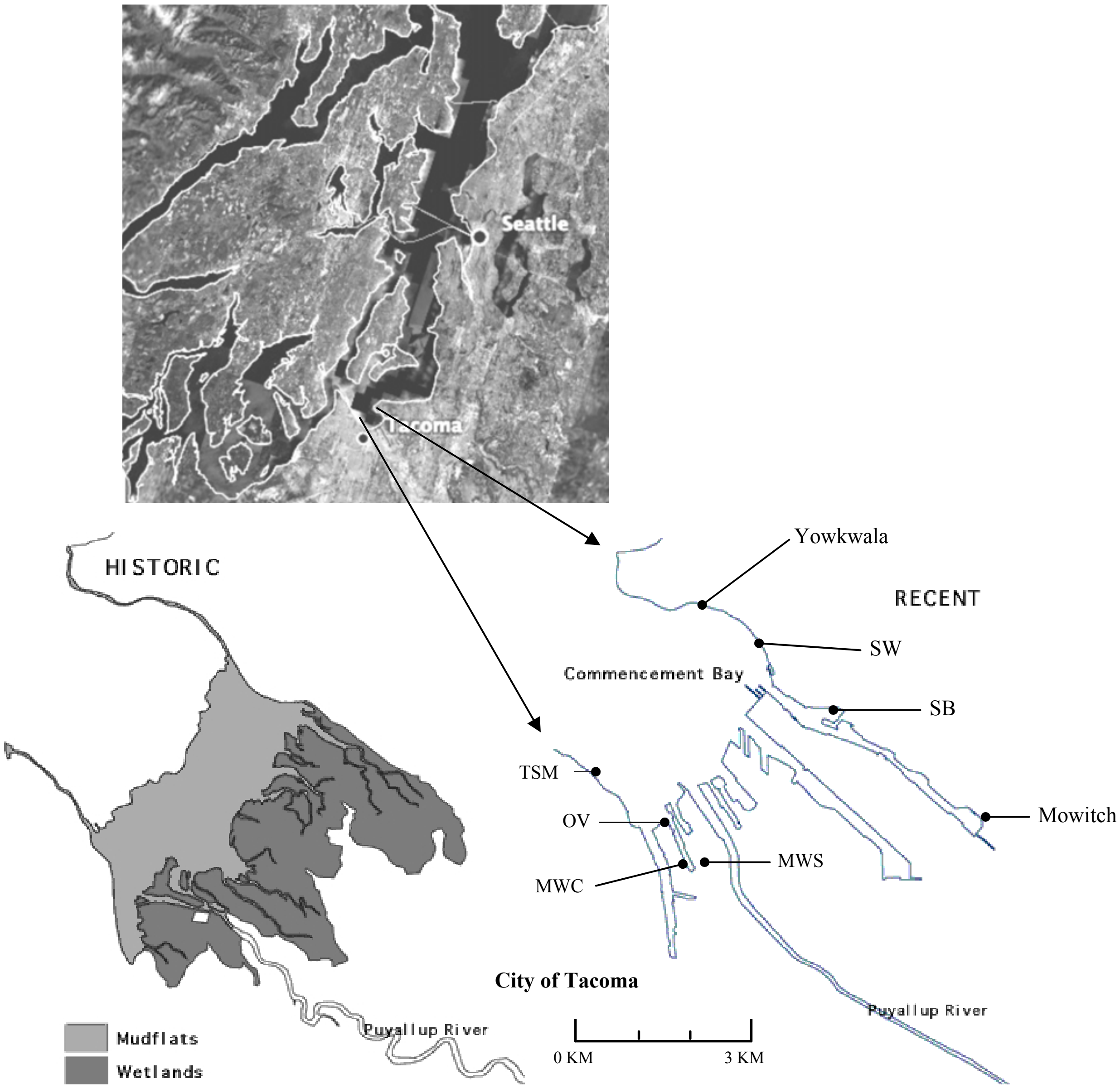

Figure 1. The historical shoreline and habitat of Commencement Bay (left) and current shoreline configuration (right) showing the restoration sites sampled during the 2002 and 2003 field seasons. Legend: TSM = Tahoma Salt Marsh, OV = Olympic View, MWC = Middle Waterway at City, MWS = Middle Waterway at Simpson, SB = Squally Beach, SW = Skookum Wulge. (Maps from http://courses.washington.edu/uwtoce03/webg3.)

chum salmon (*O. keta*) utilizing the Hylebos Waterway, at concentrations associated with immunosuppression and other health effects in previous studies (Collier et al. 1998a, Collier et al. 1998b, Stehr et al. 2000, Arkoosh et al. 2001). English sole (*Parophrys vetulus*) from the Hylebos Waterway had high prevalences of toxicopathic liver lesions and reproductive abnormalities (Collier et al. 1998a, Collier et al. 1998b, Johnson et al. 1999).

As the damage assessment case progressed, the trustees entered into settlement agreements with several potentially responsible parties, who agreed to fund or carry out projects to restore injured resources and habitats. In 1997 a bay-wide restoration plan was adopted (online at http://www.darrp.noaa.gov/pacific/cbay/). Currently, 16 sites in the Commencement Bay area have been or are in the process of being restored. The major goals of these projects are to 1) enhance fish habitat for juvenile salmonids including intertidal areas and migration corridors, 2) establish backwater pools, 3) establish areas for salt marsh vegetation, and 4) otherwise protect the sites for natural resources.

Of these 16 restoration sites, 8 principal nearshore and intertidal sites were selected for monitoring in this study. These sites included Yowkwala, Skookum Wulge, Squally Beach, Tahoma Salt Marsh, Mowitch, Olympic View, Middle Waterway at Simpson (MWS), and Middle Waterway at City (MWC) (Figure 1). These sites range from a set of constructed side channels at a freshwater creek mouth, to an industrialized mud and sand beach where contaminated sediments were removed, to unmanipulated open intertidal beaches. Restoration was completed at all sites by 2002 with the exception of Tahoma Salt Marsh (completed in 2004) and Skookum Wulge (an undeveloped site).

The following is a brief description of the characteristics and restoration history of the sites; more detailed information is available at the NOAA Damage and Restoration Program Web site (http://www.darrp.noaa.gov/pacific/cbay/).

- The Mowitch site (12,464 m^2, restoration completed in 2000), on the Hylebos Waterway, is at the end of this heavily industrialized saltwater channel. The primary onsite activities prior to restoration were log storage and log sorting. The site was maintained with crushed rock and slag from the Asarco smelter during this time, so remedial actions were required to reduce levels of arsenic, copper, lead, and zinc at the site before restoration could proceed. Currently, the sediment in the intertidal area of Mowitch consists of fine sand and mud, with boulders and log debris placed throughout the site during restoration efforts. At low tide, the entire site goes dry. The project site includes a portion of Hylebos Creek that was modified by past dredging, straightening, and filling. As part of the restoration process, the straight stream channel was modified to form three backwater pools and a secondary stream, and native saltwater marsh and riparian plants are being reestablished.
- The MWS site (13,355 m^2, restoration completed in 2000) is located on the Northeast bank of the Middle Waterway on land owned by the Simpson Kraft Company. Historically, the Middle Waterway was heavily industrialized; however, many of the original industries are no longer present. Restoration activities were carried out between 1998 and 2000 to reestablish intertidal, salt marsh, and riparian habitat at the site. As part of the restoration process, the formerly filled land was excavated and contoured to create a natural shoreline with hummocks and other natural marsh features. The top two feet of sand was removed from portions of the site and replaced with amended soil, native vegetation was planted, and anchoring logs were added to prevent erosion and to create a beach environment. The beach and intertidal sediments consist of fine to hard packed mud.

- The MWC site (7,487 m^2, restoration completed in 2001), located at the southwest end of the Middle Waterway, is an intertidal marsh and riparian buffer, created through excavation and regrading of vacant upland property adjacent to the southwest shore of the Waterway. The beach and intertidal sediments consist of fine to hard packed mud. A small creek entering the middle of the site adds an insignificant amount of fresh water.
- Skookum Wulge (4,047 m^2, acquired as an existing undeveloped intertidal site) is located just outside the mouth of the Hylebos Waterway. The beach consists of gravel, cobble-sized rocks, and small boulders (approximately 20–40 cm) from natural unconsolidated glacial till. Residential buildings surround it and the site itself has been undeveloped since 1938. The water area bordering the west side of the site is often used as a log holding area. Active restoration at this site is not being conducted at this time.
- The Yowkwala site (60,702 m^2, restoration completed in 2000) is at the base of a steep bluff between the Tyee Marina and Brown's Point. There is no current development of any kind in the immediate vicinity of the restoration area, although residential buildings line the top of the bluff. Two derelict barges were removed from the shoreline, along with a dilapidated dry dock and other woody debris. Woody debris was placed in front of the marsh to provide a protective barrier and to encourage growth of marsh vegetation. In 2003 fire swept through the area. The beach and intertidal sediments consist of gravel and cobble-sized rocks, with occasional large boulders. This site is also subjected to the strongest tidal currents of all the restoration sites sampled in 2003.
- Olympic View (47,348 m^2, restoration completed in 2002) is located between the ends of two highly industrialized waterways (Middle Waterway and Thea Foss Waterway), and was partially covered by a plywood company building—now removed. Pilings were also removed and the shoreline was softened to enhance the intertidal habitat and to create a riparian buffer. In 2002 the U.S. EPA removed dioxin-contaminated sediments and backfilled the area with clean sediment. The beach and intertidal area now consist of sand and fine gravel. Eelgrass (*Zostera marina*) beds exist below the low tide zone.
- The Squally Beach site (2,671 m^2, restoration completed in 2000) is located along the northern shoreline of the Hylebos Waterway in a relatively unimpacted area of the heavily industrialized Tacoma Tideflats. The project site includes an upland area occupied by hardwood trees and blackberry bushes (*Rubus discolor*), and a strip of intertidal salt marsh vegetation (e.g., Lingbye's sedge [*Carex lyngbyei*]). Prior to restoration, parts of the site were paved with asphalt pads indicating the presence of historical structures, and pilings, logs, and downed wood were present, indicative of previous log storage activities in the vicinity.
- The Tahoma Salt Marsh site (7,891 m^2, restoration completed in 2004) is located on the Ruston Way shoreline within the City of Tacoma and Commencement Bay. The beach and intertidal sediments consisted of large gravel, cobble size rocks, and riprap (before restoration in 2004). The newly created marsh was formed by excavation or regrading of 1.95 acres and the planting of native marsh and riparian vegetation. In addition, a tidal channel was excavated to connect the newly created marsh and the restored beach to permit tidal inundation of the marsh.

A primary goal of aquatic habitat restoration is the recovery and sustainability of specific fisheries resources such as anadromous salmon; a broader goal is to return sites and ecosystems as much as possible to their predevelopment, overall ecological condition. Unfortunately, restoration projects often fail to include the monitoring programs necessary to evaluate success, particularly with respect to long-term biological data (Rice et al. 2005). Determination of fish assemblages can be an excellent tool for the assessment and monitoring of water resources (Simon 1999). In addition to measuring presence or absence and abundance data of fish at restored sites, it is also useful to assess the health status of individual fish and degree of exposure to chemical contaminants in resident fish. This is especially important at sites like Commencement Bay, where not only physical habitat alterations have occurred, but also where chemical contaminants have been shown to adversely affect fish health (Collier et al. 1998a).

In 2001 the National Marine Fisheries Service (NMFS or NOAA Fisheries Service) Northwest Fisheries Science Center's (NWFSC) Environmental Conservation Division began a cooperative fish monitoring program with the NOAA Restoration Center and Ridolfi Inc. to monitor fish use patterns and contaminant exposure in fish at selected restoration sites. Overall, the aim was to help determine whether different restoration projects and techniques were improving habitat and supporting healthy and sustainable fish populations. Monitoring of the restoration sites was initiated in spring of 2002. All sites were sampled for fish assemblage composition and salmonid diets (stomach content analysis). In addition, samples were collected for analysis of chemical contaminant concentrations in sediments, prey (stomach contents) and tissues of a subset of fish species (Chinook, coho [*O. kisutch*], pink [*O. gorbuscha*] and chum [*O. keta*] salmon, and Pacific staghorn sculpin [*Leptocottus armatus*], hereafter referred to as PSS). This technical memorandum presents the results (fish habitat use and contaminant exposure) of the first two years of the ongoing restoration monitoring at these Commencement Bay restoration sites.

Materials and Methods

Study Design

As part of the study design, fish use of restoration sites was assessed by analysis of catch data. Fish collection was conducted from spring to early fall. Because of relatively low fish habitat use in nearshore Puget Sound during late fall and winter (Wingert and Miller 1979, Borton 1982), no fish sampling was planned during those seasons. Species richness, abundance, and catch-per-unit-effort for all species were recorded at all sites, as were length frequency distributions for salmonids and other selected species. In addition, salmonids were examined for fin clips and coded wire tags (CWTs) in order to determine the proportions of marked (known hatchery origin) and unmarked (potentially wild) fish.

To assess contaminant exposure, tissue samples (whole bodies, bile, and stomach contents) were collected from juvenile salmon and a resident fish species (e.g., English sole or PSS) for chemical analysis (Krahn et al. 1986, Krahn et al. 1994, Sloan et al. 2004, Ylitalo et al. 2005a). These body burden and dietary contaminant data were compared to available fish toxicity guidelines, such as residue effects threshold values recently calculated for protecting salmon health (Meador et al. 2002). Taxonomic analysis of fish stomach contents (Cailliet 1977) was also conducted to determine the types and abundance of food organisms that are available to juvenile Chinook salmon.

Sediment samples were collected from all sites for chemical contaminant analysis and compared to sediment quality guidelines and clean-up levels, as well as tissue burdens from fish collected at these sites. Water quality parameters (temperature and salinity) were also measured at each site during each sampling event.

Field Collection

Due to the wide variation in topography among the restoration sites, several types of sampling gear were required to effectively sample all areas. Fish were collected using either a 37 m × 2.4 m (10 mm mesh size) floating "Puget Sound" beach seine (with 20 m polypropylene lines attached at either end) at five sites (Skookum Wulge, Yowkwala, Olympic View, Tahoma Salt Marsh, and MWS), and with block nets at the remaining three sites (Squally Beach, Mowitch and MWC). Two basic block net designs, with a lead line at the bottom of the net, a float line at the top, and equipped with a floating live box, were deployed. Four 37 m × 4.1 m nets made of 10 mm mesh were used at Mowitch, and 9.7 m × 1.8 m nets made of 5 mm mesh in various configurations (e.g, additional wings) were used at MWC (one net), and Squally Beach (four nets). Beach seine sets were deployed using a 17 ft. (5.2 m) Boston Whaler, with three sets performed at all sites per sampling time, as conditions allowed (e.g., tidal current, amounts of drift macroalgae present) except at Skookum Wulge (two sets in 2002, one set in 2003) and Tahoma Salt Marsh (one set).

Due to the presence of a log boom in 2003, the Skookum Wulge site was too small to allow for more than one beach seine set without resampling the same area. Similarly, the Tahoma Salt Marsh site was too small to allow more than one set per sampling period. In addition, due to the formation of a tidal channel that allowed some fish to escape under the net, the block net at the MWC site was abandoned in August of 2003 in favor of the beach seine, with a resulting increase in the numbers of fish captured. The site was too small to allow more than two sets to be made in the same sampling period.

Fish collected were immediately identified to the species level, measured for length (up to 30 specimens for each species measured), and returned live to the water. Fish species with forked tail fins were measured to fork length; total length was used for all other fish measured. Coho salmon, Chinook salmon and steelhead (*O. mykiss*) were also checked for fin clips or CWTs to distinguish between unmarked and hatchery-reared fish. Selected juvenile salmonids (coho, Chinook, and chum) and PSS were collected for stomach contents taxonomy and chemical analyses. Initially, the proposed target species for contaminant analyses were juvenile salmonids and English sole. English sole were to be sampled for chemical analysis as a resident benthic species, but as abundances of English sole were very low at all sites in 2002, it was not possible to obtain enough samples. As a result, PSS were added in 2003 as a replacement resident benthic species. Fish collected for these purposes were usually transported live to the NWFSC facility in Seattle for necropsy and sample collection (samples for stomach taxonomy only were taken on-site). Stomach contents were collected for taxonomic identification of food prey; additional stomach contents, bile and whole bodies were collected for chemical analyses. Fish returned to the NWFSC facility were also measured for length and weight.

At all sites, physical data including water temperature, salinity, and weather conditions were observed and recorded using a thermometer and a refractometer. In addition, a Hydrolab Datasonde 4a water quality multiprobe (Hach Environmental Sales, Loveland, Colorado) was deployed at high tide at the block net sites to collect pH, water temperature, and salinity data at the surface and bottom.

Sampling of all sites began in April of 2002 and 2003, and continued on a biweekly basis until August in 2002 and until September in 2003. Sampling continued on a monthly basis through October of both years. Sampling at the Mowitch site had to be discontinued after the end of August in 2003 because of poor tide conditions. A total of 11 sampling rounds were conducted in both 2002 and 2003, involving 145 and 149 sets respectively (beach seines and block nets, collectively).

Fish Necropsy

In both 2002 and 2003, stomach taxonomy and whole body chemistry samples were taken for juvenile Chinook, coho, chum, and pink salmon at nearly all time points and sites when available. Unmarked Chinook were only taken as a result of incidental catch mortality, so the majority of Chinook sampled for chemical analysis represent hatchery stocks. In addition, in 2003 chemical analyses were expanded to include bile and stomach contents chemistry in Chinook, coho, and PSS. Fish were measured for length and weight, salmonids were checked for signs of hatchery origin (e.g., presence of fin clips or CWTs), and samples of bile, stomach chemistry, or taxonomy, as well as whole bodies, were taken and archived for future chemical analysis following protocols described in Stehr et al. (2000) and Collier et al. (1998a). Stomach taxonomy samples were preserved in 10% neutral buffered formalin for later analysis (Cordell et al. 2001). Samples for chemical analysis were kept on ice while necropsies were conducted, and then transferred to a –80°C freezer where they were stored until analyses could be performed. As no salmon were captured at Squally Beach in 2002 and the site was not sampled in 2003, there were no collections of tissue for chemical analysis for fish from this site.

Sediment Chemistry

Sediment samples for chemical analysis were collected at six of the eight sites (MWS, MWC, Olympic View, Squally Beach, Mowitch, and Skookum Wulge) in 2002. Because the beach and intertidal sediments at Tahoma Salt Marsh and Yowkwala consisted of gravel, cobble, or riprap, sediments for chemical analysis could not be collected. Sediment was collected from three positions, consisting of each end and the middle of the physical area of the site. At each position, three spoonfuls of the top 2 cm of sediment were collected using a large stainless steel spoon rinsed with water and isopropyl alcohol between uses. The samples were placed in an isopropyl rinsed stainless steel bowl and thoroughly mixed before placing approximately 100 gm of the composite sample in solvent-rinsed 4 oz. glass jars. Sediment samples were stored at –20°C until analysis.

Chemical Analyses

Various analytical methods were used to measure chemical contaminants in sediments and fish tissue samples. Sediments and fish stomach contents were analyzed for both a large suite of organochlorines (OCs) and PAHs using a gas chromatography/mass spectrometry (GC/MS) method *(*Krahn et al. 1988, Sloan et al. 2004). Whole bodies of fish were analyzed for dioxin-like PCBs, other selected PCBs, DDTs, and hexachlorobenzene (HCB) using high-performance liquid chromatography with photodiode array detection (HPLC/PDA) (Krahn et al. 1994, Ylitalo et al. 2005a). It was found that the OC data, including summed PCB (∑PCB) and summed DDT (∑DDT) concentrations, obtained by the GC/MS and HPLC/PDA methods, are in good agreement for a wide range of marine biota (Krahn et al. 1994, Ylitalo et al. 2005a). Fish bile samples were analyzed for metabolites of PAHs using high-performance liquid chromatography with fluorescence detection (HPLC/UVF) (Krahn et al. 1986).

GC/MS Analysis of Sediments and Stomach Contents

Persistent organic pollutants (POPs) and PAHs were extracted with dichloromethane from sediments and composite stomach content samples (10–15 fish stomach contents per composite) of Chinook salmon from Yowkwala, Mowitch, Tahoma Salt Marsh, and Skookum Wulge using an accelerated solvent extractor (ASE). The POPs and PAHs were separated from lipid and other biogenic compounds using high-performance size exclusion liquid chromatography and the analytes were analyzed by GC/MS for PAHs, PCB congeners, DDTs, and chlorinated pesticides as described by Sloan et al. (2004). A minimum mass of approximately 0.30 g of fish stomach content was necessary to obtain the desired limits of detection (LODs) for both OCs and PAHs using the GC/MS method. Therefore, dependent upon the stomach content mass of each salmon necropsied in the current study, a salmon stomach content composite sample contains stomach contents of 10–15 salmon in order to attain the minimum mass requirement for GC/MS analysis.

Summed hexachlorocyclohexanes (∑HCHs) were calculated by adding the concentrations of alpha-HCH, beta-HCH, and lindane. ∑PCBs were calculated by adding the concentrations of 17 PCB congeners (PCBs 18, 28, 44, 52, 66, 101, 105, 118, 128, 138, 153, 170, 180, 187, 195, 206, and 209) and then multiplying the sum by 2 as recommended by the NOAA National Status and Trends (NSandT) Program (Lauenstein et al. 1993). This method for estimating total PCB concentrations was found to provide similar values as those determined by summing PCBs based on chlorination level. The ∑DDTs levels were calculated by summing the concentrations of *p,p*'-DDT, *p,p*'-DDE (dichlorodiphenyldichloroethylene), *p,p*'-DDD (dichlorodiphenyldichloroethane), *o,p*'-DDD, *o,p*'-DDE, and *o,p*'-DDT. Summed chlordanes (∑CHLDs) were determined by adding the concentrations of heptachlor, heptachlor expoxide, alpha-chlordane, gamma-chlordane, oxy-chlordane, *cis*-nonachlor, *trans*-nonachlor, and nonachlor III.

Low (2–3 ring) and high (4–6 ring) molecular weight aromatic compounds were also measured in sediments and stomach contents samples using capillary column GC/MS, as described above (see Appendix C for list of analytes). Summed low molecular weight PAHs (∑LMWAHs) were determined by adding the concentrations of biphenyl, naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, 2,6-dimethylnapththalene, acenaphthene, fluorene, phenanthrene, 1-methylphenanthrene, retene (a wood-product derivative often associated with pulp mill effluent and logging operations), and anthracene. Summed high molecular weight PAHs (∑HMWAHs) were determined by adding the concentrations of fluoranthene, pyrene, benz[a]anthracene, chrysene + triphenylene, benzo[b]fluoranthene, benzo[j]fluranthene + benzo[k]fluoranthene, benzo[a]pyrene, benzo[e]pyrene, perylene, indenopyrene, dibenz[a,h+a,c]anthracene, and benzo[ghi]perylene. Summed PAHs were determined by adding the concentrations of ∑LMWAHs and ∑HMWAHs.

Whole body samples of salmon and PSS from various sites in Commencement Bay (see Appendices D, E, F, and G for description of compositing scheme) were analyzed for selected OCs, including dioxin-like PCBs and DDTs, by HPLC/PDA (Krahn et al. 1994). For the salmon samples, tissue (3 g), hexane/pentane (1:1 v/v), sodium sulfate (5g), and a surrogate standard were mixed using a tissue homogenizer described in Krahn et al. (1994). OCs were extracted from the PSS with dichloromethane by ASE (see Sloan et al. 2004 for details). These analytes were separated from interfering compounds (e.g., lipids, aromatic compounds) on a gravity flow cleanup column that contained neutral, basic, and acidic silica gels eluted with hexane/dichloromethane (1:1 v/v). Prior to the cleanup step, a 1 ml aliquot of each sample extract was removed for lipid quantitation by thin layer chromatography with flame ionization detection (TLC/FID) (Krahn et al. 2001, Ylitalo et al. 2005b). Dioxin-like PCB congeners (PCBs 77, 105, 118, 126, 156, 157, 169, 189) were resolved from eight other selected PCBs (PCBs 101, 128, 138, 153, 170, 180, 190, 200) and other selected OCs [*o,p'*-DDD, *p,p'*-DDD, *p,p'*-DDE, *o,p'*-DDT, *p,p'*-DDT, HCB] by HPLC on two Cosmosil PYE analytical columns. These OCs were measured by an ultraviolet (UV) photodiode array detector.

∑PCBs were calculated using the following formula: ∑PCBs = ∑concentrations of 16 PCBs listed above (based on individual response factor) + ∑concentrations of other PCB congeners (calculated by summing areas of peaks identified as PCBs and using an average PCB response factor). ∑DDTs concentrations were calculated by summing the concentrations of five DDTs (*o,p′*-DDD, *p,p′*-DDD, *p,p′*-DDE, *o,p′*-DDT, *p,p′*-DDT). Using the concentrations of the individual dioxin-like PCBs, toxic equivalent (TEQ) values were calculated by multiplying the molar concentration of each dioxin-like PCB by the appropriate toxic equivalency factor (TEF) recommended by the World Health Organization (van den Berg et al. 1998). Proportions of lipid classes (i.e., wax esters, triglycerides, free fatty acids, cholesterol, polar lipids) and percent lipid concentrations of fish whole body samples were determined by thin-layer chromatography with TLC/FID. Total lipid concentrations were calculated by adding the concentrations of five lipid classes (i.e., wax esters, triglycerides, free fatty acids, cholesterol, and polar lipids) for each sample and were reported as percent total lipid.

Quality Assurance

To monitor the accuracy of the GC/MS, TLC/FID, and HPLC/PDA methods, a National Institute of Standards and Technology (NIST) standard reference material (blue mussel homogenate SRM1974b, fish homogenate SRM1946, or sediment SRM1941b) was analyzed with each sample set and results met laboratory criteria (Sloan et al. 2006). Approximately 10% of the samples were analyzed in duplicate to measure precision of the method, and the laboratory quality assurance criteria were met for all analytes detected in the samples. Method blanks also met laboratory criteria. The percent recoveries of the surrogate standards in all samples analyzed ranged from 68 to 113%, which are within the acceptable range listed in NWFSC's Laboratory Quality Assurance Plan (Sloan et al. 2006). For sediment samples analyzed by GC/MS, the lower limits of quantitation (LLOQ) for PCB congeners ranged from <0.17 to <0.77 ng/g dry wt, for DDTs ranged from <0.17 to <0.40 ng/g dry wt, for chlordanes ranged from <0.17 to <1.9 ng/g dry wt, and for other OC pesticides ranged from <0.17 to <6.9 ng/g dry weight. In stomach contents the LLOQ for PCBs ranged from <0.15 to <1.4 ng/g wet wt, for DDTs ranged from <0.63 to 2.1 ng/g wet wt, for chlordanes ranged from <0.61 to <2.1 ng/g wet wt, and for other OC pesticides ranged from <0.61 to <5.6 ng/g wet wt.

Determination of PAH Metabolites in Fish Bile

In 2003 bile samples were collected from juvenile Chinook salmon at two Commencement Bay sites (Yowkwala and Skookum Wulge). Additionally bile samples were collected from PSS at five Commencement Bay sites (Mowitch, Yowkwala, Skookum Wulge, MWS, and MWC). The minimum volume of bile needed for PAH metabolite analysis is 25 μL. The volume of bile of each juvenile Chinook salmon was quite small (<3 μL), and thus bile samples from 10 to 15 individual salmon (based on site) were composited to achieve the minimum sample volume of 25 μL. The volume of bile collected from individual PSS, on the other hand, was >25 μL; therefore bile samples of individual PSS were analyzed in the current study. Bile samples were analyzed by high performance liquid chromatography (HPLC) for PAH metabolites as described in Krahn et al. (1986). In this method, bile is injected directly onto a C18 reverse-phase column (Phenomenex Synergi Hydro, Torrance, California) and eluted with a linear gradient from 100% water (containing a trace amount of acetic acid) to 100% methanol at a flow of 1.0 mL/min. Chromatograms were recorded at the following wavelength pairs: 1) 260/380 nm where several 3–4 ring compounds (e.g., phenanthrene, dibenzothiophene) fluoresce and 2) 380/430 nm where 4–5 ring compounds (e.g., benzo[a]pyrene [BaP]) fluoresce.

Biliary PAH metabolites fluorescing at phenanthrene (PHN) wavelengths were considered an indicator of exposure to low molecular weight PAHs, while metabolites fluorescing at BaP wavelengths were considered as an indicator of exposure to high molecular weight PAHs.

Statistical Methods

Intersite differences in tissue, stomach contents, and bile contaminant concentrations were determined by analysis of variance (ANOVA) and tested for significance difference using Tukey–Kramer honestly significant difference (Zar 1999). The Tukey–Kramer test was employed to test for differences among means for this particular set of data because it is robust for analyses in which sample sizes are small and unequal (Zar 1999). Species composition, abundance, and lengths (juvenile salmonids and PSS) at the sampling sites were analyzed for statistical differences using one-way ANOVA and using Fishers Protected LSD test for significant differences (Zar 1999). The significance level for all analyses was set at $\alpha = 0.05$.

Results

Physical Parameters

Surface water temperatures at all sites increased from April to July or August, and then declined through the end of the sampling year in 2002 (Table 1). Temperatures among sites ranged from 8–10°C in April to 14–23°C in July and August. Surface temperatures in 2003 (Table 2) also increased from April to August at all sites except Olympic View, where the temperature peaked in July. Temperatures in April were slightly warmer in 2003 than in 2002 (11.5 vs. 9.5°C), but otherwise were very similar to temperatures in 2002 throughout the sampling year.

Surface salinity was generally in the 15–30 parts per thousand (ppt) range (Table 3 and Table 4), with lowest values in April. Surface salinity in 2002 ranged from a low of 5 ppt at Mowitch in April to a high of 33 ppt at Tahoma in August. Mowitch and Tahoma Salt Marsh had the lowest and highest average salinity (12.4 ppt and 25.1 ppt, respectively) in 2002. Overall, salinity tended to rise from April to August, then fluctuated slightly thereafter.

Table 1. Surface temperatures (°C) at all sites by month in 2002 (NS = site not sampled, NR = data not recorded).

Site	April	May	June	July	August	September	October
Middle Waterway at City	10.00	11.50	16.00	20.00	17.20	15.00	12.00
Middle Waterway at Simpson	9.80	12.50	15.00	20.00	NS	16.00	12.00
Mowitch	8.70	12.50	15.10	17.10	17.60	NS	NS
Olympic View	9.00	11.50	16.00	15.50	15.50	13.00	NR
Skookum Wulge	9.50	11.30	15.50	16.00	17.20	15.00	12.50
Yowkwala	8.25	11.30	15.00	17.00	17.20	13.00	NR
Squally Beach	NS	NS	19.00	23.00	NS	16.00	12.50
Tahoma Salt Marsh	NS	12.50	15.00	14.00	15.50	NS	NR

Table 2. Surface temperatures (°C) at all sites by month in 2003 (NS = site not sampled, NR = data not recorded).

Site	April	May	June	July	August	September	October
Middle Waterway at City	9.70	10.30	15.50	14.90	19.00	15.50	13.30
Middle Waterway at Simpson	NR	11.90	16.80	16.50	19.00	15.50	13.30
Mowitch	11.50	12.40	15.70	17.20	18.80	NS	NS
Olympic View	10.50	13.10	15.20	17.00	17.00	16.10	14.40
Skookum Wulge	10.80	11.45	15.00	15.50	17.50	16.60	13.90
Yowkwala	10.00	13.80	15.00	16.10	17.50	16.60	14.40

Table 3. Salinity (parts per thousand) at each site by month for 2002 (NS = site not sampled, NR = data not recorded).

Site	April	May	June	July	August	September	October	Average
Middle Waterway at City	24.00	24.00	20.00	20.00	28.00	24.00	28.00	24.00
Middle Waterway at Simpson	15.00	24.00	28.00	20.00	NS	22.00	25.00	22.30
Mowitch	5.00	8.50	14.90	19.80	14.00	NS	NS	12.40
Olympic View	20.00	17.00	17.50	26.50	31.00	14.00	NR	21.00
Skookum Wulge	16.00	23.70	16.00	23.00	20.00	19.00	23.00	20.10
Yowkwala	18.00	15.00	16.50	21.00	20.00	22.00	NR	18.80
Squally Beach	NS	NS	14.00	19.00	NS	24.00	23.00	20.00
Tahoma Salt Marsh	NS	18	20.30	30.00	32.00	NS	NR	25.10

Table 4. Salinity (parts per thousand) at each site by month for 2003 (NS = site not sampled, NR = data not recorded).

Site	April	May	June	July	August	September	October	Average
Middle Waterway at City	20.40	28.70	17.90	20.40	28.00	34.00	31.00	25.80
Middle Waterway at Simpson	NR	28.70	20.50	20.00	28.00	34.00	31.00	27.00
Mowitch	14.90	20.95	22.00	20.10	21.80	NS	NS	20.00
Olympic View	27.80	26.50	19.70	25.00	24.50	33.00	34.00	27.20
Skookum Wulge	22.50	21.25	16.10	18.50	23.00	30.00	30.00	23.10
Yowkwala	20.00	21.80	18.25	19.90	24.90	27.00	31.00	23.30

Mowitch and Olympic View had the lowest and highest salinity (20 ppt and 27.2 ppt, respectively) in 2003. Overall, salinity increased slightly from April to May, then decreased in June at all sites except Mowitch before increasing to maximum level in September and October. Of the sampling sites, Mowitch (due to Hylebos Creek which it borders) and Yowkwala tended to have the lowest salinities, and the Middle Waterway sites, Olympic View, and Tahoma Salt Marsh had the highest.

Fish Habitat Use

Species Richness

All sites were utilized by fish, but the number and types of fish present varied with time and site (Figure 2 and Figure 3). A complete list of all species captured in both sampling years is presented in Appendix A.

Overall, the total number of species captured was lower in 2003 than 2002. If all sites sampled are included, the decline was from 38 to 32 species; excluding Squally Beach and Tahoma Salt Marsh (which were not sampled in 2003), the decline was from 37 to 32 species. This occurred even though, excluding Squally Beach and Tahoma Salt Marsh, more sampling sets were deployed in 2003 than in 2002 (149 vs. 123 respectively). However, six of the species

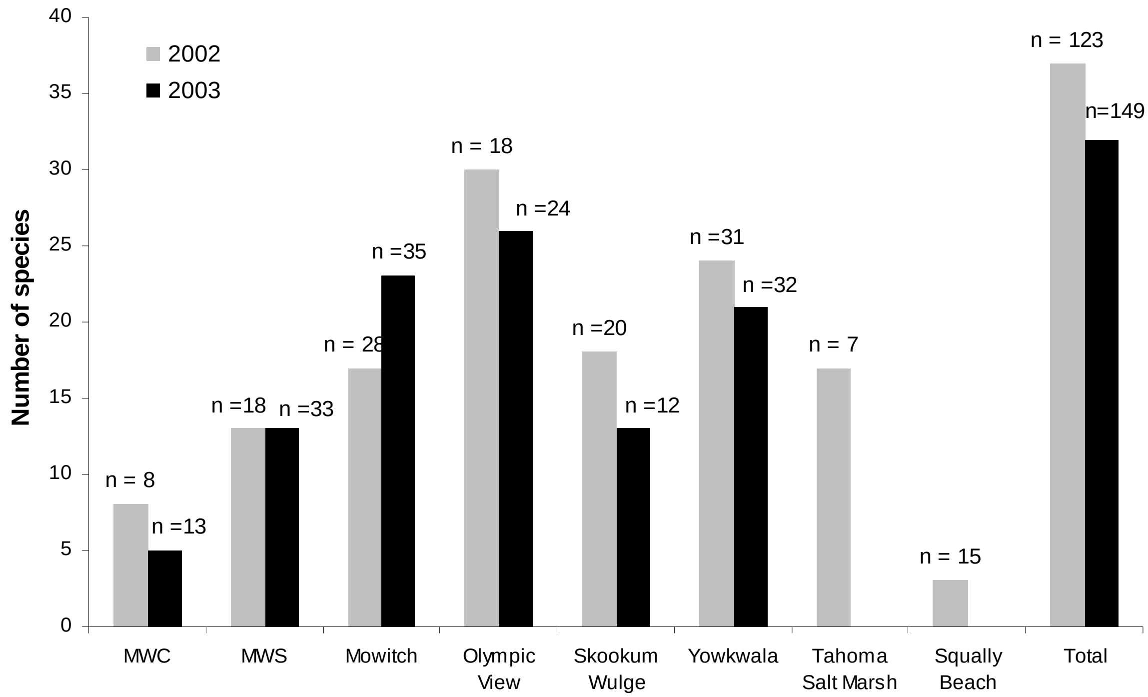

Figure 2. The total number of species captured at each site in 2002 and 2003 (not adjusted for fishing effort, n = number of samples included in the analysis, Tahoma Salt Marsh and Squally Beach were not sampled in 2003).

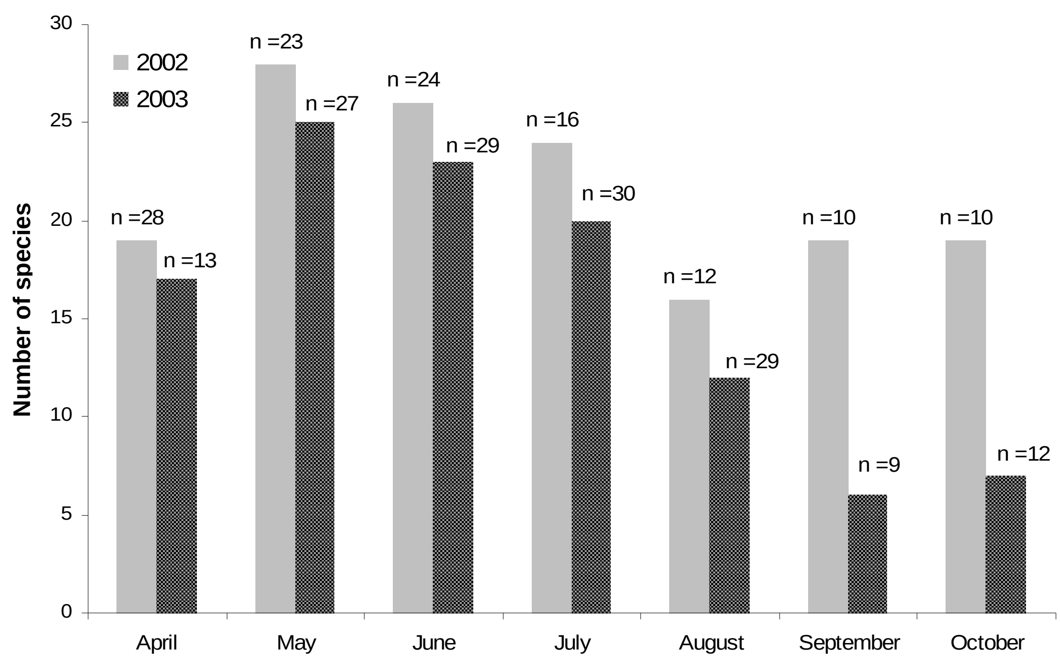

Figure 3. The number of species captured over time in 2002 and 2003 (excluding Squally Beach and Tahoma Salt Marsh, not adjusted for fishing effort, n = number of samples included in the analysis).

captured in 2002 consisted of only one individual for each species, while two of the species captured only in 2003 consisted of one individual. The number of species captured in the two years is much more consistent (31 species vs. 30 species) if these rarely captured species are excluded from the comparison. At the individual sampling sites, the only increase in number of species in 2003 was at Mowitch, with no change at MWS, while at all other sampling sites the number of species collected declined in 2003 compared to 2002 (Figure 2 and Figure 3).

Overall, Squally Beach (2002) had the lowest total number of species (three) captured at any site. At sites sampled in both years, the total number of species caught was lowest at the MWC site where 5–8 species were observed, and highest at Olympic View where 26–30 species were observed. The sites in increasing order of species richness (the number of species captured) at sites sampled in both years were MWC, MWS, Skookum Wulge, Mowitch, Yowkwala, and Olympic View (Figure 2 and Figure 3).

To reduce potential bias introduced by differences in sampling effort, species counts were analyzed per unit effort (per 1,000 square meters) to determine species density (Gotelli and Caldwell 2001). Species density (Figure 4) was significantly lower ($p = 0.0013$) in 2003 than in 2002 (excluding Squally Beach and Tahoma Salt Marsh for year-to-year comparisons). Olympic View had the highest mean species density and Mowitch the lowest of any sites for either sampling year. Olympic View, Yowkwala, and MWC had a higher species density in 2002 than 2003, while MWS, Skookum Wulge, and Mowitch had a higher species density in 2003 than 2002. Species density was significantly lower in 2003 than in 2002 at Olympic View ($p = 0.0001$) and Yowkwala ($p = 0.0166$).

In both 2002 and 2003 total species and species density peaked in late spring and early summer, then steadily declined (Figures 5 through 7) to April levels by August. In 2002 species density remained stable at August levels in September and October, while in 2003 it continued to decline until September. The number of species at all individual sites except MWC in 2003 generally increased in both years from May to June or July, then declined through October (Figures 5 through 7). Olympic View (2002) and MWC (2003) species densities tended to rise until September and October, respectively. Overall, species density generally declined from 2002 to 2003 (Figures 5 through 7) in all sampling periods at all sites except Skookum Wulge.

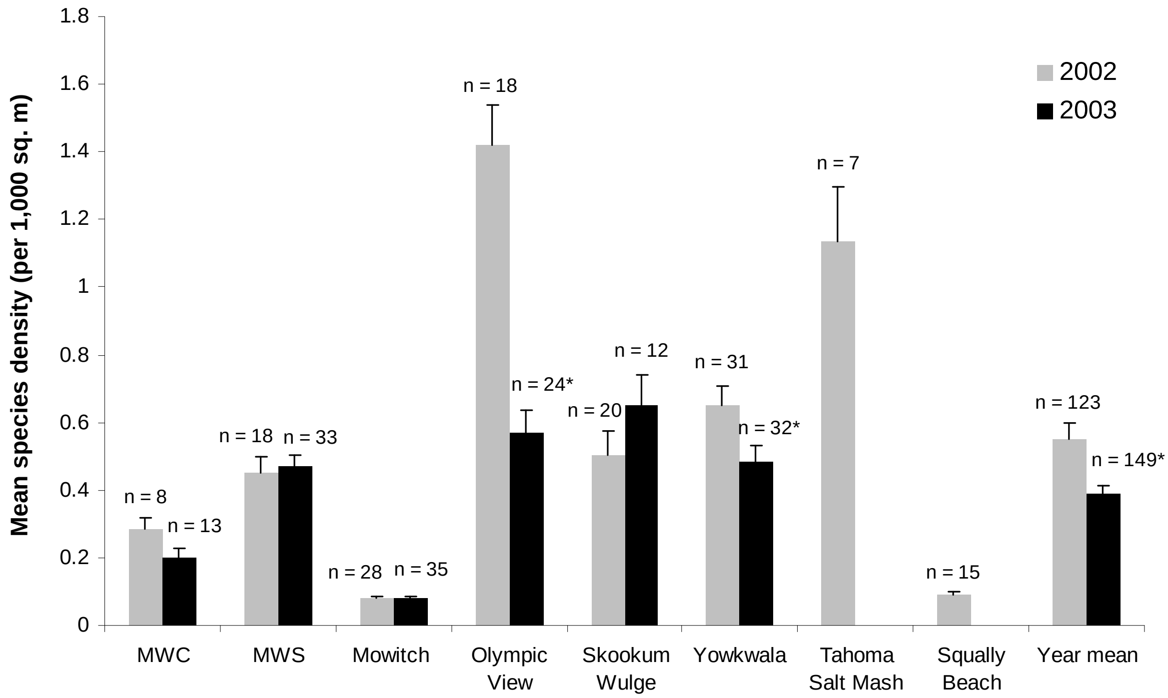

Figure 4. The number of species captured per unit effort (species density) in 2002 compared to 2003 (± Standard Error, asterisk = significantly lower than prior year, Tahoma Salt Marsh and Squally Beach not included in the year mean, n = number of samples included in the analysis).

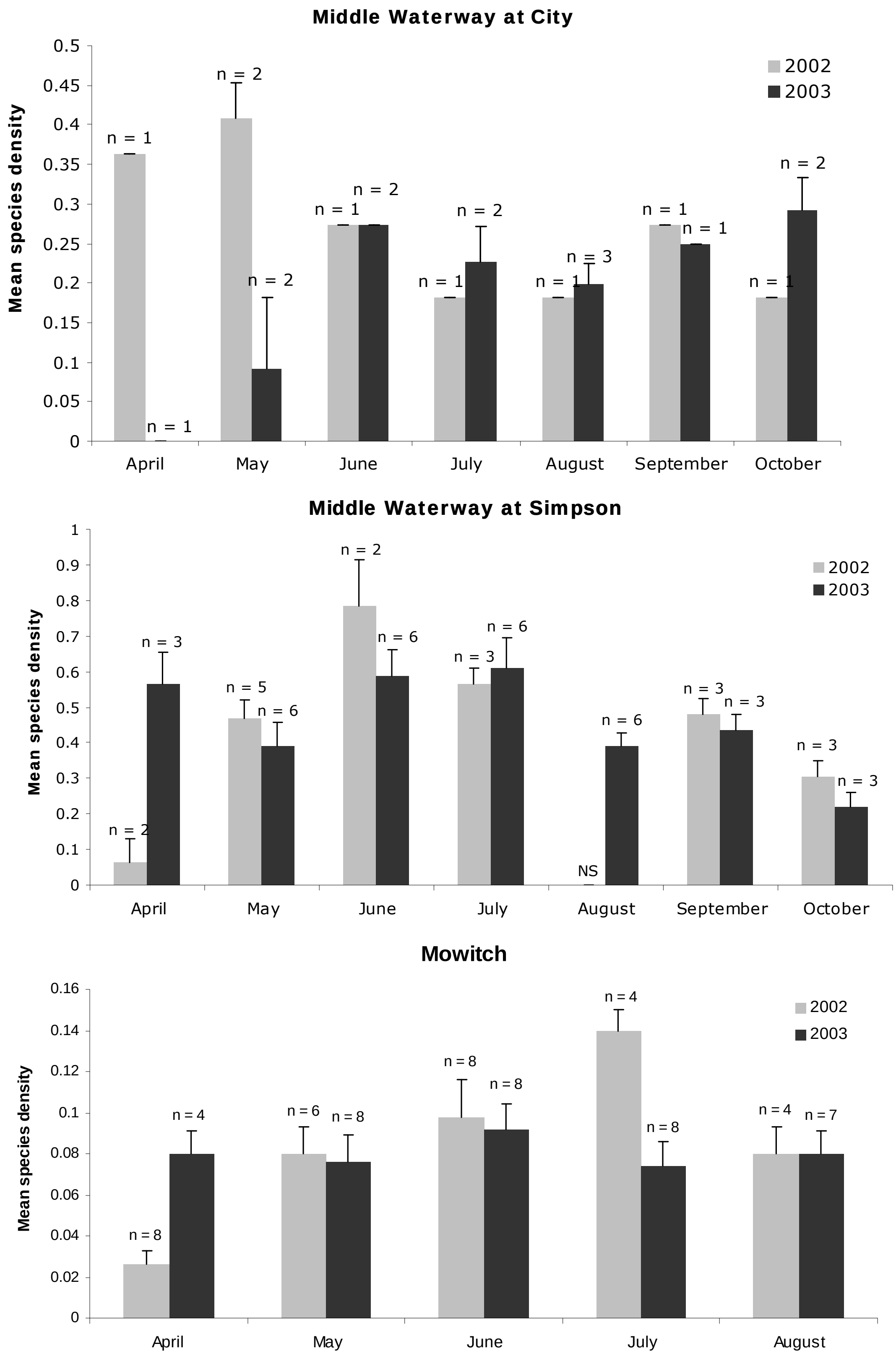

Figure 5. The mean species density (per 1,000 sq. m) captured at three sites by month in 2002 and 2003 (NS = not sampled, n = number of samples included in the analysis).

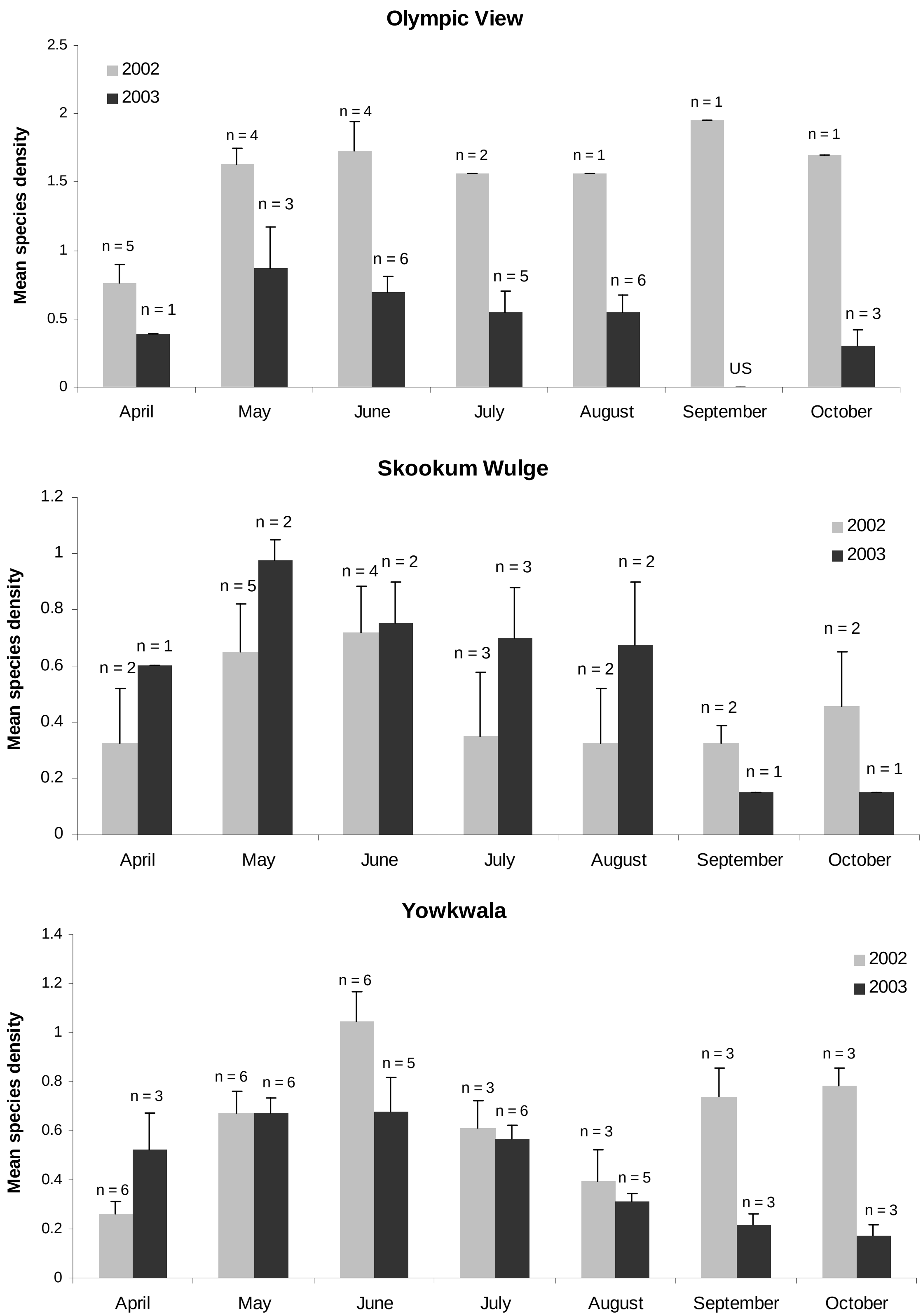

Figure 6. The mean species density (per 1,000 sq. m) captured at three sites by month in 2002 and 2003 (US = unable to sample, n = number of samples included in the analysis).

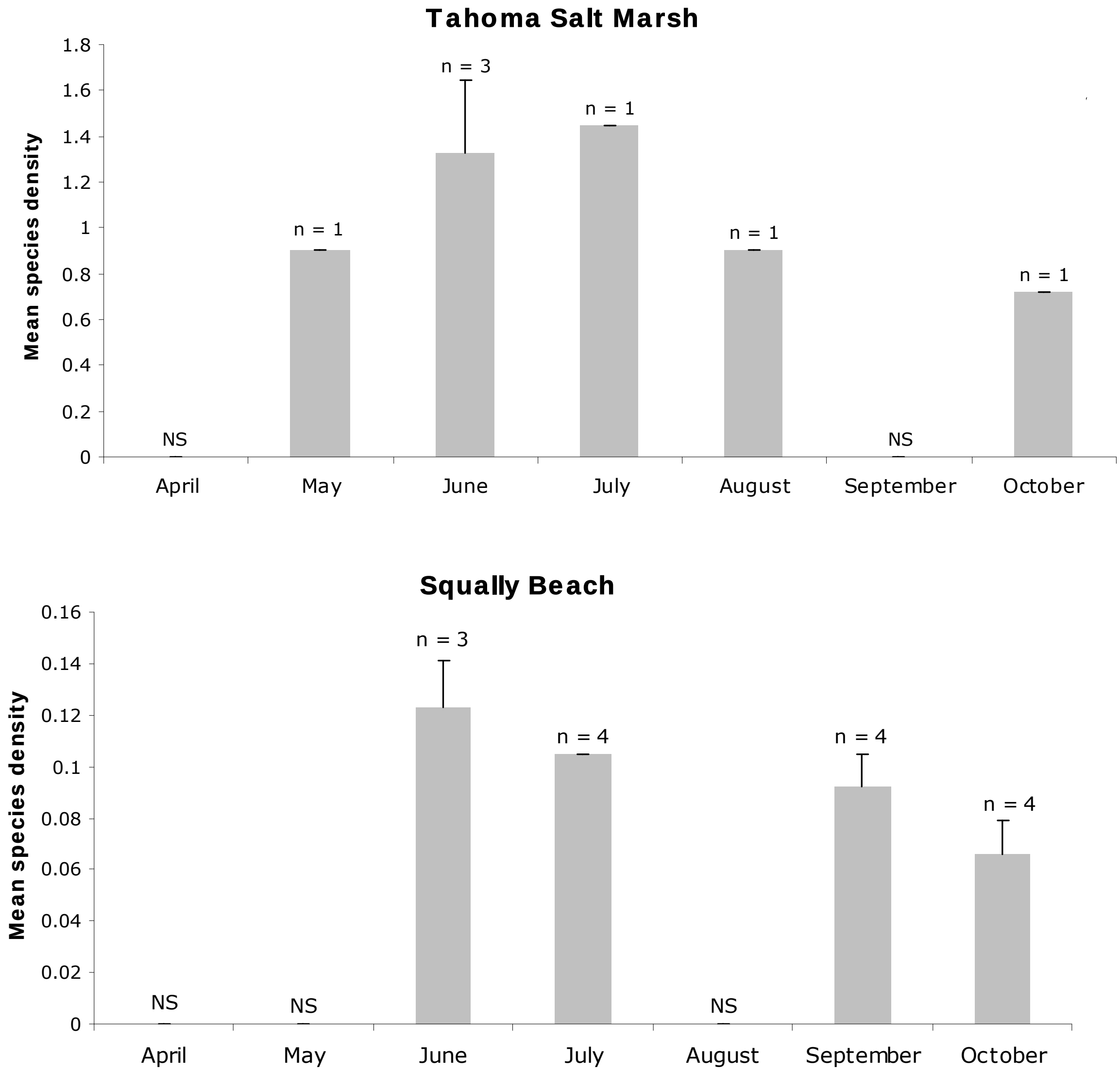

Figure 7. The mean species density (per 1,000 sq. m) captured at two sites by month in 2002 (NS = not sampled, n = number of samples included in the analysis).

Fish Abundance

The total number of individual fish captured at the Commencement Bay sampling sites ranged from a low of 603 at Skookum Wulge in 2003 to a high of 24,220 at Mowitch in 2003 (Figure 8, top panel). Fish abundance was greatest at MWS and Mowitch, lowest at Skookum Wulge and intermediate at the other three sites. Overall, the total number of individual fish captured was higher in 2003 than 2002. On an individual site basis, fish abundance increased from 2002 to 2003 at MWS and Mowitch, but declined at all the other sites.

When adjusted for fishing effort (number of fish captured per 1,000 sq. meters [Figure 8 bottom panel]), the fish density was highest at MWC, MWS, and Mowitch, and lowest at Squally

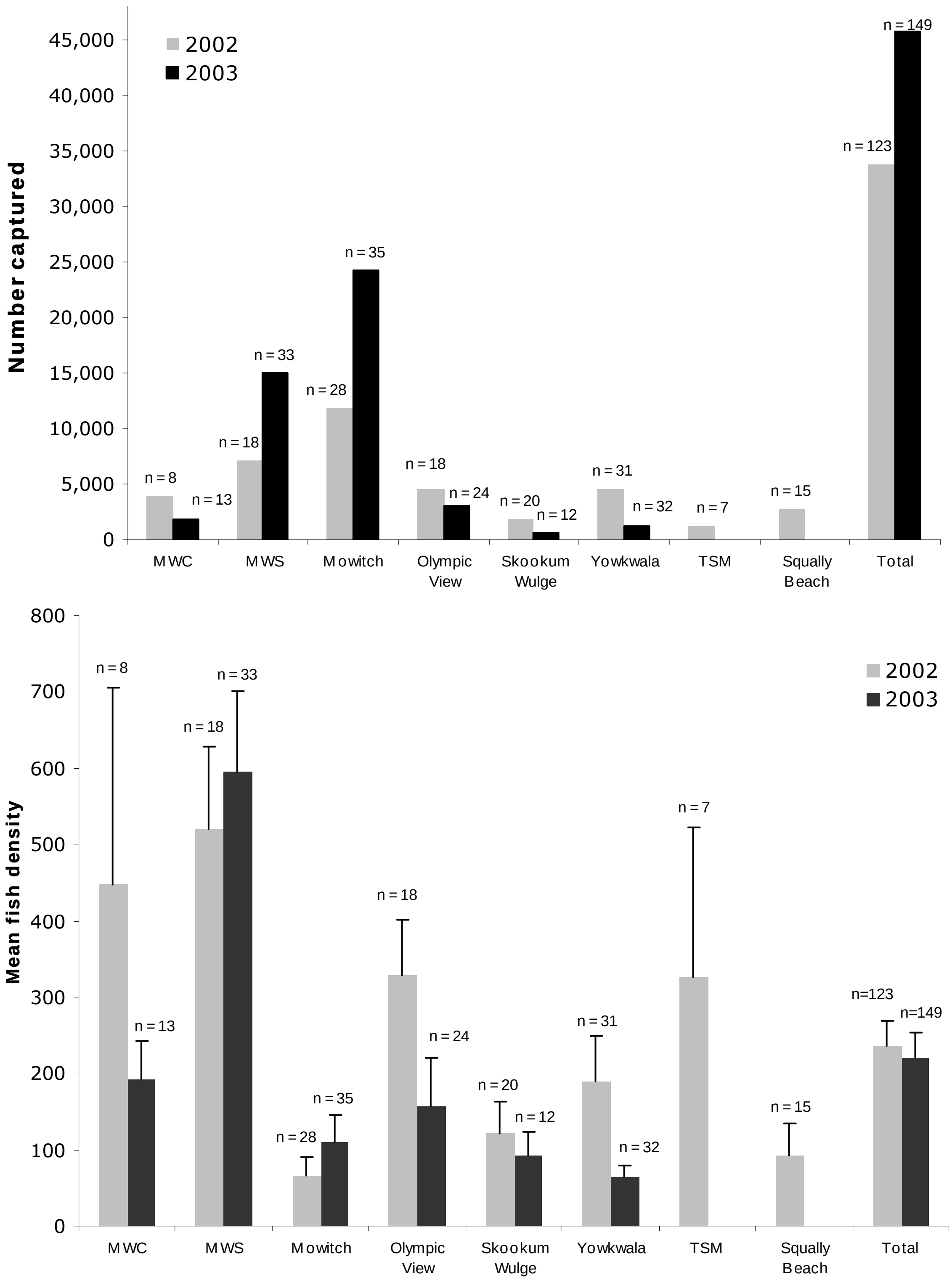

Figure 8. Top panel, the total number of individual fish of all species captured by site in 2002 and 2003 (not adjusted for fishing effort). Bottom panel, the mean fish density (per 1,000 sq. m) of all fish captured by site in 2002 and 2003 (± SE, TSM and Squally Beach not included in year-to-year comparison, n = number of samples included in the analysis).

Beach, Skookum Wulge, and Yowkwala in 2002. Density patterns were similar between 2002 and 2003. In 2003 the density of fish captured was highest at MWS and MWC, and lowest at Skookum Wulge and Yowkwala. At MWS and Mowitch, fish density increased from 2002 to 2003 but declined at all the other sampling sites, with the most dramatic change at the MWC site.

The increase in total fish density at Mowitch and MWS from 2002 to 2003 was due mainly to the capture of shiner perch (*Cymatogaster aggregata*), which comprised by far the largest number of individual fish captured in both sampling years (Figure 9). Between 2002 and 2003, the number of shiner perch collected increased dramatically, from around 22,000 to almost 40,000, although the total number of sets was higher in 2003 than in 2002. In contrast, the numbers of most other species captured including salmonids declined. The total fish density was not significantly different in 2002 and 2003. However, if shiner perch are removed from the comparison, the fish density in 2002 was significantly higher ($p = 0.0144$) than in 2003 (Figure 9).

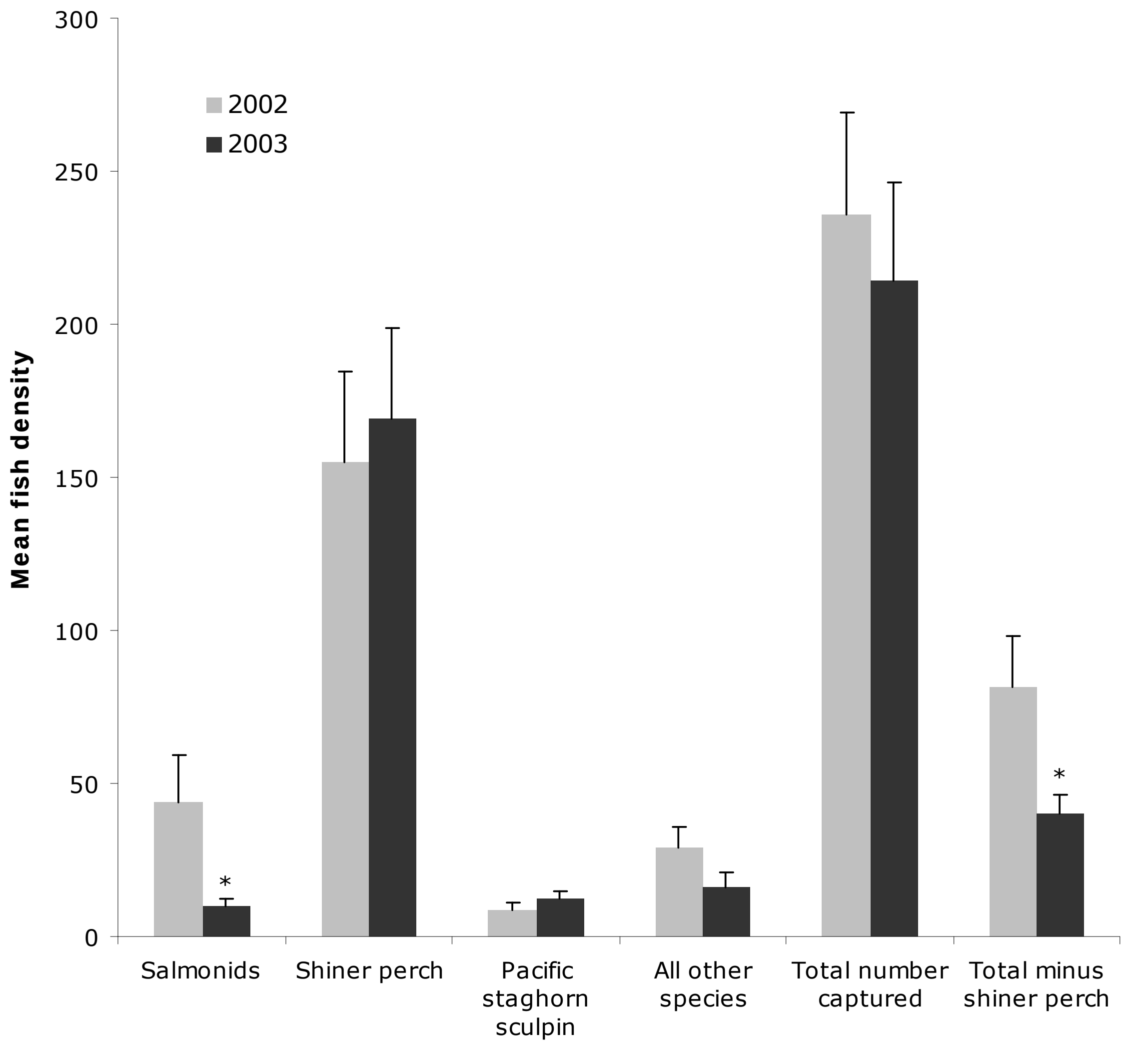

Figure 9. A comparison of mean fish density (per 1,000 sq. m) of individuals of various groups of species captured in 2002 compared to 2003 (2002 data does not include Squally Beach or Tahoma Salt Marsh for year-to-year comparisons, ± SE, asterisk = significantly lower than prior year).

Species Composition

Appendix A lists all species caught at each site for the two sampling years, and summary data are presented in Figure 10 and Figure 11. The most frequently captured species in both 2002 and 2003 were shiner perch, PSS, juvenile coho salmon, and surf smelt (*Hypomesus pretiosus*). Shiner perch were the dominant species at the MWC, MWS, Mowitch, and Olympic View sites. Other prevalent fish species at these four sites included PSS and surf smelt. While juvenile salmonids were observed at these four sites, they made up a very small proportion of the catch (5% or less). Shiner perch were less prevalent at Skookum Wulge and Yowkwala. Juvenile salmonids accounted for a high proportion of fish collected at these two sites. Surf smelt were also relatively common at MWC (2003), MWS, Olympic View (2003), Skookum Wulge, and particularly at Yowkwala.

Like the total number of species, fish abundance generally peaked in June at all sites combined in 2002 and 2003 (Figure 10, top panel). When adjusted for fishing effort, the peak fish density occurred in July of 2002 (Figure 10, bottom panel) and in June of 2003. At individual sites, in 2002 fish density peaked at most sites except Olympic View between May and July (Figure 11, top panel), declined until August, then remained relatively stable to the end of the sampling year. Similarly, in 2003 the fish density at individual sites peaked between May and July, then declined at all sites until August, except at MWC (Figure 11, bottom panel), where fish density remained fairly stable through October.

The species collected were similar in 2002 and 2003, but some changes in species composition at the sampling sites were observed (Figures 12 through 15). The most dramatic change in species from 2002 to 2003 occurred with shiner perch (declined from 94% to 62% at MWC and increased from 70% to 94% at Mowitch), PSS (increased from 2% to 25% at MWC and declined from 23% to 2% at Mowitch), surf smelt (increased from none to 12% at MWC, declined 28% to 12% at Skookum Wulge, and increased from 5% to 22% at Yowkwala), coho salmon (decreased from 54% to 31% at Yowkwala), and all salmonids (increased from 26% to 36% at Skookum Wulge).

Analysis of the functional grouping (as defined by Toft et al. 2004, see Appendix B herein) of fish species again demonstrates that in both sampling years, surfperches (*Embiotocidae spp.*) were the major species collected at the restoration sites (Figure 16). Salmonids and other demersal fish represented the next largest functional groupings, while forage fish, flatfish, and other nearshore species groupings were a relatively small percentage in both years (Squally Beach and Tahoma Salt Marsh, sampled in 2002 only, were excluded from the year-to-year comparison). Surfperches and flatfish were the only functional groups that increased as a percentage of the total species composition from 2002 to 2003. However, differences existed at individual sites between 2002 and 2003 (Figure 17). Surfperches dominated the species composition at the Middle Waterway sites, Squally Beach, Mowitch, and Olympic View, while juvenile salmon and forage fish were a large percentage of the groupings at Skookum Wulge, Yowkwala, and Tahoma Salt Marsh (Figure 17). While some year-to-year variation existed, the sites were consistent in the functional groups of species captured between 2002 and 2003.

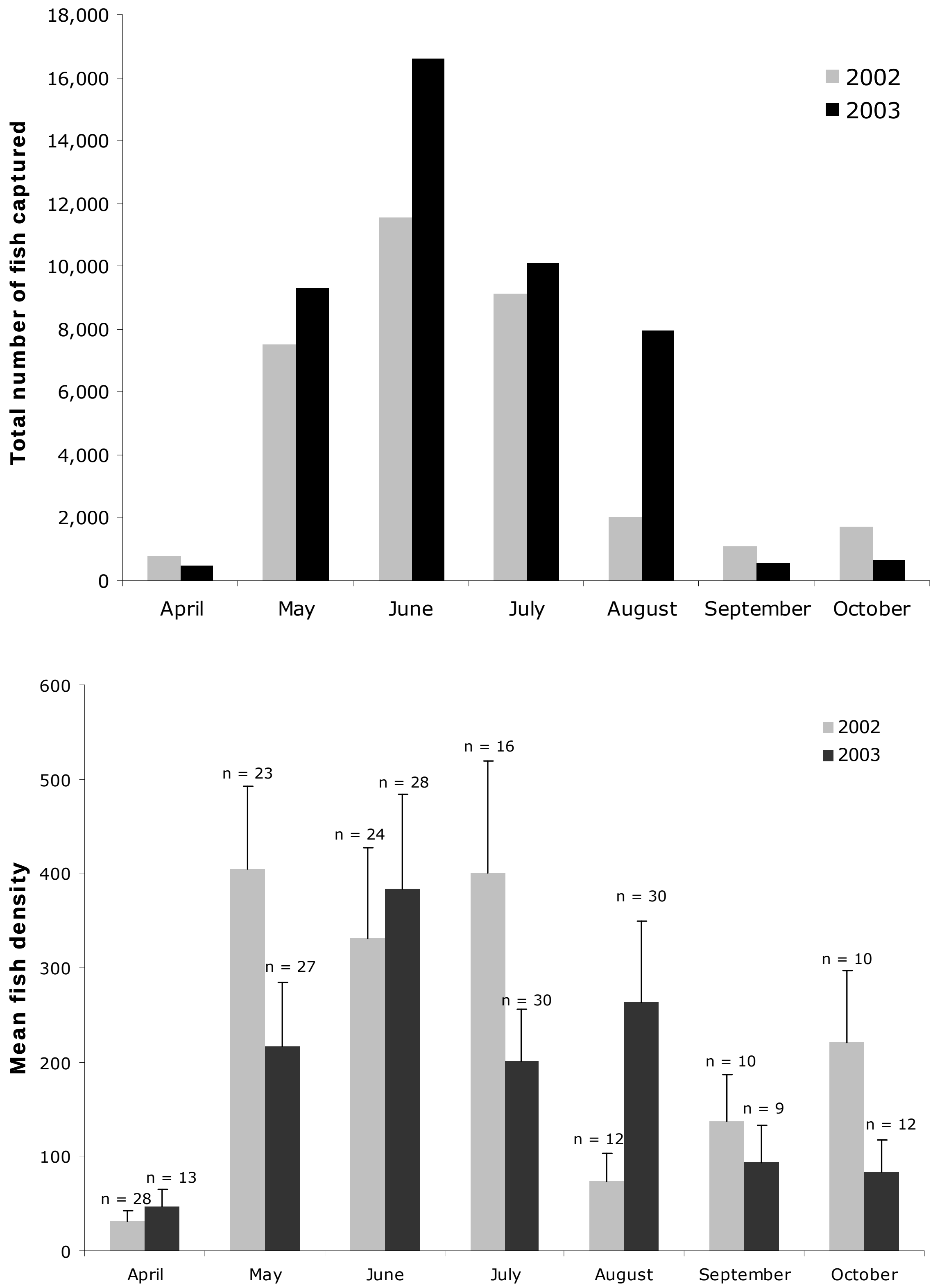

Figure 10. The total number of fish captured by month (top panel) and the mean fish density by month (bottom panel) in 2002 and 2003 (± SE, Squally Beach and Tahoma Salt Marsh not included for year-to-year comparisons, n = number of samples included in the analysis).

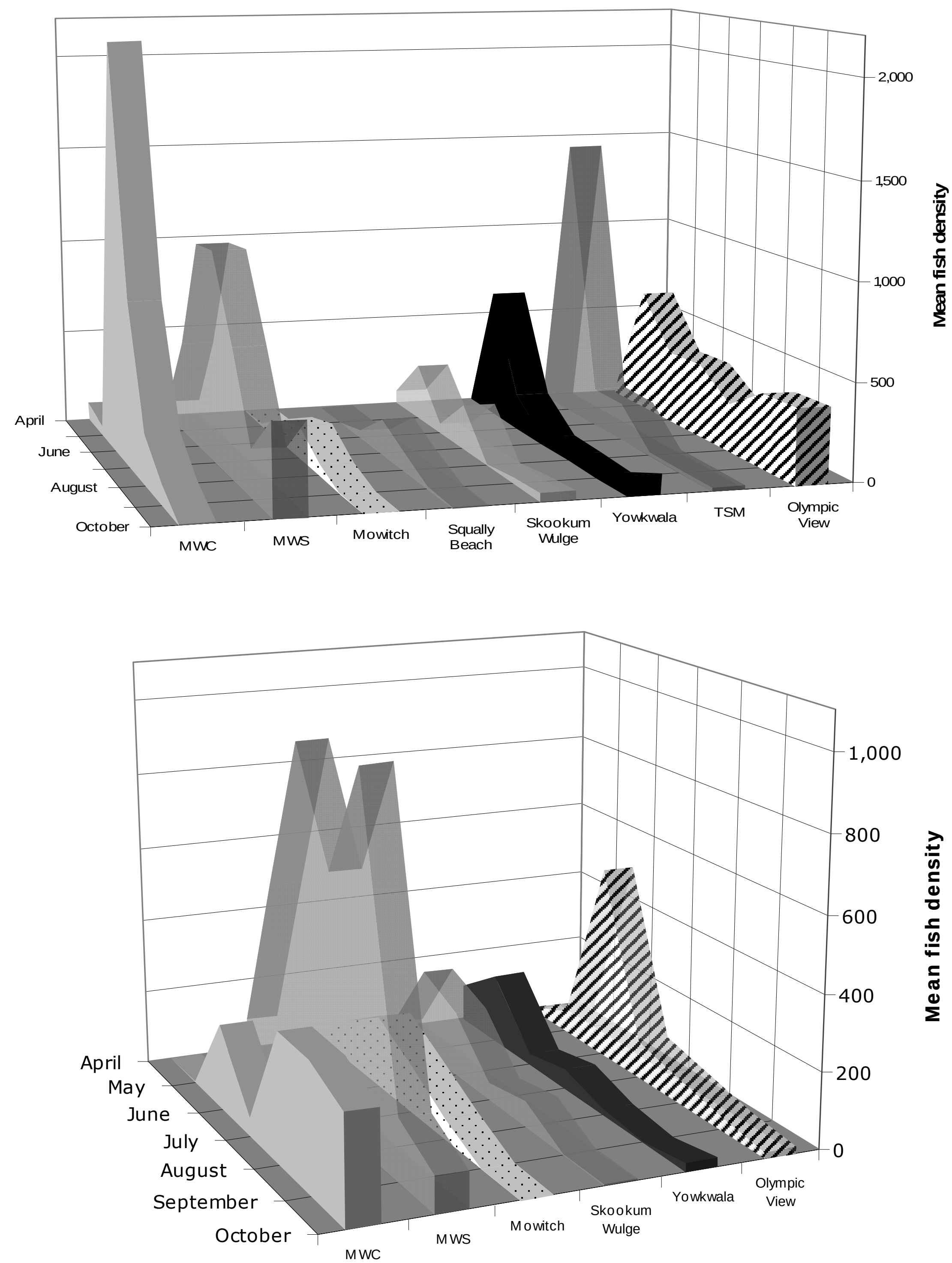

Figure 11. The mean fish density (per 1,000 sq. m) by site and month in 2002 (top panel) and 2003 (bottom panel).

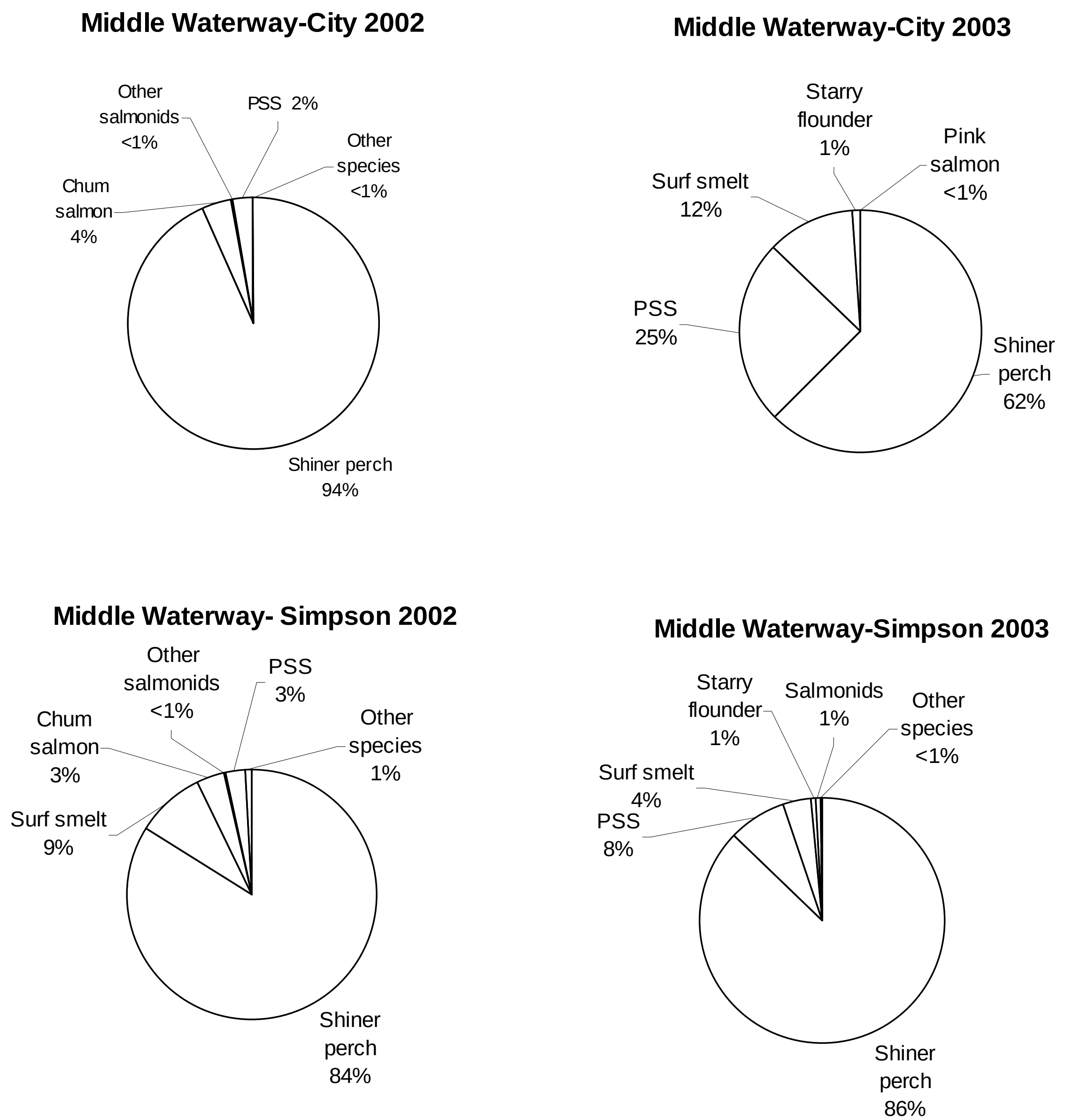

Figure 12. The total number of each species captured as a percentage of the total number of all individual fish captured at Middle Waterway-City and Middle Waterway-Simpson in 2002 and 2003 (PSS = Pacific staghorn sculpin).

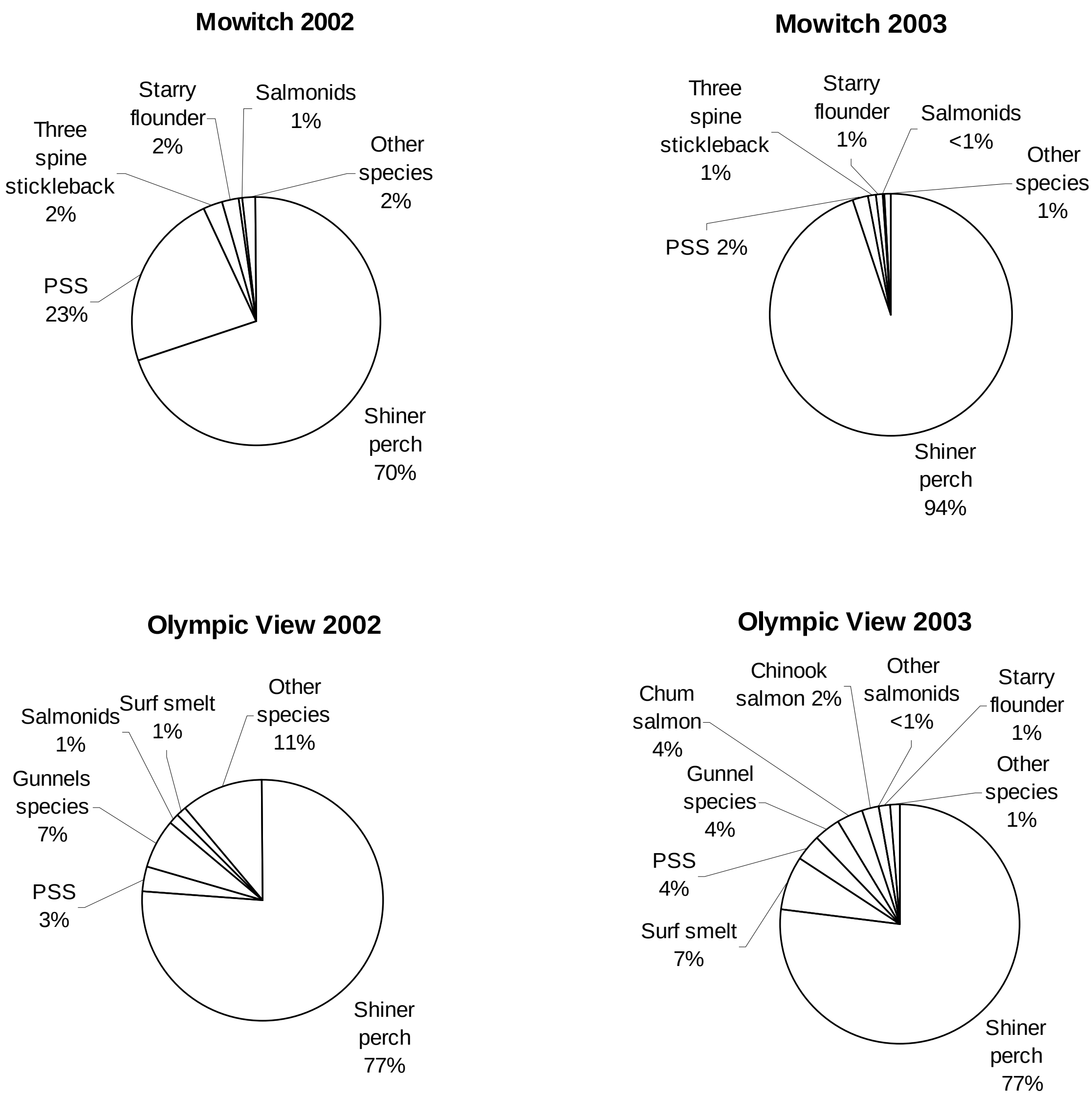

Figure 13. The total number of each species captured as a percentage of the total number of all individual fish captured at Mowitch and Olympic View in 2002 and 2003 (PSS = Pacific staghorn sculpin).

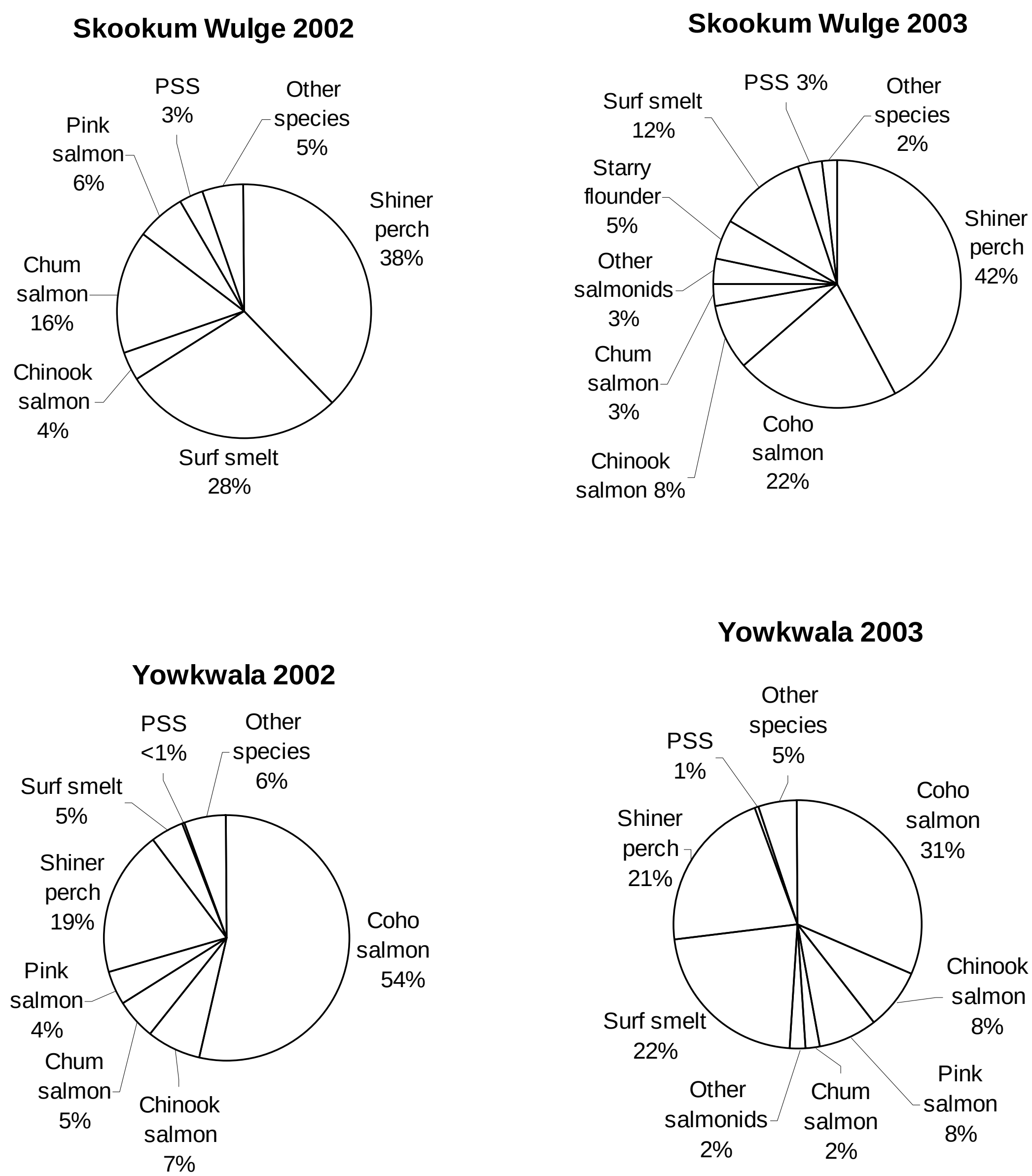

Figure 14. The total number of each species captured as a percentage of the total number of all individual fish captured at Skookum Wulge and Yowkwala in 2002 and 2003 (PSS = Pacific staghorn sculpin).

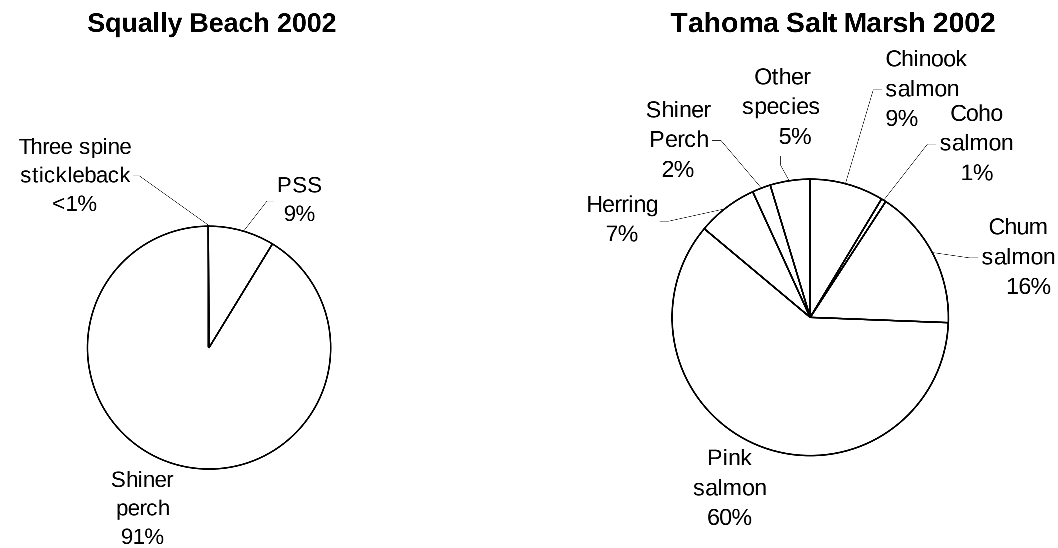

Figure 15. The total number of each species captured as a percentage of the total number of all individual fish captured at Squally Beach and Tahoma Salt Marsh in 2002 (PSS = Pacific staghorn sculpin).

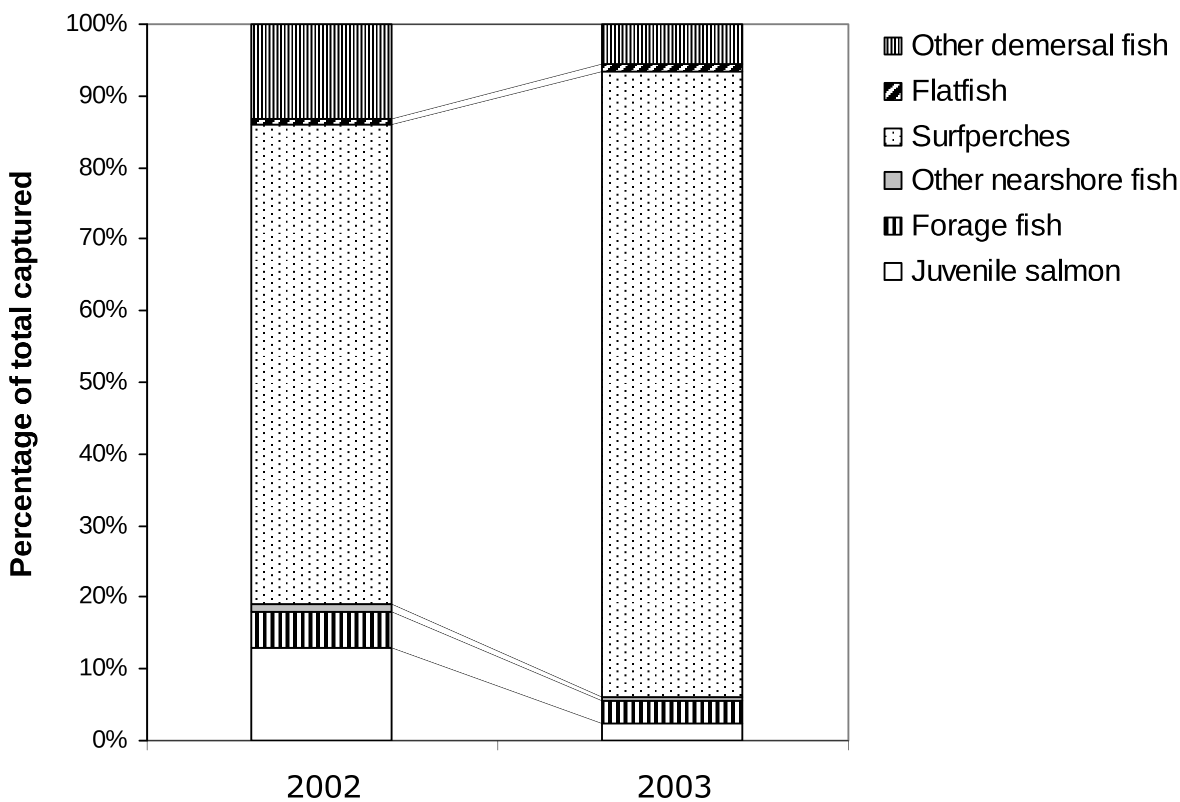

Figure 16. The functional grouping of species captured in 2002 (excluding Squally Beach and Tahoma Salt Marsh) compared to 2003.

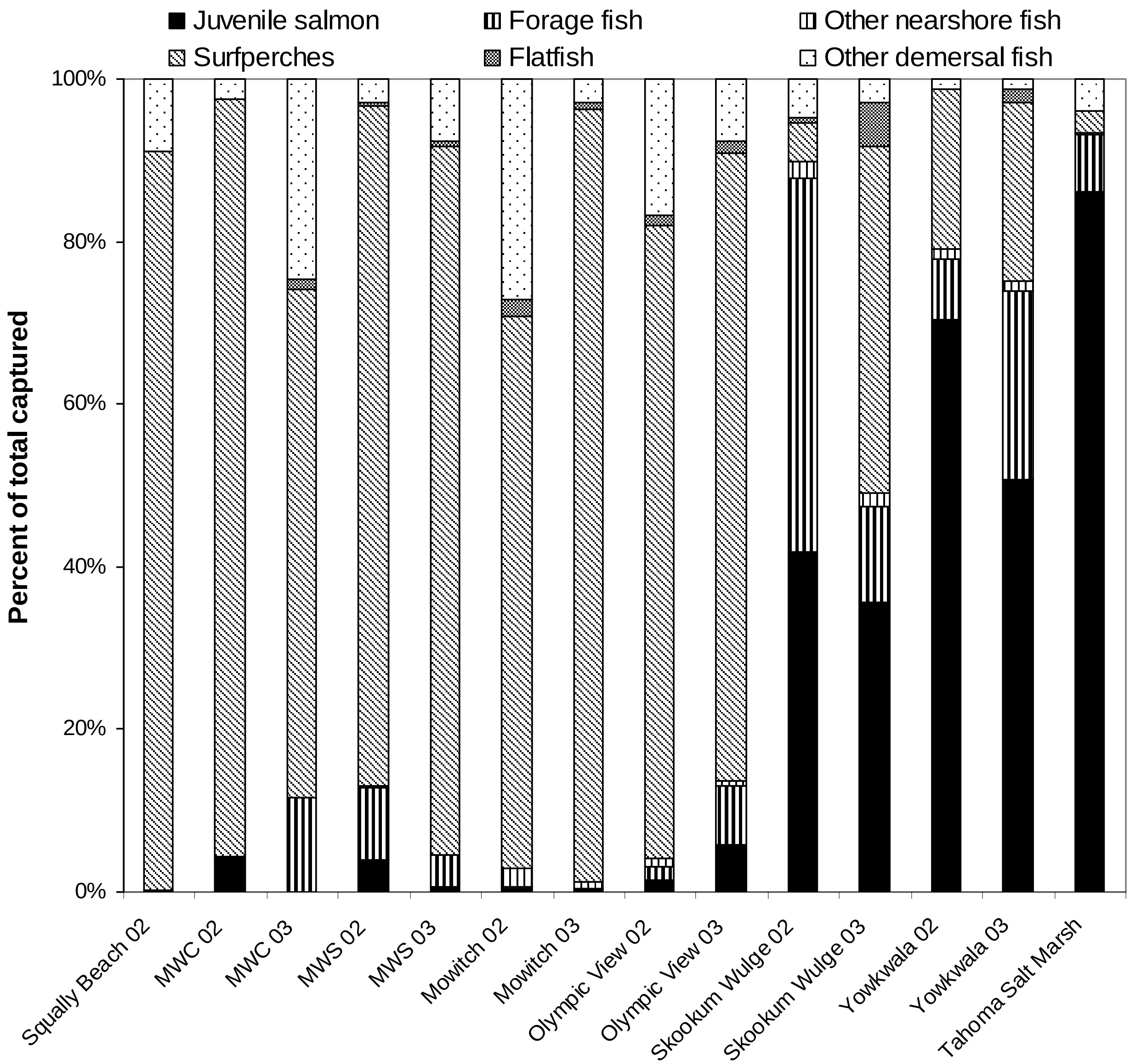

Figure 17. The functional grouping of species captured in 2002 compared to 2003 by site.

Habitat Use by Juvenile Salmonids

Juvenile salmonids were observed at all sampling sites except Squally Beach. Yowkwala yielded the highest number of all salmonids in both sampling years (Figure 18), with Tahoma Salt Marsh being the second highest in 2002, followed by Skookum Wulge, MWS, MWC, Olympic View, and Mowitch in 2002, and Skookum Wulge, Olympic View, MWS, Mowitch, and MWC in 2003. This pattern was similar on a density basis (Figure 18), which shows that Tahoma Salt Marsh has the highest salmonid density. Salmonid density was significantly lower ($p = 0.0001$) at Yowkwala in 2003 than 2002. Skookum Wulge, Tahoma Salt Marsh, and Yowkwala were the only sites where salmonids made up a large percentage of fish collected.

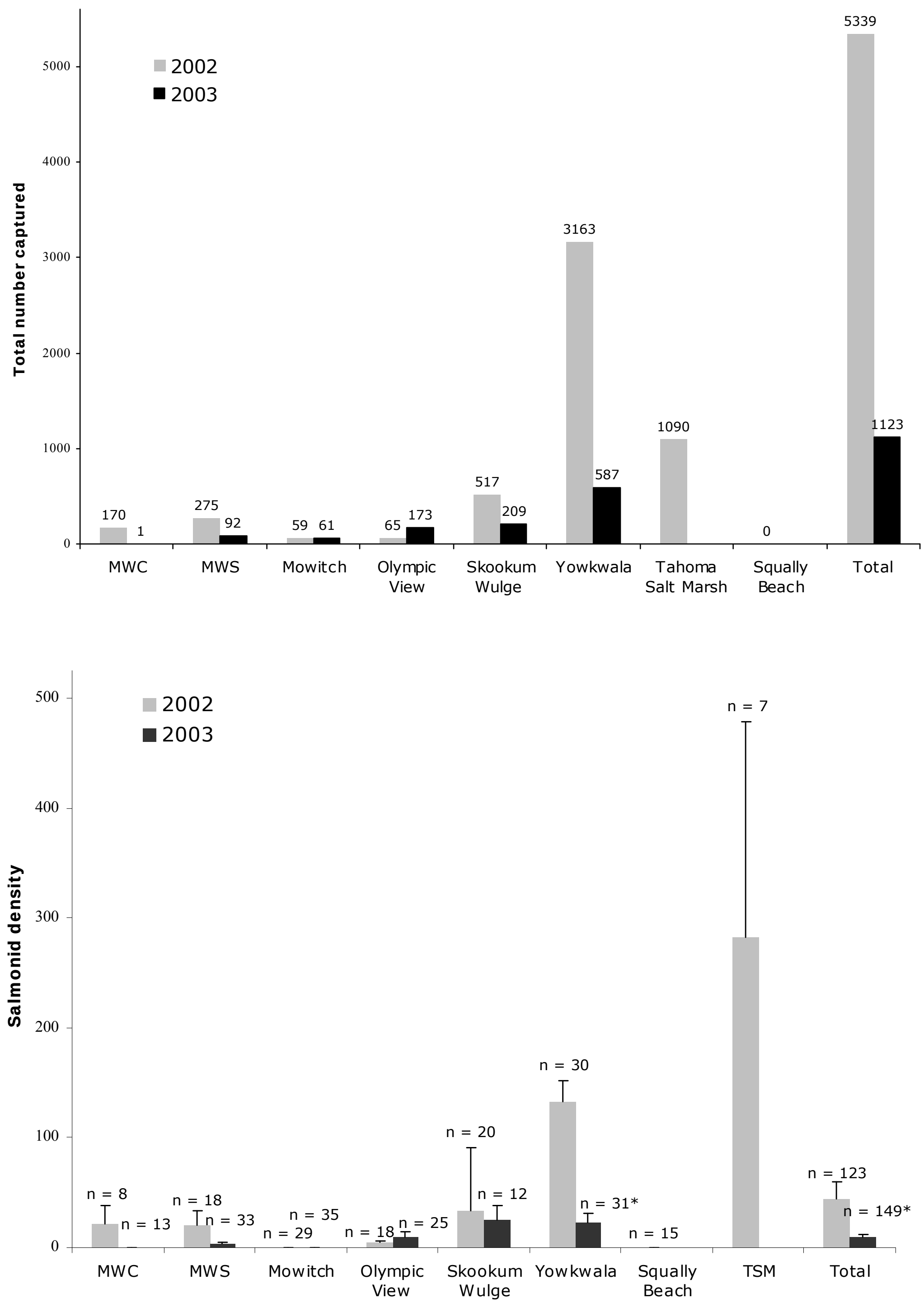

Figure 18. Top panel, the total number of all salmonid species captured at each site in 2002 and 2003. Bottom panel, the salmon density in 2002 and 2003 (per 1,000 sq. m, yearly total excludes Squally Beach and Tahoma Salt Marsh for year-to-year comparison, ± SE, asterisk = significantly different from prior year, n = number of samples included in the analysis).

Salmon species made up 26–41% of fish collected at Skookum Wulge, and 51–70% of fish collected at Yowkwala, but less than 7% at the other sites (Figures 12 through 15).

Major salmon species collected included Chinook, coho, chum, and pink. Of these species, coho salmon were the most abundant in both years (46% of the species captured, Figure 19). However, this species was found in large numbers only at Yowkwala in 2002, and at Yowkwala and Skookum Wulge in 2003. Other salmon species (in order of abundance) were chum (22% and 21% in 2002 and 2003, respectively), Chinook (11% and 24%), and pink (20% and 10%). The number of juvenile salmonids varied between sites and years. In 2002 chum salmon dominated the catches at MWC, MWS, and Skookum Wulge, while chum and Chinook salmon were the major salmon species collected at Mowitch. In 2003 chum salmon were the dominant species at Olympic View and Mowitch, Chinook salmon were most abundant at MWS, while coho salmon were the most abundant species at Skookum Wulge and Yowkwala.

Salmonid densities generally peaked in April or May at most sites in both sampling years (Figure 20), then declined rapidly until very few were caught at any site after June. Coho salmon peaked very sharply in May of both years, then fell drastically, while Chinook salmon had the broadest temporal distribution in both years.

The density of salmonids was significantly lower ($p = 0.0158$) in 2002 than in 2003 (Figure 9 and Figure 18 bottom panel). Most of this difference is explained by 2,347 coho captured in one sampling set at Yowkwala in 2002, bringing the total coho catch to 2,433 for that year, as compared with 499 in 2003. No other sampling set in either year for any salmonid species captured more than 10% of that amount of juvenile coho salmon. The total numbers of juvenile Chinook, chum, and pink salmon were also much lower in 2003 compared to 2002 (Figure 20). Generally, the density of each species declined at individual sites from 2002 to 2003, except for the density of Chinook salmon collected at MWS and Olympic View, coho salmon at MWS, chum salmon at Mowitch and Olympic View (Figure 19), and pink salmon at MWS.

Marked Chinook and coho salmon greatly outnumbered unmarked Chinook and coho salmon at all sites and sampling times except in August of 2002 (Figure 21). Marked juvenile Chinook salmon comprised 86.1% of the Chinook catch in 2002 and 78.5% in 2003. The number of both unmarked and marked Chinook generally peaked in June and then dropped markedly to only rare catches in July–October.

For juvenile coho salmon, marked fish comprised 68.5% of coho caught in 2002 and 75.6% in 2003. The number of unmarked and marked coho peaked in May, then dropped drastically thereafter in both sampling years. The total numbers of both marked and unmarked Chinook and coho salmon captured were much higher in 2002 than in 2003.

The percentage of unmarked compared to marked Chinook and coho salmon varied by month in both sampling years (Figure 22), from 10–20% or less at times of peak releases of hatchery fish to 70–100% early and late in the season. Generally, the percentage of unmarked Chinook dropped from April to May (2002) or June (2003), then increased the rest of the year. The percentage of unmarked coho rose from May to June in 2002, and highly fluctuated in 2003 until August (however, only one coho was captured in August).

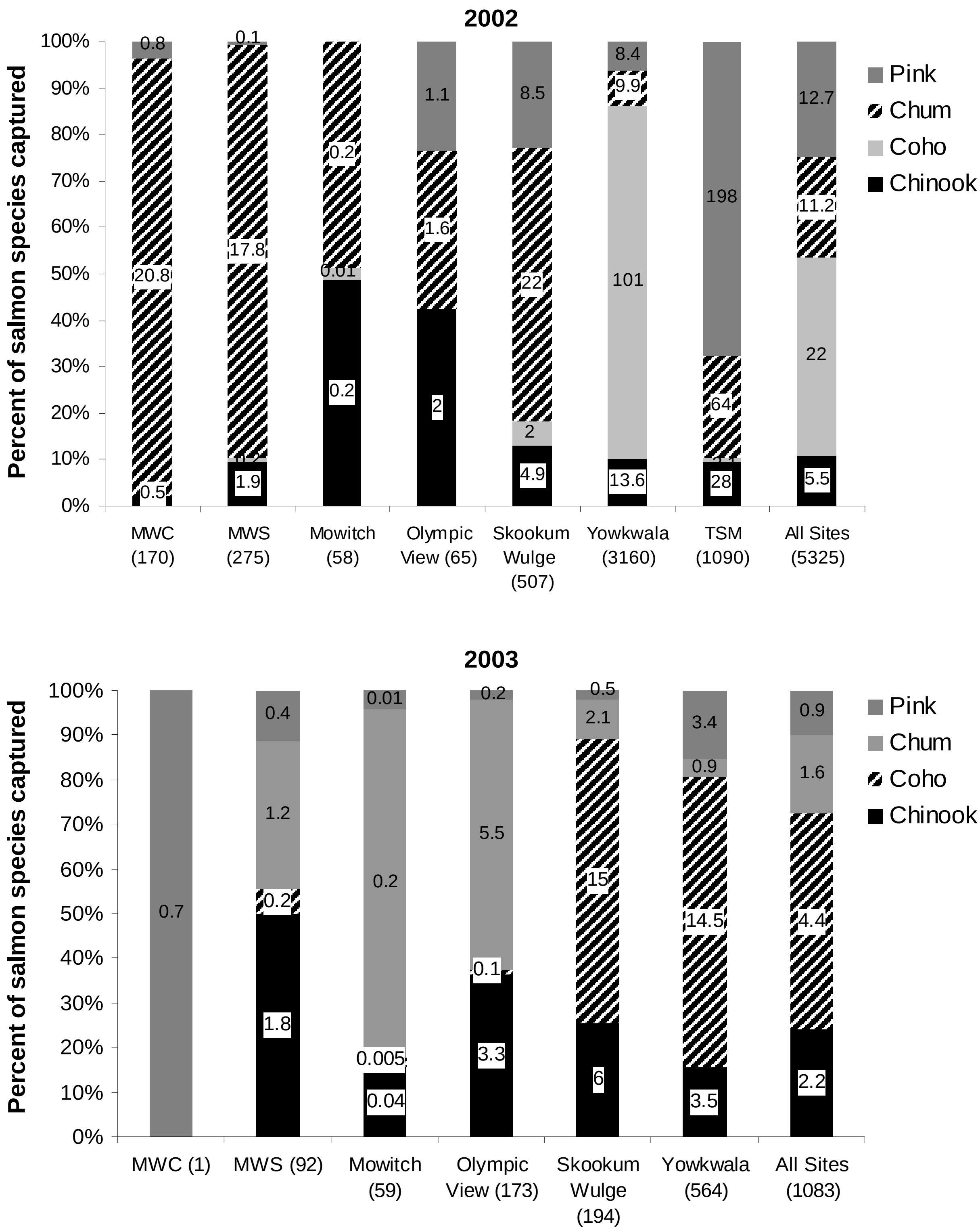

Figure 19. The percentage by density of each salmon species captured by site in 2002 (top panel) and 2003 (bottom panel). Total salmonids captured, excluding steelhead and cutthroat trout (*Oncorhynchus clarkii clarkii*), are in parentheses.

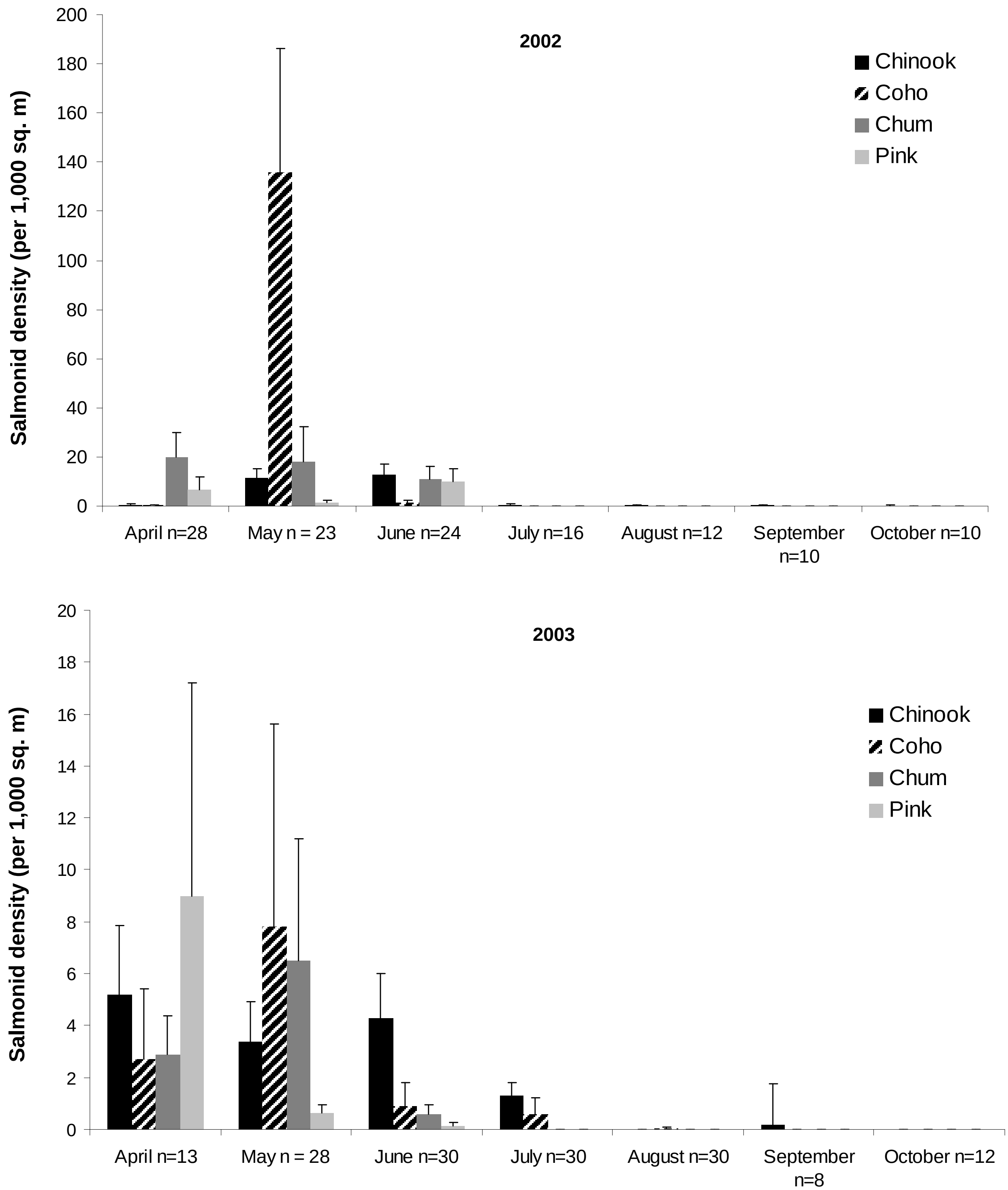

Figure 20. Salmonid densities by month in 2002 (top panel) and 2003 (bottom panel). Bars = standard error, n = number of samples included in the analysis.

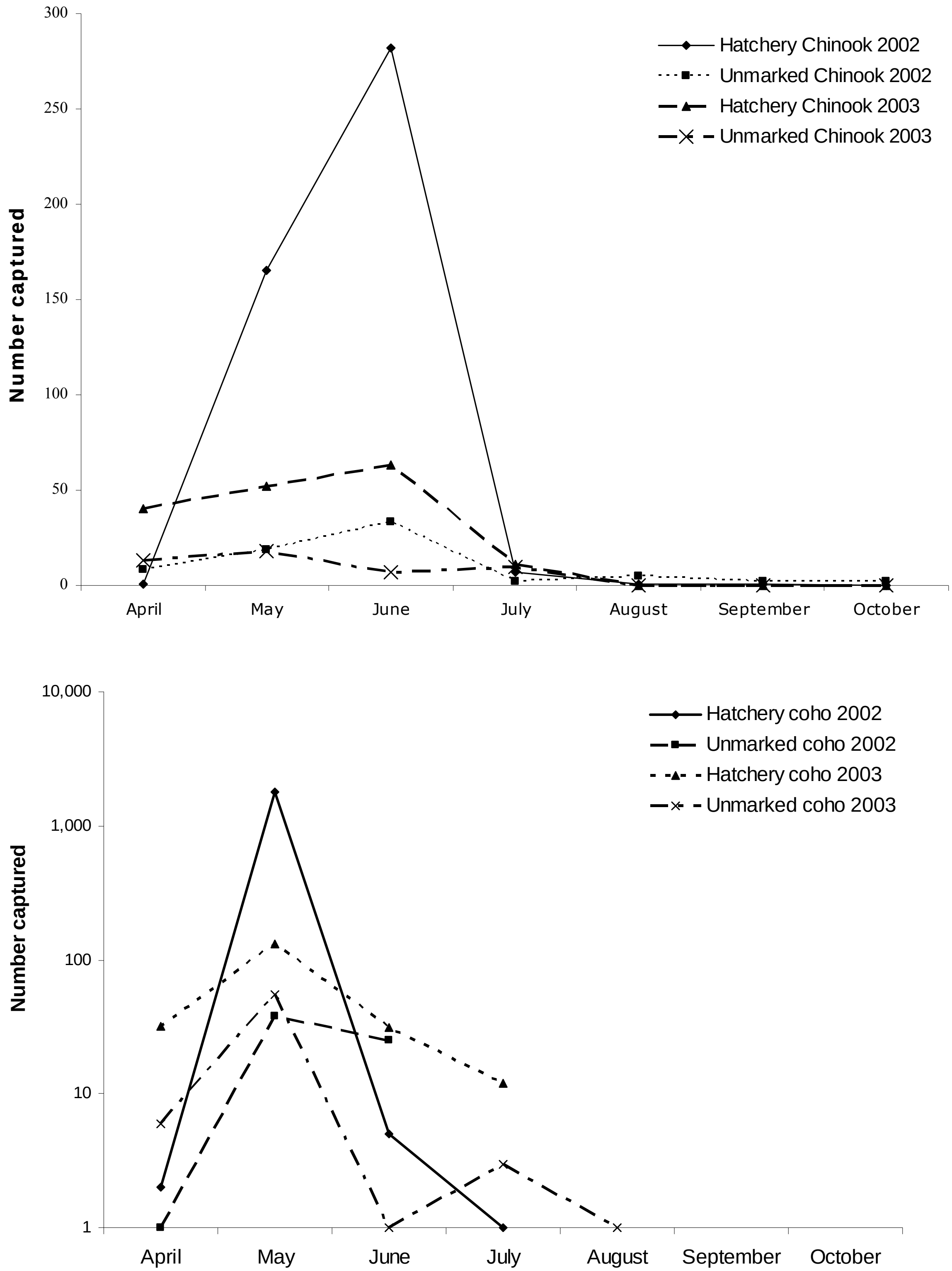

Figure 21. The number of marked versus unmarked Chinook and coho salmon captured in 2002 and 2003 over time (top panel) and log scale (bottom panel), not adjusted for fishing effort.

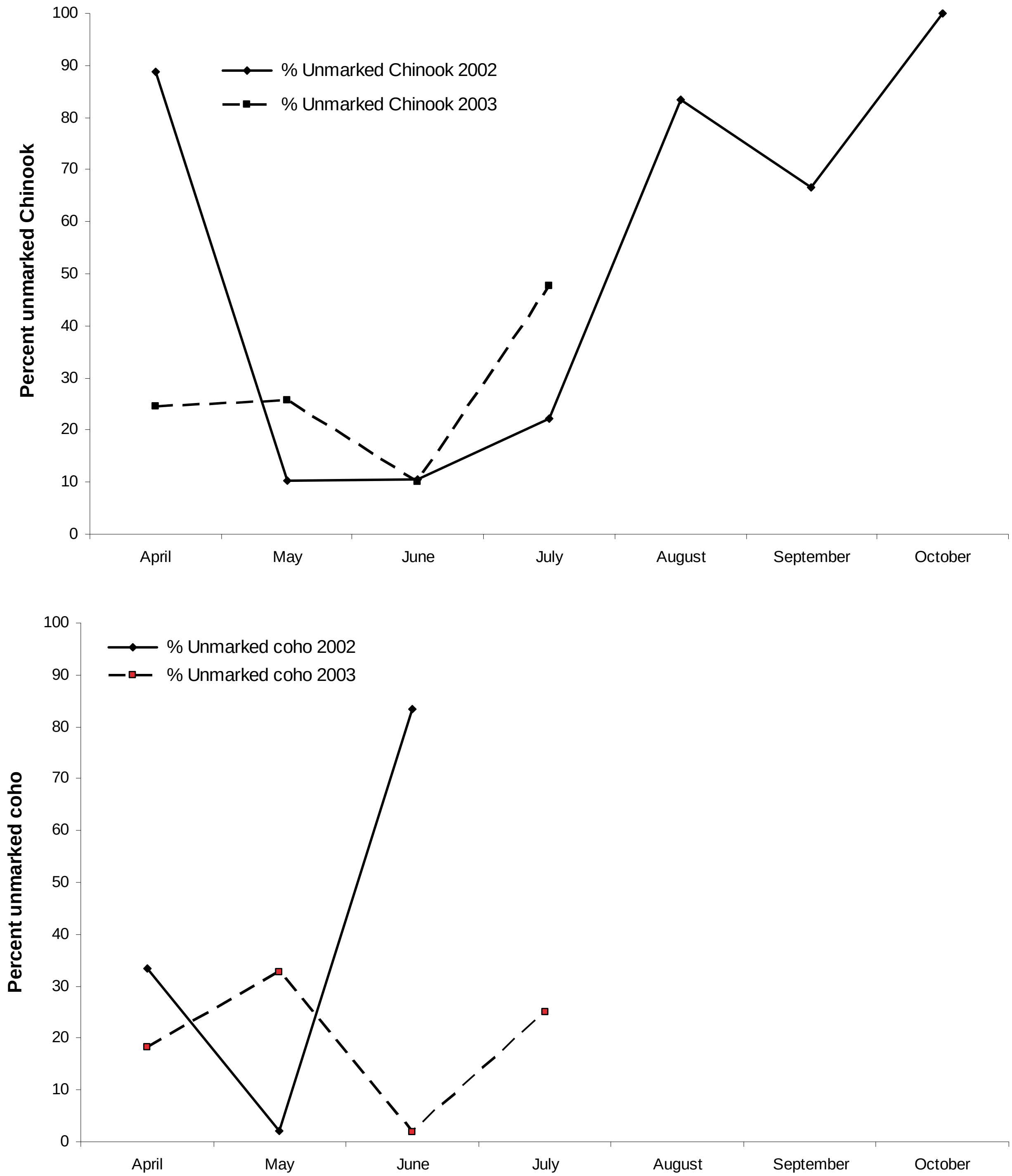

Figure 22. The percentage of unmarked Chinook (top panel) and unmarked coho salmon (bottom panel) captured in 2002 compared to 2003 (not adjusted for fishing effort).

Juvenile Salmonid Lengths

Mean fork length of Chinook salmon was significantly higher ($p < 0.05$) at Yowkwala than at all sites except Olympic View in 2002, and all sites in 2003 (Figure 23). The mean length of Chinook was significantly lower ($p < 0.05$) at MWS in 2002 and at Mowitch in 2003 than at all other sites. Overall, there was no significant difference in lengths of Chinook captured in 2002 compared to 2003. Marked Chinook were significantly larger than unmarked Chinook in 2002 ($p = 0.0488$); however, no statistically significant differences in length occurred between marked and unmarked Chinook in 2003 (Figure 24). Except in April of 2002, the mean length of Chinook was generally higher during April and May, declined until June, then increased through the remainder of the sampling year (Figure 25).

The largest coho salmon (based on fish length) were captured at Yowkwala in both years, and were significantly larger ($p < 0.05$) than at all other sites except Skookum Wulge in 2002. Overall, there was no significant difference in the length of coho between the two sampling years. As with Chinook, marked coho were significantly larger ($p = 0.0460$) than unmarked coho in 2002, but no significant differences were detected in 2003 (Figure 24). Coho were larger at the start of the sampling season (April), then decreased significantly in size ($p < 0.05$) through July in both years (Figure 25), but increased significantly in size ($p = 0.0034$) after June in 2003.

Chum salmon overall were significantly larger ($p = 0.0001$) in 2002 than in 2003 (Figure 23), regardless of capture site. Chum salmon were significantly larger ($p < 0.05$) at Yowkwala and Olympic View (except MWS) in 2002 than all other sites, and at Olympic View, MWS, and Yowkwala than all other sites in 2003 (Figure 23). They also increased in size (Figure 25) from May to June in both 2002 ($p = 0.0001$) and 2003 ($p = 0.0008$).

As with coho and chum salmon, juvenile pink salmon were significantly larger ($p = 0.0001$) in 2002 than in 2003. Pink salmon (Figure 23) were significantly larger ($p < 0.05$) at Tahoma Salt Marsh than all other sites in 2002 and at Olympic View than all other sites in 2003. Juvenile pinks increased significantly in size ($p < 0.05$) over the course of the sampling season (Figure 23).

Chinook Diet Analysis

Taxonomic analysis of juvenile Chinook salmon stomach contents was completed for fish collected only in the first year (2002) of the current study. Stomach contents were collected from April to the end of July, with most samples collected during May and June. As the majority of fish collected at these time periods were marked hatchery fish, the diet composition is more representative of feeding patterns of released hatchery fish than of wild Chinook salmon. When analyzed by ecology of prey (% gravimetric composition), site of capture, and size of juvenile salmon, a clear trend occurred with Chinook larger or smaller than 90 mm. Chinook less than or equal to 90 mm in size predominantly preyed on terrestrial-riparian species (Figure 26) at MWC, MWS, Mowitch, and Olympic View, while marine planktonic-neritic and marine benthic-epibenthic species were the predominant prey at Skookum Wulge, Tahoma Salt Marsh, and Yowkwala. At fish lengths greater than 90 mm (Figure 27), the predominant prey species shifted to marine planktonic-neritic at all sites except Mowitch and Olympic View (sample size at Olympic View and MWS was only one).

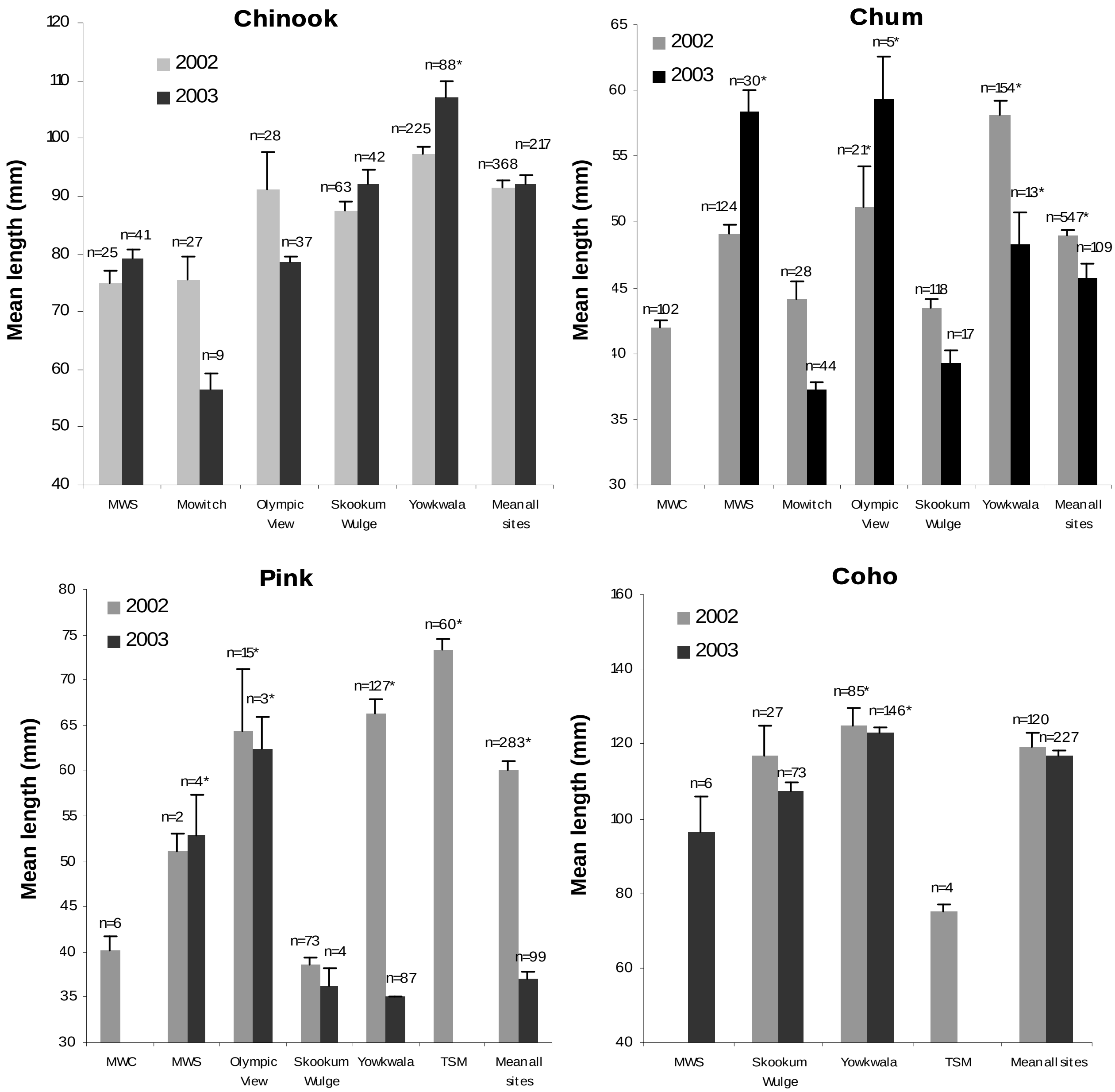

Figure 23. The mean length of four species of juvenile salmon captured in 2002 and 2003 by site (± SE, asterisk = significantly larger than other sites without an asterisk in the same year and between years for all sites combined, n = number of samples included in the analysis).

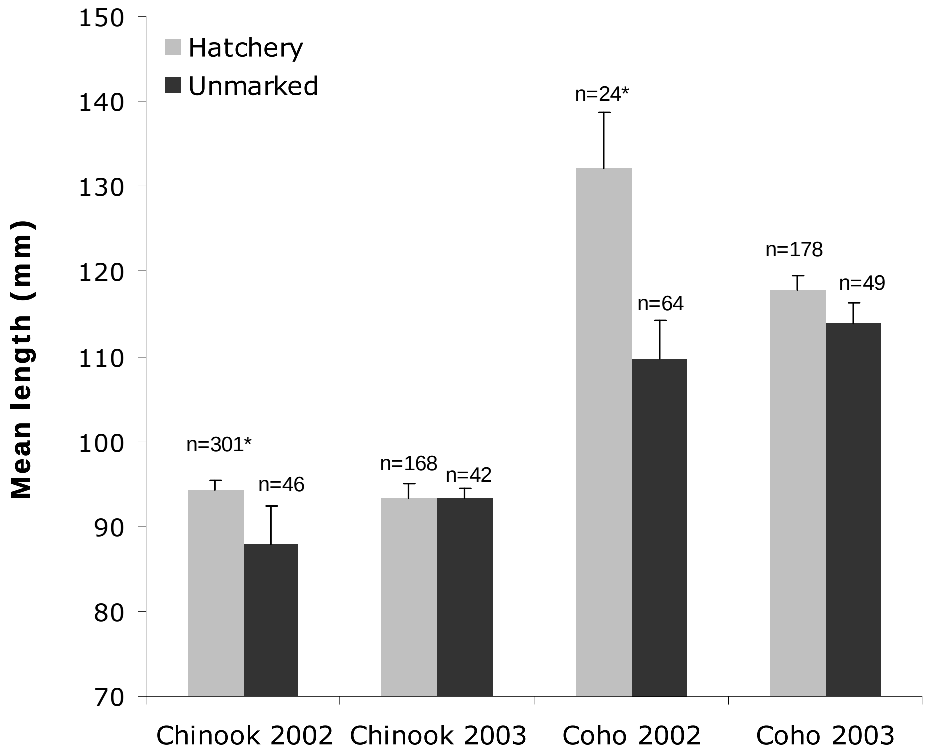

Figure 24. The mean length of marked compared to unmarked Chinook and coho salmon by year (± SE, asterisk = significantly larger, n = number of samples included in the analysis).

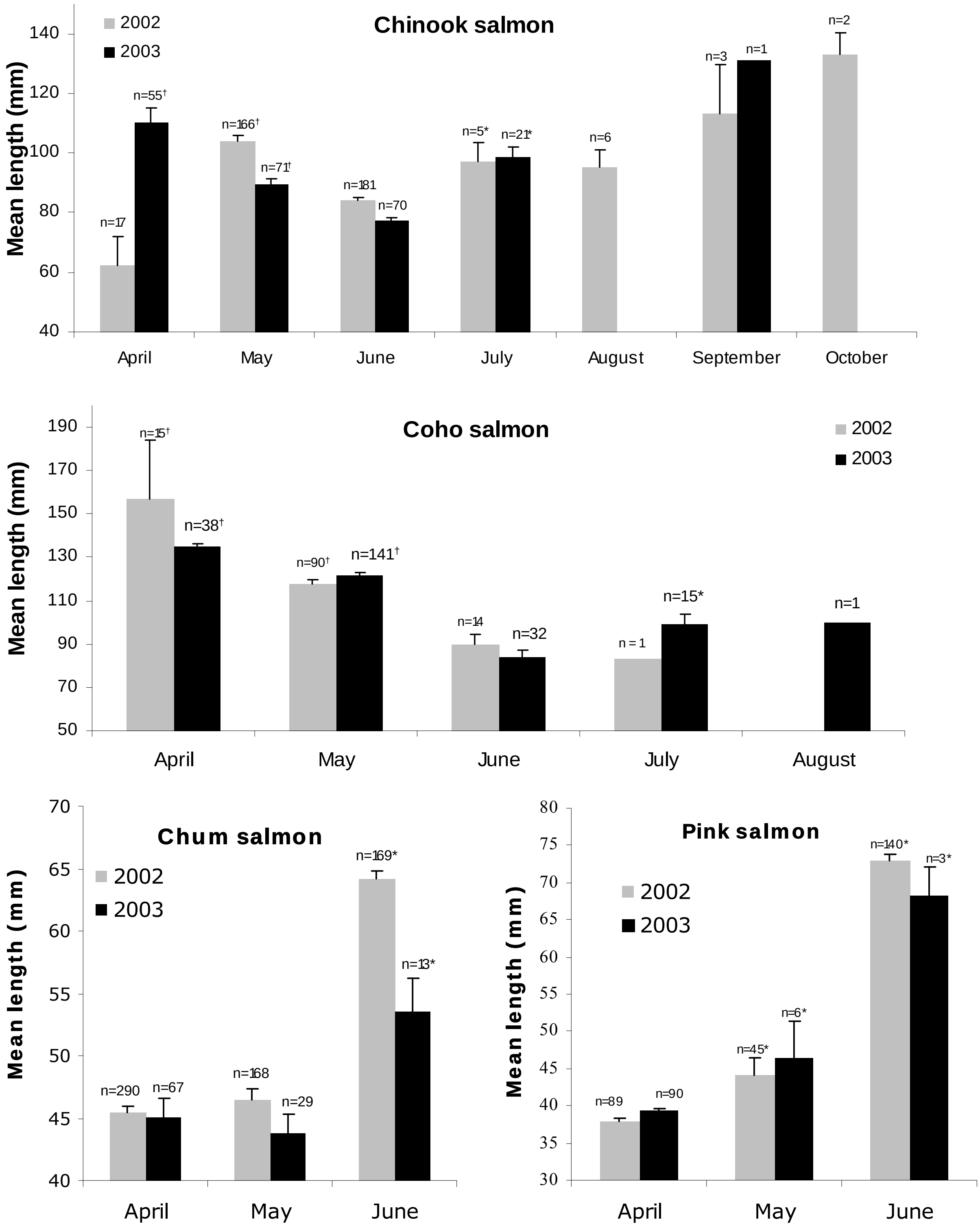

Figure 25. The mean length of juvenile Chinook, coho, chum and pink salmon from all sites combined by month in 2002 and 2003 (± SE, asterisk = significantly larger than preceeding month, dagger = significantly larger than following month, n = number of samples included in the analysis).

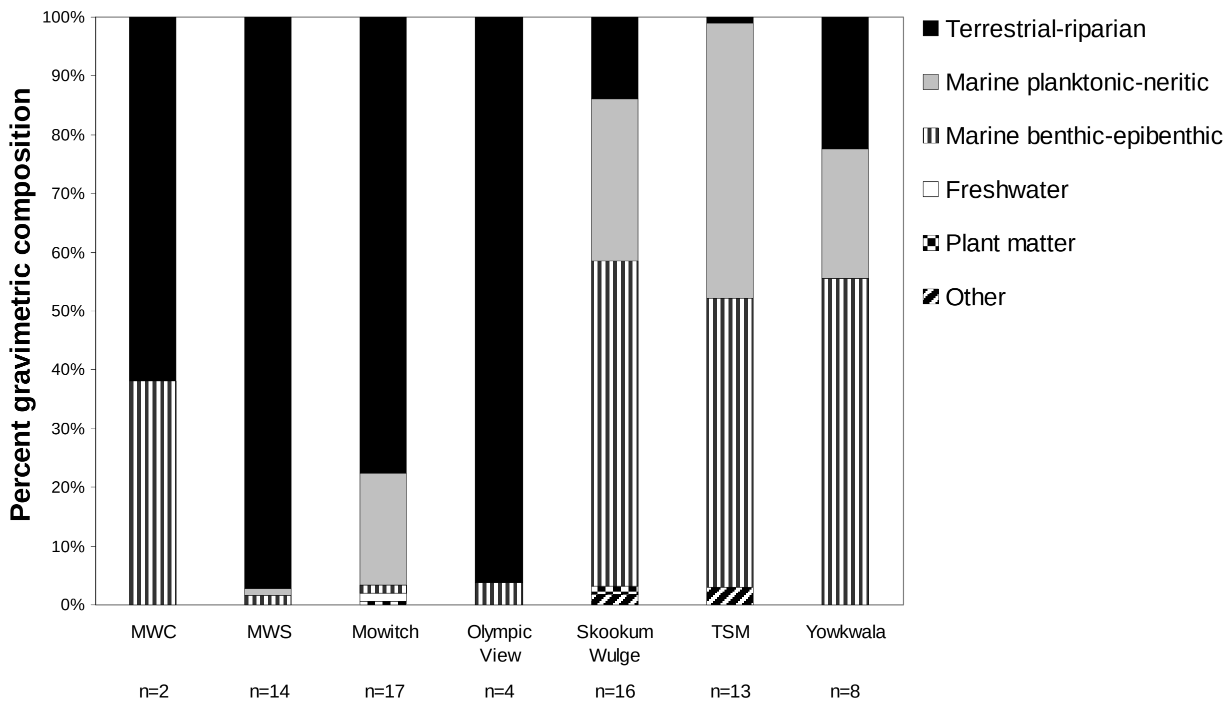

Figure 26. The diet of juvenile Chinook with a fork length less than or equal to 90 mm by site and ecology of prey (n = number of samples included in the analysis).

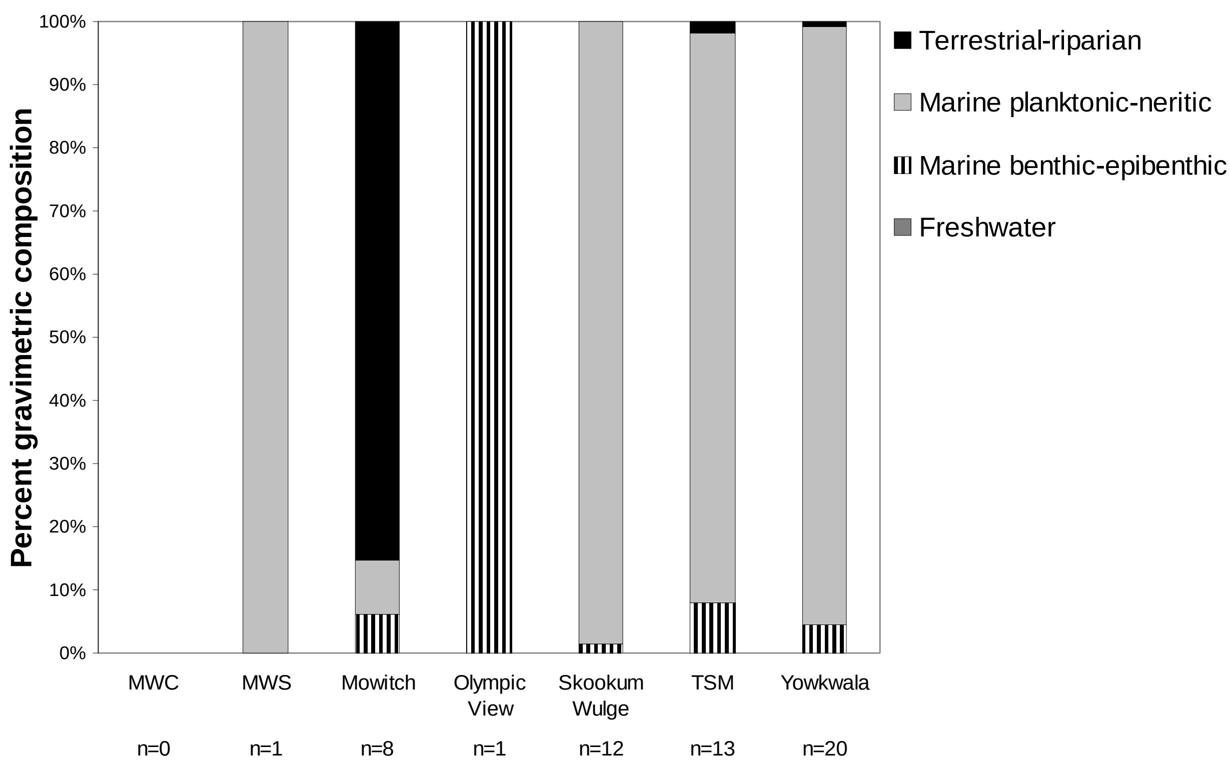

Figure 27. The diet of juvenile Chinook with a fork length greater than 90 mm by site and ecology of prey (n = number of samples included in the analysis).

Additional analysis of diet composition revealed that, overall, fish larvae were the single most predominant prey type (% gravimetric composition) of juvenile Chinook salmon (Figure 28) at MWS, Olympic View, Yowkwala, and Skookum Wulge. Diptera species (*Tethinidae* sp., *Ephydridae* sp., *Ceratopogonidae* sp., and others) were the predominant prey species at MWC and Mowitch, and crustaceans were the almost exclusive prey species at the Tahoma Salt Marsh. Juvenile Chinook less than or equal to 90 mm in length (Figure 29) fed predominantly on dipterids at MWC, MWS, and Mowitch, marine crustacea at Tahoma Salt Marsh and Yowkwala, Formicidae and other Insecta at Olympic View, and had a varied diet at Skookum Wulge. For fish lengths greater than 90 mm (Figure 30), juvenile Chinook fed primarily on marine fish (Pacific sandlance and other Osteichthyes) at all sites except Mowitch (primarily Diptera species) and Tahoma Salt Marsh (primarily marine crustacea).

Pacific Staghorn Sculpin

PSS were collected in 2003 to assess contaminant uptake in a resident benthic species. PSS were found at all sampling sites, with total catch ranging from 3 at Tahoma Salt Marsh (2002) to 2,639 at Mowitch (2002). Generally they made up from less than 1% of the catch at sites (Figures 12 through 15) such as Yowkwala (with only seven captured in both sampling years) to 25% of the catch at Mowitch (2002) and MWC (2003). The total number of PSS in 2003 was lower than in 2002 (Figure 9).

There was a year-to-year difference in mean length of PSS, with those captured in 2002 being significantly smaller ($p = 0.0001$) than in 2003. The mean length of PSS from Yowkwala was significantly greater ($p < 0.05$) than all other sites except Olympic View and Tahoma Salt Marsh (2002) in both sampling years (Figure 31). PSS were significantly smaller ($p < 0.05$) at MWC than all sites except Squally Beach, Mowitch, and Skookum Wulge in 2002, and compared to all sites in 2003. The ages of PSS sampled for chemical contaminant analyses ranged from 6 months to 1.4 years (based on length/age calculations as determined by Orsi 1999), with mean ages ranging from 9 months to 1 year (Appendix G). This is comparable in age to juvenile Chinook (yearlings and subyearlings) sampled for similar analyses.

Chemical Contaminants

Concentrations of Chemical Contaminants

Sediments

Average concentrations of ∑LMWAHs in sediments from the Commencement Bay restoration sites ranged from 380 ng/g dry wt at Mowitch to 4,600 ng/g dry wt at MWS (Figure 32, Table 5). Significantly different concentrations of ∑LMWAHs ($p = 0.0257$) were measured in sediments among some of the six Commencement Bay sites (Tahoma Salt Marsh and Yowkwala could not be sampled because sediments at these sites consisted of rocks and cobbles). MWS sediments had a significantly higher mean ∑LMWAH level than the mean value of the Mowitch samples. No other significant differences in mean ∑LWMAH concentrations were found in samples from the other sites. At all sites, the most abundant ∑LWMAHs were naphthalene and phenanthrene—accounting for 42% (Mowitch) to 50% (MWC and Squally

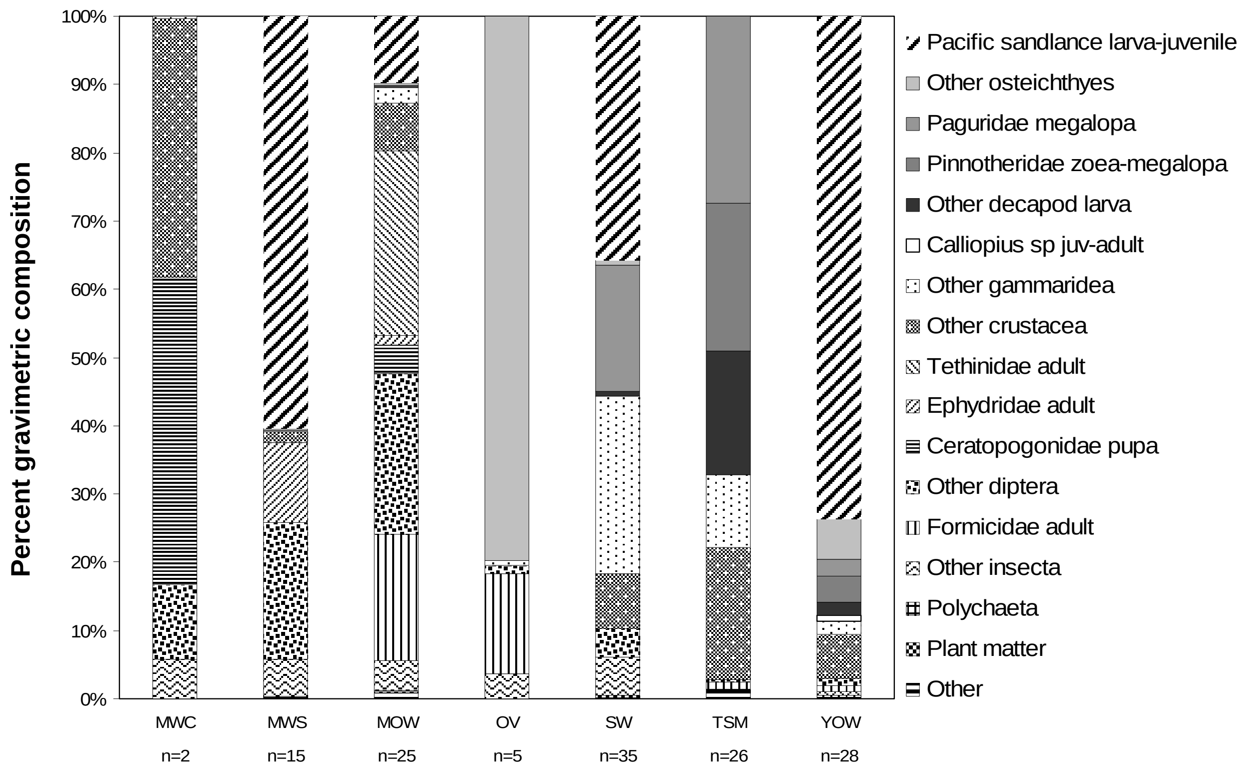

Figure 28. Comparison of the Commencement Bay juvenile Chinook diet in 2002 by site (n = number of samples included in the analysis).

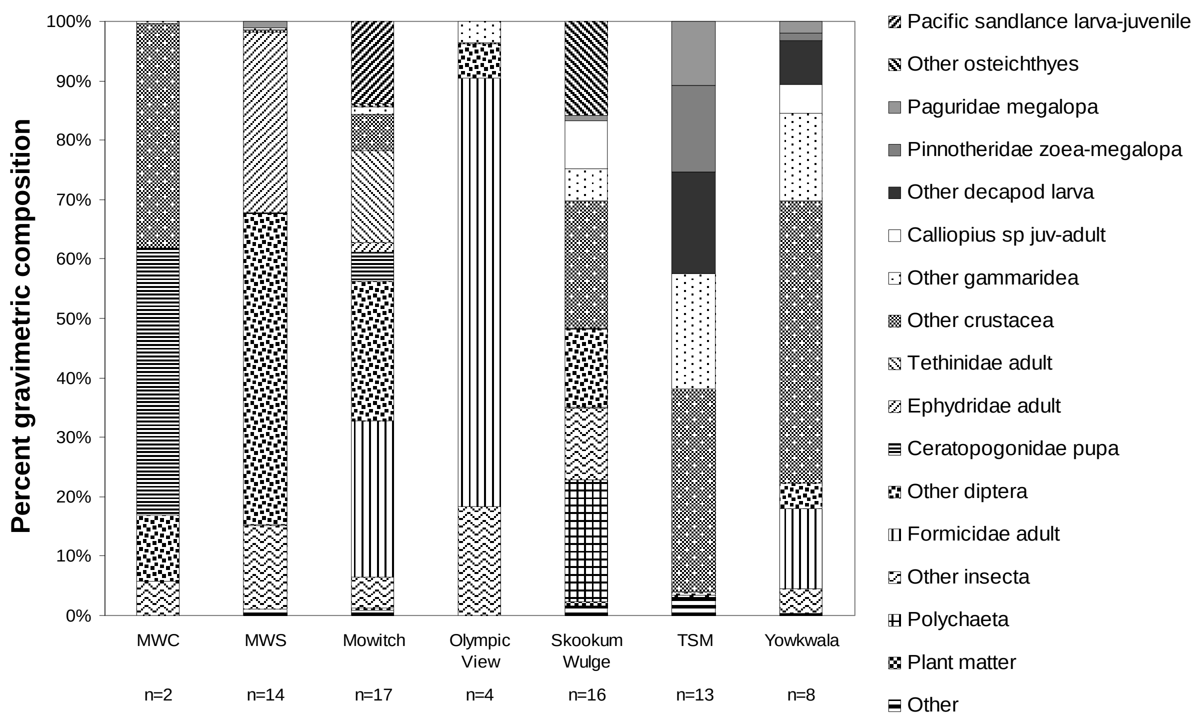

Figure 29. The diet of juvenile Chinook in 2002 with a fork length less than or equal to 90 mm by prey taxa and site of capture (n = number of samples included in the analysis).

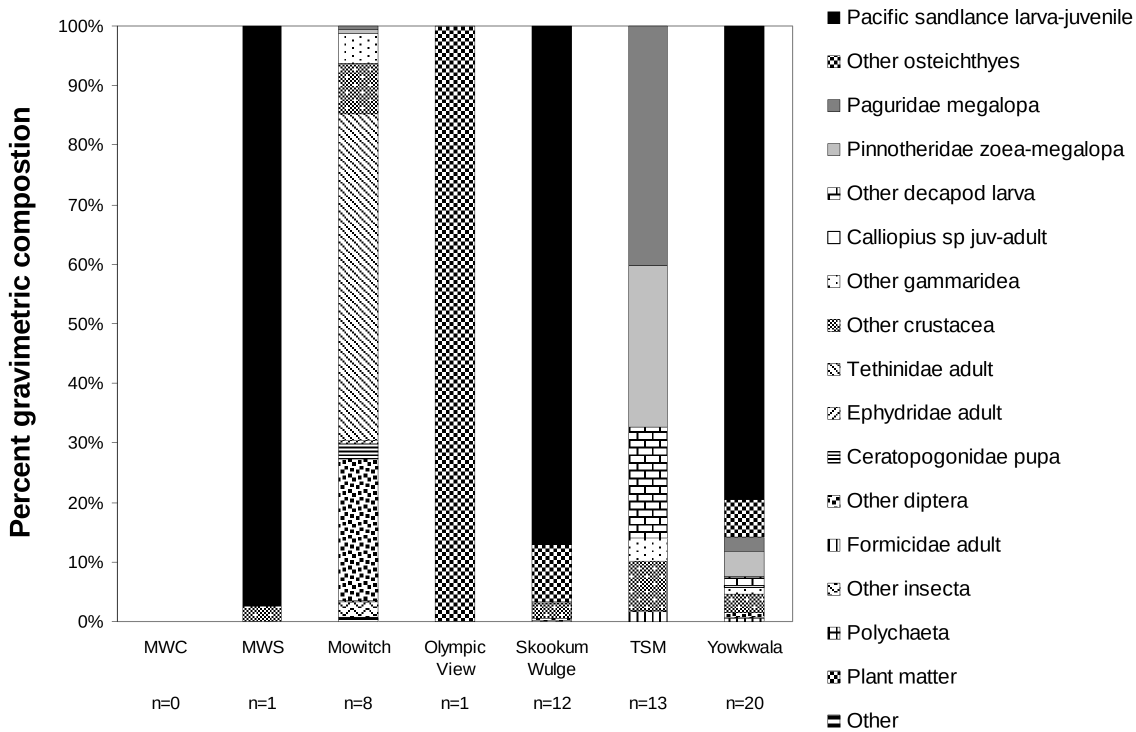

Figure 30. The diet of juvenile Chinook in 2002 with a fork length greater than 90 mm by prey taxa and site of capture (n = number of samples included in the analysis).

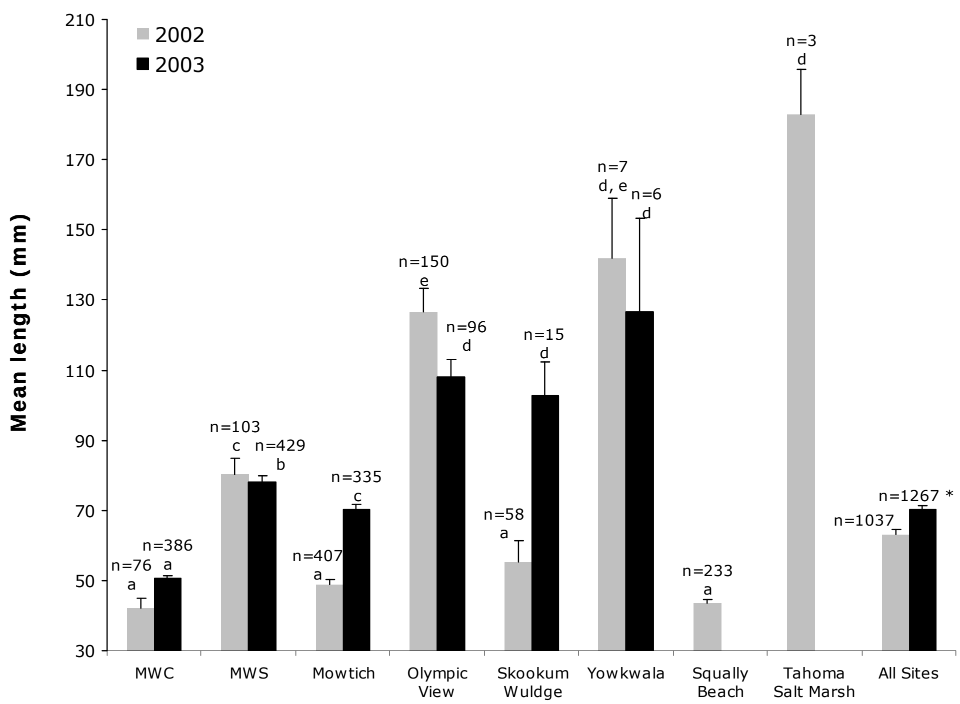

Figure 31. Mean length of Pacific staghorn sculpin in 2002 compared to 2003 by site (± SE, columns with dissimilar letters are significantly different than other sites in the same year, asterisk = significantly different than corresponding year, n = number of samples included in the analysis).

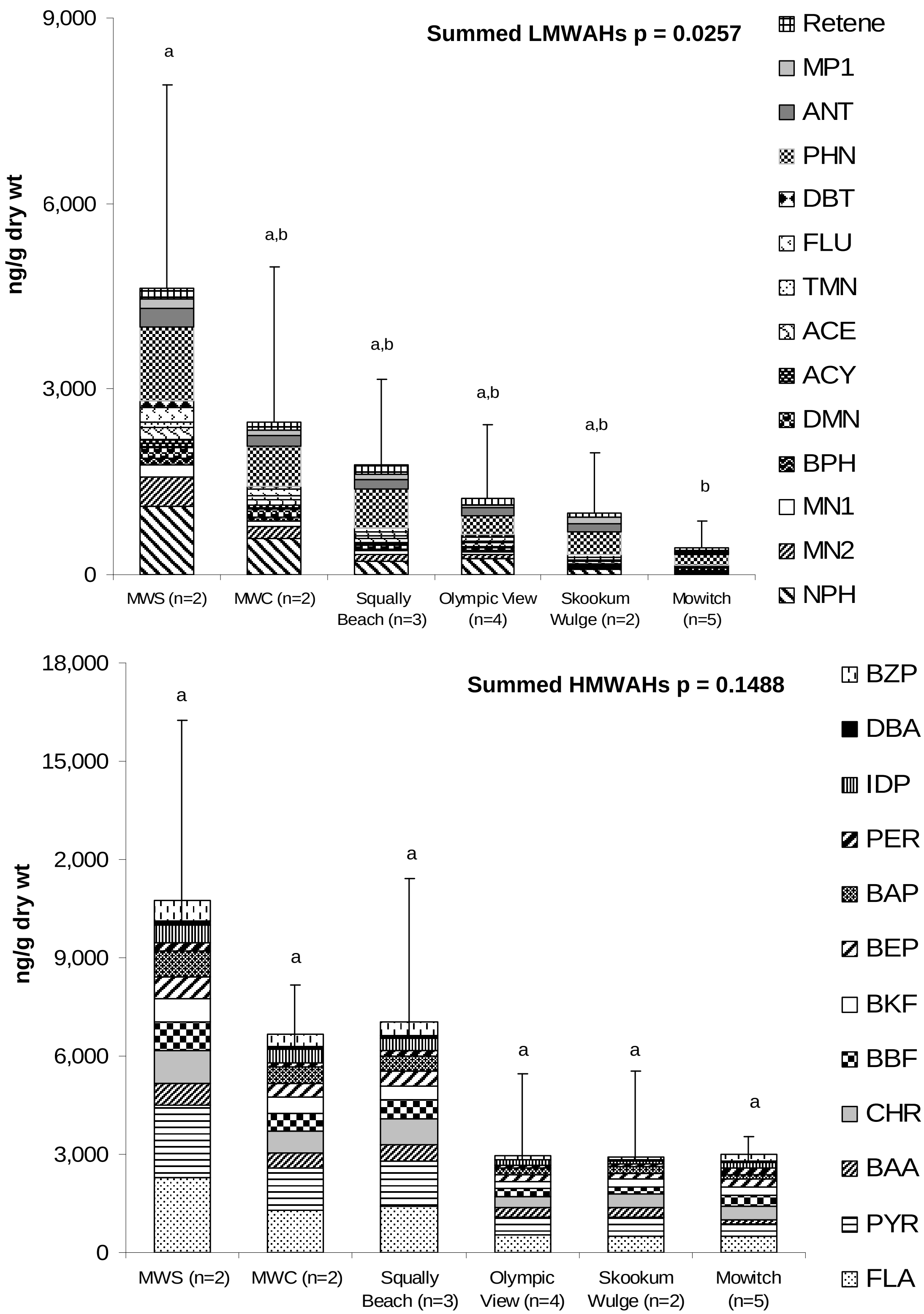

Figure 32. The concentrations of low molecular weight PAHs (top panel) and high molecular weight PAHs (bottom panel) in Commencement Bay sediments (± SE, bars with unlike letters differ significantly using Tukey-Kramer honestly significant difference test, $p < 0.05$, n = number of samples included in the analysis, legend abbreviations are defined in Appendix C).

Table 5. The concentrations of PAHs in Commencement Bay restoration site sediments in 2002 (asterisk = significantly higher than other sites, n = number of samples included in the analysis).

	ng/g dry wt	
Site	∑LMWAHs	∑HMWAHs
Middle Waterway at City (n = 2)	2,500 ± 1,100	6,700 ± 1,900
Middle Waterway at Simpson (n = 2)	4,600* ± 3,300	11,000 ± 6,100
Mowitch (n = 5)	380 ± 96	2,800 ± 750
Olympic View (n = 4)	1,100 ± 730	2,900 ± 2,500
Skookum Wulge (n = 2)	910 ± 830	2,800 ± 2,600
Squally Beach (n = 3)	1,700 ± 1,300	6,900 ± 4,800

Beach) of ∑LMWAHs (Figure 32). Retene, a LMWAH wood product associated with pulp mills or logging operations, contributed 5% (MWC) to 15% (Mowitch) of the ∑LMWAHs found in Commencement Bay sediments.

Mean levels of ∑HMWAHs in Commencement Bay sediments ranged from 2,900 ng/g dry wt at Mowitch to 8,700 ng/g dry wt at MWC (Figure 32, Table 5). No significant differences ($p = 0.1488$) in mean ∑HMWAH concentrations were found among sediments from the six restoration sites. Fluoranthene and pyrene were the most prevalent HMWAHs measured in the sediments, contributing an average of 29% (Mowitch) to 43% (Skookum Wulge) of the ∑HMWAHs (Figure 32). Mean levels of ∑PAHs (LMWAHs + HMWAHs) ranged from 3,500 to 15,000 ng/g dry wt, with highest levels at MWC and MWS. The ∑PAHs measured in the Commencement Bay sediments were comprised mostly of HMWAHs (Figure 32). For example, at all sites, the average contributions of ∑HMWAHs to ∑PAHs ranged from 66% at Olympic View to 88% at Mowitch. As with ∑HMWAHs, we found no significant differences ($p = 0.1127$) in mean levels of ∑PAHs measured in sediments from the six sites.

DDTs and PCBs were the most abundant OCs measured in the Commencement Bay sediments (Table 6). Mean concentrations of ∑DDTs ranged from 2.2 ng/g dry wt at Olympic View to 110 ng/g dry wt at Mowitch, whereas average levels of ∑PCBs ranged from 28 ng/g dry wt at Skookum Wulge to 150 ng/g dry wt at Squally Beach (Table 6). Levels of DDTs in sediments were significantly higher at Mowitch ($p < 0.0001$) than all other sediment sites. The mean levels of ∑DDTs were not significantly different between sediments collected from the other sites. The DDT metabolites p,p'-DDE and p,p'-DDD contributed approximately 39% (Skookum Wulge) to 82% (Squally Beach) of the ∑DDTs except at Skookum Wulge (Figure 33). Surprisingly, unlike the sediment collected at the other sites, we found that the parent DDT (p,p'-DDT) contributed greater than 50% to the ∑DDTs at Skookum Wulge. In contrast to the mean ∑DDT levels, we found no significant differences ($p = 0.0714$) in mean levels of ∑PCBs in sediments among the five sites (Figure 34). The moderately chlorinated congeners (those containing 5–7 chlorine atoms), such as PCBs 118, 138, and 153, were the most abundant PCB congeners found in the Commencement Bay sediments.

Table 6. The mean (± SE) concentrations of OCs in sediments collected from various Commencement Bay restoration sites in 2002 (asterisk = significantly higher than other sites, superscript a significantly higher than superscript b, n = number of samples included in the analysis).

Site	ng/g. dry wt.				
	∑DDTs	∑PCBs	∑CHLDs	Lindane	HCB
Middle Waterway at City (n =2)	3.2 ± 1.1	67 ± 21	9.6* ± 6.2	25*[a] ± 5.7	0.26 ± 0.16
Middle Waterway at Simpson (n =2)	5.8 ± 5.0	66 ± 45	14* ± 8.7	29* ± 30	0.52 ± 0.37
Mowitch (n = 5)	120* ± 26	77 ± 9.7	14* ± 4.0	6.4*[b] ± 0.99	4.2* ± 5.2
Olympic View (n = 4)	2.2 ± 2.6	62 ± 49	<0.24	<0.25	0.18 ± 0.17
Skookum Wulge (n = 2)	6.0 ± 5.4	28 ± 9.2	1.4 ± 0.064	<0.21	3.2 ± 3.7
Squally Beach (n = 3)	13 ± 2.0	150 ± 72	3.4 ± 1.3	<0.25	7.8* ± 6.5

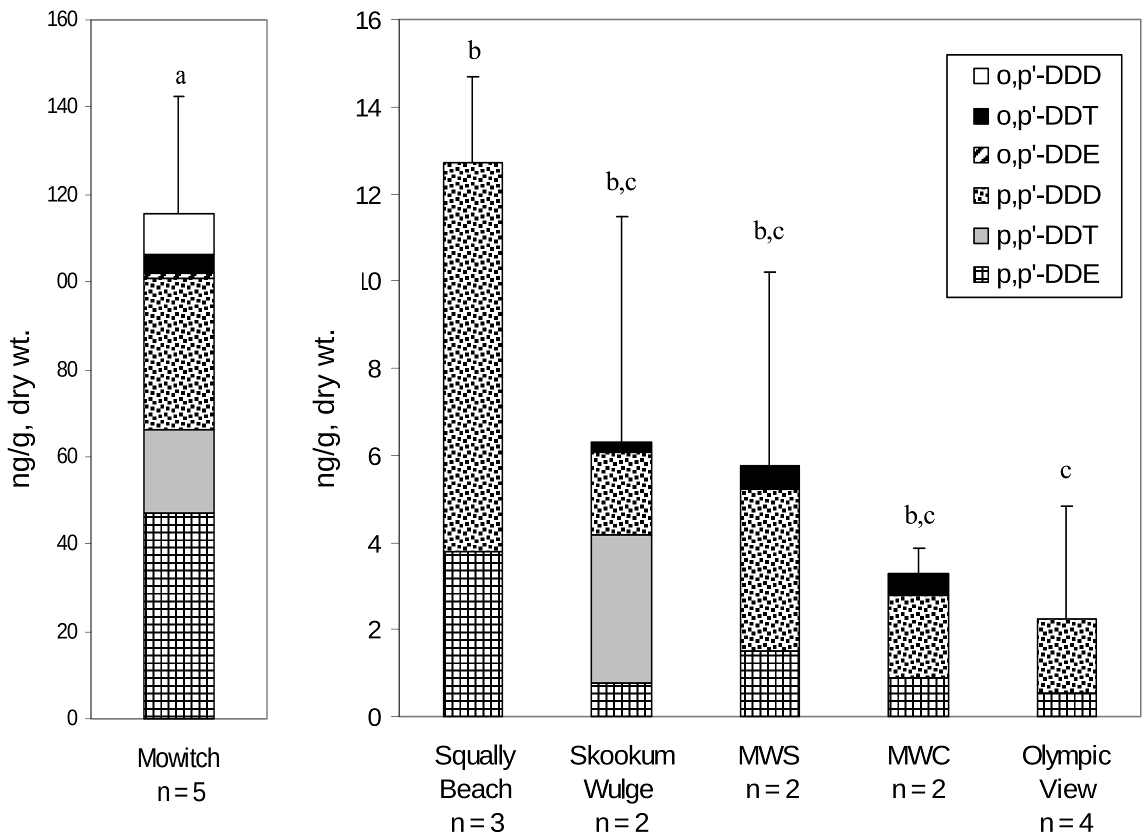

Figure 33. The concentrations of DDT isomers in the sediments from Commencement Bay sites (± SE, bars with unlike letters differ significantly using Tukey-Kramer honestly significant difference test, $p < 0.05$, n = number of samples included in the analysis).

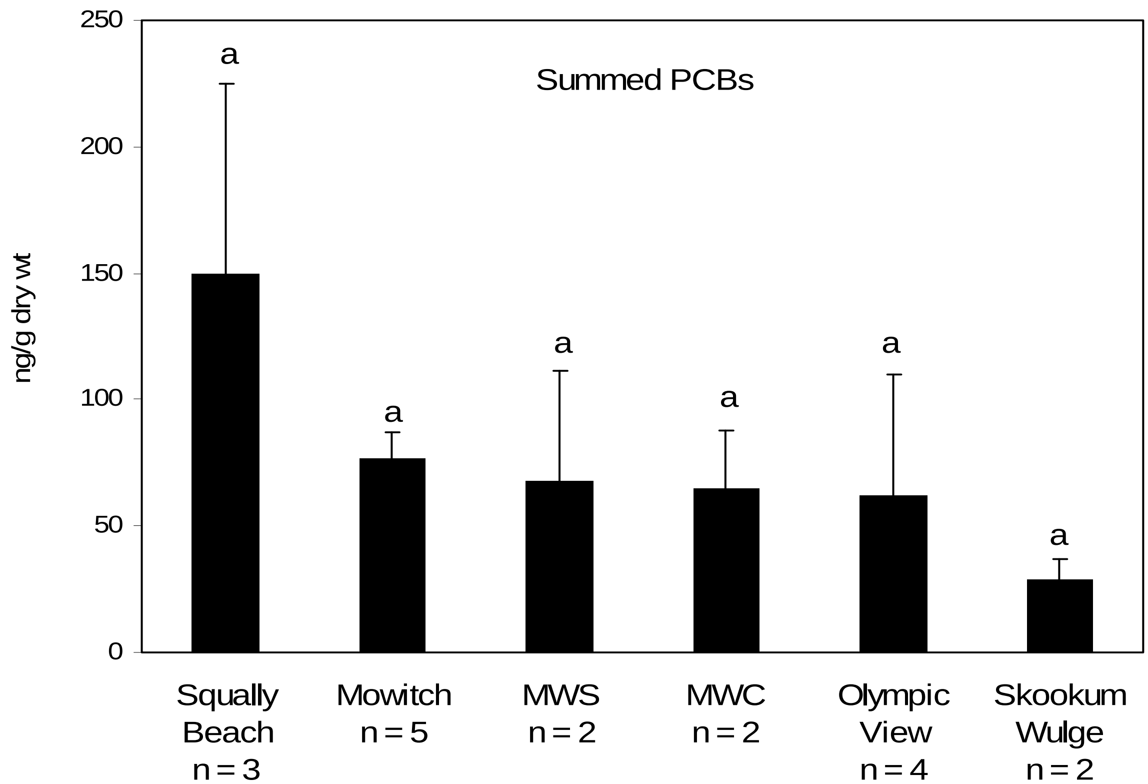

Figure 34. The concentrations of summed PCBs in the sediments from Commencement Bay sites (± SE, bars with unlike letters differ significantly using Tukey–Kramer honestly significant difference test, $p < 0.05$, n = number of samples included in the analysis).

Other OCs were also measured in the Commencement Bay sediments (Table 6). For example, mirex was measured in all 18 sediments analyzed in this study, with average concentrations ranging from 3.5 ng/g dry wt (Skookum Wulge) to 8.6 ng/g dry wt (MWC). We found no significant differences ($p = 0.4155$) in mean levels of mirex from the six sites. Average HCB concentrations ranged from 0.18 ng/g dry wt at Olympic View to 7.8 ng/g dry wt at Squally Beach. Significant differences in HCB levels ($p = 0.0366$) were found among the six sediment sites. For example, samples from Squally Beach and Mowitch had higher average HCB concentrations than sediments from Olympic View. No significant differences in mean HCB levels were found among the other sediment collection sites.

We found significant differences in levels of ∑CHLDs ($p < 0.0001$) in these sediments, with concentrations ranging from below LLOQ (< 0.24 ng/g dry wt) at Olympic View to 14 ng/g dry wt at Mowitch (Table 6). Sediments from Mowitch and MWS had mean ∑CHLDs that were comparable (14 ng/g dry weight) and were higher than the mean values in sediments from the other four sites. The mean ∑CHLD level in sediments from Squally Beach was comparable to the mean value of Skookum Wulge samples, but higher than the mean level in Olympic View sediments. At both Middle Waterway sites (MWC and MWS), the primary chlordane measured in the sediments was *oxy*-chlordane. Among other sites where chlordanes were measured in sediment, the most prevalent analytes were alpha-chlordane, gamma-chlordane, and *trans*-nonachlor.

Lindane was the only HCH measured in the Commencement Bay samples (Table 6). This OC was found in sediments from three sites (MWC, MWS, and Mowitch), with mean concentrations ranging from 6.4 ng/g dry wt at Mowitch to 29 ng/g dry wt at MWS. Significantly different concentrations of lindane were found in samples collected among the six sampling sites. The average levels of this compound in sediments from MWS, MWC, and Mowitch were significantly higher ($p < 0.0001$) than the mean levels in sediments from Olympic View, Squally Beach, and Skookum Wulge (Table 6). Furthermore, the mean lindane level in sediments from MWC was significantly higher ($p < 0.0001$) than the Mowitch value. No other differences were found in sediments collected from the other Commencement Bay sites. Significantly different levels of ∑ELSFs (endosulfans, $p < 0.0001$) were found among the various sites, with Mowitch samples containing the only measurable levels of ∑ELSFs (6.2 ng/g dry wt). Aldrin and dieldrin were below the LLOQ in all sediments analyzed in this study except a sediment sample from Squally Beach (3.7 ng/g dry wt dieldrin).

Salmon stomach contents

Composite samples of Chinook stomach contents collected at MWS, Olympic View, Skookum Wulge, and Yowkwala contained a wide range of PAH concentrations. Levels of ∑LMWAHs ranged from 390 ng/g wet wt at Skookum Wulge to 4,300 ng/g wet wt at Yowkwala (Table 7). We found no clear trends in contributions of individual LMWAHs to ∑LMWAHs in stomach contents at the four collection sites. For example, phenanthrene and 2,6-dimethylnaphthalene comprised 57% of the ∑LMWAHs in Olympic View stomach contents, whereas these compounds only contributed 27% to the ∑LMWAHs in Chinook samples collected at both Skookum Wulge and Yowkwala. At all sampling sites, the contribution of retene to the ∑LMWAHs in samples of Chinook stomach content was much lower (<2%) than percentages of retene measured in sediments (5–15%). ∑HMWAHs concentrations ranged from 820 ng/g wet wt at MWS to 2,300 ng/g wet wt at Olympic View (Table 7). In general, the most prevalent HMWAHs found in the Chinook stomach contents were fluoranthene and chrysene + triphenylene, comprising between 35 and 50% of the ∑HMWAHs. The HMWAHs contributed greater than 50% to the ∑PAHs in Chinook stomach contents at all sites except Yowkwala. At this site, the HMWAHs made up a much smaller percentage (18%) of ∑PAHs compared to samples from the other sites.

Table 7. The summed PAHs and OCs in stomach contents of Chinook salmon from Commencement Bay sites (number of samples per site included in the analysis = 1).

	ng/g wet wt						ng/g wet wt	
Site	∑PCBs	∑DDTs	∑HCHs	∑CHLDs	Lindane	Mirex	∑LMWAHs	∑HMWAHs
Middle Waterway–Simpson	150	66	12	2.6	9.6	3	760	820
Olympic View	70	11	26	15	16	1.7	180	2,300
Skookum Wulge	31	6.9	3.2	0	1.2	2.1	390	1,700
Yowkwala	7.7	8.2	22	0	6.4	<1.1	4,300	950

The Chinook salmon stomach contents also contained a wide range of ∑PCB and ∑DDT concentrations, with ∑PCBs and ∑DDTs ranging from 7.7 ng/g wet wt (Yowkwala) to 150 ng/g wet wt (MWS), and 6.9 ng/g wet wt (Yowkwala) to 66 ng/g wet wt (MWS), respectively (Figure 35, Table 7). Similar to the Commencement Bay sediment samples, the moderately chlorinated PCBs (e.g., PCBs 118, 138, 153) were the most prevalent congeners measured in the Chinook stomach contents. The DDT metabolites *p,p'*-DDE and *p,p'*-DDD (Figure 36) were the predominant DDTs measured in the stomach content samples. These DDTs contributed approximately 42% (Skookum Wulge) to 84% (Olympic View) to the ∑DDTs. In addition, the Chinook stomach content sample from Skookum Wulge, like the Skookum sediments, contained an appreciable level of *p,p'*-DDT (1.3 ng/g wet wt), contributing 52% to the ∑DDTs measured in this sample (Figure 36).

Overall, other OCs were found in much lower concentrations than ∑PCBs or ∑DDTs in the Chinook stomach samples. For example, levels of ∑HCHs ranged from 3.2 ng/g wet wt at Skookum Wulge to 26 ng/g wet wt at Olympic View (Table 7) and were 2–11 times lower than the ∑PCBs at the same sites at all sites except Yowkwala. At this site, the stomach sample had a higher level of ∑HCHs (22 ng/g wet wt) compared to the concentrations of either ∑PCBs (7.7 ng/g wet wt) or ∑DDTs (7.9 ng/g wet wt). Lindane was the major contributor to ∑HCHs at Olympic View (62%) and MWS (80%), whereas alpha-HCH was the most abundant HCH at Skookum Wulge (63%) and Yowkwala (71%). The ∑CHLDs in the Chinook stomach contents ranged from below LLOQ (at both Skookum Wulge and Yowkwala) to 15 ng/g wet wt at Olympic View. Mirex was detected in samples collected at all sites except Yowkwala, whereas HCB was measured in samples from Skookum Wulge and Yowkwala but not in stomach contents from the other two sites. The Chinook stomach sample from Skookum Wulge was the only sample that that contained measurable levels of dieldrin, whereas aldrin and the endosulfans were not detected in any of the stomach content samples analyzed in this study.

Whole bodies of salmon

Because the number of unmarked juvenile Chinook salmon at the Commencement Bay restoration sites was very small, it was not possible to collect enough unmarked fish to characterize their level of contaminant exposure. Consequently, the contaminant concentrations in stomach contents and whole bodies reported here are representative of concentrations in released hatchery fish. Concentrations in unmarked (and presumably wild) stocks may be somewhat different from those in released hatchery fish, depending on their migration history and residence times of both groups of fish within the estuary and the extent to which contaminants found in hatchery fish were accumulated during hatchery rearing. With these limitations in mind, the data from this study provide useful information about the potential for contaminant uptake by outmigrant juvenile salmon utilizing these restoration sites.

In Chinook salmon, average concentrations of ∑DDTs (based on wet weight) ranged from 5.5 ng/g wet wt at Tahoma Salt Marsh to 37 ng/g wet wt at Mowitch, while mean ∑PCB levels ranged from 24 ng/g wet wt at Tahoma Salt Marsh to 44 ng/g wet wt at Yowkwala (Table 8). The mean HCB and ∑PCB TEQ concentrations ranged from 0.28 ng/g wet wt (Tahoma Salt Marsh) to 0.91 ng/g wet wt (Mowitch), and 0.10 pg/g wet wt (Tahoma Salt Marsh) to 0.39 pg/g wet wt (Mowitch), respectively. Significant differences ($p < 0.0001$) in concentrations of ∑DDTs (based on wet wt) were found in Chinook salmon from Mowitch,

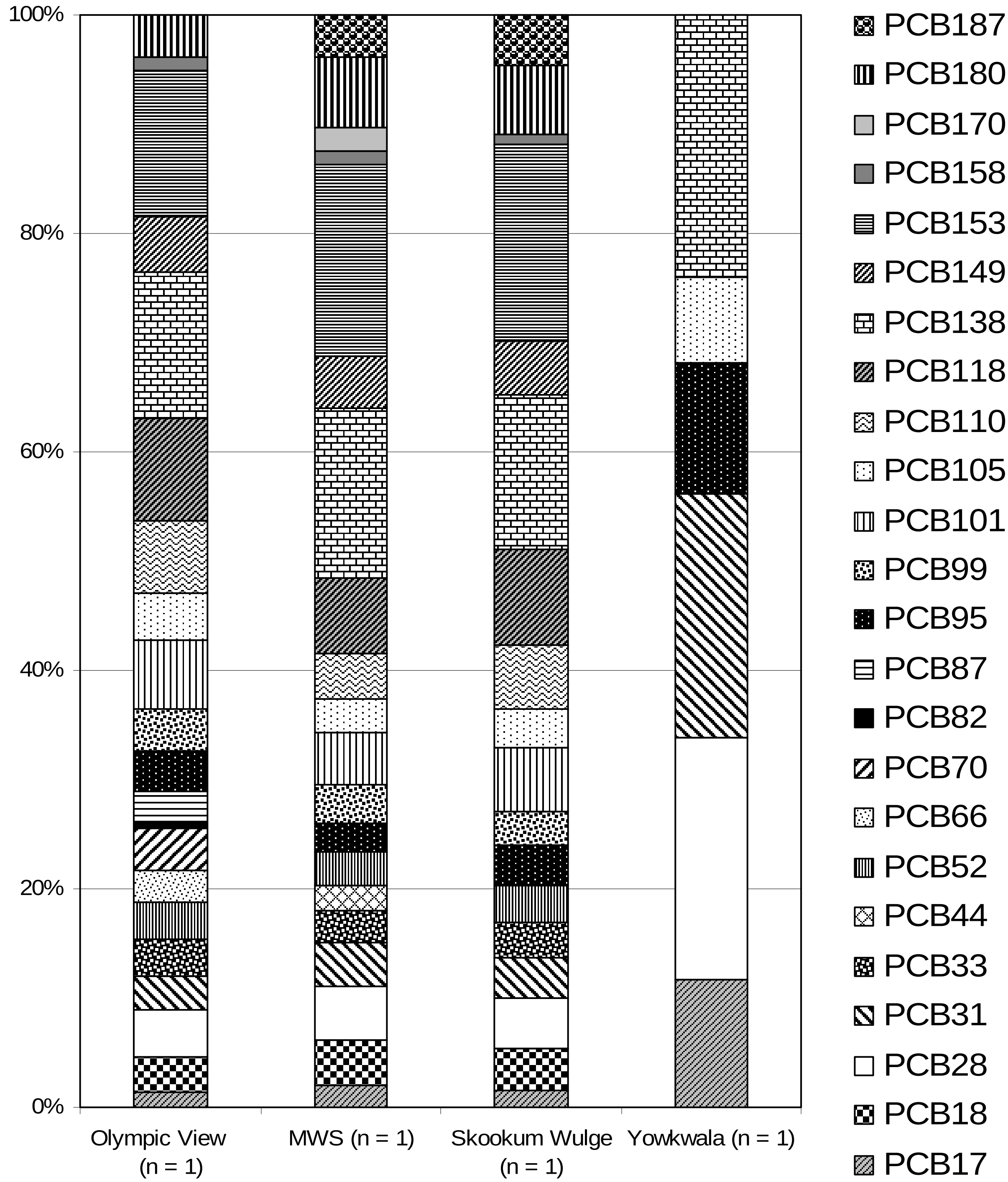

Figure 35. The percentage of individual PCB congeners contributing to summed PCBs in the stomach contents of juvenile Chinook salmon (n = number of samples included in the analysis).

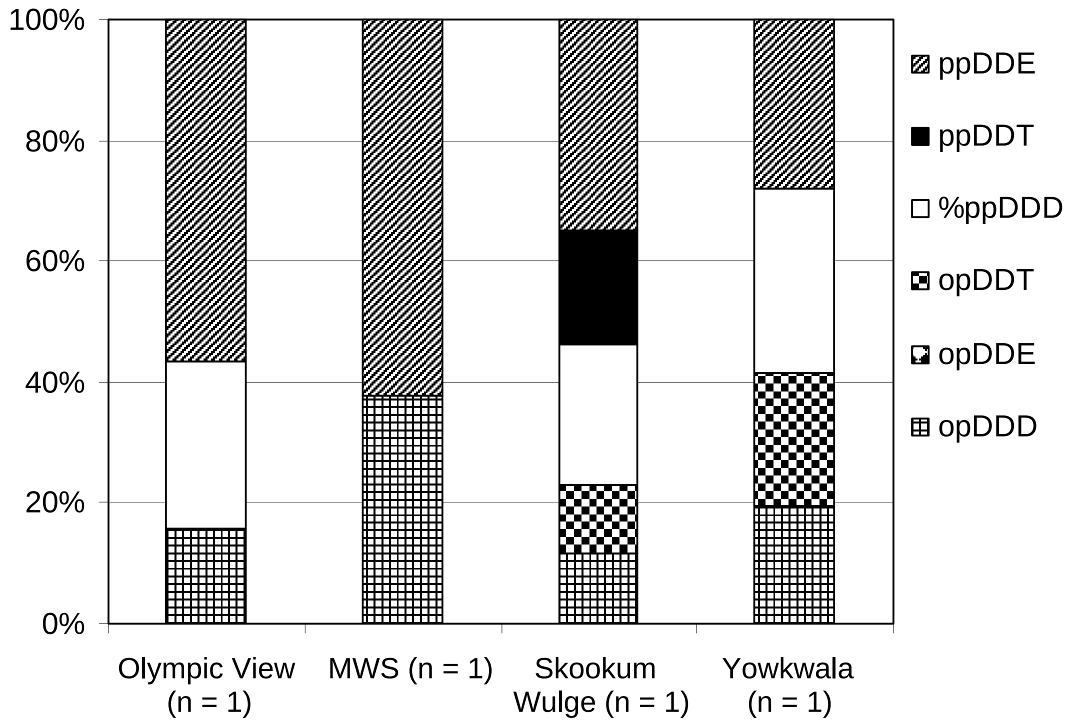

Figure 36. The percentages of individual DDT isomers contributing to summed DDTs in the stomach contents of juvenile Chinook salmon (n = number of samples included in the analysis).

Tahoma Salt Marsh, and Yowkwala. The mean ∑DDT level in Chinook from Mowitch was higher than those in fish from the other two sites. Chinook from Yowkwala had a significantly higher ($p < 0.0001$) mean concentration of ∑PCBs than fish from Tahoma Salt Marsh and Mowitch, but no significant differences in mean values were found in fish from these two sites. The mean HCB level was higher ($p < 0.0001$) in Mowitch Chinook compared to the mean levels in fish collected at the other two sites, and Yowkwala fish had a higher mean HCB value than the Tahoma Salt Marsh Chinook.

When we compared the Chinook salmon OC concentrations on a lipid weight basis, average concentrations of ∑DDTs ranged from 400 ng/g lipid at Yowkwala to 1,700 ng/g lipid at Mowitch, whereas the mean ∑PCBs ranged from 1,200 ng/g lipid at Mowitch to 2,800 ng/g lipid at Yowkwala. Chinook from Mowitch contained a significantly higher ($p < 0.0001$) mean ∑DDT level than did fish at the other two sites. Fish from Yowkwala had higher ($p = 0.0005$) ∑PCB values than Tahoma Salt Marsh and Mowitch Chinook. The average levels of HCB ranged from 31 ng/g lipid at Tahoma Salt Marsh to 42 ng/g lipid at Yowkwala. The mean concentrations of ∑PCB TEQs ranged from 11 pg/g lipid (Tahoma Salt Marsh) to 25 pg/g lipid (Yowkwala). No significant differences were found in mean whole body levels of HCB ($p = 0.566$) or ∑PCB TEQs ($p = 0.743$) among the three sites. Mean percent lipid values of Chinook whole bodies ranged from 0.90% at Tahoma Salt Marsh to 2.5% at Mowitch. Chinook salmon from this site had a significantly higher ($p = 0.0056$) mean lipid concentration than fish from Tahoma Salt Marsh. No other differences in mean percent lipid values were observed.

Table 8. The concentrations (± SE) of HCB, ∑PCBs, ∑DDTs, and ∑PCB TEQs in whole body samples from Commencement Bay sites (PSS = Pacific staghorn sculpin, superscript a = significantly different than other sites, same species, superscript b = significantly different than other salmonid species with dissimilar or no letters, n = number of samples included in the analysis).

Species	**Site**	**% lipid**	**HCB**	**∑PCBs** (ng/g wet wt)	**∑PCB TEQs** (pg/g wet wt)	**∑DDTs** (ng/g wet wt)	**∑DDTs/ ∑PCBs**
Chinook (n = 14)	Mowitch	2.5 ± 0.69	0.91[a] ± 0.42	24.9 ± 4.0	0.39 ± 0.11	36.9[a] ± 7.3	1.51 ± 0.32
Chinook (n = 2)	Tahoma Salt Marsh	0.90 ± 0.0	0.28 ± 0.028	24.0 ± 1.4	0.10 ± 0.002	5.5 ± 0.24	0.23 ± 0.024
Chinook (n = 5)	Yowkwala	1.7 ± 0.67	0.67[b] ± 0.22	44.0[a,b] ± 12	0.36 ± 0.35	6.1 ± 2.9	0.15 ± 0.094
Chum (n = 4)	Tahoma Salt Marsh	1.5 ± 1.3	0.49 ± 0.18	22 ± 5.6	0.13 ± 0.049	2.7 ± 1.4	0.14 ± 0.11
Coho (n = 6)	Yowkwala	4.4 ± 1.1	0.63[b] ± 0.061	6.7 ± 0.72	0.069 ± 0.027	4.2 ± 0.46	0.64 ± 0.082
Pink (n = 2)	Yowkwala	0.80 ± 0.14	ND	15 ± 0.71	0.048 ± 0.022	0.90[a,b] ± 0.29	0.061 ± 0.017
Pink (n = 2)	Tahoma Salt Marsh	1.6[a] ± 0.28	0.27[a] ± 0.035	15 ± 5.0	0.084 ± 0.056	1.8[a] ± 0.78	0.12 ± 0.013
PSS (n = 3)	Middle Waterway at Simpson	0.54 ± 0.045	0.54 ± 0.47	82 ± 47	1.2 ± 0.66	2.8 ± 1.5	0.047 ± 0.020
PSS (n = 4)	Middle Waterway at City	0.86 ± 0.045	0.25 ± 0.12	67 ± 31	1.1 ± 0.68	2.8 ± 1.5	0.041 ± 0.008
PSS (n = 3)	Mowitch	0.57 ± 0.015	7.1[b] ± 0.90	91.3 ± 16	2.0 ± 0.58	40[a] ± 5.6	0.45 ± 0.054
PSS (n = 4)	Olympic View	0.61 ± 0.15	0.71 ± 0.27	100[c] ± 30	2.4[a] ± 0.73	4.1 ± 1.3	0.041 ± 0.003
PSS (n = 3)	Skookum Wulge	0.63 ± 0.095	3.30[b] ± 1.3	36[a] ± 9.3	0.62 ± 0.33	5.6 ± 1.6	0.15 ± 0.021
PSS (n = 1)	Yowkwala	1.2	3.0[b]	30	0.40	2.4	0.08

OCs were also measured in whole bodies of non-Chinook salmonids captured in Commencement Bay (Table 8). The mean levels of ∑DDTs (based on wet wt) ranged from 0.90 ng/g wet wt in pink salmon from Yowkwala to 4.2 ng/g wet wt in coho salmon collected at this same site. The average ∑PCB and ∑PCB TEQs concentrations ranged from 6.6 ng/g wet wt (Yowkwala coho) to 22 ng/g wet wt (Tahoma Salt Marsh chum) and 0.048 pg/g wet wt (pink salmon from Yowkwala) to 0.13 pg/g wet wt (chum from Tahoma Salt Marsh), respectively. The average levels of HCB ranged from below LLOQ in pink salmon from Yowkwala to 0.63 ng/g wet wt in coho from the same site.

When based on lipid weight, the mean concentrations (Figure 37) of ∑DDTs ranged from 100 ng/g lipid in Yowkwala coho to 230 ng/g lipid in chum collected at Tahoma Salt Marsh, while the mean ∑PCB levels ranged from 160 ng/g lipid in Yowkwala coho to 2,600 ng/g lipid in chum from Tahoma Salt Marsh. The average HCB and ∑PCB TEQ concentrations ranged from below LLOQ in Yowkwala pink salmon to 17 ng/g lipid in Tahoma Salt Marsh pink salmon and from 1.6 pg/g lipid (Yowkwala coho) to 15 pg/g lipid (Tahoma Salt Marsh chum), respectively. We found a wide range of percent lipid values in these fish from 0.80% in whole bodies of Yowkwala pink salmon to 4.4% in Yowkwala coho.

Other than Chinook, pink salmon were the only salmon species collected at more than one site (Tahoma Salt Marsh and Yowkwala). Based on wet weight, we found that there was a significantly higher ($p = 0.0072$) mean HCB concentration in pink salmon from Tahoma Salt Marsh compared to the mean levels in Yowkwala fish. The mean levels of ∑PCBs, ∑PCB TEQs, and HCB in whole bodies showed no significant differences ($p = 0.917$, $p = 0.506$, and $p = 0.248$, respectively) between the two sites.

When based on lipid weight, an appreciably higher ($p = 0.048$) mean HCB level was found in Tahoma Salt Marsh salmon compared to Yowkwala pink salmon. Similar to the OC levels based on wet weight, mean values of ∑DDTs, ∑PCBs, and ∑PCB TEQs (based on lipid weight) were not significantly different ($p = 0.928$, $p = 0.200$, and $p = 0.819$, respectively) between the two sites. In addition, the average percent lipid value in the Tahoma Salt Marsh fish (1.6%) was higher than that of the Yowkwala pink salmon (0.80%), but this difference was not significant ($p = 0.0625$).

Proportions of individual DDTs and PCBs contributing to the summed ∑DDTs and ∑PCBs in whole bodies of salmon are shown in Figure 38 and Figure 39, respectively. At all sites, the most abundant PCB congeners measured in the Chinook salmon whole bodies were the moderately chlorinated congeners (e.g., PCBs 101, 153); these two congeners accounted for 35–46% of the summed PCBs. Similarly, PCBs 101 and 153 were the predominant PCBs measured in the whole bodies of other species of salmon, contributing 38–43% to the summed PCBs (Figure 39). The most abundant DDTs measured in the salmon bodies were p,p'-DDE and p,p'-DDD, accounting for greater than 90% of the summed DDTs at all sites except Mowitch (Figure 38). Interestingly, at this site, p,p'-DDT, the parent DDT, contributed 31% to the summed DDT values in Chinook salmon. In juvenile Chinook collected at all sites, the predominant dioxin-like PCB congeners that contributed to the ∑PCB TEQs were PCBs 105, 118, and 156, accounting for 74–100% of the total ∑PCB TEQs (data not shown). PCB 118 was the major dioxin-like congener measured in whole bodies of chum, coho, and pink salmon, contributing 53–100% to the summed TEQ.

We compared the mean OC levels and lipid content of various species of salmon collected from the same sites (Tahoma Salt Marsh and Yowkwala) in 2002 (Table 8). At Tahoma Salt Marsh, we found that pink salmon contained lower mean levels of OCs than did other species of salmon, but these differences ($p > 0.05$ for all species) were not significant. For example, the mean ∑PCB levels of Chinook (24 ng/g wet wt) and chum salmon (23 ng/g wet wt) were higher than the mean level in pink salmon (15 ng/g wet wt). Similarly, mean values of ∑PCB TEQs, ∑DDTs, and HCB (based on wet weight) in pink salmon were lower than those in Chinook or chum salmon, but these differences were not significant ($p > 0.05$ among all species). When based on lipid weight, the mean concentration of ∑DDTs (120 ± 69 ng/g lipid) in pink salmon was significantly lower ($p = 0.022$) than the mean levels in Chinook salmon (610 ± 27 ng/g lipid) and chum salmon (230 ± 79 ng/g lipid).

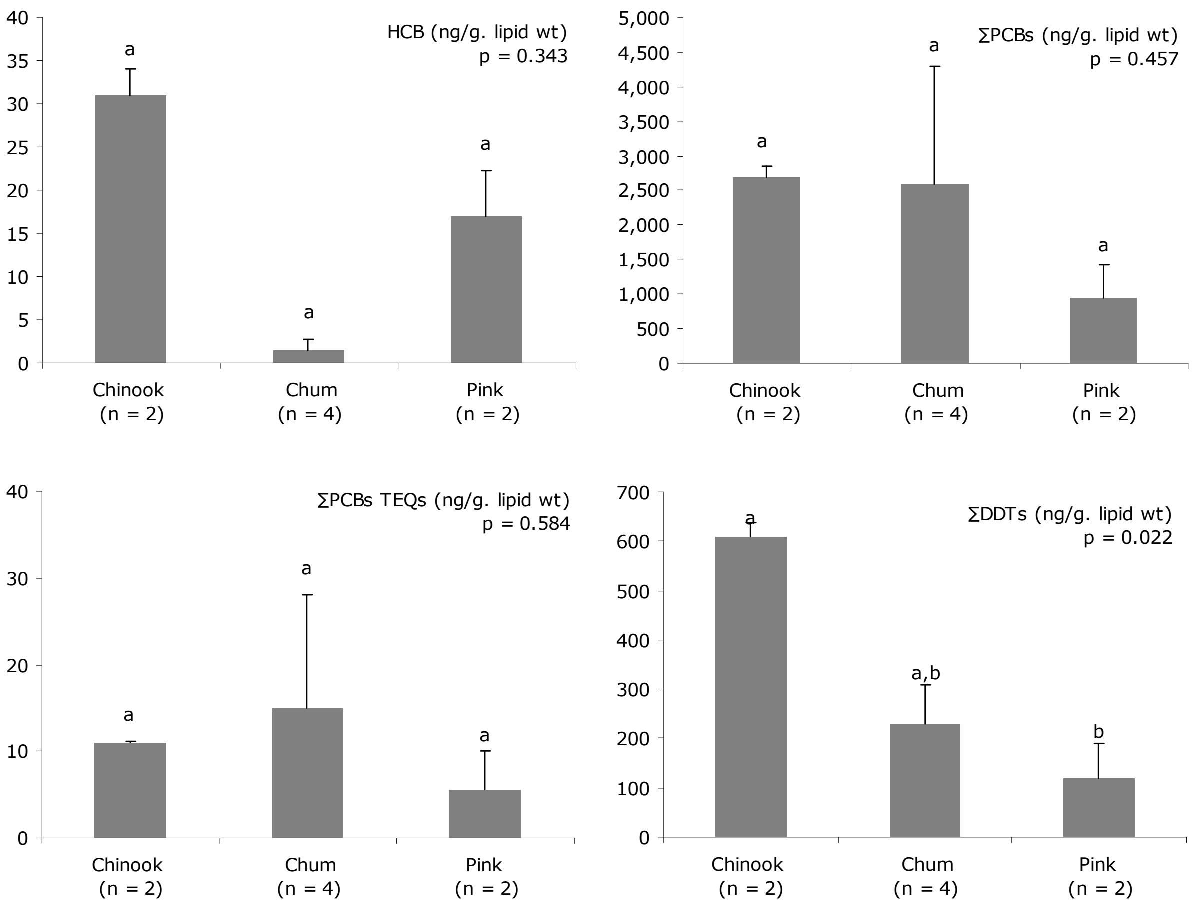

Figure 37. The mean concentrations (± SE) of hexachlorobenzene (HCB), ∑PCBs, ∑PCB TEQs, and ∑DDTs measured in whole bodies of three species of juvenile samonids collected at Tahoma Salt Marsh (2002). Bars with unlike letters differ significantly using Tukey-Kramer honestly significant difference test, $p < 0.05$, and n = number of samples included in the analysis.

In contrast to ∑DDTs, no significant differences ($p = 0.343$, $p = 0.457$, and $p = 0.584$, respectively) were found in mean HCB, ∑PCB, and ∑PCB TEQ levels (based on lipid weight) among the three salmon species captured at Tahoma Salt Marsh. Furthermore, no significant differences ($p = 0.717$) were found in percent lipid values among Chinook, chum, and pink salmon from this site. At Yowkwala, the wet weight mean levels of HCB and ∑DDTs in Chinook and coho salmon were significantly higher ($p < 0.0001$) for both OC groups than the mean concentrations in pink salmon (Table 8). Chinook also had significantly higher mean levels of ∑PCBs and ∑PCB TEQs (based on wet weight) than either pink or coho salmon. On a lipid-adjusted basis, Chinook salmon collected at Yowkwala had higher mean concentrations of HCB, ∑PCBs, and ∑DDTs ($p < 0.0001$, $p < 0.0001$, and $p = 0.0008$, respectively) than the average levels in coho from this site (Figure 40). Juvenile Chinook also had higher average levels of HCB and ∑DDTs than pink salmon. The mean percent lipid value of coho (4.4 ± 1.1%) was significantly higher ($p < 0.0001$) than the mean values of Chinook salmon (1.7 ± 0.67%) and pink salmon (0.80 ± 0.14%).

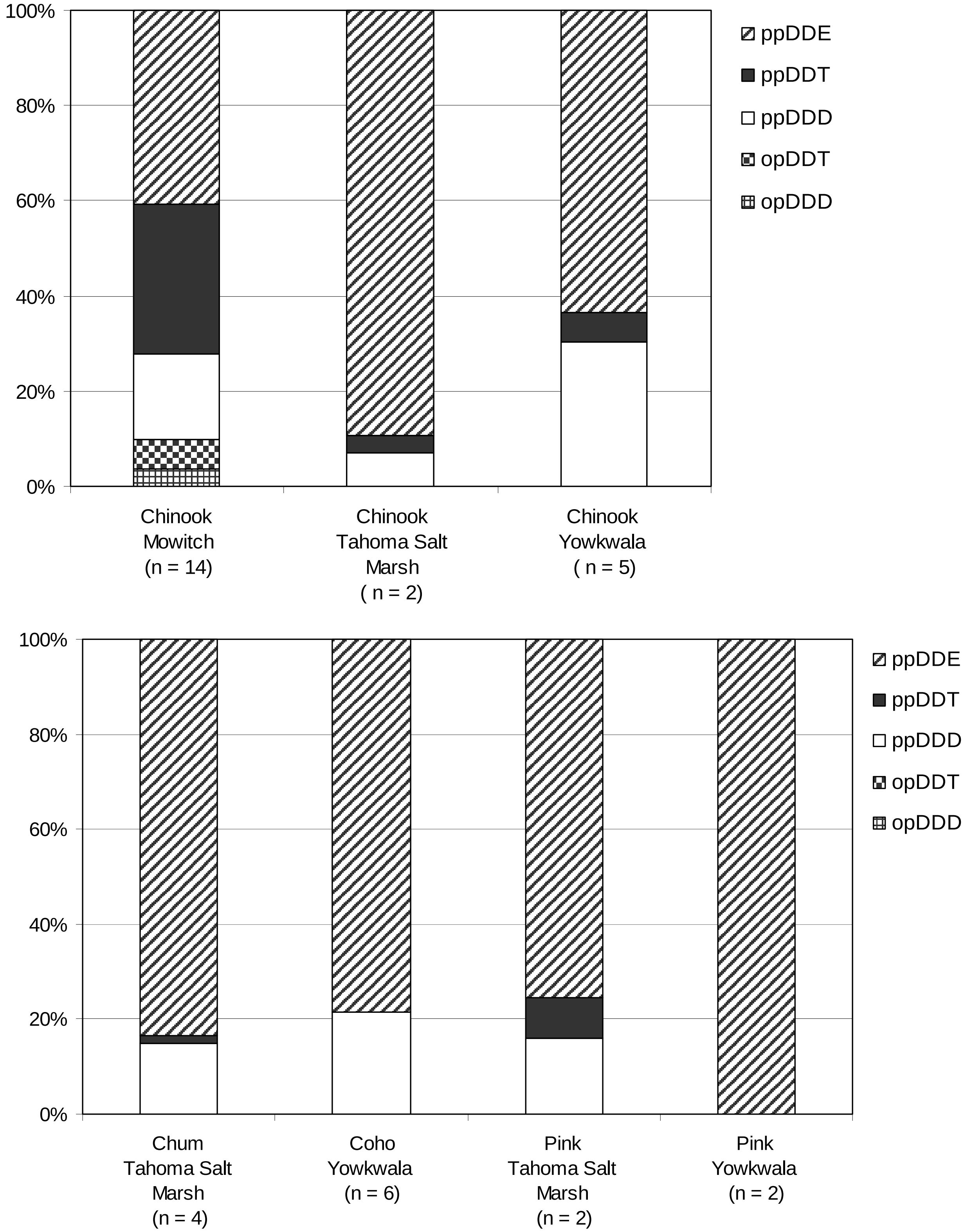

Figure 38. The percentages of individual DDT isomers contributing to summed DDTs in the whole bodies of juvenile salmon (n = number of samples included in the analysis).

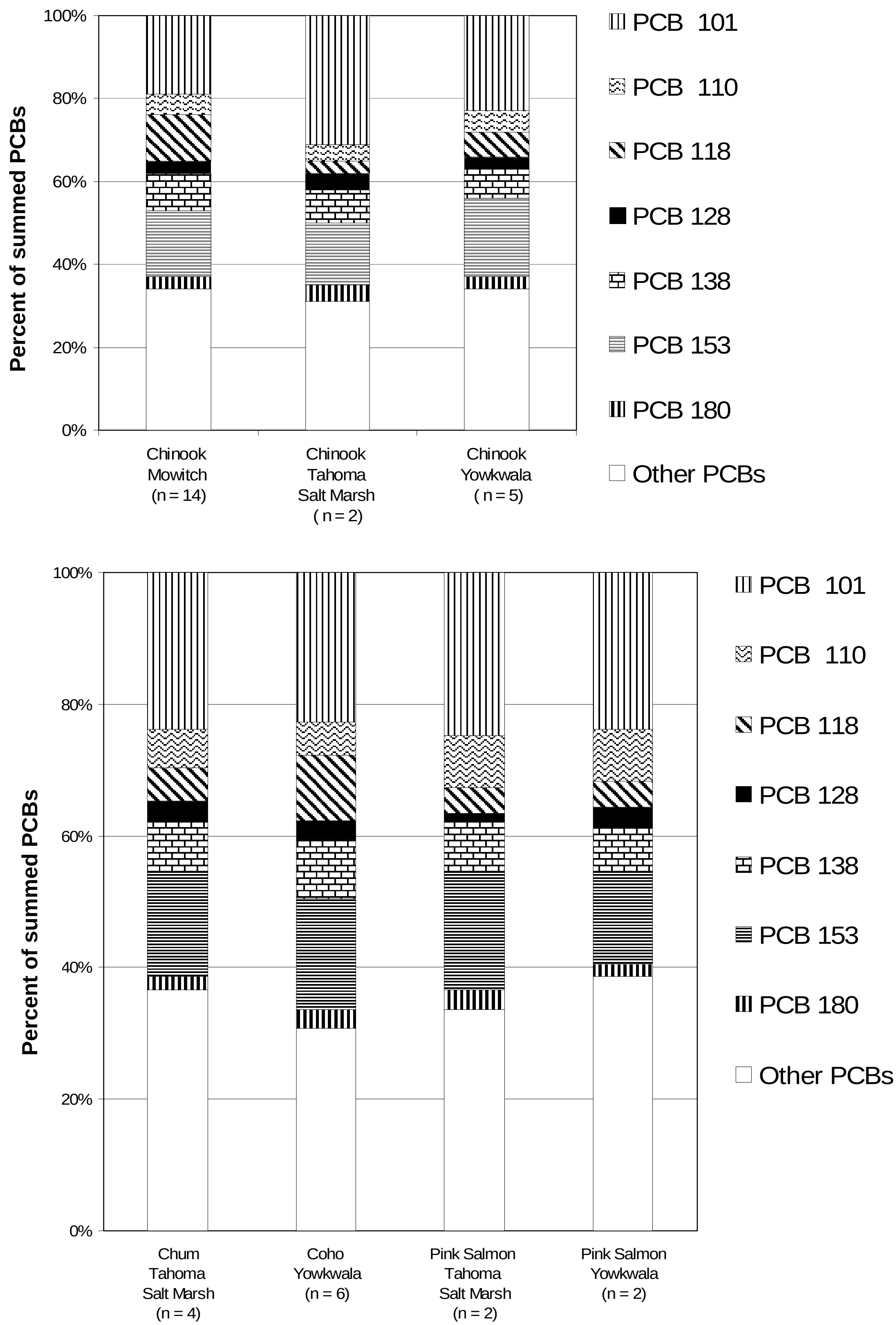

Figure 39. The percentages of individual PCB cogeners contributing to summed PCBs in the whole bodies of juvenile salmon (n = number of samples included in the analysis).

Whole bodies of Pacific staghorn sculpin

At all sites, PCBs were the OCs found in the highest concentrations in the whole bodies of Commencement Bay PSS. Average ∑PCB levels (based on wet wt) in PSS from Mowitch, Olympic View, and MWC were comparable, ranging 80–100 ng/g wet wt (Table 8, Figure 40) whereas the ∑PCB level in fish from MWS was slightly lower (67 ng/g wet wt). Fish collected at Skookum Wulge and Yowkwala had the lowest ∑PCB concentrations, ranging 30–35 ng/g wet wt. However, these concentrations were not significantly different among sampling sites, except for PSS collected at Olympic View and Skookum Wulge (significantly greater and lower, respectively, $p = 0.02$). For the lipid-normalized ∑PCB concentrations, PSS from Olympic View, Mowitch, MWC, and MWS had comparable mean levels of ∑PCBs (ranging 9,600–16,000 ng/g lipid) and all were significantly higher than the levels measured in fish from Yowkwala (2,500 ng/g lipid). Fish from Olympic View and Mowitch had significantly higher higher mean levels of ∑PCBs (16,000 ng/g lipid at each site) than fish from Skookum Wulge (5,600 ng/g lipid) and Yowkwala. No significant differences ($p > 0.05$) in ∑PCBs were found in fish from the other sites. PCBs 101, 138, and 153 were the most abundant congeners measured in the PSS whole bodies, contributing 52–58% to the ∑PCBs.

The average concentrations of ∑PCB TEQs in the PSS ranged from 0.40 pg/g wet wt at Yowkwala to 2.4 pg/g wet wt at Olympic View. PSS from Olympic View had a significantly higher ($p = 0.0378$) mean ∑PCB TEQ level than did fish from Skookum Wulge. No other significant differences were found among fish from other sites. The lipid-normalized average ∑PCB TEQ concentrations in the Commencement Bay PSS ranged from 33 pg/g lipid (Yowkwala) to 380 pg/g lipid (Olympic View). The mean TEQ values in PSS from Olympic View and Mowitch were not significantly different from one another (380 pg/g lipid and 350 pg/g lipid, respectively) but were significantly higher ($p = 0.005$) than the levels measured in fish from MWC and Skookum Wulge. No significant differences ($p > 0.05$) in levels of ∑PCB TEQs were found among fish from among the other sites. Three dioxin-like PCB congeners (PCBs 105, 118, and 156) were measured in all PSS samples analyzed in the current study. PCB 118 comprised 40–62% of ∑PCB TEQs in these tissues.

Mean levels of ∑DDTs were fairly similar in fish collected at MWC, MWS, Yowkwala, Skookum Wulge, and Olympic View, ranging 2–5 ng/g wet wt and 200–820 ng/g lipid. At Mowitch, in contrast, the mean concentrations of ∑DDTs (40 ng/g wet wt and 7,100 ng/g lipid) were approximately an order of magnitude higher than the average values at the other sites. The mean concentration of ∑DDTs (based on wet wt) in PSS from Mowitch was significantly higher ($p = 0.0001$) than the average values in PSS from any other sites. When comparing mean levels on a lipid weight basis, it was found that the ∑DDT level in PSS from Mowitch was significantly higher ($p = 0.0001$) than the mean levels in fish from the other collection sites. Similar to the case with juvenile salmon, the predominant DDTs measured in the PSS whole body samples were *p,p'*-DDE and *p,p'*-DDD. These compounds accounted for 75% (Skookum Wulge) to 100% (MWC, MWS, Olympic View, Yowkwala) of the ∑DDTs.

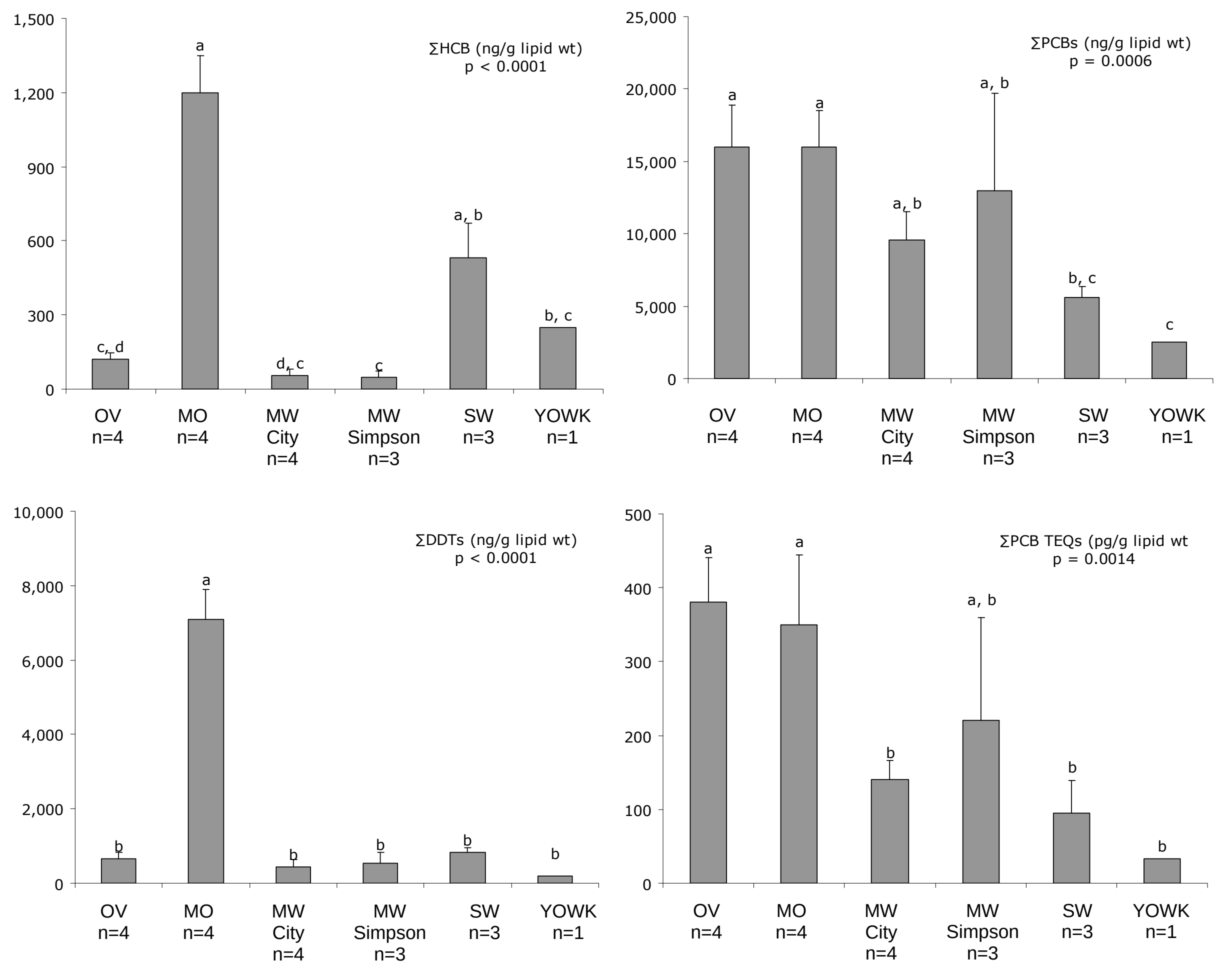

Figure 40. The mean concentrations (± SE) of hexachlorobenzene (HCB), ∑PCBs, ∑PCB TEQs, and ∑DDTs measured in whole bodies of PSS (2003). Bars with unlike letters differ significantly using Tukey-Kramer honestly significant difference test, $p < 0.05$, and n = number of samples included in the analysis. Site abbreviations are OV = Olympic View, MO = Mowitch, MW City = Middle Waterway at City, MW Simpson = Middle Waterway at Simpson, SW = Skookum Wulge, and YOWK = Yowkwala.

Average HCB levels in PSS collected at the six Commencement Bay sites ranged from 0.25 ng/g wet wt at MWS to 7.1 ng/g wet wt at Mowitch (Table 8, Figure 40). The HCB levels in fish from Mowitch and Skookum Wulge were not significantly different, but they were significantly higher ($p < 0.0001$) than the mean values of fish from MWC, MWS, and Olympic View. In addition, fish from Mowitch and Skookum Wulge had mean HCB levels that were significantly higher than PSS from Olympic View and MWC. When based on lipid weight, mean concentrations of HCB ranged from 48 ng/g lipid at MWS to 1,200 ng/g lipid at Mowitch (Figure 40). Average HCB levels in PSS from Mowitch and Skookum Wulge were significantly higher ($p < 0.0001$) than the mean concentrations in fish from Olympic View and the two Middle Waterway sites. The HCB level in PSS from Yowkwala was significantly higher than the mean levels in fish from the two Middle Waterway sites, but was not significantly different than the mean HCB values of fish from Skookum Wulge and Olympic View.

Differences in OC Levels between Pacific Staghorn Sculpin and Chinook Salmon

At Yowkwala and Mowitch, where both salmon and PSS were collected, the concentration of OCs in both species was highest at Mowitch. However, within a site, average concentrations of OCs tended to be higher in PSS than in salmonids, with the most marked differences in fish from the Mowitch site. For example, at Mowitch, the PSS ∑PCB level (91 ng/g wet wt) was higher than the Chinook PCB concentration (25 ng/g wet wt), but at Yowkwala, the levels were similar (44 compared to 30 ng/g wet wt). Similar patterns were seen for PCB-TEQs and HCB. At Mowitch, the PSS HCB concentration was 7.1 ng/g wet wt compared to 0.91 ng/g wet wt for Chinook, whereas at Yowkwala, the HCB level was 3.0 ng/g wet wt for PSS and 0.67 ng/g wet wt for Chinook. However, the relationship was quite different for DDTs, where no species differences ($p > 0.05$) in levels were observed.

The percent lipid content of the PSS (0.57 ± 0.015%) was significantly lower ($p = 0.0002$) than the Chinook value (2.5 ± 0.69%). However, even on a lipid-weight basis, OC concentrations were higher in PSS than in salmon, especially at the Mowitch site. For example, the mean PSS ∑PCB level at Mowitch was 16,000 ± 2,900 ng/g lipid, whereas in Chinook, the mean value was 1,200 ± 650 ng/g lipid. At Yowkwala, differences in OC concentrations between these two species were less pronounced. The ∑PCB levels in Chinook and PSS were 2,800 ng/g lipid and 2,500 ng/g lipid, respectively. Similar trends were found for ∑PCB TEQs and ∑DDTs. However, the HCB concentrations were different between PSS (250 ng/g lipid) and Chinook (42 ng/g lipid) collected at Yowkwala.

Levels of PAH Metabolites in Bile of Salmon and Pacific Staghorn Sculpin

Levels of PAH metabolites could only be measured in Chinook salmon from two sites, Skookum Wulge and Yowkwala, because sufficient bile could not be collected at the other sites. Concentrations of both high and low molecular weight metabolites were similar in fish from the two sites (Table 9). For example, the concentrations of PHN equivalents (47,000 ng/g bile at Skookum Wulge and 53,000 ng/g bile at Yowkwala) were comparable between the two sites. The mean level of benzo[a]pyrene (BaP) equivalents determined in Chinook from Skookum Wulge (940 ng/g bile) was similar to the average in fish from Yowkwala (1,200 ng/g bile at Yowkwala).

Similar to the situation with Chinook bile findings, moderate levels of PAH metabolites were measured in the bile of individual PSS. The mean concentrations of PHN equivalents ranged from 63,000 ng/g bile at Olympic View to 75,000 ng/g bile at Mowitch, while the BaP equivalents ranged from 980 ng/g bile at Olympic View to 1,300 ng/g bile at Mowitch (Table 9). No significant differences were found in mean concentrations of either PHN ($p = 0.871$) or BaP equivalents ($p = 0.734$) in bile of fish collected among the four sites.

Table 9. Bile FACs in juvenile Chinook salmon and Pacific staghorn sculpin (PSS). Concentrations in parts per billion (ppb) (ng/g ± SE) based on total area compared to the fluorescence of a PHN standard at PHN wavelengths (260/380 nm), a BaP standard at BaP wavelengths (380/430 nm), or napthalene (NPH) wavelengths (293/335 nm) (n = number of samples included in the analysis).

		Equivalents (ng/g)		**Protein corrected equivalents (ng/mg)**	
Species	**Site**	PHN	BaP	PHN	BaP
Chinook	Skookum Wulge (n = 1)	47,000	940	15,000	300
	Yowkwala (n = 1)	53,000	1,200	10,000	230
PSS	Middle Waterway at Simpson (n = 6)	67,000 ± 19,000	1,100 ± 330	9,600 ± 3,200	150 ± 53
	Mowitch (n = 6)	75,000 ± 38,000	1,300 ± 630	10,000 ± 5,400	170 ± 85
	Olympic View (n = 6)	63,000 ± 23,000	980 ± 400	11,000 ± 7,000	170 ± 86
	Yowkwala (n = 2)	67,000 ± 16,000	1,000 ± 370	11,000 ± 6,100	180 ± 120

Discussion

Coastal and nearshore estuarine ecosystems play a vital role as rearing habitat for early life stages of large numbers of marine fish species (Levy and Northcote 1982, Day et al. 1989, Beck et al. 2001, Beck et al. 2003, Rice et al. 2005). Some of these areas may contribute disproportionately to adult populations, with even small areas providing a significant number of recruits (Beck et al. 2001). Marine and estuarine ecosystems can be particularly important in the recovery of species at risk (Feist et al. 2003), such as listed Pacific Northwest salmon stocks that all use estuaries as juveniles (Conley 1977, Meyer 1979, Levy and Northcote 1982, Simenstad et al. 1982, Aitkin 1998, Cederholm et al. 2000, Cornu and Sadro 2002, Gray et al. 2002, Rice et al. 2005).

While estuarine and coastal ecosystems are some of the most diverse and complex habitats in the marine environment, they are also some of the most heavily impacted by human activities (Shreffler et al. 1990, Cederholm et al. 2000, Beck et al. 2001, Beck et al. 2003, Rice et al. 2005). Changes to these habitats can result in complex and cumulative effects on estuarine fish populations that may be especially significant for threatened and endangered species. Restoration of degraded marine environments has become a major focus of regulatory and nonregulatory responses to these anthropogenic alterations (Rice et al. 2005).

Restoration of degraded marine and estuarine areas tends to focus on reproduction of the physical attributes of the original system (Gray et al. 2002), with more limited attention to problems associated with anthropogenic contamination. Moreover, for projects involving physical habitat restoration, sediment contaminant remediation, or both, there has generally been insufficient monitoring of the success or failure of these efforts (Michener 1997, Roni et al. 2002, Beck et al. 2003). Many of these monitoring efforts are poorly designed, lack long-term data collection, have no initial baseline data, or are deficient in appropriate reference sites. Toxicant exposure of biota at restoration sites in contaminated environments is often not measured. However, monitoring of restoration activities is an important step in determining their effectiveness, particularly in light of the large sums of money spent to improve degraded habitat (Roni et al. 2002, Steel et al. 2003); for example, $3,364,929 were spent for restoration of five sites in the Commencement Bay area (online at: www.darp.noaa.gov/northwest/cbay).

Restoration activities in Commencement Bay included sediment remediation to remove heavy metals at the Mowitch site and dioxins at Olympic View; removal of pilings and structures; modification of stream channels and excavation of sites to create more natural shorelines; and revegetation with native marsh and riparian plants. The specific focus of most of these efforts was on restoring physical habitat attributes that would make these sites suitable for outmigrant juvenile salmon. We are currently monitoring the effects of these efforts by measuring habitat use by salmonids and other fish species, the chemical contaminant levels in sediment, fish and their prey, and by monitoring the physical attributes (salinity and temperature) of the sites. With the constraints of the original sampling design and only two years of data thus far, definitive conclusions on the effectiveness of these restoration activities cannot yet be made. However, data collected in this study do provide a useful characterization of the current status of these Commencement Bay restoration sites, identifying features that either enhance or detract from their desirability as habitat for listed fish, and providing a basis for evaluation in the future.

General Patterns of Fish Habitat Use

The current study showed that all restoration sites are being utilized by fish, with dominant species including surfperches, juvenile salmonids, sculpins, and surf smelt. These assemblages of nearshore and estuarine fish are similar to those at other Puget Sound sites (Pearce et al. 1982, Conley 1977, Miller et al. 1977, Fresh et al. 1979, Fresh 1979, Wingert and Miller 1979, Miller et al. 1980, Mavros and Brennan 2001, Hirschi et al. 2003, Brennan and Higgins 2004, Toft et al. 2004), as well as other Pacific Northwest estuaries (Roegner et al. 2004). Pearce et al. (1982), for example, found that in the Nisqually River estuary (located about 30 miles south of Commencement Bay), the most abundant species were PSS and shiner perch, followed by starry flounder (*Platichthys stellatus*) and surf smelt. The biggest difference in species composition at the restoration sites in comparison to other Puget Sound beach seine studies was the low number of flatfish species captured at the restoration sites (Moulton et al. 1974, Miller et al. 1975, Miller et al. 1977, Fresh et al. 1979, Fresh 1979, Toft et al. 2004).

In this study, the total number of species, as well as the total number of individual fish, followed seasonal patterns, generally increasing from April to June, and then declining until the end of the sampling year, as has been found at other beach seining sites in Puget Sound (Moulton et al. 1974, Miller et al. 1975, Conley 1977, Miller et al. 1977, Fresh et al. 1979, Fresh 1979, Miller et al. 1980, Borton 1982, Monaco et al. 1990). Much of the declining trend is due to juveniles of several species maturing and moving to deeper waters later in the season to complete their life cycles, and to outmigrant juvenile salmon continuing their migration off shore and to the ocean environment (Conley 1977, Fresh 1979, Levy and Northcote 1982, Fisher and Pearcy 1990, Beamish et al. 1998, Simenstad 2000, Beck et al. 2003). High summer temperatures, which are above the range preferred by some fish species, including salmonids (Duffy 2003, Bottom et al. 2005a), could also contribute to the decline in fish abundance in August and September. This could especially have affected the Middle Waterway and Mowitch sites where summer water temperatures were the highest.

Patterns of Habitat Use by Salmon

The numbers of juvenile salmon collected were lower in 2003 than in 2002, whether by total numbers or, significantly, by catch per unit effort (CPUE). This difference is not explained by hatchery releases of juvenile Chinook, coho, or chum salmon on the Puyallup River system. Washington State and Native American hatchery releases were similar between 2002 and 2003 (3,666,760 vs. 3,260,502 Chinook; 2,231,885 vs. 2,266,351 coho; and 3,294,425 vs. 3,591,324 chum salmon in 2002 and 2003, respectively).[1] Although there are no hatchery releases of pink salmon in this river system, 2003 should have been a low abundance year for naturally spawning

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[1] K. Henderson, Washington Dept. Fish and Wildlife, Olympia, WA. Pers. commun., July 2005.

pink salmon.[2] The variation of total numbers of juvenile salmonids between years may have more to do with sampling timing and year-to-year differences in habitat utilization in the Commencement Bay system than with hatchery practices.

Regardless of collection year, salmon abundance peaked in May and June, primarily due to the influx of hatchery release fish in central Puget Sound. These patterns of juvenile salmonid abundance alongshore are consistent with those observed in other estuaries in the Pacific Northwest (Mavros and Brennan 2001, Beamer et al. 2003, Duffy 2003, Brennan et al. 2004), including the Cambell River Estuary (Levings et al. 1986), north Hood Canal (Hirischi et al. 2003), Everett Bay (Conley 1977), the Nisqually Reach (Fresh et al. 1979, Pearce et al. 1982), the Lower Columbia River Estuary (Dawley et al. 1986, Roegner et al. 2004, Bottom et al. 2005a, Kagley et al. 2006), Coos Bay, Oregon (Fisher and Pearcy 1990), and the Salmon River Estuary (Bottom et al. 2005b). Meyer et al. (1981) found a later peak catch of juvenile chum salmon (June) at sites near Yowkwala and Skookum Wulge, while peak catches of juvenile Chinook salmon were similar to those found in this study.

Overall and during periods of peak salmon abundance, hatchery salmon dominated catches at the Commencement Bay restoration sites. This is typical of beach seine surveys in southern Puget Sound (Duffy 2003, Toft et al. 2004). Unmarked fish used habitat from April to October, making up a larger percentage of the catch in April, August, September, and October, when hatchery releases were not occurring. This indicates an extended period of estuarine habitat use by wild salmon, as compared to hatchery stocks, as has been found in other Puget Sound studies (Mavros and Brennan 2001, Rice et al. 2003, Ruggerone and Jeanes 2004, Toft et al. 2004). A more extensive sampling (between Bellingham Bay in the north and Nisqually Reach in the south) of neritic waters by Rice et al. (in prep.) found clear differences in seasonal patterns of hatchery and unmarked Chinook salmon, with a much more protracted presence and later peak abundance of unmarked fish in Skagit, Padilla, and Bellingham Bays compared with all areas sampled to the south.

While hatchery Chinook and hatchery coho salmon tended to be larger than unmarked juveniles in both 2002 and 2003, they were only significantly larger in 2002. This is similar to other Puget Sound sites (Mavros and Brennan 2001, Duffy 2003) where, even though marked fish were slightly larger than unmarked fish, no significant differences were detected. Levings et al. (1986) found hatchery Chinook to be larger than unmarked fish in the Campbell River estuary in British Columbia, while Toft et al. (2004) found unmarked Chinook larger than hatchery fish, with no difference between unmarked and hatchery coho salmon. These consistencies are likely due, in part, to the incomplete marking of hatchery fish in Puget Sound. In the Skagit River system, where there are high marking rates and relatively high abundances of wild Chinook salmon, marked fish are consistently larger than unmarked fish across estuarine habitats and seasons (Beamer et al. 2007).

The rapid passage of other salmonids (coho, chum, and pink salmon) through shallow nearshore and intertidal areas, with peak abundance between May and June, which we observed in Commencement Bay, is also characteristic of other Pacific Northwest estuaries (Conley 1977,

[2] C. A. Rice, Northwest Fisheries Science Center, Seattle, WA. Pers. commun., June 2005.

Fresh et al. 1979, Fresh 1979, Levy and Northcote 1982, Shreffler et al. 1990, Cederholm, et al. 2000, Gray et al. 2002, Duffy 2003, Ruggerone and Jeanes 2004, Bottom et al. 2005a, Bottom et al 2005b). Hirschi et al. (2003) found a slightly earlier peak (April–May) of juveniles captured in north Hood Canal (Puget Sound).

It is possible that some juvenile salmonids, especially Chinook salmon, may be present in Commencement Bay for a longer period than our survey indicates. Deeper nearshore areas and channels were not sampled in the current study, although evidence from other studies indicates that these habitats are heavily used by juvenile salmon, especially Chinook, as they increase in size (Myers and Horton 1982, Dawley et al. 1986, Healy 1991, Beamish et al. 1998, Beamer et al. 2003, Duffy 2003, Beamer and Rice 2007, Rice et al. in prep.). Beamish et al. (1998) found an increase in the number of juvenile salmonids (Chinook, coho, and chum salmon) in the surface zone of deeper waters of Puget Sound from May to September, and Duffy (2003) found similar patterns in juvenile Chinook (from the Snohomish River estuary and south) from May to July. The increasing size over time in all salmon except coho in this study suggests extended estuarine rearing by most species.

Overall, the chum and pink salmon collected as part of this study increased in size over the sampling season, suggesting that they are growing during their stay at the Commencement Bay estuary sites, although it is not clear how their growth rates might compare with other estuaries, and no clear changes were observed between 2002 and 2003. Coho salmon, on the other hand, decreased in size from April to May, then their size remained fairly stable. Chinook salmon decreased in size from April to June, then increased in size again from June to September.

The larger size of Chinook and coho salmon collected in April and May is likely because, in the early season, catches were dominated by yearling juveniles passing through the system. These fish can spend up to a year in freshwater before outmigrating to saltwater (Healey 1991, Cederholm et al. 2000) along with subyearlings. Yearling Chinook in Puget Sound generally range in size from approximately 95 mm to 175 mm (Ward et al. 2002, Beamer et al. 2005) while yearling coho generally range from approximately 90 mm to more than 200 mm (Maahs 1997, Weitkamp 2001). The mean length of hatchery-release Chinook (80 mm and 77 mm in 2002 and 2003, respectively, all released were subyearlings) and coho (125 mm and 121 mm in 2002 and 2003, respectively, all released were yearlings) also indicates a contribution from prior year subyearlings still in the system.[3]

The difference in length between hatchery released coho in April of both years (125 mm and 121 mm in 2002 and 2003, respectively) and the mean length of captured coho (157 mm and 135 mm in 2002 and 2003, respectively) probably represents naturally spawned yearlings (and growth between release and capture) passing through the system. The abrupt reduction in length of juvenile Chinook and coho salmon after May reflects the short residence of yearlings in the estuary and the ongoing migration of subyearlings into the system (Healey 1991). The increase in length, at least in Chinook, after May indicates an increase in growth of the subyearlings

[3] See footnote 1.

remaining in the system. The smaller coho, captured after May, would represent subyearling river spawned coho, which show a slight increase in growth towards the end of the summer.

At individual restoration sites within Commencement Bay, juvenile salmon tended to be most abundant and largest at the Yowkwala sampling site, suggesting that this is where their performance is best. However, we did not see consistent differences in salmon size that could be clearly linked to site condition or changes from 2002 to 2003 to indicate improvements in fish performance. Juvenile salmonid growth is likely to be highly variable among estuaries and sites, depending on a number of factors (Fisher and Pearcy 1990), including temperature (Banks et al. 1971, Brett et al. 1982, Healy 1991, Mortensen and Savikko 1993, Hinrichsen 1994, Duffy 2003) and food supply, that in some cases can have confounding effects. Increased predation may also drive juvenile salmon into deeper waters. As juvenile marine fish inhabiting shallow waters reach sizes of 75–100 mm and above, they become increasingly likely to prey on juvenile salmonids (Beamer et al. 2003). The number of PSS over 75 mm, a predator of juvenile salmon (Beamer et al. 2003, Duffy 2003), increased in numbers at all sites from April to August and may have affected the distribution of juvenile salmon. These factors plus the limitations of the study design make interpretation of our fish size data problematic.

Actual growth rates of juvenile salmon within the estuary are difficult to evaluate from beach seine surveys alone. While some beach seine surveys have shown apparent growth (increases in size of juvenile salmonids captured) over the spring to summer sampling period (Fisher and Pearcy 1990, Mavros and Brennan 2001, Toft et al. 2004), others did not document consistent increases in size over time (Reimers 1973, Fresh et al. 1979, Fresh 1979, Roegner et al. 2002). To better characterize individual growth in the estuary, it is necessary to conduct a more detailed (mark-recapture, otolith analysis) and comprehensive (other gear and habitat types) sampling.

Chinook Salmon Diet Analysis

Taxonomic analysis of juvenile Chinook stomach contents revealed strong associations between prey type and salmon size class. For juveniles less than 90 mm, dominant prey types were of terrestrial-riparian origin, and included Diptera and other insect species. Marine crustaceans were also consumed at some of the sites with greater saltwater influence. In Chinook larger than 90 mm, a dramatic shift to almost exclusively marine planktonic-neritic and benthic-epibenthic prey occurred. A shift to larval fish prey at lengths greater than 90 mm was also observed, and may indicate a general shift to larger prey species with an increase in size of the juvenile Chinook salmon.

The size of juvenile salmon does have a strong influence on diet as has been noted in other systems (Cederholm et al. 2000, Duffy 2003). Similar shifts in prey type and size class (e.g., a shift to greater piscivory with larger size) have been reported for outmigrant juvenile salmon from other estuaries in Puget Sound and the Pacific Northwest (Healy 1991, Hinrichsen 1994, Beamish et al. 1998, Cederholm et al. 2000, and Duffy 2003, Bottom et al. 2005a, Bottom et al. 2005b). For example, in the Columbia River estuary, diets of smaller salmon inhabiting marsh sites with high freshwater inputs are dominated by insects similar to those found in terrestrial riparian habitats, whereas diets of older fish are dominated by amphipod species (Roegner et al. 2004, Bottom et al. 2005a).

Other studies have found a time shift in prey composition in juvenile Chinook salmon, with decreasing consumption of terrestrial riparian prey and increasing consumption of fish and marine prey from spring to fall, although they did not do size comparisons (Fresh et al. 1979, Fresh 1979, Fresh et al. 1981, Healy 1991, Shreffler et al. 1992, Hinrichsen 1994, Beamish et al. 1998, Tanner et al. 2002,). The availability of prey could also be a factor in movement of juvenile salmonids into deeper waters (Brennan and Higgins 2004). The shift to fish prey in larger juvenile Chinook, and a reduction in fish abundance at all sites from May to August may reflect the distribution to deeper channels and offshore areas that may have greater fish prey availability.

The prey organisms in stomach contents of juvenile salmon from the sites where they were observed in the highest numbers (Skookum Wulge, Tahoma Salt Marsh, and Yowkwala) were distinctive, being largely marine planktonic-neritic and marine benthic-epibenthic in origin, with marine crustaceans being the dominant prey item. This would be expected considering Skookum Wulge, Tahoma Salt Marsh, and Yowkwala are the most marine sites sampled in this study. Salmon of all size classes consumed these types of prey at these sites. In contrast, prey was predominantly of terrestrial-riparian origin at MWC, MWS, Mowitch, and Olympic View for salmon less than 90 mm. At these sites, the predominate prey items of smaller Chinook were Diptera and other insects, rather than the crustacea consumed by fish of this size class at Tahoma Salt Marsh, Yowkwala, and Skookum Wulge. This might reflect a dietary preference, but it is also possible that marine crustaceans have not yet become established at these recently reengineered sites.

The predominance of terrestrial and riparian prey at Olympic View is surprising in that this site has the least riparian influence of all the sites sampled. However, in contrast to Skookum Wulge, Tahoma Salt Marsh, and Yowkwala (also having a more open marine and less riparian habitat), it is located between the ends of the Middle Waterway and Thea Foss Waterway, so may receive terrestrial prey species on outgoing tides. It is also possible that Olympic View receives a significant input of terrestrial-riparian prey species as they drift from the Puyallup River, a major freshwater river that empties into Commencement Bay less than a kilometer north of Olympic View. The tidal and circulation patterns move freshwater flotsam to the south past Olympic View (O. P. Olson field observation) and this could have a major influence on prey species available.

As noted earlier, at all sites except Mowitch, Chinook salmon prey shifted to almost exclusively marine planktonic-neritic and benthic-epibenthic organisms for fish larger than 90 mm, and fish larvae became an important part of the diet at all sites except Mowitch. The apparent preference in smaller Chinook salmon for Diptera and other insect species at MWC, MWS, Mowitch, and Olympic View, as compared to marine crustacea at Skookum Wulge, Tahoma Salt Marsh, and Yowkwala, and the continued preference for a more terrestrial-riprarian diet even in larger fish at Mowitch may simply be a reflection of the type of prey available. Similarly, the failure of salmon greater than 90 mm to shift to fish larvae as prey at Mowitch and Tahoma Salt Marsh may be due to the absence of such prey. The freshwater influence of Hylebos Creek next to the Mowitch site may keep juvenile Pacific sandlance (*Ammodytes hexapterus*), the principal prey species for Chinook over 90 mm, from inhabiting this site. It also would provide a greater source of terrestrial insects from upstream, which correlates with the terrestrial-riparian diet of Chinook of all size classes at Mowitch. Additionally, tidal reversal may keep these prey items more concentrated at this end of the waterway (Cederholm et al. 2000).

Seasonal variations in diet do occur in other salmon species, but overall, juvenile Chinook consume a more varied diet (Fresh et al. 1979, Fresh 1979, Cordell et al. 1999, Gray et al. 2002). The lack of a consistent shift to a fish diet in larger Chinook salmon at all sites may reflect this opportunistic and varied feeding behavior of juvenile Chinook compared to other juvenile salmonids (Healy 1991, Cordell et al. 2001, Tanner et al. 2002, Duffy 2003). It should be noted there was only one juvenile Chinook greater than 90 mm captured at both MWS and Olympic View and none at MWC, making it difficult to interpret the data from those sites. A larger sample size from fish of different size classes may have revealed more consistent relationships between fish site, habitat type, and prey choice.

With the existing information, it is unclear whether the prey composition of salmon stomach contents reflects dietary preferences or prey availability at the sites where they were collected. However, diet analysis can provide useful information on how juvenile salmon utilize a site, how successful foraging is, and whether a restored site is providing prey consumed by developing salmon and how the prey base at these sites changes over time. Future sampling should concentrate on early season sampling to acquire larger sample sizes when more fish are present. The sampling of prey organisms in the environment as well as in fish stomach contents would also help to clarify the dietary preferences of juvenile Chinook salmon.

Site-specific Habitat Use

The surveys of fish assemblages thus far indicate that restoration sites fall into two distinct groups on the basis of species composition: 1) MWC, MWS, Mowitch, Olympic View, and Squally Beach sites, which were dominated by surfperches (predominantly shiner perch), dermersal fish (predominantly sculpins) and occasionally forage fish, and which were not heavily used by juvenile salmonids (5% of catch or less); and 2) Yowkwala, Skookum Wulge, and Tahoma Salt Marsh, where juvenile salmonids and forage fish accounted for a substantial proportion of the catch (26–86%). The abundance of fish was generally higher in the first group of sites, in part because of the large number of shiner perch that made up the bulk of the catch at these locations. Species density, on the other hand, tended to be higher at the second group of sites. An exception was Olympic View, which, although it was not heavily utilized by salmon, had a particularly high species density, probably due to the presence of eelgrass. The two Middle Waterway sites, MWS and MWC, as well as Squally Beach had the lowest species density of any of the sites sampled in both years, and were not heavily used by salmonids.

A number of factors can contribute to the differences in species density, fish density, and fish species assemblages at estuarine sites, including elevation, sediment type, beach slope, tidal influence, shoreline vegetation, accessibility and connectivity, and temperature and salinity regimes, all of which may play an important role in providing diverse habitat structure to attract fish species (Miller et al. 1980, Beck et al. 2001, Cornu and Sadro 2002, Rice et al. 2005). A muddy to sandy sediment predominated at the sites in the first grouping, where surfperches and demersal fish (primarily PSS) dominated the species composition. In contrast, the sites in the second grouping, where salmonids and forage fish dominated the species composition, were high energy (tidal current) sites with a gravel, cobble, and boulder bottom type.

At the Commencement Bay restoration sites, the average temperatures of surface waters, while periodically higher at MWS and Mowitch, did not appear to heavily influence species richness or fish abundance and density. Surface salinity also fluctuated between sites on a monthly basis in both sampling years and does not appear to have affected species richness (as defined by species density) or abundance. These temperature and salinity profiles were typical of nearshore estuarine environments in the Pacific Northwest (Conley 1977, Dawley et al. 1986, Levings et al. 1986, Duffy 2003). The Mowitch site, which had the lowest salinity of all the restoration sites, had intermediate species richness. This is perhaps not surprising, as Miller et al. (1980) also did not find any predictable relationship between temperature, salinity, and species richness.

Structural complexity, vegetation, and accessibility all appeared to influence species richness and species assemblages at the restoration sites. These features may have made Skookum Wulge, Yowkwala, and Tahoma Salt Marsh especially attractive to salmonids and Olympic View attractive to other juvenile marine fishes. Skookum Wulge, Yowkwala, and Tahoma Salt Marsh are accessible to salmon and also may provide greater protection from predators due to the presence of cobble and larger boulders, an important habitat benefit for juvenile salmon (Beck et al. 2001, Tanner et al. 2002, Toft et al. 2004). Olympic View did not have great structural complexity but the presence of eelgrass beds and other macroalgae may have led to the greater species richness found at this site. Moulton et al. (1974) found a higher species richness correlated to the presence of eelgrass beds. Submergent vegetation is considered an important feature of nursery areas for many juvenile marine fish (Beck et al. 2001, Beck et al. 2003).

The two Middle Waterway sites—which had the lowest species richness of any of the sites sampled in both years, and few juvenile salmonids—have relatively high mean elevation and limited structural complexity. They were also characterized by a low sloping beach and intertidal area, a muddy substrate, with very little boulder or log debris. Their location at the far end of a saltwater channel may also make them less accessible to fish than some of the other sampling sites. The entrance to these sites is often partially obstructed by large ships that may limit access to these sites and discourage their use by juvenile fish. Juvenile salmon in particular will avoid swimming under overhead structures during their migration through, and residency in, estuarine systems (Toft et al. 2004).

Gear type may also have contributed to differences in species richness at the MWC site, which was generally sampled by block net, and other sites, which were sampled by beach seine. Beach seining is an active sampling method while block netting is a passive, set and wait technique that may introduce sampling bias when comparing the two. Potential biases arising between the use of beach seines and block nets could include any of the following: the area and volume sampled, different mesh sizes allowing the potential escape of smaller fish, and the failure of the floating beach seine to sample the bottom when in a depth greater than 2.4 m.

All of these could influence species composition, number, and size of fish captured, but the assumption is that the larger mesh size in the wings of the moving seine tends to “herd” fish into the smaller mesh in the center of the net. Still, the low number of species caught at the MWC site may have been due to the use of block nets for fish capture. The block net occasionally failed to make complete contact with the bottom due to a channel formed by a small stream/tidal channel running through the site, allowing some fish to escape at low tide. A beach seine was used starting in August, with a small increase in the number of species caught, although this may have resulted from a slightly larger sampling area (less than 10%) with the use of the beach seine.

The Mowitch site had a similar sediment type to the Middle Waterway sites, but had substantially higher species richness, possibly because of the boulder and log debris placed at Mowitch during restoration efforts, thus increasing habitat diversity, cover, and possibly greater food diversity for resident fish populations. It also is located at the mouth of a freshwater creek, has deeper channels in close proximity, and is located on a much larger waterway, providing greater accessibility to a wider variety of fish species. The relatively low numbers of salmonids at the Mowitch site was somewhat surprising, although Duker et al. (1989) also found low numbers of juvenile Chinook, coho, and chum salmon during beach seine studies very near (within 50 m) the Mowitch site. It was postulated that, with the largest freshwater input of any of the sites, juvenile salmonids would utilize this area in their acclimation to marine environment (Simenstad 2000). However, there has not been significant usage of the Mowitch site by salmon species, at least in the two years of monitoring reported here.

With the Commencement Bay system being dominated by hatchery fish from the Puyallup River hatcheries, few hatchery fish may make the long journey (4.5 km) up the Hylebos Waterway to the head of this system where the Mowitch site is located. However, the few juveniles captured at Mowitch, whose size was typical of wild fish, may represent naturally spawned Chinook and chum salmon from upstream spawning in this small stream system. This could make sites such as Mowitch very important habitats to species at risk such as Chinook. It may be that increased habitat use of the Mowitch site by juvenile salmonids may not occur for several years hence, when the positive effects of restoration become better established (Michener 1997, Cederholm et al. 2000, Cordell et al. 2001).

Olympic View, with its high density of other species and presence of eelgrass beds, might be expected to support higher number of salmonids. However, other studies from Puget Sound suggest that eelgrass beds are not the preferred habitat for juvenile salmonids. Borton (1982) found very few juveniles in eelgrass beds. Wingert and Miller (1979) found the highest abundance of juvenile salmonids over cobble and shallow riprap substrates, with none occurring over eelgrass or sandy areas, while Toft et al. (2004) found that the total number of salmonids increased from cobble to sand to riprap substrates. Forsberg et al. (1976) also observed that juvenile Chinook prefer a rocky substrate. Similarly, there was no evidence from this study that the presence of eelgrass encouraged salmonid usage of the Olympic View site. Differences in prey items available at the restoration sites may also have contributed to differences in use of the restoration sites by salmon and other fish, especially at sites such as Mowitch and Olympic View that had other favorable habitat characteristics.

Temporal Trends in Habitat Use

With only two years of data, it is premature to attempt to evaluate long-term trends in habitat use at the restoration sites. While site-specific differences occurred, species density, fish density, and species assemblages were similar in 2002 and 2003, with no sign of increased use of sites by salmonids or other fish species. The variability in the number of salmonids captured at the Yowkwala and Olympic View sites during 2002 and 2003 may show the rapid transitory migration of juvenile salmon, and sampling of the site may or may not occur concurrently with the rapid passage of some salmonids. Pink salmon in particular pass through estuarine systems rapidly (Levy and Northcote 1982, Cederholm et al. 2000), with coho, then chum and Chinook salmon residing the longest before outward migration (Levy and Northcote 1982, Shreffler et al. 1990, Cederholm et al. 2000). The similarity in the size of hatchery releases between the two years did not appear to influence the number of salmonids captured in the present study.

The causes of these observed differences in species and fish densities remain open to conjecture, but probably reflect measurement error and normal year-to-year variation in species composition and fish abundance. Similar year-to-year variations have been found at other Puget Sound nearshore sites (Conley 1977, Miller et al. 1980, Tanner et al. 2002). Assessing the results of restoration efforts upon juvenile salmon is a complicated task given the intermittent usage and rapid passage they typically exhibit (Tanner et al. 2002).

Our efforts to evaluate the success of restoration activities in Commencement Bay are also hampered by the lack of prerestoration baseline data for these restoration sites. A major problem with restoration projects of this type is that little or no consideration is given to evaluating conditions before the restoration efforts begin (Cordell et al 2001, Roni et al. 2002, Rice et al. 2005). It would also be beneficial to have reference sites with which to make comparisons, although it can be very difficult to find equivalent reference areas within the same embayment (Michener 1997, Cordell et al. 2001, Weinstein et al. 2001, Cornu and Sadro 2002). The difficulties in finding an appropriate reference site for this study cannot be overemphasized. Since almost every restoration site differs in some capacity, finding matching reference sites would nearly double the sampling effort, and moreover, finding such sites in an environment as altered as Commencement Bay is basically impossible. Consequently, we relied on historical data from Puget Sound sites for comparisons of biological and chemical analyses.

Additional years of sampling will be needed to determine the true effects of restoration efforts in Commencement Bay. The amount of variability in the year-to-year data at the restoration sites emphasizes the need for much longer periods of monitoring. Because of the complexity of estuarine environments and the unpredictable recovery rates of restored habitat, it has been suggested that monitoring duration should be at least five years to decades (Michener 1997, Weinstein et al. 2001, Rice et al. 2005).

Contaminants

Sediments

As part of the Commencement Bay restoration and remediation process, the Commencement Bay trustees established sediment cleanup goals (Table 10) for active restoration sites (USDI et al. 1997; see http://www.darp.noaa.gov/pacific/cbay/sediment.html). For PAHs, PCBs, and tributyltin, these objectives were based on NOAA sediment guidelines developed for the protection of salmon and benthic fish (Johnson et al. 2002, Meador et al. 2002). For other compounds, the trustees recommended the lower of either the USEPA's sediment quality objectives (SQOs; USEPA 1997, USEPA 1989), or Washington State's Sediment Management Standards and Marine Sediment Quality Standards (MSQSs) (WAC

Table 10. Sediment cleanup goals for active restoration sites in Commencement Bay for contaminant classes measured in sediments in the current study, and comparison with other sediment quality objectives and guidelines. Cleanup goals for PAHs and PCBs are based on NOAA Fisheries sediment guidelines for fish injury (Meador et al. 2002, Johnson et al. 2002). Other values are based on Washington State Sediment Quality Standards and AET values associated with 5% service loss in the Hylebos Waterway Habitat Equivalency Analysis (Wolotira 2002).

Contaminant class	**Trustee cleanup goal (ng/g dry wt)**	**NOAA threshold (ng/g dry wt)[a]**	**Washington State SQG (ng/g dry wt; assumes TOC of 1%)[b]**	**EPA SQO (ng/g dry wt)[c]**	**Washington DOE[d] apparent effects threshold (ng/g dry wt)[e]**
Total PAHs	2,000	2,000	ND[f]	ND	ND
HAHs[g]			9,600	17,000	7,900
LAHs[h]			3,600	5,200	1,200
Total PCBs	200	200	120	300	130
ΣDDTs	12	ND	ND	ND	11
ppDDD	16	ND	ND	16	16
ppDDE	9	ND	ND	9	9
ppDDT	12	ND	ND	34	12
Chlordane	ND	ND	ND	ND	2.8
Lindane	ND	ND	ND	ND	4.8
Hexachlorobenzene	22	ND	ND	22	6
Hexachlorobutadiene	11	ND	ND	11	1.3
Heptachlor	ND	ND	ND	ND	0.3

[a]Johnson et al. 2002, Meador et al. 2002.
[b]WAC 1730-204 (Washington State sediment quality guidelines [SQGs]).
[c]USEPA 1989 (EPA sediment quality objectives [SQOs]), USEPA 1997.
[d]DOE = Department of Ecology.
[e]Barrick et al. 1988.
[f]ND = not detected.
[g]HAH = high molecular weight aromatic hydrocarbons.
[h]LAH = low molecular weight aromatic hydrocarbons.

1995). The latter are based on Apparent Effects Thresholds (AETs) derived from a set of seven invertebrate bioassays (Barrick et al. 1988, Gries and Waldow 1996). Using the same information, the trustees established sediment guidelines for assessment of injury to natural resources in the Hylebos Waterway (Wolotira 2002).

One of the important findings of this study was the presence of chemical contaminants in sediments from the restoration sites, in some cases at concentrations that exceeded restoration site cleanup goals or resource injury guidelines. PAHs were especially widespread. The concentrations of total PAHs at all sites were above the NOAA Fisheries threshold levels for effects on benthic fish (Johnson et al. 2002) and the sediment cleanup goal (Table 10) adopted by the trustees (USDI et al. 1997, CBNRT 2002) of 2,000 ppb dry wt total PAHs. Total PAH concentrations at three of these sites (Mowitch, Skookum Wulge, and Olympic View) were between 3,000 and 4,000 ng/g dry wt, which is comparable to levels observed at Commencement Bay sites used as reference areas for the Hylebos Waterway damage assessment sediment evaluation studies (EVS 1996), and substantially cleaner than sediments in the heavily industrialized sections of the Hylebos Waterway (EVS 1996). However, at the remaining three sites (MWS, MWC, and Squally Beach), total PAH concentrations were in the 8,000–15,000 ppb range, comparable to concentrations measured at some of the more contaminated sites in the Hylebos Waterway as part of the damage assessment study (EVS 1996, Collier et al. 1998a). The sediments are below USEPA SQOs of 5,200 ng/g dry wt for total LMWAHs and 17,000 ng/g dry wt for HMWAHs, but these guidelines were developed for the protection of benthic invertebrates and may not be protective against sublethal effects in fish (Johnson et al. 2002).

Concentrations of PCBs were somewhat lower in restoration site sediments in comparison with established sediment quality guidelines and objectives. At all sites where sediments were analyzed, levels of PCBs were below the 200 ppb dry wt guideline established by the trustees and similar to levels at Commencement Bay reference sites sampled during the Hylebos damage assessment study (60–70 ppb dry wt). Squally Beach was the only site that approached the 200 ppb dry wt level. In comparison, PCB concentrations measured in the Hylebos Waterway during damage assessment were 400–600 ppb dry wt at most sites; only one site was below 200 ppb dry wt.

Similarly, concentrations of DDTs in sediments were low to moderate (2–6 ng/g dry wt) at most of the restoration sites (i.e., MWC, MWS, Skookum Wulge, and Olympic View). These concentrations are similar to or slightly higher than those measured at the Commencement Bay reference sites during damage assessment (1–3 ng/g dry wt), and below USEPA's SQOs and Washington State's MSQSs, as well as the screening level guidelines used by the Army Corps of Engineers in its Dredged Material Management Framework (6.9 ng/g dry wt; USACE 1998). DDT levels at Squally Beach were somewhat higher (13 ng/g dry wt), approaching or exceeding most screening guidelines and within the higher range of DDT concentrations recorded in the Hylebos Waterway during damage assessment (EVS 1996). At Mowitch, DDT concentrations (120 ng/g dry wt) were an order of magnitude higher than concentrations measured at any of the restoration sites or at any sites sampled in the Hylebos Waterway or Commencement Bay during damage assessment (EVS 1996). Mowitch DDT concentrations exceeded any of the sediment quality guidelines or objectives developed by USEPA or the trustees.

HCB levels were above the lowest reported AET in sediments from one site, Squally Beach, but were below EPA's SQOs and Washington State's SQGs, and the value associated with 5% injury loss by the trustees. Although sediment cleanup goals or injury thresholds were not established for chlordane, heptachlor, or lindane; sediment concentrations above the lowest observed AETs from Washington State DOE's sediment management database (Barrick et al. 1988, Gries and Waldow 1996, Wolotira 2002) were observed at the MWC, MWS, and Squally Beach sites.

With the exception of the guidelines for PAHs and PCBs recommended by the trustees, which are based on effects in benthic fish and juvenile salmon (Johnson et al. 2002, Meador et al. 2002) or humans (USEPA 1989, USEPA 1997), the sediment quality guidelines used to develop sediment cleanup goals for Commencement Bay were developed primarily for the protection of benthic invertebrates. As such, they may not necessarily be appropriate for juvenile salmon or marine fish species utilizing the restoration sites. However, exceedances do indicate potential risks to the prey base of salmon and other fish species, which may have indirect effects on their health and survival.

Sources of contaminants in restoration site sediments are not known. They were probably present at some of these sites before restoration activities were initiated. Sediments from sites in the vicinity of Skoolum Wulge, Mowitch, and Squally Beach were sampled as part of the Hylebos damage assessment studies (EVS 1996, Collier et al. 1998) and showed contaminant concentrations generally comparable to those measured in the present study. Contaminants may also have been present in fill material placed at some of the sites. This is especially likely for PAHs, which were present at concentrations that the trustees deemed harmful to aquatic life, but which were still within screening guidelines for open water disposal established by the Puget Sound Dredge Disposal Analysis Program (Barrick et al. 1988, Gries and Waldow 1996).

The very high DDT concentrations in sediment from the Mowitch site could represent localized contamination from a spill or from disposal of toxic waste, as DDTs do not appear to be present at similar concentrations at other sites in Commencement Bay. Sediments from a site near Mowitch were analyzed during damage assessment, but DDTs concentrations were substantially lower (≈35 ng/g dry wt). It is also possible that there may have been some recontamination of the restoration sites from nonpoint source pollution, particularly for the PAHs, which are present in urban runoff. More extensive sediment sampling, including analysis of sediment cores, would help to determine how widespread these contaminants are, and whether they are due to recent contamination.

Juvenile Salmon

Species differences in salmonid contaminant uptake

Patterns of contaminant uptake by different juvenile salmon species were consistent with those observed in other studies (Stein et al. 1995, Stehr et al. 2000, Johnson et al. 2007a). Chinook and chum salmon generally had the highest contaminant body burdens of both PCBs and DDTs, while concentrations in coho and pink salmon were lower. These differences are likely related to differences in life history and habitat use, as well as diet and metabolism. Assuming that the estuary is an important source of contaminants for outmigrant salmonids, higher body burdens are consistent with the more prolonged period of estuarine residence in Chinook and chum salmon (Reimers 1973, Levy and Northcote 1982, Simenstad et al. 1982, Healey 1991, Healy and Prince 1995).

Increased bioaccumulation in Chinook and chum salmon may also indicate they are feeding at a higher trophic level than coho or pink salmon. This would be consistent with dietary studies showing that, while there is considerable overlap in the diet of juvenile coho and Chinook salmon, coho tend to consume a lower proportion of juvenile and larval fish and a higher proportion of invertebrates than Chinook (Brodeur and Pearcy 1990, Schabetsberger et al. 2003). As noted earlier, hatchery rearing may also influence contaminant body burdens in some of these stocks.

The ratio of DDTs to PCBs can be useful in identifying patterns of contaminant uptake or contaminant sources in different groups of organisms. In prerelease juvenile Chinook salmon collected from Pacific Northwest hatcheries, for example, $\sum$DDT/$\sum$PCB ratios ranged 0.7–0.8 (Johnson et al. 2007a). Of the four species sampled in Commencement Bay, coho salmon had the ∑DDT/∑PCB ratio most similar to the hatchery populations (0.6), which is consistent with their high lipid content and the likelihood that they were part of a recent hatchery release. In the other species, DDT/PCB ratios were much lower, an average of 0.1–0.2, suggesting substantial PCB uptake from the environment following hatchery release. The exception was Chinook salmon from Mowitch, which had a DDT/PCB ratio of 1.5, reflecting the very high concentrations of DDTs found in sediments at this site.

Whole body lipid concentrations were similar for all salmon species except coho. The average lipid concentrations (≈1.5%) in Chinook, pink, and chum salmon sampled during this study were similar to lipid levels measured in predominantly wild Chinook and coho salmon from other Pacific Northwest estuaries (1–2%, Johnson et al. 2007a), and much lower than levels measured in juvenile Chinook salmon sampled directly from Pacific Northwest hatcheries (6–9% range) (Johnson et al. 2007a). The low lipid levels in Commencement Bay Chinook salmon of known hatchery origin suggest the fish had spent some time rearing in the estuary before they were captured and during this period their lipid levels declined. Rapid drops in lipid content have been documented in other hatchery stocks following hatchery release. The lipid content (≈4%) of coho salmon collected as part of this study was higher than that of other salmonids sampled, as well as higher than levels measured in predominantly wild juvenile coho from other Pacific Northwest estuaries (Johnson et al. 2007a). This may be because the sampled coho were part of a large, recent hatchery release, collected mainly from the unusually abundant catch taken in the June 2002 sampling.

Contaminant concentrations in whole bodies and prey of Chinook salmon

Because they are widely distributed in Commencement Bay and other Pacific Northwest estuaries, and have a more extended period of estuarine residence than some other species, our salmon contaminant monitoring focused mainly on juvenile Chinook.

Measurable concentrations of PAHs, DDTs, PCBs, and other OC pesticides were found in stomach contents of juvenile Chinook salmon from all sites where stomach contents samples were analyzed and collected (Yowkwala, Olympic View, MWS, and Skookum Wulge), indicating that diet is one source of exposure for juvenile salmon. In comparison with concentrations of PCBs and DDTs that have been measured in stomach contents of juvenile Chinook salmon from other areas in the Pacific Northwest (Figure 41 and Figure 42), concentrations of these contaminants in juvenile salmon from the MWS site were among the highest levels observed. Levels of PCBs and DDTs were comparable to concentrations in juvenile Chinook from the Duwamish Waterway and the Lower Columbia River (Stein et al. 1995, Johnson et al. 2007a), and higher than concentrations measured in juvenile salmon from Hylebos Waterway during the NRDA fish injury studies (Stehr et al. 2000).

In salmon from Olympic View, PCB and DDT concentrations in stomach contents were similar to those measured in juvenile Chinook from the Hylebos Waterway, while in fish from the Yowkwala and Skookum Wulge sites, stomach contents concentrations of PCBs and DDTs were lower than those previously measured in Hylebos fish and comparable to levels observed in juvenile Chinook salmon from other minimally to moderately developed estuaries in the Pacific Northwest (Johnson et al. 2007a). Interestingly, the Chinook stomach content samples from Skookum Wulge and Yowkwala contained the parent DDT isomers (*o,p'*- and *p,p'*-DDT) that

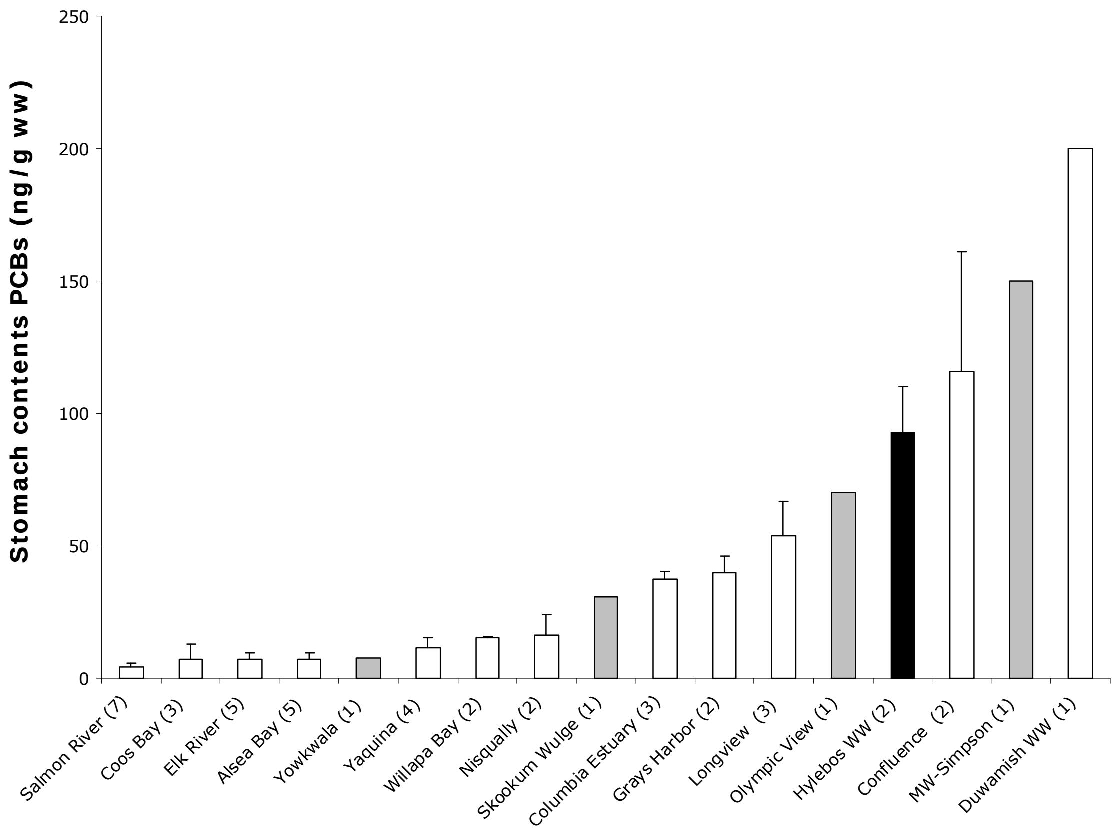


Figure 41. Concentrations (ng/g wet wt, ± SE) of ∑PCBs in stomach contents of juvenile Chinook salmon from Commencement Bay restoration sites (indicated in gray) compared to concentrations in juvenile Chinook salmon from the Hylebos Waterway (indicated in black) (Stehr et al. 2000) and other Pacific Northwest estuaries (Johnson et al. 2007a). Numbers in parentheses indicate number of composite samples analyzed. Each composite sample contained stomach contents from 10 to 15 fish.

were not present in stomach contents of juvenile salmon from any of the other restoration sites. The presence of these parent compounds suggests DDT input from a relatively recent source (ATSDR 2002a).

In contrast to OC compounds, total PAH concentrations in stomach contents were highest in fish from Yowkwala and lowest in fish from the MWS site, bracketing concentrations measured in juvenile Chinook salmon in the Hylebos during the fish injury studies (Stehr et al. 2000, Figure 43). While substantially higher than total PAH concentrations measured in juvenile Chinook salmon from nonurban estuaries, they did not approach concentrations measured in salmon from the Duwamish Waterway or the Lower Columbia River (Johnson et al. 2007a, Figure 43).

Contaminant concentrations in salmon stomach contents did not necessarily reflect sediment contaminant concentrations at the sites where the fish were collected. For example, although fish from the MWS site had the highest concentrations of DDTs and PCBs in stomach

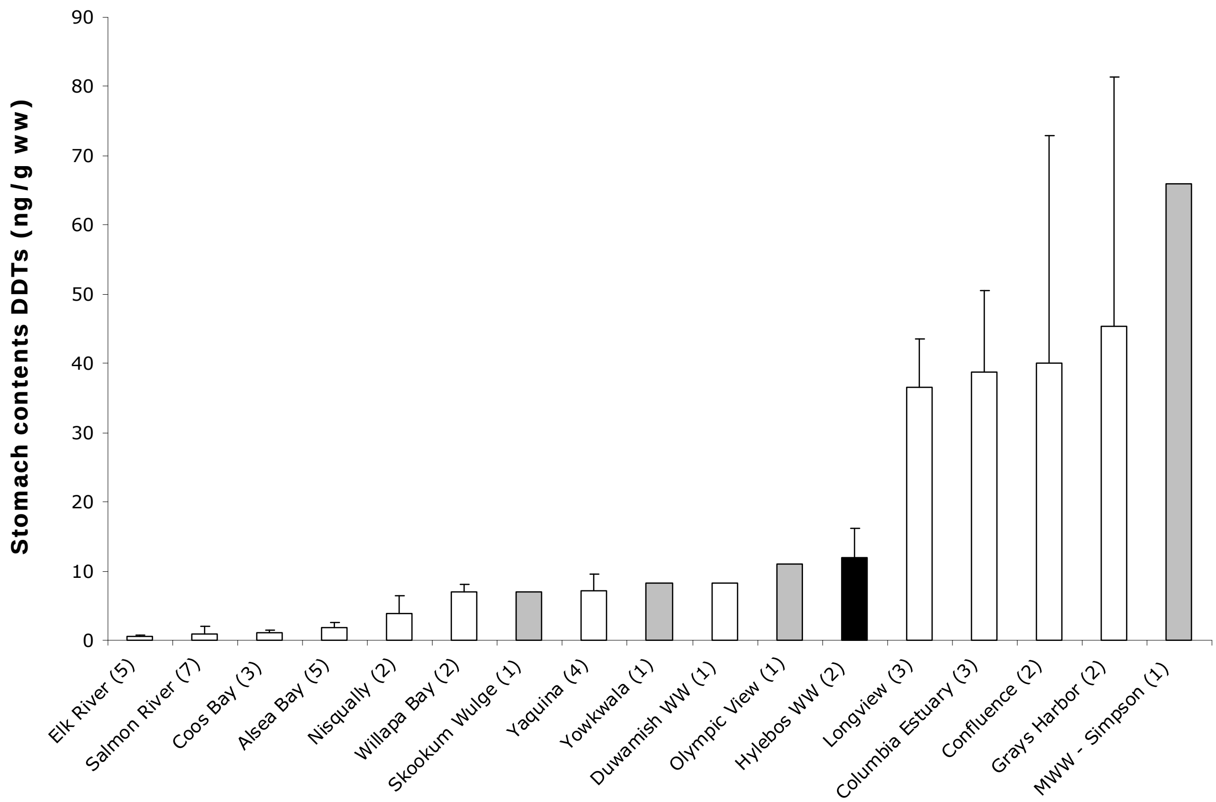


Figure 42. Concentrations (ng/g wet wt, ± SE) of ∑DDTs in stomach contents of juvenile Chinook salmon from Commencement Bay restoration sites (indicated in gray) compared to concentrations in juvenile Chinook salmon from the Hylebos Waterway (indicated in black) (Stehr et al. 2000) and other Pacific Northwest estuaries (Johnson et al. 2007a, Johnson et al. 2007b). Numbers in parentheses indicate number of composite samples analyzed. Each composite sample contained stomach contents from 10 to 15 fish.

contents, PCB and DDT concentrations in sediments from this site were similar to those at Olympic View. Conversely, although sediment PAH concentrations were higher at the MWS site than at Skookum Wulge or Olympic View, stomach content PAH concentrations were lower. Furthermore, contaminant profiles in stomach contents often differed from those in sediments. For example, retene, a LMWAH wood product associated with pulp mills or logging operations, contributed up to 15% of ∑LMWAH determined in restoration sites sediments, but less than 2% of ∑LMWAHs in stomach contents of fish from these sites. Also, proportions of ∑LMWAHs were generally higher in stomach contents than in sediments.

These differences reflect, in part, the differences in bioavailability of LMWAHs and HMWAHs present in sediment, as well as differential uptake and metabolism of PAHs by the prey organisms upon which salmon are feeding (Meador et al. 1995). However, a more important factor may be the source of the prey in stomach contents samples, that is, whether they were of marine or terrestrial and riparian origin. Additionally, since juvenile salmonids are highly mobile, the stomach contents may include some prey consumed at nearby sites and not be totally reflective of the sediment contaminant levels at specific sites.

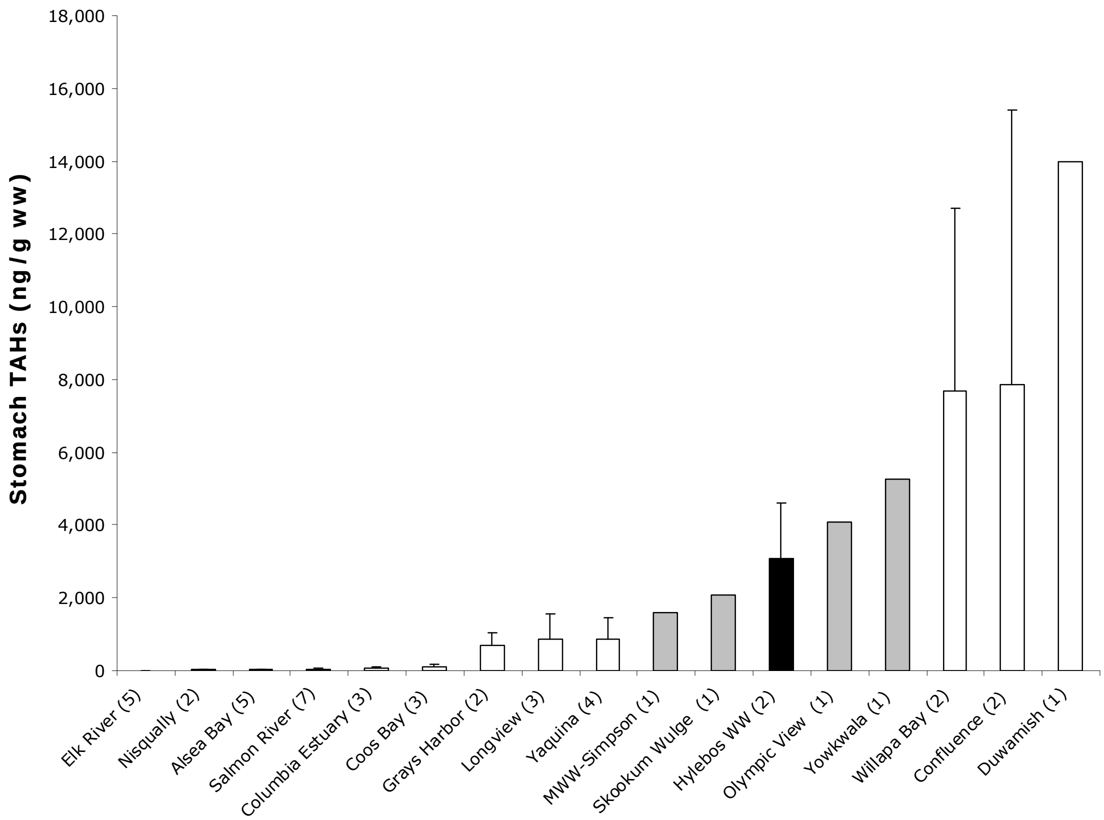


Figure 43. Concentrations (ng/g wet wt, ± SE) of total AHs (TAHs) in stomach contents of juvenile Chinook salmon from Commencement Bay restoration sites (indicated in gray) compared to concentrations in juvenile Chinook salmon from the Hylebos Waterway (indicated in black) (Stehr et al. 2000) and other Pacific Northwest estuaries (Johnson et al. 2007). Numbers in parentheses indicate number of composite samples analyzed. Each composite sample contained stomach contents from 10 to 15 fish.

The MWS, Olympic View, and Mowitch restoration sites all have substantial riparian influence, and taxonomic analyses of stomach contents in juvenile Chinook salmon from these sites showed a high proportion of terrestrial and riparian prey. Thus the concentrations of contaminants in salmon stomach contents from these sites may be more reflective of contaminants present in riparian or terrestrial environments than of contaminants in subtidal sediments. This may explain why retene, which was prominent in sediments at the Mowitch site, was found only at low concentrations in stomach contents of salmon from this site. It also suggests that at the MWS site, the riparian, upland environment may be an important source of PCBs and DDTs. In fish from Skookum Wulge and Yowkwala, on the other hand, prey in stomach contents were primarily of marine origin, so would be expected to more accurately reflect sediment contaminant concentrations at the restoration sites. This seemed to be true for the Skookum Wulge site, where contaminant concentrations in both sediments and stomach contents were among the lowest levels measured; the relationship cannot be evaluated for Yowkwala, as no sediment contaminant data are available for this site.

Concentrations of PCBs, DDTs, and OC pesticides in whole bodies of juvenile Chinook salmon confirm the uptake and bioaccumulation of these compounds in fish utilizing the restoration sites. Concentrations of PCBs were especially high in Chinook from the Yowkwala and Tahoma Salt Marsh sites, comparable to concentrations reported in juvenile Chinook salmon from the Duwamish Waterway (Figure 44 upper panel) (Stein et al. 1995, Meador et al. 2002, Johnson et al. 2007a). Whole body PCB concentrations in chum salmon from the Tahoma Salt Marsh were equally high. Whole body concentrations of DDTs (Figure 44 lower panel) were especially high in juvenile Chinook from Mowitch, comparable to concentrations measured in juvenile Chinook from the Lower Columbia Estuary (Johnson et al. 2007a). The DDT profile of Mowitch salmon was also distinctive for the presence of the parent DDT isomer, p,p'-DDT, which was also present in sediments from the Mowitch site. In juvenile salmon from Yowkwala and the Tahoma Salt Marsh, DDT body burdens were similar to those measured in juvenile Chinook from undeveloped to moderately developed estuaries in the Pacific Northwest (Johnson et al. 2007b) and contained little of parent DDT isomers.

The relatively high concentrations of PCBs in whole bodies of juvenile salmon from Yowkwala and Tahoma Salt Marsh, where PCB concentrations in sediment and stomach contents were relatively low, might be due to uptake of contaminants during hatchery rearing, as the majority of the fish analyzed were hatchery marked. Several studies have documented the presence of PCBs and similar contaminants in diets of hatchery and farm-raised salmon (Karl et al. 2003, Parkins 2003, Hites et al. 2004, Johnson et al. 2007a). Even relatively low levels of these substances in feed can result in high body burdens in juvenile hatchery-reared salmon, because of the high body fat content these fish typically develop by the time they are released from the hatchery (Johnson et al. 2007a, LCREP 2007). Whole body lipid content is a controlling factor in the expressed toxicity of bioaccumulative compounds such as PCBs (Lassiter and Hallam 1990, van Wezel et al. 1995, Meador et al. 2002, Elskus et al. 2005), so mobilization of these contaminants as released hatchery fish use their lipid stores puts them at increased risk for toxic effects. Moreover, if contaminant body burdens are already moderate to high when fish leave the hatchery, they have an increased risk of reaching toxic threshold exposure concentrations during estuarine residence that could significantly reduce their likelihood of survival.

Concentrations of PAH metabolites in bile were available only for juvenile Chinook from Yowkwala and Skookum Wulge. Levels of metabolites of both high and lower molecular weight PAHs in fish from these sites were moderate in comparison with PAH metabolites levels measured in Chinook salmon from other Puget Sound sites (Figure 45). While higher than levels typical of juvenile Chinook from nonurban estuaries, they were below levels measured in juvenile Chinook from the Hylebos Waterway during the fish injury study in 1994 or in salmon from other contaminated sites in the Pacific Northwest, such as the Duwamish Waterway (Stehr et al. 2000, Johnson et al. 2007a) (Figure 46). This finding would be expected for juvenile salmon from Skookum Wulge, in view of PAH concentrations in salmon stomach contents and sediments from this site, but is somewhat surprising for the Yowkwala site, as stomach contents PAHs in fish from these site were higher than those measured in the Hylebos salmon. However, because only one composite stomach contents sample was analyzed from the Yowkwala site, it is difficult to know whether that is indicative of PAH concentrations in stomach contents of salmon from this site.

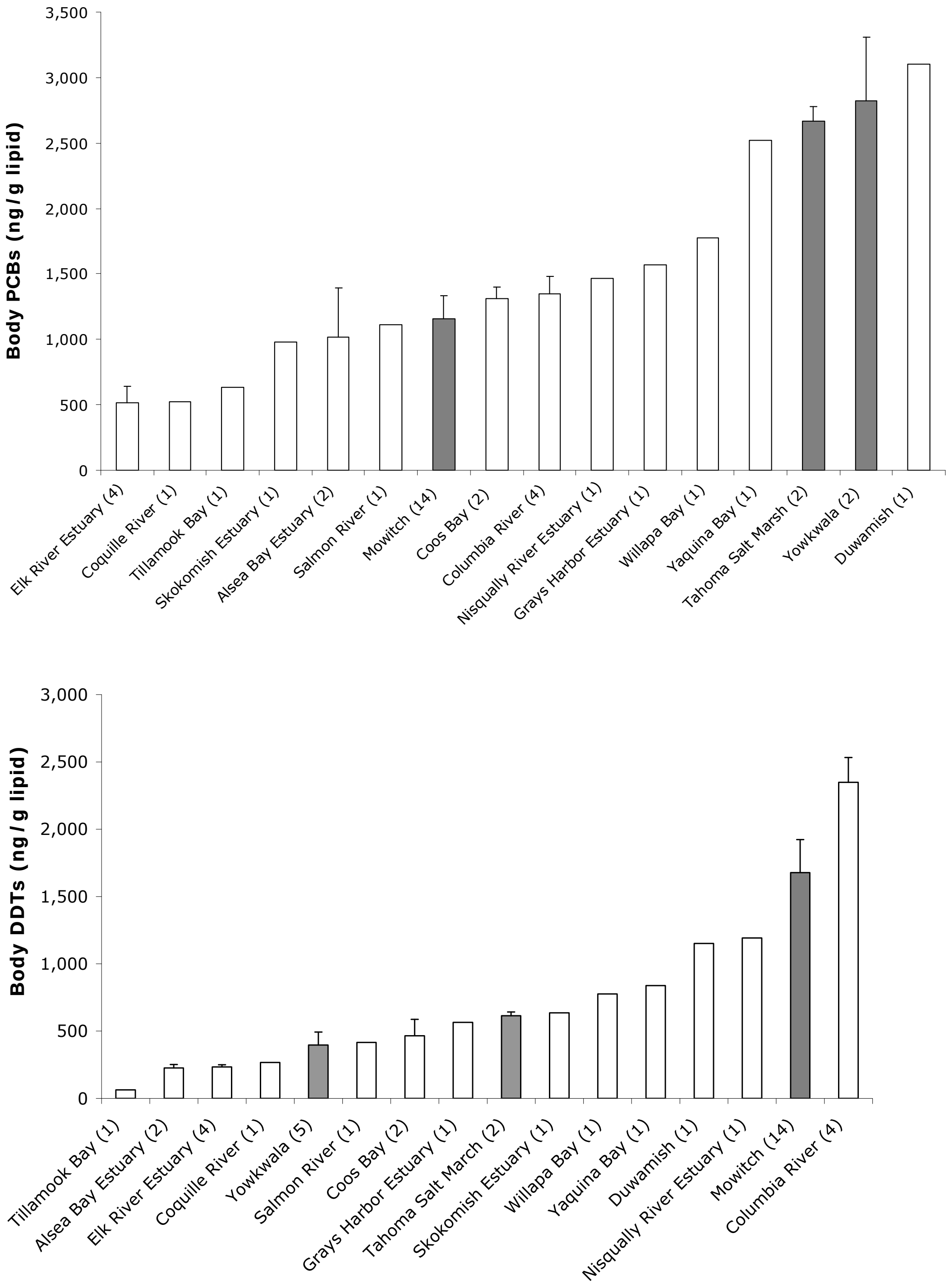
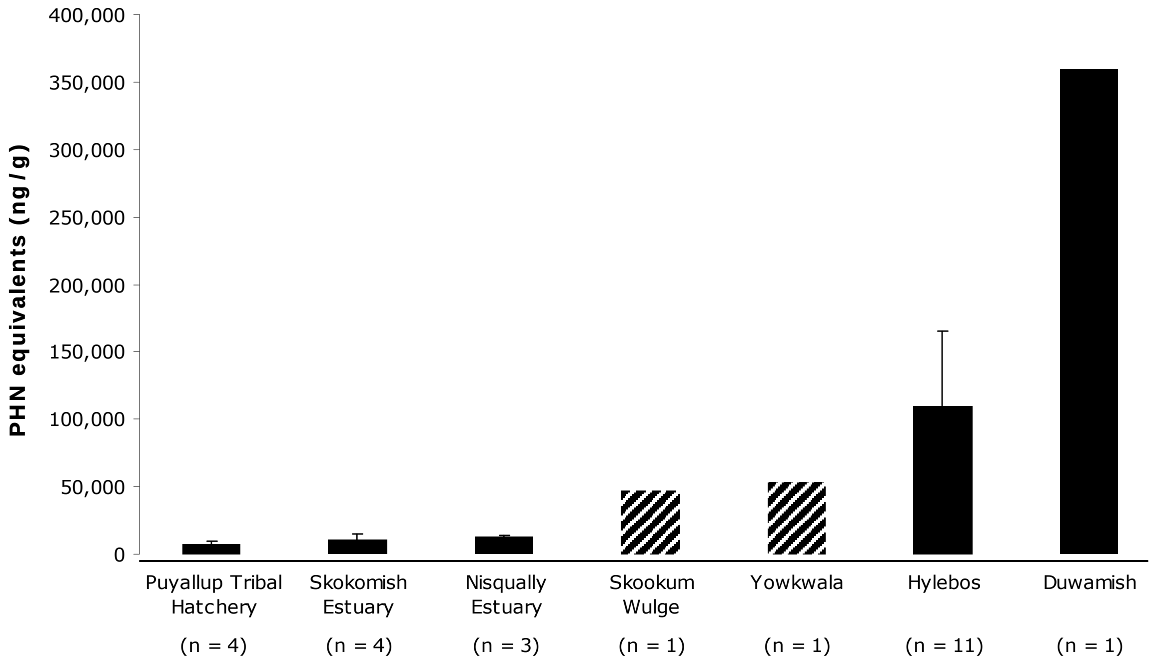


Figure 44. Concentrations (ng/g lipid, ± SE) of ∑PCBs (top panel) and ∑DDTs (bottom panel) in whole bodies of juvenile Chinook salmon from Commencement Bay restoration sites (in gray) compared to concentrations in juvenile Chinook salmon from other Pacific Northwest estuaries (Johnson et al. 2007a). Numbers in parentheses indicate number of composite samples analyzed. Each composite sample contained stomach contents from 10 to 15 fish.



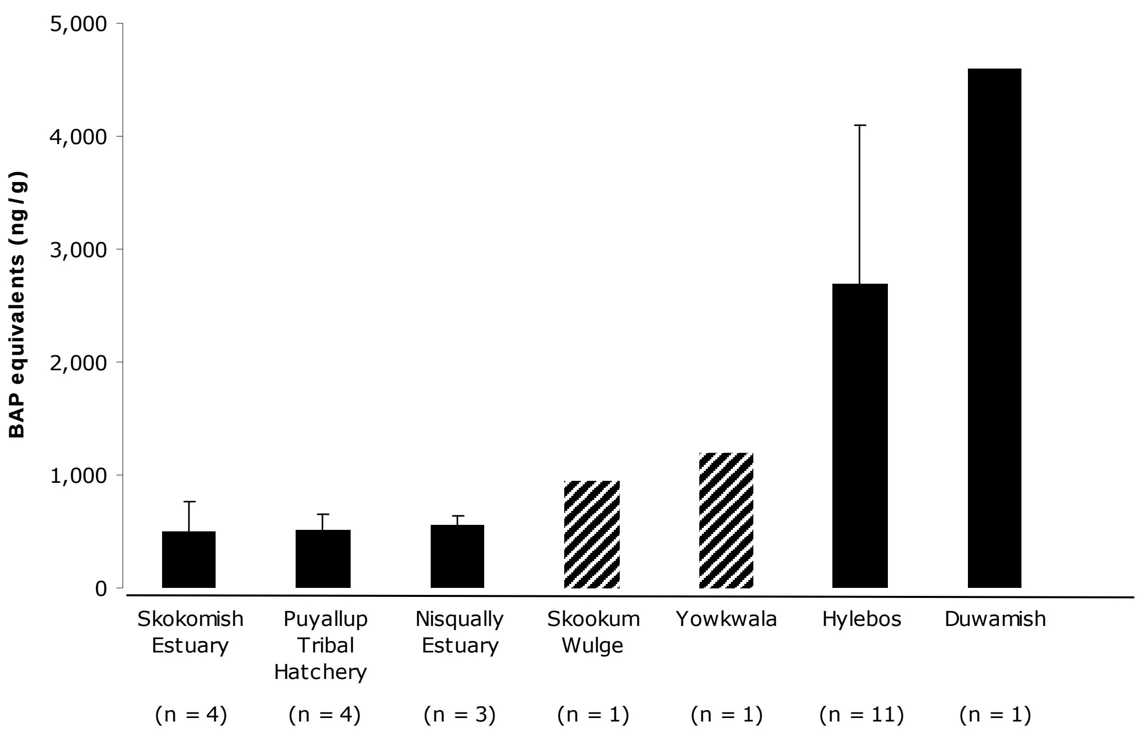


Figure 45. The mean concentrations (± SE) of PAH bile metabolites (PHN top panel, BaP bottom panel) in juvenile Chinook salmon from Commencement Bay (diagonal stripes) in comparison to various Puget Sound sites. (Columns in diagonal lines = current study, columns in black = other studies, n = number of samples included in the analysis.)

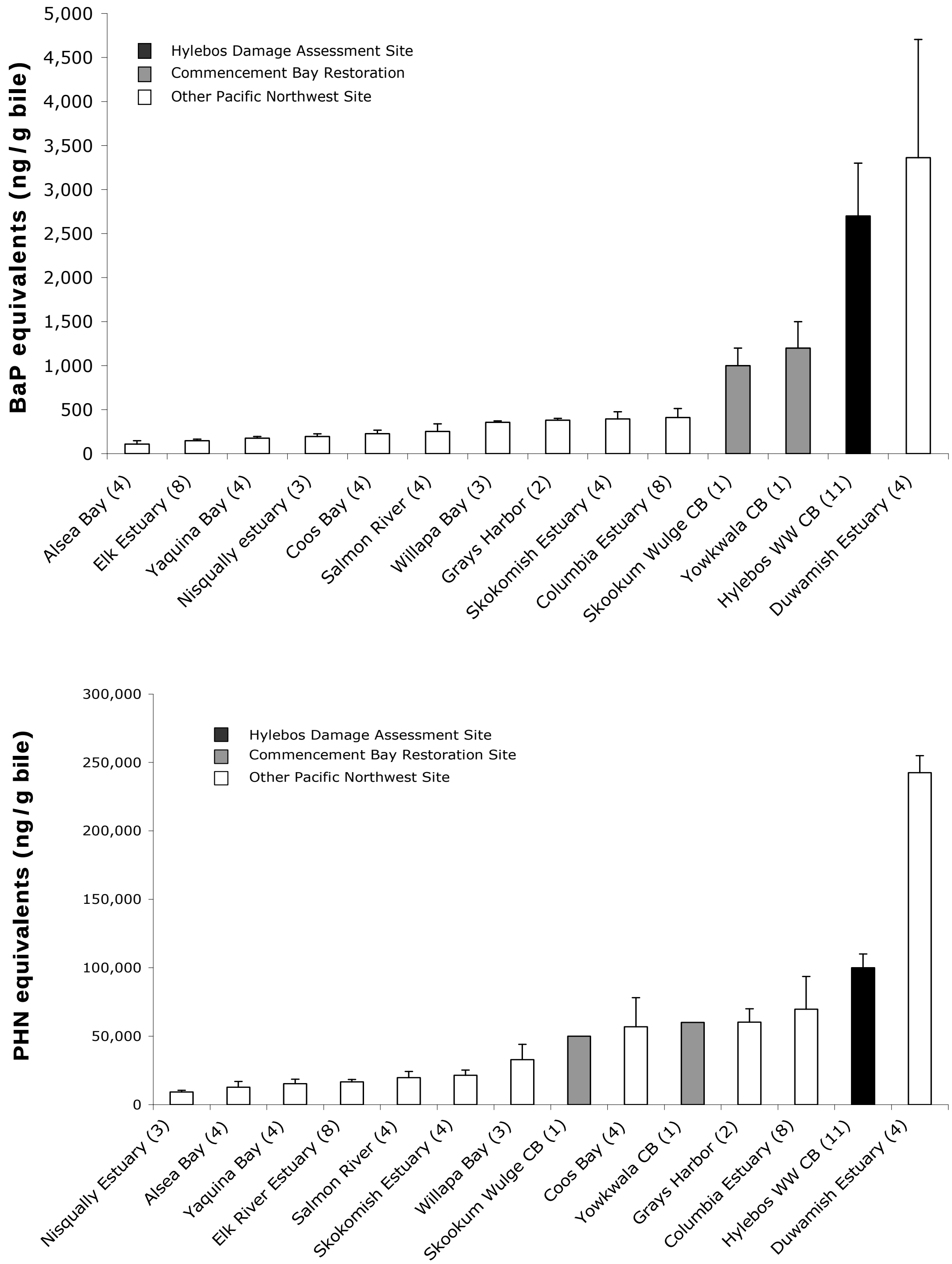


Figure 46. Mean concentrations (± SE) of metabolites of PAHs measured at BaP (top panel) and PHN (bottom panel) wavelengths in bile of juvenile Chinook salmon from Commencement Bay restoration sites (gray and black) compared to concentrations in juvenile Chinook salmon from other Pacific Northwest estuaries. Numbers in parentheses indicate number of composite samples analyzed. Each composite sample contained bile from 10 to 15 fish.

Pacific Staghorn Sculpin

Contaminant concentrations were measured in PSS to provide information on contaminant accumulation in a resident species. As expected, contaminant concentrations in PSS tended to be higher than in salmon. For example, the average whole body PCB levels in PSS ranged from 2,500 to 10,000 ng/g lipid, and whole body DDT concentrations ranged from 500 to 7,000 ng/g lipid, whereas the highest levels in salmon were in the 1,500–3,000 ng/g lipid range. The higher levels of contaminants in PSS as compared to Chinook salmon are likely related to differences in residence time at the sampling sites and differences in prey items between the two genera; fish age is probably not a major factor, as both the salmon and the PSS sampled for chemical analyses were subyearlings to yearlings (Appendix G). PSS feed primarily on organisms of benthic and marine origin (Wolf et al. 1983, Armstrong et al. 1995), whereas juvenile salmon may spend more time in the water column and consume a mixture of benthic and pelagic organisms (Fresh et al. 1979, Fresh 1979, Healy 1991, Cordell et al. 1999, Cordell et al. 2001, Gray et al. 2002, Tanner et al. 2002). Also, as noted above, terrestrial and riparian prey made up a substantial component of the diets of salmon from several of the restoration sites.

Intersite trends in contaminant levels in PSS were similar to those in salmon for DDTs, with the highest concentrations in fish from Mowitch. In the case of PCBs, however, the pattern was somewhat different; the highest PCB concentrations in PSS were in fish from Olympic View, Mowitch, MWC, and MWS, while highest concentrations were found in salmon from Skookum Wulge and Yowkwala. Generally, concentrations of PCBs and DDTs in PSS were more closely correlated to sediment contaminant concentrations than PCB and DDT concentrations in salmon. For example, whole body PCB concentrations were in the 10,000–15,000 ng/g lipid wt range in PSS from Olympic View, Mowitch, and the two Middle Waterway sites, all of which have sediment PCB concentrations in the 75–100 ng/g dry wt range, but only about 5,000 ng/g lipid in fish from Skookum Wulge, where the sediment PCB concentration was about 25 ng/g dry wt.

The correlation of contaminant body burdens in PSS with sediment contaminant concentrations at the restoration sites confirms the bioavailability of contaminants in sediments at the restoration sites, and suggests that although the sediment samples were very limited, they are reasonably representative of contaminant concentrations at the restoration sites. The high body burdens of contaminants in PSS from several of the sites are also indicative of the potential for adverse health effects on resident marine fish species using these sites.

Like juvenile salmon, PSS showed exposure to PAHs based on the presence of PAH metabolites in bile. Bile metabolite levels were similar in PSS from all restoration sites and were similar to levels observed in juvenile salmon from Yowkwala and Skookum Wulge. As in salmon, bile metabolite levels in Commencement Bay PSS were lower than those measured in English sole and juvenile salmon collected from the Hylebos Waterway during the fish injury study (Collier et al. 1998a, Collier et al. 1998b, Johnson et al. 1999, Stehr et al. 2000). However, the mean levels of PAH metabolites in Commencement Bay PSS were higher than the average concentrations measured previously, typically found in marine fish from nonurban sites (Brown et al. 1998).

Potential Contaminant Impacts on Fish Health

Whole body PCB concentrations in juvenile Chinook and chum salmon from the Tahoma Salt Marsh and juvenile chum salmon from Yowkwala approximated the NOAA Fisheries threshold of 2,400 ng/g lipid for adverse health effects in juvenile salmon (Meador et al. 2002). Whole body PCB concentrations at and above this level are associated with an increased risk of immunosuppression, impaired thyroid function, reduced growth, and delayed mortality (Meador et al. 2002). Studies suggest that lipid-normalized tissue residue effect thresholds for toxic effects are similar across a wide range of species (Shephard 1997, Shephard 2004), so it is likely that PSS would be at risk for the same effects as those seen in salmon at comparable tissue PCB concentrations.

Concentrations of ∑PCB TEQs ranged 0.048–0.39 pg/g wet wt in salmon from the Commencement Bay sites, and 0.4–2.4 pg/g wet wt in PSS. These levels are well below the tetrochlorodibenzodioxin (TCDD) tissue benchmark concentrations of 8–9 pg/g wet wt recommended for salmonids by NOAA Fisheries and EPA (NMFS 2004, USEPA 2003). However, Giesy et al. (2002) reported delayed mortality in rainbow trout (*Oncorhynchus mykiss*) at a whole body TCDD concentration of 1 pg/g wet wt. Tissue ∑PCB TEQ concentrations in salmon and PSS from several of the restoration sites are close to or above that level. Moreover, these tissue benchmark concentrations are not adjusted for lipid content, which is an important factor regulating the toxicity of bioacumulative contaminants (Lassiter and Hallam 1990, Meador et al. 2002). The low lipid levels (<2%) in the majority of juvenile salmon and PSS from Commencement Bay could put them at increased risk for toxic effects of dioxin-like PCBs and similar contaminants in comparison to fish with comparable wet weight body burdens and higher lipid concentrations.

NOAA Fisheries Service has not yet systematically determined a threshold tissue residue concentration for the impact of DDTs on listed salmon. Most reported effects in salmonids are associated with whole body tissue total DDT concentrations at or above 500 ng/g wet wt (Allison et al. 1963, Burdick et al. 1964, Buhler et al. 1969, Johnson and Pecor 1969, Poels et al. 1980). Using these and other data, Beckvar et al. (2005) proposed a total DDT effect threshold of 600 ng/g wet wt for juvenile and adult fish, including salmonids. However, this guideline may not be fully protective because of the lack of adequate data on sublethal endpoints, including endocrine-disrupting, immunotoxic, and behavioral effects of DDTs (Donohoe and Curtis 1996, Arukwe et al. 1998, Celius and Walther 1998, Khan and Thomas 1998, Faulk et al. 1999, Christiansen et al. 2000, Zaroogian et al. 2001, Milston et al. 2003, Papoulis et al. 2003). Moreover, the guideline is not normalized for lipid content, which could affect toxicity threshold values considerably (Lassiter and Hallam 1990, van Wezel et al. 1995, Meador et al. 2002, Beckvar et al. 2005, Elskus et al. 2005).

The lipid content of the salmon sampled in Commencement Bay was generally in the 1–2% range for Chinook, chum, and pink salmon, and about 4% for coho salmon, while laboratory-reared salmonids typically have a lipid content of 8–11% (Meador et al. 2002). If we assume a lipid content of 10% in the laboratory fish used to determine DDT toxicity, a threshold of 500–600 ng/g wet wt would be equivalent to 5,000–6,000 ng/g lipid. Even in salmon from the Mowitch site, whole body DDT concentrations were below these levels. However, PSS from

Mowitch had whole body DDT concentrations above 5,000 ng/g lipid, so could be at risk for adverse health effects.

Data on the toxicity of HCB to juvenile salmon are limited, but the information that exists suggests that the tissue concentrations we observed in salmon from Commencement Bay are unlikely to have significant effects on their health. Wet weight concentrations were approximately 1 ng/g in salmon, and up to 7 ng/g in PSS. Most studies show health effects only at considerably higher concentrations, in the 1,000–10,000 ng/g wet wt range or above (Dalich et al. 1982, Niimi et al. 1984). Many sublethal effects of chronic exposure to HCB have been documented in mammals, such as neurological damage, altered immune and reproductive function, porphyria, and increased cancer risk (Daniell et al. 1997, Michielsen et al. 1999, ATSDR 2002b, Alvarez et al. 2005, Randi et al. 2006). Such effects, however, have not been studied in salmonid species that have been subjected to long-term exposure to these compounds at environmentally realistic concentrations. Additionally, as with DDTs, existing tissue residue data for HCB are not lipid normalized.

Exposure to PAHs may also contribute to health risks in juvenile Chinook salmon and marine fish from some of the sampling sites. Concentrations of PAHs in sediments were above the NOAA Fisheries threshold levels for effects on benthic fish (Johnson et al. 2002) at all sites sampled, suggesting there is some risk of liver disease, reproductive impairment, reduced growth, and possible other adverse effects in bottom fish residing at these sites. In English sole, for example, residence at sites with a total PAH concentration of 10,000 ppb (comparable to levels observed at the two Middle Waterway sites and Squally Beach) was associated with a 13 fold increase in DNA damage in the liver, and 15–35% declines in egg and larval viability and the proportions of adult females undergoing ovarian maturation and spawning, as compared to fish from sites with PAH concentrations of 100 ppb or below (Johnson et al. 2002). A toxicopathic liver disease prevalence of about 40% was observed in adult fish from sites with sediment PAH concentrations in the 10,000 ppb range (Johnson et al. 2002).

Moreover, Rice et al. (2000) observed significant reductions in growth of juvenile English sole fed on polychaete worms raised in sediments containing 3,000–4,000 ppb total PAHs. Weight increased about 1% per day in fish fed on uncontaminated worms, as compared to less than 0.1% per day in fish fed on contaminated worms. Sediment contamination and its effects on growth and development of juvenile sole could be partly responsible for the low number of juvenile flatfish observed at the Commencement Bay restoration sites sampled in this study, although other factors such as sediment grain size, habitat type, salinity, and larval transport patterns are likely to be equally or more important determinants of juvenile flatfish distribution.

Sediment or stomach contents PAH concentrations in the range of those measured at the Middle Waterway, Squally Beach, and Yowkwala sites might also adversely affect juvenile salmon. Concentrations of PAHs in sediments and stomach contents of salmon from some of the restoration sites are within the range of those reported at the Duwamish and Hylebos waterways, where immunosuppression and other health effects have been observed (Arkoosh et al. 1991, Varanasi et al. 1993, Arkoosh et al. 1994, Casillas et al. 1995, Stein et al. 1995, Arkoosh et al. 1998, Casillas et al. 1998, Stehr et al. 2000, Arkoosh et al. 2001, Bravo et al. 2004). In stomach contents of salmon from the Hylebos and Duwamish sites, total PAHs ranged from 4,000 to

15,000 ng/g wet wt, while sediment concentrations are typically greater than 5,000 ng/g dry wt (Varanasi et al. 1993, Stein et al. 1995, Stehr et al. 2000).

Dietary PAH levels associated impacts on growth and immune function in laboratory exposure studies are slightly higher. For example, Meador et al. (2006) observed effects on fish weight, plasma chemistry, and lipid class profiles at dietary concentrations of 7.7 ug/g wt wt total PAHs and above. Reported no-effect doses for immunosuppressive and other physiogical effects in other studies with juvenile salmon (e.g., Palm et al. 2003) are in the 8,000–16,000 ng/g wet wt range. Even in salmon from the Yowkwala site, where contamination was the greatest, total PAH concentrations in stomach contents were below this level. However, PAHs may contribute to immunosuppression or growth-altering impacts of other contaminants in environmental mixtures, even if they are below toxicity thresholds when considered alone (e.g., see Loge et al. 2005).

Summary and Conclusions

According to the NRDA trustees, the major goals of the restoration projects being carried out in Commencement Bay are to enhance fish habitat for juvenile salmonids and to otherwise protect the sites for natural resources. The objectives of this monitoring study were to assess the effectiveness of restoration and remediation efforts in improving fish habitat accessibility and use, especially for juvenile salmon, and in reducing the impacts of contaminants on Commencement Bay biota. It should be noted that no funding was available for prerestoration analysis of either fish assemblage or chemical analysis of fish or sediments. This significantly limits our ability to detect the effects of restoration efforts. However, data collected in this study do provide a useful characterization of the current status of these Commencement Bay restoration sites, identifying features that either enhance or detract from their desirability as habitat for listed fish and providing a basis for evaluation in the future.

Three criteria were suggested by Simenstad and Cordell (2000) to assess whether restoration efforts are successful for improving juvenile salmon habitat: 1) an increase in "opportunity" for juveniles to access the site, 2) an increase in the "capacity" or improvements to the habitat structure providing improved physical parameters, refugia from predators, increased prey, etc., and 3) a "realized function" such as improved growth, survival, or physiological performance. As yet, no clear trends in habitat use at the restoration sites can be observed. While there were some yearly differences in species abundances, the differences were relatively small considering the natural year-to-year variation that is known to occur (Miller et al. 1980, Levings et al. 1986, Shreffler et al. 1990, Tanner et al. 2002). However, the mere presence of juvenile salmon at the Commencement Bay restoration sites suggests that improvements have been made with respect to opportunity and capacity. The removal and replacement of contaminated sediments, removal of barges and other debris, as well as the addition of backwater pools and native riparian vegetation certainly increased the physical capacity (quality of the habitat) and the opportunity for fish to access the sites.

Accessibility for salmon and other fish species appears to be a problem at some sites, particularly those in the Middle Waterway. Restoration of additional sites and perhaps linking of waterways with traditional tidal channels or interwaterway connections may increase the use of these sites by juvenile salmon (Simenstad 2000, Cordell et al. 2001). The addition of a larger variety of landscape features would also provide better linkage between preferred salmon habitats (Cederholm et al. 2000, Simenstad 2000, Cordell et al. 2001). With the greatly diminished historical estuarine and coastal habitat of Commencement Bay, this has become an important link in restoring juvenile salmon rearing habitat (Shreffler et al. 1992, Toft et al. 2004).

The presence of chemical contaminants complicates evaluation of the functional performance improvements in fish growth and survival as a result of these site improvements. Chemical contaminants were present in sediment, prey, and fish at all sites, and concentrations of PAHs, PCBs, and DDTs in sediment, prey, and fish at some of the restoration sites are above concentrations associated with adverse health effects in salmon and marine fish, and above cleanup guidelines established by the trustees. The effects of these contaminants on fish health represent a reduction in the "realized function" that could offset any increase due to the overall physical improvements at the sites. Furthermore, it should be noted that other toxic contaminants (e.g., heavy metals, dioxins, furans) that have been identified previously at Commencement Bay sites (EVS 1996, Brown et al. 1998, EVS 2003) were not analyzed in sediments or fish tissues samples in the current study, but could be present.

While sediment remediation activities have been carried out at some of the restoration sites, it appears that they have not been fully successful in eliminating risks associated with contaminant exposure. Analyses also indicate DDT inputs into this system from relatively recent sources, which is an additional cause for concern. Although the presence of contaminants may not deter juvenile salmonids and other fish from using these Commencement Bay restoration sites for feeding and rearing, the resultant toxicant exposure poses risks to their long-term health and survival.

The biological and chemical monitoring efforts reported in this study are an important step in the assessment of the Commencement Bay restoration efforts. Few restoration projects have studied restoration effects on a multiyear basis, particularly where there is more than a once or twice a year sampling effort (Miller and Simenstad 1997, Roni et al. 2002, Beck et al. 2003). Another positive aspect of this monitoring effort is that sampling was done over a more extended season, April to October, which can be of particular importance in assessing the benefits to outmigrant juvenile salmon. For example, a May to June monitoring would likely miss sampling of juvenile salmonids with a greater probability to add to adult populations, that is, those juveniles arriving later or staying longer in the restored sites (Rice et al. 2005). Given the historically much more extensive estuarine habitat in Commencement Bay and the fact that juvenile salmon most likely had a much longer usage of the estuary (Simenstad 2000), any efforts to retain juvenile salmon in the system would contribute to the adult population by allowing more extended acclimation to salt water and a longer growth phase before outward migration.

The monitoring of multiple species is another major accomplishment of this study, considering that many monitoring efforts concentrate on only a few species. Monitoring multiple species as well as multiple years should be an integral part of any studies to determine how much system functionality has returned due to restoration efforts (Michener 1997, Weinstein et al. 2001, Gray et al. 2002, Roni et al. 2002, Rice et al. 2005). Restoration projects should include both an increase in the quality of the physical habitat and a concurrent increase in the accessibilty of fish to the system (Rice et al. 2005), incorporating better opportunities for growth and survival. It is also apparent that the state of the chemical habitat is an important consideration, as expensive efforts to improve physical habitat characteristics may have a limited impact on productivity if toxicants capable of reducing salmon growth and survival are present. The inclusion of contaminant monitoring is a particularly valuable feature of this study and a component that is often neglected, even in restoration of urban sites.

Additional years of sampling and analysis will be needed to determine the real results of the Commencement Bay restoration projects. The amount of variability in the year-to-year data at the restoration sites emphasizes the need for much longer periods of monitoring. Because of the complexity of estuarine environments and the unpredictable recovery rates of restored habitat, it has been suggested that monitoring duration should be at least five years to decades (Michener 1997, Weinstein et al. 2001, Rice et al. 2005). The monitoring results reported here are the initial contribution to this effort.

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Appendix A: Fish Species by Year and Site

Table A-1. All fish species captured during 2002 (02) and 2003 (03) by site.

Species	MWC	MWS	Mowitch	OV	SW	SB	TSM	Yow
American shad *(Alosa sapidissima)*			02					02
Buffalo sculpin *(Enophrys bison)*			03	02, 03			02	02
Cabezon sculpin (*Scorpaenichthys marmoratus)*			03					
Chinook salmon *(Oncorhynchus tshawytscha)*	02	02, 03	02, 03	02, 03	02, 03		02	02, 03
Chum salmon *(O. keta)*	02	02, 03	02, 03	02, 03	02, 03		02	02, 03
Coho salmon *(O. kisutch)*		02, 03	02, 03	03	02, 03		02	02, 03
Crescent gunnel *(Pholis laeta)*			02, 03	02, 03				02
Cutthroat trout *(O. clarkii clarkii)*			02, 03		02, 03			02, 03
English sole *(Parophrys vetulus)*				02, 03				
Flatfish (unidentified)				02, 03	02, 03			
Great sculpin *(Myoxocephalus polyacanthocephalus)*								02
Northern anchovy *(Engraulis mordax)*					02			
Northern sculpin (*Icelinus borealis*)				03				
Pacific herring *(Clupea pallasii)*		02, 03	02, 03	02, 03	02		02	02, 03
Pacific staghorn sculpin *(Leptocottus armatus)*	02, 03	02, 03	02, 03	02, 03	02, 03	02	02	02, 03
Penpoint gunnel *(Apodichthys flavidus)*			02, 03	02, 03			02	02
Pile perch *(Rhacochilus vacca)*		02, 03	02, 03	02, 03	03		02	02, 03
Pink salmon *(O. gorbuscha)*	02, 03	02, 03	02	03	02, 03		02	02, 03
Bay pipefish *(Syngnathus leptorhynchus)*			02, 30	02, 03	02		02	03
Ratfish (*Hydrolagus colliei*)								02

Table A-1 continued. All fish species captured during 2002 (02) and 2003 (03) by site.

Species	MWC	MWS	Mowitch	OV	SW	SB	TSM	Yow
Rock Sole *(Lepidopsetta bilineata)*			02	03				03
Saddleback gunnel *(Pholis ornata)*			02, 03	02, 03			02	02, 03
Sailfin sculpin *(Nautichthys oculofasciatus)*			03	02				
Salmon (unidentified)	02				02		02	03
Sandlance *(Ammodytes hexapterus)*			02	02, 03	02		02	02, 03
Sculpin (unidentified)		02	02, 03	02, 03	02, 03			02
Sebastes (unidentified)				02				
Shiner perch *(Cymatogaster aggregata)*	02, 03	02, 03	02, 03	02, 03	02, 03	02	02	02, 03
Silverspotted sculpin *(Blepsias cirrhosus)*				02				02
Snake prickleback *(Lumpenus sagitta)*		02	02, 03	02				
Speckled sanddab *(Citharichthys stigmaeus)*				02				
Starry flounder *(Platichthys stellatus)*	02, 03		02, 03	02, 03	02, 03			02, 03
Steelhead (*O. mykiss*)								03
Striped perch *(Embiotoca lateralis)*			02, 03	02, 03			02	02, 03
Sturgeon poacher *(Podothecus acipenserinus)*				02, 03				
Surf smelt *(Hypomesus pretiosus)*	03	02, 03	02, 03	02, 03	02, 03			02, 03
Threespine stickleback *(Gasterosteus aculeatus)*		02, 03	02, 03	02, 03	02, 03	02	02	02, 03
Tidepool sculpin *(Oligocottus maculosus)*		03	03	02, 03			02	03
Tubesnout *(Aulorhynchus flavidus)*				03	02, 03			02, 03
Whitespot greenling *(Hexagrammos stelleri)*				02				02

Appendix B: Fish Species Comprising Functional Groups

Table B-1. Fish species comprising functional groups captured at Commencement Bay restoration sites.

Functional group	Species included	Scientific name
Flatfish	English sole	*Parophrys vetulus*
	Rock sole	*Lepidopsetta bilineata*
	Starry flounder	*Platichthys stellatus*
	Speckled sandab	*Citharichthys stigmaeus*
Other dermsal fish	Pacific staghorn sculpin	*Leptocottus armatus*
	Silverspotted sculpin	*Blepsias cirrhosus*
	Great sculpin	*Myoxocephalus polyacanthocephalu*
	Buffalo sculpin	*Enophrys bison*
	Tidepool sculpin	*Oligocottus maculosus*
	Sailfin sculpin	*Nautichthys oculofasciatus*
	Northern sculpin	*Icelinus borealis*
	Cabezon	*Scorpaenichthys marmoratus*
	Penpoint gunnel	*Apodichthys flavidus*
	Crescent gunnel	*Pholis laeta*
	Saddleback gunnel	*P. ornata*
	Snake prickleback	*Lumpenus sagitta*
	White spot greenling	*Hexagrammos stellei*
	Sturgeon poacher	*Podothecus acipenserinus*
	Ratfish	*Hydrolagus colliei*
Surf perches	Shiner perch	*Cymatogaster aggregata*
	Pile perch	*Rhacochilus vacca*
	Striped perch	*Embiotoca lateralis*
Other nearshore fish	Bay pipefish	*Syngnathus leptorhynchus*
	Tubesnout	*Aulorhynchus flavidus*
	Threespine stickleback	*Gasterosteus aculeatu*
Forage fish	Surf smelt	*Hypomesus pretiosus*
	Pacific sandlance	*Ammodytes hexapterus*
	Pacific herring	*Clupea pallasii*
	American shad	*Alosa sapidissima*
	Northern anchovy	*Engraulis mordax*
Juvenile salmonids	Chinook salmon	*Oncorhynchus tshawytscha*
	Coho salmon	*O. kisutch*
	Pink salmon	*O. gorbuscha*
	Chum salmon	*O. keta*
	Steelhead	*O. mykiss*
	Cutthroat	*O. clarkii clarkii*

Appendix C: Lists of Chemical Contaminants in Sediments and Tissue Samples

The following are lists of chemical contaminants measured in sediments and tissue samples of salmon (*Oncorhynchus* spp.) and Pacific staghorn sculpin (*Leptocottus armatus*) collected in Commencement Bay during 2002 and 2003.

Analytes measured by gas chromatography/mass spectrometry (GC/MS) in sediments and fish stomach contents

- *Organochlorines (OCs)*
    - Chlordanes–heptachlor, heptachlor epoxide, oxychlordane, *trans*-nonachlor, *alpha*-chlordane, *gamma*-chlordane, *cis*-nonachlor and nonachlor III.
    - DDT and DDT metabolites–*p,p*'-DDT, *p,p*'-DDE, *p,p*'-DDD, *o,p*'-DDD, *o,p*'-DDE and *o,p*'-DDT
    - PCBs (40 congeners) –PCBs 17, 18, 28, 31, 33, 44, 49, 52, 66, 70, 74, 82, 87, 95, 99, 101/90, 105, 110, 118, 128, 138, 149, 151, 153/132, 156, 158, 170/190, 171, 177, 180, 183, 187, 191, 194, 195, 199, 205, 206, 208, and 209.
- *Other organochlorines*–hexachlorobenzene (HCB), mirex, *alpha*-hexachlorocyclohexane, *beta*-hexachlorocyclohexane, lindane (*gamma*-hexachlorocyclohexane), aldrin, dieldrin, endosulfan I, endosulfan II and endosulfan sulfate.
- *Polycyclic aromatic hydrocarbons (PAHs)*
    - Low molecular weight PAHs–naphthalene (NPH), 2-methylnaphthalene (MN2), 1-methylnaphthalene (MN1), biphenyl (BPH), 2,6-dimethylnaphthalene (DMN), acenapthylene (ACY), acenaphthene (ACE), 2,3,6-trimethylnaphthalene (TMN), fluorene (FLU), dibenzothiophene (DBT), phenanthrene (PHN), anthracene (ANT), and 1-methylphenanthrene (1MP).
    - High molecular weight PAHs–fluoranthene (FLA), pyrene (PYR), benzo[a]anthracene (BAA), chrysene + triphenylene (CHR/Trphn), benzo[b]fluoranthene (BBF), benzo[j]fluoranthene + benzo[k]fluoranthene (BJK/BKF), benzo[a]pyrene (BaP), benzo[e] pyrene (BeP), dibenz[a,h]anthracene (DBA), perylene (PER), indenopyrene (IDP), dibenz[a,h+c]anthracene, and benzo[ghi]perylene (BZP).

Analytes measured by high-performance liquid chromatography/ photodiode array (HPLC/PDA) in fish whole bodies

- DDT and DDT metabolites–*p*,*p*'-DDT, *p*,*p*'-DDE, *p*,*p*'-DDD, *o*,*p*'-DDD, and *o*,*p*'-DDT.

- PCBs (15 congeners)–PCBs 77, 101, 105, 110, 118, 126, 128, 138, 153, 156, 157, 169, 170/194, 180, and 189.

Appendix D: Information on Sediment Chemical Contaminant Analysis

Table D-1. Information on sediments collected at five Commencement Bay restoration sites in 2002 for chemical contaminant analysis.

Sample number	Site	Collection date	Contaminant analysis
OV Comp #1	Olympic View	05/31/02	PAHs and OCs
OV Comp #2	Olympic View	05/31/02	PAHs and OCs
OV Comp #3	Olympic View	05/31/02	Archived
OV Jar A	Olympic View	6/24/02	PAHs and OCs
OV Jar B	Olympic View	6/24/02	Archived
OV Jar C	Olympic View	6/24/02	PAHs and OCs
MW Jar A	Middle Waterway at Simpson	6/24/02	PAHs and OCs
MW Jar B	Middle Waterway at Simpson	6/24/02	Archived
MW Jar C	Middle Waterway at Simpson	6/24/02	PAHs and OCs
MW Jar D	Middle Waterway at Simpson	6/24/02	Archived
MW Jar E	Middle Waterway at City	6/24/02	PAHs and OCs
MW Jar F	Middle Waterway at City	6/24/02	PAHs and OCs
MW Jar G	Middle Waterway at City	6/24/02	Archived
MO Jar A	Mowitch	6/24/02	PAHs and OCs
MO Jar B	Mowitch	6/24/02	PAHs and OCs
MO Jar C	Mowitch	6/24/02	Archived
MO Jar D	Mowitch	6/24/02	PAHs and OCs
MO Jar E	Mowitch	6/24/02	Archived
MO Jar F	Mowitch	6/24/02	PAHs and OCs
MO Jar G	Mowitch	6/24/02	PAHs and OCs
MO Jar H	Mowitch	6/24/02	Archived
SW Jar A	Skookum Wulge	6/24/02	Archived
SW Jar B	Skookum Wulge	6/24/02	PAHs and OCs
SW Jar C	Skookum Wulge	6/24/02	PAHs and OCs
SB Jar A	Squally Beach	6/26/02	Archived
SB Jar B	Squally Beach	6/26/02	Archived
SB Jar C	Squally Beach	6/26/02	PAHs and OCs
SB Jar D	Squally Beach	6/26/02	PAHs and OCs
SB Jar E	Squally Beach	6/26/02	Archived
SB Jar F	Squally Beach	6/26/02	PAHs and OCs

Appendix E: Composite Scheme of Whole Bodies of Salmonids Analyzed for OCs by HPLC/PDA

Table E-1. Composite scheme of whole bodies of salmonids (in chronological order) collected in 2002 and analyzed for OCs by HPLC/PDA.

Sample number	Species	Site	Collection date	No. fish in sample
SP2385	Coho	Yowkwala	5/13/02	3
SP2387	Coho	Yowkwala	5/13/02	3
SP2388	Coho	Yowkwala	5/13/02	2
SP2389	Coho	Yowkwala	5/13/02	2
SP2390	Coho	Yowkwala	5/14/02	2
SP2391	Coho	Yowkwala	5/14/02	3
SP2368	Pink	Tahoma Salt Marsh	5/28/02	3
SP2369	Pink	Tahoma Salt Marsh	5/28/02	2
SP2372	Chum	Tahoma Salt Marsh	5/28/02	6
SP2373	Chum	Tahoma Salt Marsh	5/28/02	6
SP2354	Chinook	Mowitch	5/29/02	1
SP2355	Chinook	Mowitch	5/29/02	1
SP2356	Chinook	Mowitch	5/29/02	1
SP2359	Chinook	Mowitch	5/29/02	1
SP2362	Chinook	Mowitch	5/29/02	1
SP2363	Chinook	Mowitch	5/29/02	1
SP2364	Chinook	Mowitch	5/29/02	1
SP2374	Chinook	Mowitch	5/29/02	1
SP2375	Chinook	Mowitch	5/29/02	1
SP2376	Chinook	Mowitch	5/29/02	1
SP2377	Chinook	Mowitch	5/29/02	1
SP2378	Chinook	Mowitch	5/29/02	1
SP2379	Chinook	Mowitch	5/29/02	1
SP2380	Chinook	Mowitch	5/29/02	1
SP2371	Chum	Tahoma Salt Marsh	6/24/02	2
SP2370	Chum	Tahoma Salt Marsh	6/26/02	3
SP2357	Chinook	Tahoma Salt Marsh	6/26/02	2
SP2358	Chinook	Tahoma Salt Marsh	6/26/02	2
SP2351	Chinook	Yowkwala	6/27/02	1
SP2352	Chinook	Yowkwala	6/27/02	2
SP2353	Chinook	Yowkwala	6/27/02	2

Table E-1 continued. Composite scheme of whole bodies of salmonids (in chronological order) collected in 2002 and analyzed for OCs by HPLC/PDA.

Sample number	Species	Site	Collection date	No. fish in sample
SP2360	Chinook	Yowkwala	6/27/02	3
SP2361	Chinook	Yowkwala	6/27/02	1
SP2367	Pink	Yowkwala	6/27/02	2
SP2384	Pink	Yowkwala	6/27/02	1

Table E-2. Composite scheme of whole Pacific staghorn sculpin samples (in chronological order) collected in 2003 and analyzed for OCs by HPLC/PDA.

Sample number	Species	Site	Collection date	No. fish in sample
SP2491	PSS	Olympic View	6/6/03	3
SP2493	PSS	Olympic View	6/6/03	3
SP2497	PSS	Middle Waterway at City	6/6/03	3
SP2498	PSS	Middle Waterway at City	6/6/03	3
SP2508	PSS	Middle Waterway at City	6/6/03	3
SP2499	PSS	Olympic View	6/18/03	2
SP2504	PSS	Mowitch	6/19/03	3
SP2492	PSS	Olympic View	7/17/03	3
SP2494	PSS	Middle Waterway at Simpson	7/17/03	3
SP2500	PSS	Middle Waterway at Simpson	7/17/03	3
SP2509	PSS	Skookum Wulge	7/18/03	1
SP2510	PSS	Skookum Wulge	7/18/03	1
SP2511	PSS	Skookum Wulge	7/18/03	2
SP2495	PSS	Yowkwala	7/18/03	1
SP2496	PSS	Middle Waterway at City	8/6/03	2
SP2505	PSS	Mowitch	8/15/03	3
SP2507	PSS	Mowitch	8/15/03	3
SP2506	PSS	Middle Waterway at Simpson	8/19/03	3

Appendix F: Fish Samples Analyzed from Restoration Sites

Table F-1. Fish samples analyzed from Commencement Bay restoration sites in 2002 and 2003 (Comp. = composite sample of fish, Indiv. = individual fish, PSS = Pacific staghorn sculpin).

			Matrix			
Collection year	**Collection site**	**Species**	**Bile**	**Stomach contents chemistry**	**Stomach contents taxonomy**	**Whole bodies**
2002	MWC	Chinook	No	No	Yes	No
		Chum	No	No	Yes	No
		Coho	No	No	Yes	No
		Pink	No	No	Yes	No
	MWS	Chinook	No	No	Yes	No
		Chum	No	No	Yes	No
		Coho	No	No	Yes	No
		Pink	No	No	Yes	No
	Mowitch	Chinook	No	No	Yes	11 indiv.
		Chum	No	No	Yes	No
		Coho	No	No	Yes	No
		Pink	No	No	Yes	No
	Olympic View	Chinook	No	No	Yes	No
		Chum	No	No	Yes	No
		Coho	No	No	Yes	No
		Pink	No	No	Yes	No
	Skookum Wulge	Chinook	No	No	Yes	No
		Chum	No	No	Yes	No
		Coho	No	No	Yes	No
		Pink	No	No	Yes	No
	Squally Beach	Chinook	No	No	Yes	No
		Chum	No	No	Yes	No
		Coho	No	No	Yes	No
		Pink	No	No	Yes	No

Table F-1 continued. Fish samples analyzed from Commencement Bay restoration sites in 2002 and 2003 (Comp. = composite sample of fish, Indiv. = individual fish, PSS = Pacific staghorn sculpin).

			Matrix			
Collection year	**Collection site**	**Species**	**Bile**	**Stomach contents chemistry**	**Stomach contents taxonomy**	**Whole bodies**
2002 cont.	Tahoma Salt Marsh	Chinook	No	No	Yes	2 comps.
		Chum	No	No	Yes	4 comps.
		Coho	No	No	Yes	
		Pink	No	No	Yes	2 comps.
	Yowkwala	Chinook	No	No	Yes	3 comps., 2 indiv.
		Chum				
		Coho	No	No	Yes	6 comps.
		Pink	No	No	Yes	1 comp., 1 indiv.
2003	MWC	Chinook	No	No	No	No
		Chum	No	No	No	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	Yes	4 comps.
	MWS	Chinook	No	1 comp.	Yes	No
		Chum	No	No	No	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	Yes	3 comps.
	Mowitch	Chinook	No	No	No	No
		Chum	No	No	No	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	Yes	3 comps.
	Olympic View	Chinook	No	1 comp.	Yes	No
		Chum	No	No	No	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	Yes	4 comps.
	Skookum Wulge	Chinook	1 comp.	1 comp.	Yes	No
		Chum	No	No	No	No
		Coho	No	No	No	No

Table F-1 continued. Fish samples analyzed from Commencement Bay restoration sites in 2002 and 2003 (Comp. = composite sample of fish, Indiv. = individual fish, PSS = Pacific staghorn sculpin).

			Matrix			
Collection year	**Collection site**	**Species**	**Bile**	**Stomach contents chemistry**	**Stomach contents taxonomy**	**Whole bodies**
2003 cont.	Skookum Wulge cont.	Pink	No	No	No	No
		PSS	No	No	Yes	3 comps.
	Squally Beach	Chinook	No	No	No	No
		Chum	No	No	No	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	No	No
	Tahoma Salt Marsh	Chinook	No	No	No	No
		Chum	No	No	No	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	No	No
	Yowkwala	Chinook	1 comp.	1 comp.	Yes	No
		Chum	No	No	Yes	No
		Coho	No	No	No	No
		Pink	No	No	No	No
		PSS	No	No	Yes	1 comp.

Appendix G: Pacific Staghorn Sculpin Age Determination of Individuals Sampled

Table G-1. Pacific staghorn sculpin age determination of individuals sampled for chemical analyses.

Collection site	No.	Age range (years)	Mean age (± standard error)
Middle Waterway at City	39	0.5–1.3	0.78 ± 0.04
Middle Waterway at Simpson	53	0.6–1.3	0.82 ± 0.03
Mowitch	31	0.5–0.8	0.70 ± 0.2
Olympic View	24	0.5–1.4	0.86 ± 0.04
Skookum Wulge	7	0.7–1.1	0.83 ± 0.5
Yowkwala	4	0.7–1.2	1.00 ± 0.12

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