Evaluating the Biological Condition of Puget Sound

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Abstract

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Puget Sound is a biologically rich and productive fjord-estuary of high ecological and socioeconomic significance. During the last two centuries, the Puget Sound region became a major population center, full of industrial, agricultural, and forestry activity, and subjected to intensive environmental manipulation and natural resource harvest. Today we see severe and expanding human influence throughout the Puget Sound landscape, and multiple, continuing signs of biological decline. At the same time, monitoring and research to understand, protect, and recover Puget Sound is a fragmented, uneven collection of efforts, surprisingly little of which considers Puget Sound in an ecosystem context or focuses specifically on the biological effects of human activity. As a result, we have no comprehensive, coherent narrative of how the Puget System ecosystem works, how it has been affected by human activity, and what can and should be done to restore the Sound or even halt or slow its decline.

This dissertation contributes to such a narrative by briefly summarizing our understanding of the Puget Sound ecosystem in the context of human activity; by providing new research that improves that understanding; and by suggesting future directions for monitoring and research. Chapter 1 reviews the basic ecological character of Puget Sound and the history of natural resource management and

environmental assessment. The next four chapters present results from several distinct research projects: a site-level assessment of effects of anthropogenic shoreline modification on beach microclimate and egg mortality in an intertidally spawning fish (Chapter 2); seasonal, geographic, and size distributions of juvenile hatchery and wild Chinook salmon in nearshore surface waters (Chapter 3); landscape-scale characterization of pelagic macrofauna assemblage composition in nearshore surface waters (Chapter 4); and the combination and reanalysis of data from historical and ongoing assessment and monitoring programs to explore relationships between marine bird and waterfowl assemblage composition and urbanization in the adjacent terrestrial landscape (Chapter 5). Finally, Chapter 6 uses the historical context and research results of the first five chapters to outline the primary challenges in developing more effective biological monitoring and assessment programs for Puget Sound.

Table of Contents

List of Figures

iv

List of Tables

Acknowledgements

Dedication

To my mother, Shirley Steele Rice.
Thanks, Mom.

CHAPTER 1

The Forsaken Fjord: Science, Society, and Biological Decline in Puget Sound

Introduction

R. M. Strickland's *The Fertile Fjord* (1983), an obscure paperback, now out of print, is among the best ecology writings about Puget Sound. Focusing on the lowly plankton but ranging across spatial and temporal scales from microbes to mammals, Strickland synthesized patchy information into an accessible, engaging, and surprisingly comprehensive introduction to the biological oceanography of Puget Sound. The narrative was mindful of the socioeconomic value of the ecosystem and the biological consequences of human activity. It also discussed the scientific process and pointed out many knowledge gaps. Fortunately, *The Fertile Fjord* was published. Unfortunately, it has not been updated or replaced, and its subject, the Puget Sound ecosystem, continues a long decline, in part because of what we do not know, but also because of our failure to apply what we do know. Indeed, the plankton, known to be a fundamental component of the ecosystem and the focus of *The Fertile Fjord*, are routinely ignored in Puget Sound science and management.

The Fertile Fjord is part of the Puget Sound Books series (Chasan 1981, Angell and Balcomb 1982, Bish 1982, Downing 1983, Strickland 1983, Burns 1985, Cheney and Mumford 1986), produced by Washington Sea Grant during the late 1970's and early 1980's, to make available to a general audience "useful information about Puget Sound"—its physical and biological characteristics, its history as a natural resource, and its governance. By that time, over a century of modern human activity in and around Puget Sound had radically transformed the physical and biological landscape: tidelands diked and filled (Bortleson et al. 1980, Collins and Sheikh 2005); rivers diverted, impounded, and polluted (Montgomery et al. 2003); forests cut; cities and farms built (Chasan 1981); wild salmon pushed to extinction and hatchery salmon manufactured to compensate (Lichatowich 1999, Taylor 1999, Montgomery 2003, Pess et al. 2003); nonnative species introduced (sometimes

intentionally) (Cohen et al. 1998, Ruiz et al. 2000); fisheries and wildlife in decline
(West 1997, PSAT 2007b); and on and on. Managing the consequences of these
transformations in the face of ever increasing pressure from human activity required
solid and accessible information about the ecosystem and its relationship to society.
That was the motivation for the Puget Sound Books, and while the series is
incomplete (many important topics were not addressed, and half the planned
volumes, including one on fish ecology and another on physical and chemical
oceanography, were never published), it remains the high water mark for broad
synthesis information on Puget Sound.

That the Puget Sound Books series is incomplete, much of it unavailable, and not
superseded is consistent with the larger picture of Puget Sound science and
management: a fragmented, uneven jumble of mostly small pieces, remote in the
public mind and so far unable to sufficiently prevent biological decline. In the two
decades since the publication of the Puget Sound Books, the human population of the
Puget Sound region increased by a third to over four million (WSOFM 2005), while
the number of wild Chinook salmon (*Oncorhynchus tshawytscha*) returning each
year to spawn continued declining and reached a low of less than 10% of its historic
size, eliciting protection under the Endangered Species Act (Myers et al. 1998).
Many of the Sound's other fish and wildlife species also entered problem status,
dozens of them either listed as "species of concern" or "threatened" or "endangered"
by state and federal agencies, including the most iconic Pacific Northwest wildlife
species of all, the orca (*Orcinus orca*) (Krahn et al. 2004). All this despite the
creation of multiple government and nongovernment entities to manage, monitor,
assess, study, and "restore" Puget Sound. Why?

Perhaps a large, modern human population and healthy ecosystems in Puget
Sound are irreconcilable. But ecological destruction by society is much more likely
without 1) a comprehensive understanding of the ecosystem, especially how it is
affected by human activity, 2) effective methods to evaluate its condition, and 3) the
political will to act on that accumulating knowledge when necessary. Developing all

of these for an ecosystem as large and complex as Puget Sound requires study and monitoring of the whole ecosystem and the full array of human influences that affect it, but also synthesis of the resulting data into useful information (NRC 1990, Peterson and Estes 2001, Hughs et al. 2005). Unfortunately, few scientific efforts in Puget Sound have taken, or contributed to, such a conceptually broad approach. Most have been limited to a single or few physical, chemical, or biological measures at restricted places and times. Although the biological consequences of some major human influences have been studied extensively (e.g., chemical pollutants), the consequences of others (e.g., filling of estuarine wetlands, shoreline armoring, hatcheries) have been largely ignored. Moreover, data collected in many projects are underreported, underused, and even lost or destroyed. This general lack of breadth, synthesis, communication, and continuity impairs the development of both a comprehensive understanding of the Puget Sound ecosystem and more effective biological assessment methods, undermining our ability to protect and recover the living systems of Puget Sound.

Improved Puget Sound science and management requires better information on the biological effects of natural, but more importantly, human, influences. This can be achieved through the collection of new data and the application of different analytical approaches. The four data chapters in this dissertation (Chapters 2 – 5) provide a mix of both, and to varying degrees emphasize biology, human influence, and whole ecosystem context (Table 1.1). Attention to these three elements is explicitly required by major environmental laws guiding the management of Puget Sound (e.g., the Clean Water Act, the Endangered Species Act, and most recently, the Puget Sound Initiative) yet are often missing from Puget Sound science.

Table 1.1. Biological, human influence, and ecosystem contexts of the four data chapters.

	Biology	Human Influence	Ecosystem Context
Chapter 2	Surf smelt embryo condition	Shoreline modification	Site-level study of a specific, prevalent disturbance type and a widespread species
Chapter 3	Juvenile Chinook salmon density and size	Hatcheries	Broad geographic and seasonal coverage of neritic waters including six river systems
Chapter 4	Pelagic fish & gelatinous zooplankton assemblage composition	Hatcheries	Broad geographic and seasonal coverage of neritic waters including six river systems; Multiple taxa included in analysis
Chapter 5	Marine bird & waterfowl assemblage composition	Urbanization	All shorelines and pelagic surface waters of greater Puget Sound; Multiple taxa included in analysis

In addition to new scientific information, we must understand the historical attempts to evaluate the ecological consequences of human activity, including an examination of why these efforts have not been more successful to date (Lombard 2006). Recent agency summaries of environmental issues in Puget Sound (PSP 2006, Ruckelshaus and McClure 2007) do not take this critical step. In this chapter, I provide brief background on Puget Sound's physical and biological character, its history as a natural resource, development of our scientific understanding of the ecosystem, and then evaluate some past efforts to assess its condition. Chapter 6 revisits issues about Puget Sound science raised in this chapter using key results from the data chapters and offers suggestions for future monitoring and research efforts. This information can help scientists, policy makers, and the public avoid

historical mistakes (NRC 1990) as they make choices about the future management of Puget Sound.

Ecological Setting

Puget Sound is the southern arm of an inland sea that includes the Straits of Georgia and Juan de Fuca, located on the Pacific Coast of North America and directly connected to the Pacific Ocean (Figure 1.1). The landforms of this system are primarily the result of tectonic activity and glaciation (Burns 1985, Alt and Hyndman 1995). Subduction of the Earth's Pacific plate under the North American plate formed the uplifted marine crust of the Olympic mountains to the west of Puget Sound, and the volcanic peaks of the Cascade mountains to the east (McKee 1972). Several glacial advances, the most recent of which peaked approximately 16,950 years ago and extended as far south as present day Olympia, Washington, gouged and eroded trenches and deposited sediment to form the complex topography of the Puget Sound basin (Booth 1987, Porter and Swanson 1998). After the last glacial retreat, the complex interplay of sea level rise (+90 m), isostatic rebound of the land surface (+200 m), and downcutting of rivers into glacial sediments (-10s to 100s of meters) gave rise to the present-day Puget Sound drainage basin (Beechie et al. 2001). Seawater flooded the basin and ever since, tidal action, freshwater inflow, and ocean currents have interacted to circulate and exchange dense marine water at depth from the Pacific, and less dense freshwater from the surrounding watersheds at the surface, producing a net seaward flow of water (Strickland 1983). Tides are mixed-semidiurnal: two highs and two lows in a 24–hour and 50–minute tidal day, with the highs more similar than the lows. Because of interactions between the oceanic tidal wave and the morphology of the basins, tidal ranges vary spatially, generally increasing with distance from Admiralty Inlet (Mofjeld and Larsen 1984, Mofjeld 1992) toward southern Puget Sound (Figure 2.1). For example, tidal range at the north end of Admiralty Inlet is approximately 2.5 m, but in South Puget Sound is approximately 4.5 m.

Figure 1.1. Georgia Basin and Puget Sound watershed and marine waters.
Map courtesy of Environment Canada.

The seaward boundaries of Puget Sound are the sill (a bathymetric shallow point) at Admiralty Inlet, and Deception Pass and the southern terminus of Swinomish Channel, both in Skagit Bay (Figure 1.2). Puget Sound is typically divided into four major sub-areas, or basins: Whidbey Basin, Main Basin, South Sound, and Hood Canal. Sills define the boundaries between them, except where the Whidbey Basin meets the Main Basin. Thousands of streams and rivers deliver freshwater to Puget Sound (the larger river systems are in the Whidbey and Main Basins), producing sub-estuaries within the larger Puget Sound estuary. Consequently, Puget Sound is best described as a fjord-estuary complex. Most of the water exchange in Puget Sound is through Admiralty Inlet, and the configuration of sills and deep basins results in the partial recirculation of water masses, and the retention of sediment and biota (Strickland 1983).

Puget Sound has biotic features typical of estuaries (Day et al. 1989): highly productive, often structurally complex, intertidal wetlands and submergent vegetation; and a rich fauna (Kozloff 1983, Kruckeberg 1991) that includes many temporary residents such as larval and juvenile fishes, and migratory species such as anadromous fishes, and marine birds and waterfowl. However, because it is a deep, cold, salty fjord, Puget Sound is more oceanic in character than the coastal plain estuaries to the south and west. For example, in contrast to the Columbia River estuary just 100 km away, Puget Sound is used more by many organisms typical of the coastal ocean, including marine fishes such as long-lived rockfish (*Sebastes spp.*) and large sharks (e.g., the sixgill *Hexancus griseus*); toothed and baleen whales, and porpoises (*Cetacea spp.*); and resident populations of several salmon (*Oncorhynchus spp.*) that never go to sea. Its rich biota (DeLacy et al. 1972, Miller and Borton 1980, Angell and Balcomb 1982, Kozloff 1983, Kruckeberg 1991) includes at least (accounting varies, depending on the definition of Puget Sound): 220 fish species, 100 marine bird and waterfowl species, 20 mammal species, hundreds of plant species, and thousands of invertebrate species. Marsh plants, eelgrass, and macroalgae—among the highest primary producers per unit area on Earth—line its

shores, often adjacent to one another and even intermingling. These plants provide physical habitat for many species, food for primary consumers, and are sources of the detritus that is a major input to Puget Sound food webs.
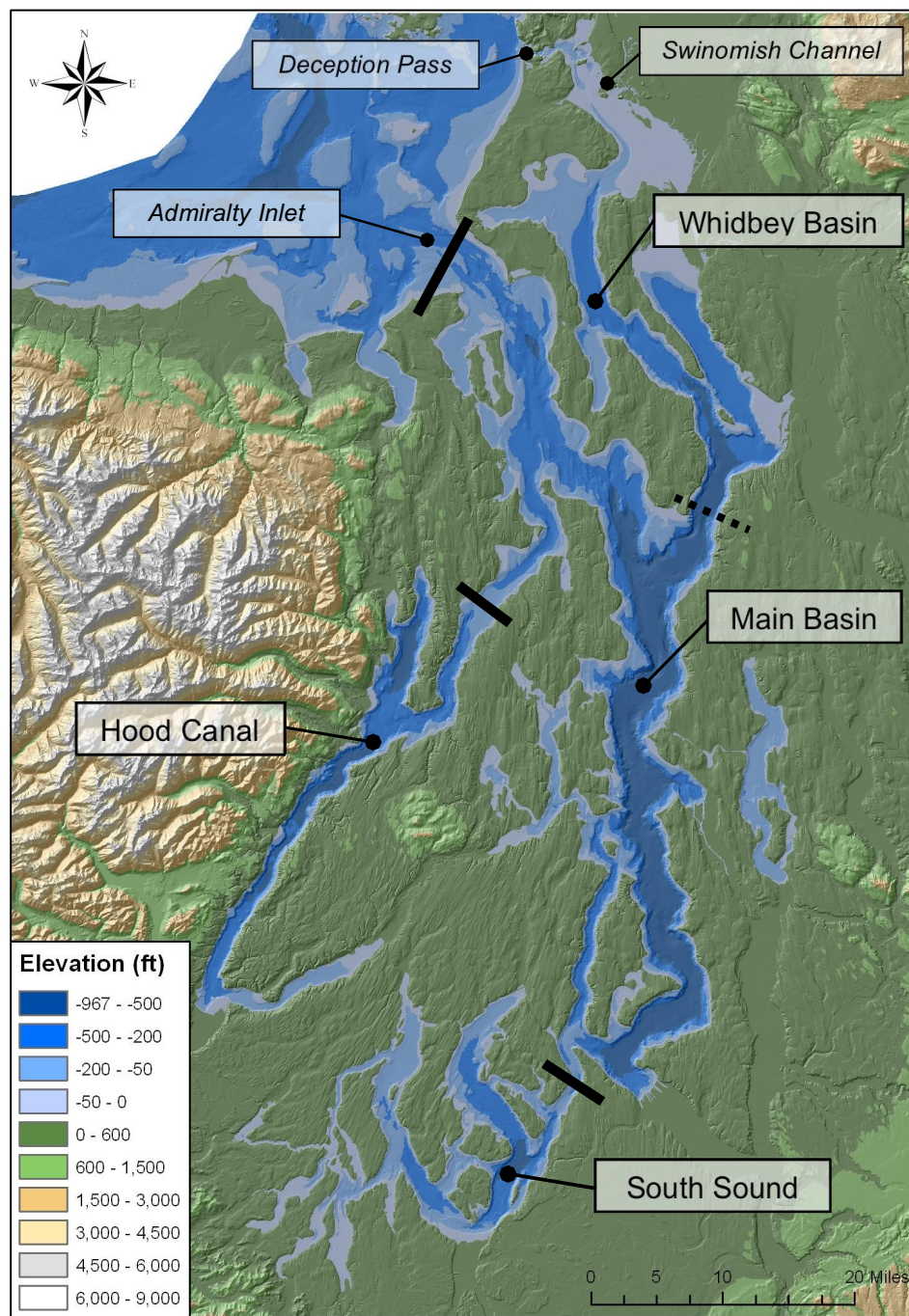
Figure 1.2. Puget Sound. Heavy black bars indicate sills (bathymetric shallows) that divide all basins except for where the Whidbey and Main Basins meet (dashed line). Base map by Jason Hall using data from Finlayson (2005).

Governance and Environmental Laws

No single governmental entity has complete jurisdiction over the environment affairs of Puget Sound, but the State of Washington is primarily responsible, with federal, local, and tribal governments also playing significant roles (Bish 1982, Lombard 2006). Management of biological resources, the physical environment and water quality, and recreation is spread across many agencies, with each having a specific focus. At the state level, biological resources are traditionally the responsibility of the Washington Department of Fish and Wildlife (WDFW), the Washington Department of Natural Resources (WDNR), and the tribes who are co-managers of fish and shellfish with the State. One inherent conflict with respect to WDFW is that it is simultaneously charged with management of resource extraction and conservation. Approximately 20 % of its budget comes from license sales, yielding a strong tendency to focus on harvested species (WDFW 2006). The departments of Ecology (DOE) and Natural Resources (DNR), and city and county governments share most of the monitoring and regulation of the physical environment and water quality. The Washington State Parks and Recreation Commission (WSPRC), WDFW, WDNR, and city and county governments manage recreation.

The more important laws influencing the management of the Puget Sound ecosystem are, at the federal level, the Clean Water Act, the Endangered Species Act, and the Marine Mammal Protection Act; and at the state level, the Growth Management Act and the Shoreline Management Act. The past two decades have seen substantial growth in the biological focus of Clean Water Act programs, although Washington is behind other states in this expansion (Chasan 2000, USEPA 2002).

History as a resource

The Puget Sound ecosystem has supported human populations for millennia (Sturtevant 1996). Aboriginal inhabitants had major effects on the ecosystem (White

1992, Flannery 2001), particularly in terrestrial environments (for example, through use of fire to cultivate certain food crops), but at the time of European settlement Puget Sound had rich, productive natural systems that supported human populations numbering at least into the tens of thousands (Sturtevant 1996). Despite extensive harvest, salmon remained the dominant staple (Lichatowich 1999, Taylor 1999, Butler and Campbell 2004), and none of the major environmental problems we struggle with today was present.

The ecological transformation of Puget Sound by Europeans began in the early 1800s when traders and trappers, in addition to introducing catastrophic diseases to the aboriginal humans (Boyd 1999), decimated beaver populations, surely reducing the extent of wetlands and associated biota (Naiman et al. 1988, Beechie et al. 2001, Pollock et al. 2004). More extensive changes came in the mid-1800s with rapid growth of the lumber industry, facilitated not only by seemingly inexhaustable forests of prime old-growth timber that often covered the landscape up to the shoreline (Ayers 1899, Gannett 1899), but by the deep, protected waters of the Sound that allowed relatively easy shipping to markets and population centers to the south, especially the gold rush boomtown of San Francisco (Chasan 1981). Significant commercial fisheries, primarily for Pacific salmon (*Oncorhychus spp.*) and native Olympia oysters (*Ostrea concaphila*), were established by the early 1870s, and, as with the lumber, much of the production was shipped south to San Francisco, in part because of over harvest and pollution of local fisheries resources in California (Chasan 1981, Lichatowich 1999, Taylor 1999).

Agriculture grew along with European colonization from the mid-1800s, often resulting in diking or filling vast areas of river floodplain and estuarine wetland (Beechie et al. 1994, Collins et al. 2003, Collins and Sheikh 2005). Arrival of the railroads in the 1880s expanded the markets for Puget Sound's natural resources and also facilitated increasing population. In the early 1900's, primarily because of the Alaskan gold rush, population in the Puget Sound region more than doubled to over a quarter million people (Chasan 1981, USCB 1995). At the same time, dam

construction on the region's rivers began to supply water to population centers and power to lumber and pulp mills.

By the 1890s declines of the two major fisheries resources, salmon and native oysters, were already underway (Chasan 1981, Lichatowich 1999, Taylor 1999). Both were over harvested, but also suffered from habitat destruction, initially from timber industry and mining activity: salmon from the destruction of freshwater spawning and rearing habitat resulting from the cutting and transport of trees and dumping of mine waste (Smith and Anderson 1922, Lichatowich 1999), and oysters from toxic pulp mill effluent (Chasan 1981). Increasing dam construction and upland land development further degraded rivers, at times blocking salmon and other anadromous species access to whole watersheds.

The first attempts to maintain fisheries production in response to declining harvests were hatcheries in the case of salmon, and in oysters, the transplantation of larger and presumably hardier and faster growing species from outside the region. The first (and unsuccessful) oyster transplant was the eastern oyster (*Crassostrea virginicus*) from the Atlantic coast. Later, the Pacific oyster (*Crassostrea gigas*) from Asia was transplanted and successfully established. Unintentionally introduced with the nonnative oysters were the Manila clam (*Venerupis philippinarum*), and two species of predatory snail (eastern oyster drill [*Urosalpinx cinerea*] and the Japanese oyster drill [*Ocinebrellus inornatus*]). To this day, Pacific oysters and Manila clams are the dominant commercial bivalve species produced throughout the Pacific Northwest.

Increasing industrial development and population growth continued in the early Twentieth Century, and with Word War II, the Puget Sound region became a more active military port for production and service of military equipment, further altering the landscape and increasing the population (Chasan 1981). After the war, recreational and commercial fishing in Puget Sound expanded, and by the mid-1970s several species were beginning to decline (Schmitt et al. 1994). Three gadoids—Pacific cod (*Gadus macrocephalus*), walleye pollock (*Theragra chalcogramma*), and

Pacific whiting, or hake (*Merluccius productus*)—are now nearly absent from Puget Sound but it is unclear what the relative roles were of natural environmental factors, biological interactions, and harvest (Schmitt et al. 1994, Palsson et al. 1997). Pressure on non-salmon fisheries further increased in the late 1970s as a result of declining salmon fisheries, growing popularity of non-salmon fish, and possibly the Boldt decision (W. Palsson WDFW personal communication). The Boldt ruling, in recognizing treaty rights, established that half of all salmon harvest (including hatchery fish) was to be granted to the tribes (Cone and Ridlington 1996). Changes in effort (Palsson et al. 1997) do roughly correspond to the timing but I know of no documentary evidence connecting a change in non-salmon fisheries management to the Boldt decision. For whatever reason, the harvest of non-salmon species increased (Palsson 1997). Only later, after rapid declines in several of these species, was it understood that the biology of many of them did not lend itself to heavy fishing pressure. For example, several rockfish (*Sebastes spp.)* species are very long lived and mature late (Love et al. 2002), so heavy fishing pressure can quickly deplete spawners that take decades to replace.

Declines in commercial and recreational fisheries did not continue entirely unabated. Harvest reductions were introduced for herring and various demersal species in the 1980s, and bottom trawling was banned entirely in Puget Sound in 1989 (although this was driven as much by owners of residential shoreline property who objected to bottom trawling on aesthetic grounds, as it was harvest managers or conservationists [Hensleigh 1994]). Most restrictions occurred only after populations were severely reduced (Schmitt et al. 1994).

As wild salmon and demersal fish populations declined, salmon production in hatcheries increased. The single overriding goal for hatcheries is to produce as many fish as possible for harvest as adults, but success in many programs was simply measured by the numbers of juveniles released rather than those that survived to harvestable size or return from the ocean and spawn (Mobrand et al. 2005). Since freshwater habitat and harvest were assumed to be primary limits to salmon, the

carrying capacity of estuarine and oceanic receiving environments was considered to be effectively limitless. Despite recent acknowledgements of the potential for adverse ecological effects of hatchery releases on estuarine environments (WDFW and PSTT 2004, Mobrand et al. 2005, SSPS 2005), almost no monitoring and research on key topics (e.g., reform of hatchery practices, competitive interactions and disease transmission between hatchery and wild fish, environmental carrying capacity) has been done.

Scientific Study

Scientific study of Puget Sound began with the Vancouver expedition in 1792. The Sound is named for one of the ship's officers, and the species name of the dominant tree in the region (Douglas fir [*Pseudotsuga Menziesii*]) is taken from the name of the expedition naturalist. In addition to searching for navigation routes, early mapping and characterization of the biota by the Vancouver expedition and those that followed was often to evaluate the natural resource potential of the area. Subsequent colonization and development continued this pattern of producing information about the Puget Sound ecosystem as a natural consequence of evaluating and delineating property, and extracting natural resources (e.g., Gannett 1899, Adair 1909). Thus, the primary drivers of information about the Puget Sound ecosystem have always been natural resource extraction and colonization; major ecological transformations occurred before serious studies of the ecosystem were initiated.

Academic study of Puget Sound dates back to the late 1800s but basic biology and oceanography really progressed in earnest with the establishment of the University of Washington's (UW) Puget Sound Marine Station (now the Friday Harbor Laboratories) in 1910, the UW College of Fisheries in 1919, and the UW Oceanographic Laboratories in 1930 (DeLacy et al. 1972, Strickland 1983, Mills and Hermans 2007). In addition to direct support from tuition and the University, this early work was often supported by creative acquisition of foundation grants by faculty such as Thomas Thompson, and also by fairly unconstrained support from

sources such as the Office of Naval Research (see Weir 2001). By the late 1950s this work had catalogued much of the native biota, and characterized the basic bathymetry, circulation, and physical and chemical attributes of the water masses (McLellan et al. 1953-1954, Collias and Andreeva 1977), including a working physical model of Puget Sound still in use today (Lincoln 1952). But more often than not, academic work on Puget Sound has been "applied" science, focused on practical problems with support from a variety of economic and governmental interests. For example, Trevor Kincaid, who founded the Puget Sound Marine Station and was instrumental in the creation of the UW School of Fisheries and Department of Zoology, and was a pioneer in the study of zoology and marine biology in Puget Sound, received substantial support from commercial shellfish interests and government fisheries agencies. For much of its 100-year life, the UW School of Aquatic and Fishery Sciences (formerly the College of, Department of, and School of, Fisheries, and the first fisheries school in the US) focused on processing and marketing of fisheries products, harvest management of ocean fisheries resources, salmon ecology in Alaska (initiated by the salmon canning industry), and fish and bivalve culture for salmon hatchery production and commercial shellfish farming (Stickney 1989).

This applied orientation, and the general reductionist tendency in science, led to a persistent pattern—while Puget Sound was at the doorstep of the University of Washington, it was seldom the focus of teaching and research in its own right. A review of the course catalogues over the last century shows that few oceanographic, fisheries, or zoology classes explicitly focus on Puget Sound, and it appears as a subject keyword in only 5% of all 2274 graduate thesis and dissertation abstracts from the College of Ocean and Fishery Science (Schools of Aquatic and Fishery Science, Oceanography, and Marine Affairs). The oceanographers and fisheries scientists often either spent their time in the laboratory, or stepped over Puget Sound on their way to sea, to Alaska, or elsewhere. Consequently, the most advanced academic work at the University of Washington was often not focusing on the unique

ecological context of Puget Sound. As a result, the effects of the intensive human activity that was occurring in Puget Sound were less likely to be detected, understood, and corrected; when they were detected, narrow conceptions and approaches dominated assessment and management.

While much scientific information on the Puget Sound ecosystem exists, many large and critical gaps remain. We have mapped it, described its geology and basic circulation, and catalogued most of the biota, but have surprisingly poor understanding of the temporal and spatial patterns of biotic composition, to say nothing of the natural and anthropogenic forces that drive them.

Environmental Assessment

Ideally, environmental monitoring and assessment should directly measure the condition of ecosystems, diagnose causes of change, assist in the development of corrective actions, and evaluate the effects of management choices (NRC 1990, Karr 1991, Karr and Chu 1999, Downes et al. 2002, Karr 2006). Unfortunately, such tools are lacking in Puget Sound, in part because of scientific failings noted earlier and explored in more detail here. In this section, I briefly summarize desirable attributes of ecological monitoring and assessment, and then review a series of environmental programs conducted in Puget Sound relative to those key attributes. The review of key attributes of monitoring and assessment programs provides a framework for evaluating where previous and ongoing efforts fall short, and therefore provides a basis for identifying key information gaps and methods to fill them. Key attributes include both the breadth and ecological relevance of ecosystem components and stressors addressed by monitoring and assessment programs, as well spatial and temporal distribution of sampling efforts.

Ecosystem monitoring and assessment should emphasize direct measurement of biological attributes that are responsive to human influence (Karr and Chu 1999, Karr 2006), and assess condition based on difference from reference [e.g., departure from biological integrity (Karr 1991]), best defined as pre-disturbance or minimally

disturbed condition. Defining reference condition requires comprehensive natural history information on the regional biota. If not measuring biology directly (e.g., environmental measurements of physical or chemical characteristics), a sound rationale for the metric based on its biological significance is necessary (NRC 2000), and since direct measure of biology is always preferable to physical/chemical surrogates (Karr and Chu 1999), biological response variables should be sought. Consideration of ecologically relevant spatial and temporal scale is also essential (Weins et al. 1986, Levin 1992, Hughs et al. 2005). Finally, the data should be synthesized into useful information to guide policy (NRC 1990, Karr 2006).

Specific terminology for kinds of monitoring (NRC 1990, Downes et al. 2002, Rice et al. 2005, Roni 2005) varies depending on the context (e.g, restoration vs. fishery management) but monitoring efforts generally fall into three main areas: 1) status and trend (tracking changes in ecosystem attributes over time); 2) diagnostic (understanding the cause of ecological changes); and 3) effectiveness (evaluating the success of management actions). The limited biological monitoring conducted in Puget Sound is nearly all status and trend monitoring (PSAT 2007b); improved diagnostic and effectiveness monitoring are pressing needs for Puget Sound management (PSAMP Management Committee 2005, Gelfenbaum et al. 2006, PSP 2006).

Human activities affect ecosystem parts and processes in many complex and interrelated ways. This gives rise to biological responses, which, despite their complexity, are often readily and predictably observed as diverse attributes of the biota (Figure 1.3; Karr 2006). It follows then, that information on the numbers and kinds of organisms collected at ecologically appropriate temporal and spatial scales (Weins et al. 1986, Levin 1992) across a range of human influence, is most informative in evaluating ecosystem condition (Karr and Chu 1999, Karr 2006). Using such information to develop ecological dose-response relationships (Figure 1.4; Karr 2006) can assist in building effective monitoring and assessment tools. Without those tools we cannot track the changing condition of living systems,

identify or diagnose the causes of degradation, or track system condition following management decisions to determine if they are actually reversing downward trends. Collectively, this situation provides ample opportunity to spend money unwisely on recovery and restoration plans that simply do not work.

Human influences
(L = local; R = regional; G = global)

Diking and filling of wetlands (L)

Fisheries (L, R, G)

Shoreline armoring/vegetation removal (L)

Addition of alien taxa (L, R, G)

Artificial propagation of organisms (L, R, G)

Land use in watersheds (L, R)

Water appropriation and regulation in watersheds (L, R)

Vessel and foot traffic (L, R)

Dredging (L)

Generation of chemical pollutants & pathogens (L, R, G)

Altered ecosystem features

Physical environment

Biotic interactions

Energy & nutrients

Chemical environment

Biological responses
(Animals)

Declining diversity within feeding and reproductive guilds

Declining diversity of food specialists

Increase in omnivores

Loss of top predators

Loss of migratory species

Loss of sensitive taxa

Declining life history diversity within species and populations

Altered size/age structure within species and populations

Altered geographic range & seasonal distributions of taxa and life history types

Increased frequency of tumors, lesions, etc.

Decline of harvestable species and biomass

Figure 1.3. Schematic representation of how multiple human influences alter features of Puget Sound, resulting in likely changes in biota. (Adapted from Karr 2006).
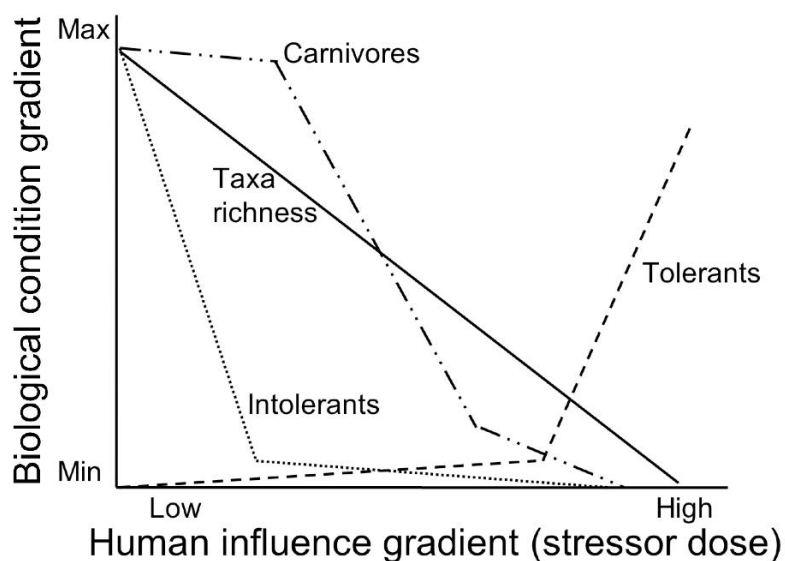
Figure 1.4. Ecological dose-response curve showing several commonly observed changes in biota along gradients of human influence. (From Karr 2006).

Environmental Assessment in Puget Sound

Aside from the declines in Pacific salmon and shellfish abundance that were apparent by the early 1900s, concerns about the effects of human activity on the Puget Sound ecosystem did not begin until the 1960s and 70s, when declines in some commercial and recreational fisheries became obvious and national and international environmental awareness expanded. Like most early environmental efforts, Puget Sound programs tended to focus on water quality and chemical pollutants. This growing awareness led to increasing point source environmental regulation and studies of environmental impacts from accidents such as oil spills (e.g., Long 1982), or the construction and operation of pubic works projects such as wastewater treatment (e.g., English and Thorne 1977) and power generation facilities (e.g., Stober and Salo 1973). Consequently, much of the ecological study of Puget Sound has not focused on the whole ecosystem, or on the broad array of human stressors that affect its condition, and has not been sustained over long time periods.

The best example of this is the Marine Ecosystem Analysis (MESA) Program of the late 1970s and early 1980s (Long 1982). MESA was a national program created by the National Oceanic and Atmospheric Administration (NOAA) to "…focus scientific research upon specific problems related to human activities in estuarine and coastal environments" (NOAA 1978). The first system studied was the New York Bight, and emphasized chemical pollution. Puget Sound was the second system studied, and while the Puget Sound MESA project planners sometimes spoke in broad terms about the full array of human influences (consistent with major revisions of the Clean Water Act in 1972 that broadened the definition of pollution), the specific language always emphasized chemical contaminants (NOAA 1978). A main driver of the Puget Sound MESA Program was the proposal to open up greater Puget Sound to supertanker traffic. A small portion of the work was done in Puget Sound proper studying basic circulation (primarily to understand the transport and dilution of pollutants) and conducting laboratory toxicity tests. The majority of work focused on cataloguing the geographic and seasonal composition and food web relationships of the biota in the Strait of Juan de Fuca and the San Juan Islands, to establish "baseline" conditions that could be used to evaluate ecosystem damage in the event of a major oil spill. These "north Puget Sound" MESA studies included plankton; demersal, littoral, and neritic fishes; benthic macroinvertebrates; marine birds and waterfowl; and marine mammals. Much of the work was characterized by considerable depth and detail. The fish studies, for example, collected complete taxonomic, abundance, and size data from the catch, but also analyzed the diets of most fish species encountered (Miller et al. 1980). Diet information was combined with existing data to produce detailed conceptual models of food web relationships for the many environments of the ecosystem (Simenstad et al. 1979). MESA work emphasized consistent application of sampling protocols, and full compatibility, reporting, and archiving of data. But because the emphasis was on "baseline" conditions to evaluate supertanker traffic and oil refining in the Straits, most of the

work was done outside of Puget Sound proper; the entire project spanned five years but individual elements collected data for no more than three years.

The MESA studies left us better prepared for municipal waste handling and catastrophic oil spills (that thankfully have not happened, in no small part because of the deft legislative maneuverings of Senator Warren Magnuson who effectively blocked the supertanker traffic in Puget Sound [Chasan 1981]), and provided some of the most comprehensive natural history information ever collected in the region (see references in Long 1982). But what if the north Puget Sound MESA studies had been expanded to all of Puget Sound (and better still, the Georgia Basin as well), broadened to focus on the full array of human influences, continually refined to improve sampling designs and address emerging management and research needs, and the results regularly synthesized and communicated to policy makers and the public? We would have today what we need and do not have—a much more comprehensive, coherent, accurate, and shared understanding of the Puget Sound ecosystem, and much better monitoring and assessment tools.

Before and after MESA, many separate projects have focused on a limited number of single stressors, and were typically narrow in temporal, spatial, and ecological scope. The individual studies satisfied their project obligations and ended, and the researchers moved on to other funding sources and topics. These efforts undoubtedly produced useful information about their specific topics, but the scientific results rarely reached the peer-reviewed literature or the general public, and were seldom integrated into broad syntheses.

Several programs are partial exceptions to this fragmented, project-oriented picture. The largest is the Puget Sound Assessment and Monitoring Program (PSAMP, and until early 2007 named the Puget Sound Ambient Monitoring Program). PSAMP is a component of the Puget Sound Water Quality Authority (PSWQA), created by the State Legislature in 1983. The stated purpose of PSWQA is to "develop a comprehensive plan for water quality protection in Puget Sound to be implemented by existing state and local government agencies" (RCW 90.70.001).

Despite this explicit water quality focus in the legislation, PSWQA was also assigned broader tasks, including "… assessment of the Sound's resources," and "… the protection, preservation, and restoration of wetlands, wildlife habitat, and shellfish beds" (PSWQA 1987). At its creation, PSWQA recognized the fragmented and patchy nature of Puget Sound monitoring and assessment (PSWQA 1988), and created PSAMP in 1988 as a "comprehensive environmental monitoring program for Puget Sound" (PSWQA 1987). PSAMP would coordinate existing efforts, create new ones where necessary, integrate the results, and communicate them to policy makers and the public. It is the largest environmental assessment program ever undertaken in Puget Sound, and despite its laudable goals, it shares many of the common problems of earlier efforts. While the monitoring is ostensibly supposed to have broad geographic and ecological scope, it has been heavily oriented toward water quality and chemical contaminants, often limited in spatial and temporal extent and resolution, and has paid little attention to major portions of the ecosystem (e.g., the pelagic zone) and not included crucial components (e.g., plankton) and attributes (e.g., trophic status) of the biota. Content of reports (Figure 1.5) of PSAMP over the years illustrates the subject emphasis of the program.
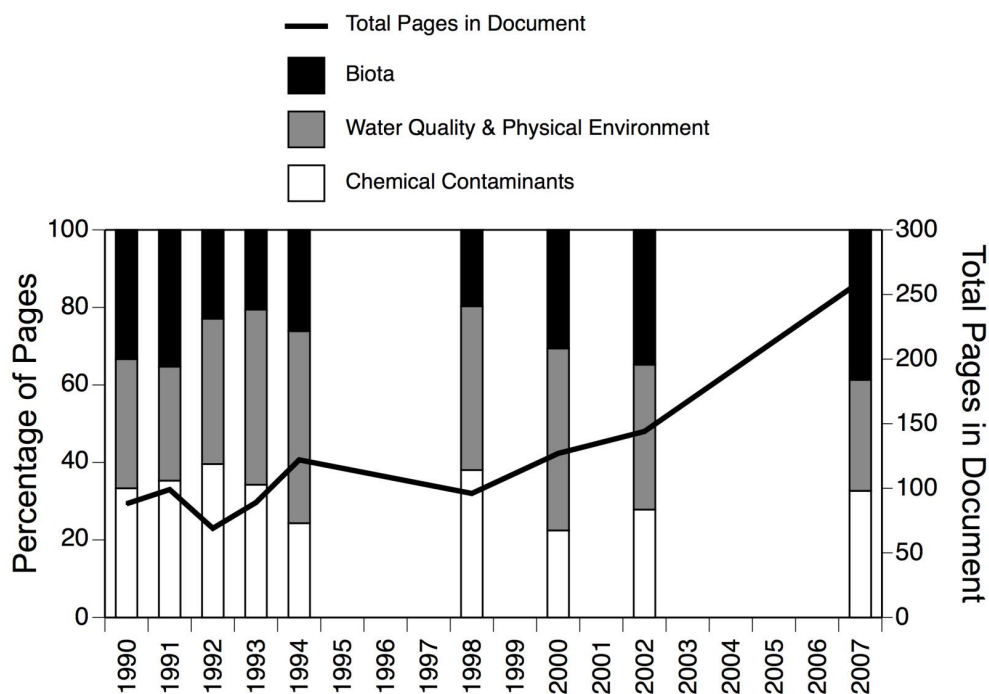
Figure 1.5. Percentage of pages in the Puget Sound Assessment and Monitoring Program's Puget Sound Updates devoted to chemical contaminants, water quality and physical environment, and the biota (non contaminant). Black line represents the total number of pages in the document. Although the total discussion and percentage of the document devoted to the biota has expanded in the most recent edition, overall emphasis continues to be on chemical contaminants and water quality.

Many of the PSAMP metrics (see PSAT 2007b) focus on physical and chemical properties of water, sediment, and biota, and only a few (e.g., incidence of liver lesions in flatfish, assemblage composition in benthic invertebrates) actually attempt to evaluate biological responses to such phenomena. When biological measurements are not focused on toxicology (in demersal fish, marine bird, and marine mammal programs, for example), they focus primarily on abundance estimates of single species over time. Little multispecies analysis has been done, and little attempt has been made to relate changes in the overall composition of the biota to human

influences. Few of the individual elements are optimized for statistical power, or integrated with other elements of the program.

The first comprehensive review of PSAMP (Shen 1995) pointed out these and other deficiencies. Responding to criticisms that the program lacked a comprehensive ecosystem view and did not integrate its components well, PSAMP began development of a conceptual model for the monitoring program (Newton et al. 2000), but the actual components and operation of the program changed little. Another review (less comprehensive and involving fewer outside reviewers) occurred in 2005 and again pointed out problems with scope and integration, as well as poor diagnostic and performance monitoring capabilities. Nevertheless, PSAMP gave itself high marks for "delivering data and analysis for the assessment of the health of Puget Sound" even though little had changed about the program since its last review (PSAMP Management Committee 2005). The lack of even a qualitative explanation of the overarching goal of "health," and omission of most of the primary biological PSAMP components (marine birds and waterfowl, marine mammals, and demersal fishes) from the 2005 review is indicative of the lack of a coherent, biologically- and human influence-focused framework for Puget Sound monitoring and assessment. While the most recent summary report on PSAMP (PSAT 2007b) has a much more comprehensive listing of biotic components and ongoing and future directions for monitoring and research, it still fails to integrate well, or explain the causes (especially the human ones) of most of the "status and trend" information it reports.

This is a natural consequence of the approach to data collection and analysis taken by PSAMP—a fragmented collection of efforts that rarely attempts to relate a broad range of human and natural environmental influences explicitly to biological responses. A comprehensive diagnosis of ecosystem condition remains to be completed, and as a result, management priorities (PSAT 2007a) are set in the absence of carefully framed scientific analyses, and it is unclear whether either assessment or restoration efforts are focused on the most important stressors. For

example, as in most estuarine (Lotze et al. 2006), and freshwater ecosystems (Karr and Dudley 1981b, Karr 1991, 2006), chemical contaminants are not likely to be the primary cause of most of the declining populations of Puget Sound fish and wildlife (West 1997), yet the majority of funds for addressing environmental problems are devoted to contaminant and water quality issues (PSP 2006, PSAT 2007a). Other potential stressors, including hatcheries (ostensibly a solution and dependent on major public support), may be aggravating some of these problems but receive little or no study.

The most comprehensive current effort to understand the oceanography of the whole system came from a grant competition at the University of Washington (and not at the direction of the designated stewards of the ecosystem). The Puget Sound Regional Synthesis Model (PRISM) was established in 2001 with goal of defining "…the movement of water across the air-, land- and seascape of the Puget Sound basin. Our goals are to predict the dynamics of water and its associated biology and chemistry under natural and anthropogenically modified conditions, and to convey this information to multiple users in a timely fashion." But while PRISM has produced impressive visual representations of the physical environment and some measures of human disturbance in the landscape, it has been slow to develop working ecosystem models, particularly those relating human activity and natural physical changes to biological responses in the Puget Sound ecosystem. Field sampling efforts to help refine models collect little biological data.

Another ecosystem science effort to emerge outside of PSAMP is the Puget Sound Nearshore Restoration Program (PSNERP). Addressing the lack of programs explicitly focused on restoration of the "nearshore" (an inconsistently used term that generally refers to shoreline environments from the backshore and bluffs to the lower extent of the photic zone), the US Army Corps of Engineers created the program to "Restore and protect the nearshore habitat of Puget Sound for the benefit of the biological resources and the integrity of the ecosystem, including the functions and natural processes of the basin" (PSNSERP 2004). Beginning with much promise for

comprehensive ecosystem restoration on the scale of efforts in the Everglades and the Chesapeake Bay, major funding cuts occurred by the second year of the initial planning phase of the program, reducing its scope and slowing its progress. Despite the setbacks, PSNERP has produced some of the most thorough conceptual work on the Puget Sound ecosystem (e.g., Simenstad et al. 2006) and is now beginning to develop priorities and plans for actual projects, and plans to develop specific monitoring tools to evaluate restoration success.

Periodically, efforts emerged that attempted to summarize knowledge about the character and health of the ecosystem (Wilson et al. 1994, West 1997, Ruckelshaus and McClure 2007). But as well intentioned as these efforts were, they too often fail to distinguish between what is actually supported by data and what is not. Still not rigorously demonstrated are many presumed relationships between human activity and biological condition. Several examples follow.

Shoreline alteration is often cited as a known threat to the Puget Sound ecosystem yet the ecological consequences are poorly understood and documented. One common narrative is that shoreline modification can reduce abundance of small pelagic fishes through degradation of spawning habitat, which in turn can reduce food for salmon, which in turn can reduce food for whales. While this is logical reasoning (i.e., beach spawning habitat is a requirement for dominant pelagic fish species and complete loss of it would eliminate that portion of the salmon prey base), the implication is that shoreline armoring is known to affect whale populations and that restoring armored shorelines will help recover those populations, yet this set of relationships has not been established. We simply do not know if armoring has significantly affected populations of beach-spawning pelagic fishes, and until we evaluate that relationship we risk misdiagnosing causes of the perceived changes in biology, and not addressing true causes of declines.

Shoreline armoring is also often cited as an adverse influence on eelgrass in Puget Sound (Thom and Shreffler 1994, PSP 2006, Puget Sound Action Team 2007b, Ruckelshaus and McClure 2007) but little supporting data exists. And when a Sound-

wide monitoring program was established in 2000 as part of PSAMP (Gaeckle et al. 2007), it did not explicitly include armoring in the sampling design. If it had been included, the program might have been able to say something about this critical issue in the first year of the study, rather than having to wait for a time series of multiple years to even have a chance at evaluating such effects. After seven years the program still has nothing to say about the effects of shoreline armoring on eelgrass ecosystems, and little to say about other potential anthropogenic stressors, although some additional focus studies are being done (Gaeckle et al. 2007). The one current research project that is addressing the effects of shoreline armoring on eelgrass is being done independent of PSAMP, on a shoestring budget in Hood Canal (Simenstad et al. 2007). Without understanding these relationships, we are less able to identify priorities for conservation and restoration, or track the effectiveness of management actions such as restrictions on shoreline development, or restoration of degraded shorelines.

Another widely cited but poorly documented issue in Puget Sound (PSP 2006, PSAT 2007b, Ruckelshaus and McClure 2007) is perceived changes in the condition of populations of several small pelagic fishes: Pacific herring (*Clupea harengus pallasi*), surf smelt (*Hypomesus pretiosus*), Pacific sand lance (*Ammodytes hexapterus*), and northern anchovy (*Engraulis mordax*). Often narrowly conceived of as "forage fish" they are the focus of much concern and speculation in Puget Sound (for example, that declining populations of these small pelagics may be related to declines in many of their fish, bird, and mammal predators), yet we have very limited understanding of the biology of these species, especially with respect to human activity. Small pelagic fishes exert major bottom-up and top-down control in pelagic food webs, particularly in upwelling ecosystems (ASG 1997, Cury et al. 2000). Environmental tolerances, biogeography, and demographics of Puget Sound populations—fundamental in understanding the role of these species in the Puget Sound ecosystem—are poorly understood in herring, and barely known at all in surf smelt, sand lance, and anchovy. Instead, only spawning aggregations of adult herring

are targeted in acoustic surveys, and estimates of spawning biomass are calculated from roe surveys (Stick 2005). No information is collected on adults (or other life stages) during the rest of the year. Surf smelt, sand lance, and anchovy are not monitored at all in Puget Sound, except for limited presence or absence surveys of beach spawn in surf smelt and sand lance (e.g., Penttila 1995), and some recreational catches of surf smelt (Lemberg et al. 1997). Further, research and monitoring on the structure (e.g., composition of plankton and fish assemblages) and dynamics (e.g., seasonal productivity) of pelagic food webs across Puget Sound, and their natural and anthropogenic influences, is virtually absent. Continued failure to improve our knowledge of the effects of natural and human influences on these dominant pelagic species impairs efforts to prudently manage them, and the many other species that interact strongly with them.

Serious problems also exist with monitoring and assessment of other components of the biota. For example, the only non-fishery survey of demersal fishes comes from WDFW bottom trawl surveys, and only a fraction of the data are reported or made available, usually in the form of brief and infrequent agency reports (e.g, Palsson et al. 1997, PSAT 2007b), and emphasizing abundance trends of selected single species. Surveys of marine birds and waterfowl (see Chapter 5) are conducted throughout greater Puget Sound but again, analysis is of abundance estimates of singe species over time, and effects of human activity are not explicitly considered. And while reduced abundance and altered spatial distributions of several marine mammal species have been noted (PSAT 2007b), the overwhelming emphasis of Puget Sound surveys has been on harbor seals (*Phoca vitulina*), one of the most abundant and tolerant species.

Conclusion

The environmental history of Puget Sound leaves us with a heavily altered ecosystem, and poor understanding of the natural and anthropogenic factors that determine its biological character. Such understanding is critical for diagnostic and

performance monitoring and assessment in Puget Sound, and can be developed through research that provides new data and introduces alternative analytical approaches. The next four chapters are a combination of both, and are examples of several kinds of research historically neglected in Puget Sound: documentation of ecological effects of human activity, basic natural history of highly valued species in a presumed critical habitat, the combination and reanalysis of historical and ongoing monitoring and assessment efforts, and exploratory analysis to identify potential monitoring and assessment metrics. The simplest study (Chapter 2) provides basic information on ecological effects of shoreline modification, a pervasive disturbance widely cited as a major ecological concern but barely studied. A much larger field study (Chapter 3) examines the use of pelagic surface waters across Puget Sound by wild Chinook salmon, one of the region's most imperiled and iconic species. Seasonal and geographic patterns of density and size are compared with those of the hatchery fish that are manufactured to compensate for depleted wild Chinook production, and to aid in wild Chinook recovery despite scant consideration of adverse effects in estuarine environments. In addition, biomass data on fish and gelatinous zooplankton collected opportunistically as part of the salmon study are used to explore seasonal and geographic patterns in the macrofaunal composition of nearshore surface waters (Chapter 4). These mid-level consumers are a major factor in determining the overall biotic character of coastal ecosystems and are responsive to a variety of human influences but are poorly understood in Puget Sound. The final project (Chapter 5) combines several existing data sets to examine interannual, seasonal, and geographic patterns in the composition of marine bird and waterfowl assemblages, specifically with respect to urbanization. Chapter 6 revisits the key results of the four data chapters in the context of the Puget Sound science problems identified in this chapter, and offers a set of recommendations for monitoring and research in Puget Sound. Together these studies enrich the ecological narrative of Puget Sound, and assist in the development of improved tools for evaluating the health of Puget Sound.

CHAPTER 2

**Effects of Shoreline Modification on a Northern Puget Sound Beach:
Microclimate and Embryo Mortality in Surf Smelt (*Hypomesus pretiosus*)**

Summary

Human alteration of Puget Sound shorelines is extensive yet its ecological consequences are largely undocumented. This study evaluates differences between a natural and a heavily modified beach in terms of microclimate and one aspect of biological condition. Electronic data loggers were placed at a tidal height of approximately 3.7 m (12 ft) above mean lower low water (MLLW) during the period of July 16-20, 2001, to monitor light intensity, substrate and air temperature, and humidity. Substrate samples were collected at the end of the monitoring period to evaluate condition and density of eggs from surf smelt (*Hypomesus pretiosus*), a small pelagic species that spawns on gravel- sand beaches in the upper intertidal zone. The modified beach had significantly higher daily mean light intensity, air temperature, and substrate temperature, and significantly lower daily mean relative humidity. Particularly striking were differences in substrate temperature, which, on the natural beach, ranged from 12.1°C to 18.2°C (mean = 14.1°C), and on the modified beach ranged from 14.4°C to 29.4°C (mean = 18.8°C). In addition to these different means and more extreme values, microclimate conditions on the modified beach were more variable, indicative of a less "buffered" environment. The proportion of smelt eggs containing live embryos on the altered beach was approximately half that of the natural beach.

Introduction

The nearshore ecosystems of Puget Sound are crucial in the life cycle of many fish and wildlife species (Simenstad et al. 1979; Kozloff 1983; Simenstad 1983; Phillips 1984; Kruckeberg 1991). They also are subjected to many human influences, including shoreline armoring and removal of terrestrial vegetation, two of the most

prevalent and severe anthropogenic disturbances in the region (Williams and Thom 2001; PSWQAT 2002). While little is known about the ecological consequences of anthropogenic shoreline modification in Puget Sound, available studies suggest impairment (Thom and Shreffler 1994; Levings and Jamieson 2001; Penttila 2001; Romanuk and Levings 2003; Sobocinski 2003). This study examines effects of shoreline modification on beach microclimate (light, air and substrate temperature, and humidity) and one aspect of biological condition (embryo mortality in an intertidally spawning fish). The implications of shoreline alteration on the overall ecology of nearshore Puget Sound, including cumulative effects at the landscape level, are briefly considered also.

Generally, nearshore environments include the region between the lower extent of light penetration on the seaward side of the shoreline, and on the landward side, the extent of direct interaction in the form of sediment supply from adjacent bluffs, or shading and bank stabilization by terrestrial vegetation (Williams and Thom 2001); that is, the sublittoral photic, littoral, and supralittoral zones combined. As an ecotone between terrestrial and aquatic estuarine ecosystems, the nearshore performs a number of distinctive ecological functions including the generation, accumulation, and decomposition of detritus that can be an important part of estuarine and terrestrial food webs (Day et al. 1989; Polis and Hurd 1996; Colombini and Chelazzi 2003; Dugan et al. 2003); and as foraging, spawning, rearing, and migration habitat for a rich variety of organisms (Day et al. 1989, Brennan and Culverwell 2004). Many of these species are recreationally and commercially important. In Puget Sound these include five species of Pacific salmon (*Oncorhynchus spp.*) and four species of anadromous trout (*Oncorhynchus spp.* and *Salvelinus spp.*) comprising many distinct population segments, three of which (Puget Sound Chinook salmon (*O. tshawytscha*), Hood Canal summer chum salmon (*O. keta*), and bull trout (*S. confluentus*))*,* are listed as threatened under the Endangered Species Act (Johnson et al. 1997; Myers et al. 1998; USFWS 1999). Another salmonid (Puget Sound coho (*O. kisutch*)) is proposed for listing (Weitkamp et al. 1995).

Puget Sound shorelines have experienced major impacts as result of human activity, including diking, filling, armoring, and vegetation removal for purposes of agricultural, industrial, and residential development. Indeed, at least 1/3 of the linear shoreline has been modified by humans (Puget Sound Water Quality Action Team 2002). Physical effects of these activities (hardening and deepening of the shoreline, loss of structural complexity, and loss of connectivity between aquatic and terrestrial environments) occur immediately and can persist for decades (Macdonald et al. 1994; Williams and Thom 2001). Long-term effects are largely the result of changes in sediment dynamics and the reduction or elimination of the supply of organic matter from the adjacent terrestrial environment. The substrate can coarsen, the beach slope steepen, and the structural complexity and organic debris accumulation decline (Macdonald et al. 1994; Williams and Thom 2001).

Because physical shoreline conditions are a primary influence on biological processes (Ricketts et al. 1985; Day et al. 1989; Ricklefs and Miller 2000; Knox 2001), extensive physical modifications can potentially affect species composition, abundance, and distribution; the flow of nutrients and organic matter; and many other factors. Anthropogenic alterations of freshwater riparian ecosystems have such effects (Kelsey and West 1998; Naiman et al. 1998). While little empirical evidence exists on the ecological consequences of shoreline modification in Puget Sound, available information suggests significant adverse effects can occur, including loss or degradation of spawning substrate and food resources for fishes (Thom and Shreffler 1994; Levings and Jamieson 2001; Penttila 2001; Brennan and Culverwell 2004). One specific example is the reduction in taxonomic richness and abundance of invertebrate assemblages on armored beaches when compared to natural ones (Romanuk and Levings 2003; Sobocinski 2003). Such effects are likely to be partially due to changes in microclimate, the local suite of climatic conditions near the ground (Geiger 1965; Brosofske et al. 1997; Chen et al. 1999). Drastic changes in light, thermal, and moisture conditions can have severe biological consequences (Ricklefs and Miller 2000). On estuarine shorelines, for example, removal of

overhanging supralittoral vegetation increases beach exposure to sunlight, thereby increasing temperature and evaporation, and drying out beach environments. Reductions in structural complexity and accumulation of organic debris on altered beaches may reduce capacity for water retention, further contributing to drying. Given the sensitivity of intertidal organisms (Pugh and Macalister 1994; Rafaelli and Hawkins 1996) and geochemical processes (Valiela 1995) to drivers such as temperature and moisture conditions (Jedrzejczak 2002), microclimatic changes from physical disturbance at the shoreline could influence the distribution and behavior of organisms, and the flux of energy and material in the nearshore. Unfortunately, these and other potential effects of anthropogenic changes in estuarine and coastal marine shoreline microclimate are poorly documented.

Shoreline Modification and Surf Smelt Spawning

One major concern regarding shoreline modification is adverse effects on essential nearshore fish habitats. Intertidal beaches provide spawning habitat for a number of fish species (DeMartini 1999; Martin and Swiderski 2001). Potential benefits of intertidal spawning include refuge of embryos from aquatic predators, and increased oxygenation and rate of development of embryos while not immersed by the tide. Potential risks of intertidal spawning include time and energy expended during tidal migration of spawning adults, exposure of spawning adults and embryos to terrestrial and avian predators, and physiological stress of embryos while not immersed. Embryos of intertidal spawning fishes often have broad thermal tolerances and plasticity in incubation duration that allow them to persist in the variable and harsh intertidal environment (DeMartini 1999; Smyder and Martin 2002). However, environmental extremes can be hazardous to developing embryos. Some upper-intertidal spawning species protect embryos from thermal and desiccation stress by burial in the substrate, or deposition in beach debris or abandoned burrows of other organisms (Middaugh et al. 1983). Similarly, intertidally spawning fishes in Puget Sound may, to some degree, depend on shade and debris in

the upper-intertidal to protect their incubating embryos. Anthropogenic changes in shoreline microclimate will change the intertidal incubating environment, potentially altering developmental rates or increasing physiological stress in fish embryos.

In Puget Sound, most concern in this regard is focused on the surf smelt (*Hypomesus pretiosus*), a recreationally important pelagic fish that is a common food item for many fish and wildlife species. Surf smelt occur throughout Puget Sound and spawn on upper-intertidal, gravel-sand beaches (Schaefer 1936; Thompson et al. 1936; Loosanoff 1937). Except for Pacific sand lance (*Ammodytes hexapterus*), which also spawn on upper-intertidal beaches but only during winter months, surf smelt are unique among Puget Sound fishes in their obligate spawning use of these habitats. Surf smelt spawn at various times of year, but in much of northern Puget Sound, summer is the peak spawning season (Loosanoff 1937; Penttila 1973; Penttila 1978; Penttila 1995). Spawning typically occurs on extreme high tides during evening or night. At spawning, eggs adhere to substrate particles and incubate for approximately 10 to 21 days before hatching when immersed and agitated by tidal inundation.

Temperature and moisture conditions of the substrate influence the survival and rate of development in surf smelt and other intertidal spawners (Yap-Chiongco 1941; Frank and Leggett 1981a; DeMartini 1999; Smyder and Martin 2002; Chris Lee personal communication). Because of warmer weather and higher light levels during the summer months, it has been postulated that smelt embryos incubating on intertidal beaches in summer would be most exposed to excessive thermal stress and desiccation, especially at armored spawning beaches that have no terrestrial vegetation (Schaefer 1936; Penttila 1973). Historical data from Puget Sound summer beach spawn surveys showed that beaches without terrestrial shoreline vegetation had significantly lower proportions of live smelt embryos (Figure 2.1; Penttila 2001). Thermal stress and desiccation were proposed by Penttila (2001) as the cause of lower embryo survival on the unvegetated beaches; however, no detailed assessment

of the physical environment was made, nor was anthropogenic alteration of beaches an explicit factor in the analysis.
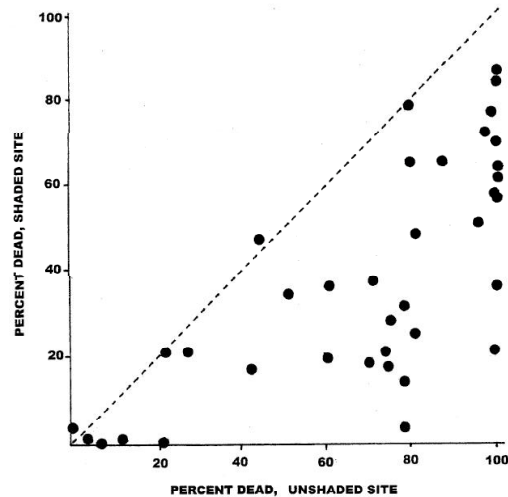
Figure 2.1. Scatter diagram of surf smelt embryo mortality at 37 sets of paired shaded and unshaded beaches. From Penttila (2001); reprinted with permission.

To evaluate the potential effects of shoreline modification on summer shoreline microclimate and surf smelt embryo survival, I compared an armored beach with no terrestrial shoreline vegetation to an unarmored, naturally vegetated beach. The specific null hypotheses tested were: 1) there were not significant differences between the natural and modified beach during periods of sunny summer weather in terms of light intensity, substrate temperature, air temperature, and relative humidity; and 2) surf smelt egg density and proportion of eggs containing live surf smelt embryos would not be significantly lower on the modified beach.

Methods

Study Sites

   The two adjacent beaches used in this study are located at the northern end of
Camano Island, Washington (Figure 2.2), a known area of summer surf smelt
spawning (Loosanoff 1937; Penttila 1995). Similar to much of Puget Sound, the
shoreline in the study area consists primarily of unconsolidated glacial material with
mixed sand-gravel beaches and upland banks that, under natural conditions, are often
forested with a mix of conifers and deciduous hardwoods, depending on the slope
and exposure (Downing 1983; Kruckeberg 1991). One beach has no overhanging
terrestrial vegetation and is armored with a vertical concrete bulkhead at
approximately + 4 m (13 ft.) relative to mean lower low water (MLLW). The
adjacent beach is not armored and has extensive terrestrial vegetation dominated by
mature big leaf maple (*Acer macrophyllum*). Study sites were monitored during the
period of July 16-20, 2001, likely the early or middle part of the spawning season
(Loosanoff 1937), and a period of sunny but not exceptionally hot weather for
summer in the region.
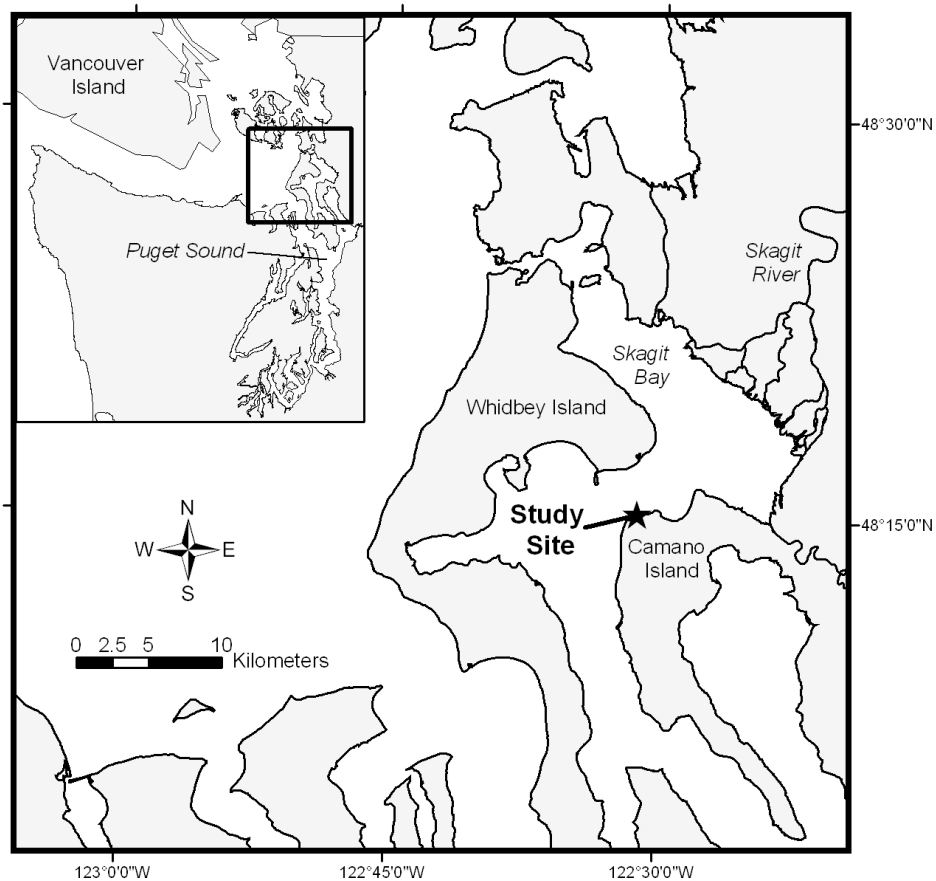
Figure 2.2. Map of study location on Camano Island in northern Puget Sound,
 Washington. Adjacent modified and unmodified (one of each), north-
 facing beaches were used.

Data Collection

At one location in the middle of each beach at a tidal elevation of approximately

+3.7 m (12 ft) MLLW, electronic data loggers were installed at the start of the five-

day monitoring period. Combination temperature and relative humidity loggers

(Onset HOBO® model H08-032-08) enclosed in radiation shields (Onset model RSI)

were mounted at 1-m above the ground on 1.9-cm galvanized pipes hammered into the substrate. Light intensity loggers (Onset HOBO® model HLI) enclosed in plastic petri dishes were glued to the tops of the radiation shields, and digital temperature loggers (Onset HOBO® model H01-001-01) in white, waterproof cases were pressed into the substrate and tied to the base of the mounting poles so that they were flush with the substrate surface. All data were recorded at 5-minute intervals. After five days all loggers were removed from the beaches, and the data downloaded and separated into day (sunrise to sunset) or night (sunset to sunrise) groups. Due to incomplete data collection on the first day, only four complete sets of data from day and night periods were analyzed.

At the end of the study period, five surficial (top 3 cm) substrate samples were collected at the same tidal elevation as the loggers. Glass 4 oz jars were used to scoop bulk substrate samples at approximately 2 m intervals along a transect parallel to the shoreline and centered on the logger locations. These samples were preserved in Stockard's solution (5% formaldehyde and 4% acetic acid). One subsample (0.5 cm deep layer of sediment in 8 cm diameter Petri dish) from each replicate substrate sample was then examined under a dissecting microscope for counts of total, "live," and "dead" smelt eggs. Eggs were considered dead if they were opaque, obviously desiccated, broken open, or no intact embryo was visible in the egg.

Data Analysis

Statistical analyses consisted of pairwise (by day), two-tailed t-tests on daily minimum, maximum, and averages of the environmental variables, and unpaired, one-tailed t-tests on smelt egg data. One-tailed tests on the egg data were justified based on the results of Penttila (2001), that showed a significantly lower proportion of live versus dead embryos on unshaded beaches. To reduce the effects of nonnormal data distribution and heteroscedasticity, egg count data (eggs cm^{-3}) were transformed using the log (x + 1) transformation, and proportion live data were

transformed using an arcsine square root transformation (Zar 1996). Significance criterion was set at a ≤ 0.05.

Results

Three of the four physical variables measured (substrate temperature, air temperature, and relative humidity) were different between the two beaches for nearly the entire study period, even at night (Figure 2.3). The obvious exception was light intensity, which was higher on the altered beach during the day but showed no difference between the two beaches at night due to the absence of sunlight. The observed differences suggest that enough heat was absorbed by the altered beach during the day to maintain higher substrate and air temperatures through the night. It is impossible to know to what extent the lower humidity on the altered beach is the result of these higher temperatures or by the lack of vegetation but it is probably a combination of both. Daily mean values for all four physical variables were significantly different ($p \leq 0.04$; Table 2.1). The altered beach had significantly higher daily maximum light intensity, significantly higher daily maximum and minimum substrate temperature, significantly higher maximum daily air temperature, and significantly lower daily minimum relative humidity (Table 2.1, Figure 2.3). Particularly striking were the differences in substrate temperature (Figures 2.3 and 2.4), where grand mean and nightly minimums were approximately 2° C higher on the altered beach but peak daytime values averaged nearly 11° C higher. Maximum substrate temperatures approached 30° C on the altered beach but always remained less than 20° C on the natural beach. In addition to more extreme values and different means for all physical variables, the altered beach showed broader distributions (Figure 2.4) indicative of a more variable environment.
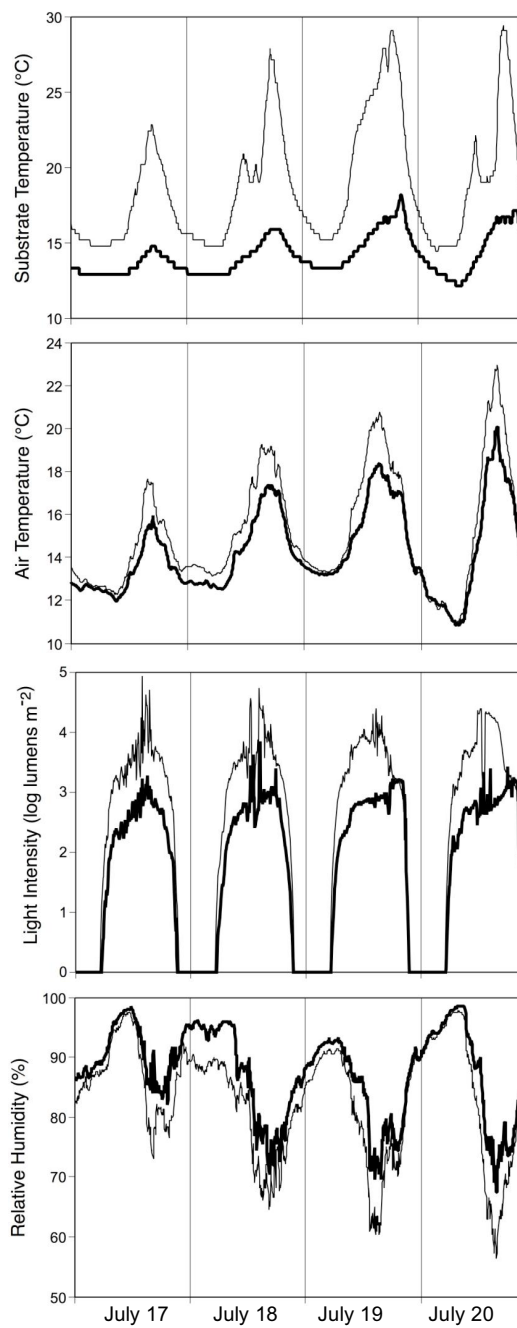
Figure 2.3. Light intensity, substrate temperature, air temperature, and relative humidity at the natural (thick line) and modified (thin line) beach over four sunny days in July, 2001.

Table 2.1. Average (± sd) daily minimum, maximum, and mean values for physical measurements over four days at the natural and modified beach. Asterisk (*) indicates statistically significant difference between the natural and modified beach at p ≤ 0.004 (n = 4).

	Light Intensity (log lumens m^{-2})		Substrate Temperature (°C)		Air Temperature (°C)		Relative Humidity (%)	
	Natural	Modified	Natural	Modified	Natural	Modified	Natural	Modified
Daily Minimum	-1.8 ± 0.0	-1.9 ± 0.0	12.8 ± 0.5*	14.8 ± 0.3*	12.1 ± 1.0	12.4 ± 1.1	72.4 ± 6.6*	63.6 ± 7.2*
Daily Maximum	3.5 ± 0.3*	4.6 ± 0.3*	16.5 ± 1.5*	27.3 ± 3.1*	18.0 ± 1.7*	20.2 ± 2.3*	96.7 ± 2.5	94.3 ± 3.9
Daily Mean	1.2 ± 0.3*	1.8 ± 0.3*	14.1 ± 0.6	18.8 ± 1.5*	14.3 ± 0.8*	15.2 ± 0.9*	87.5 ± 2.4*	83.0 ± 3.0*
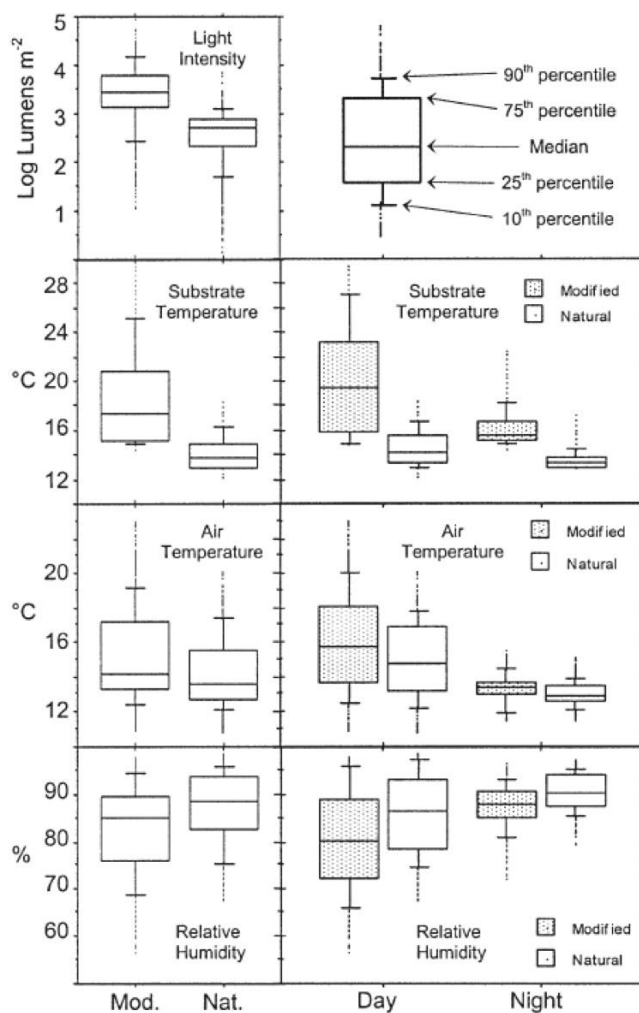
Figure 2.4. Daytime light intensity, and combined day and night substrate temperature, air temperature, and relative humidity at the modified and natural beach over the entire 4 day study period (left column); and during day and night hours (right column). In right column, black boxes are for the modified beach; white boxes are for the natural beach. Box plots show 10th, 25th, 50th (median), 75th, and 90th percentiles.

Both the proportion of eggs containing live embryos and total egg density at the altered beach were approximately half that of the natural beach (Table 2.2). Differences in mean proportion of eggs with live embryos were statistically significant (p = 0.048) but differences in total egg density were not (p = 0.18), probably as a result of very low statistical power ($1 - \beta = 0.26$ for total density) due to the high variability inherent in the patchy distribution of eggs, and the small sample sizes. Sample size power analysis indicated that increasing sample size to at least 11 samples per beach would be required to achieve conventional power of 0.8.

Table 2.2. Surf smelt egg density per sample and percent of eggs containing live embryos at the natural and modified beach. Difference in mean percentage of eggs with live embryos was statistically significant between beaches (p = 0.048; power ($1- \beta$) = 0.74) but total egg density was not (p = 0.18; power ($1- \beta$) = 0.26).

	Smelt egg density (eggs per cm^{-3}; n = 5)		Percent of smelt eggs containing live embryos (n = 5)	
	Natural	Modified	Natural	Modified
Mean (± sd)	15.7 ± 17.6	7.9 ± 4.7	49.8 ± 13.6	24.8 ± 26.3
Median	5.7	7.9	51.7	12.1
Range	1.0 – 35.7	2.3 – 14.1	32.8 – 63.5	1.9 – 59.4

Discussion

Understanding the specific relationships between shoreline modification and changes in biological condition is necessary for successful shoreline management. Data from this study demonstrate that anthropogenic shoreline alteration can make shoreline environments in Puget Sound brighter, hotter, drier, and less suitable for

surf smelt embryos. Although these data are limited in scope (measuring only four environmental variables at two sites at one time of year, and directly addressing only one aspect of biology), they demonstrate the potential changes in abiotic and biotic conditions that occur on modified shorelines throughout Puget Sound.

That the larger physical differences between the heavily altered and natural beach in this study occurred during daylight hours indicates that changes in beach exposure to sunlight is the primary cause of differences between beaches. Removal, then, of natural shoreline structure, including shade-providing terrestrial shoreline vegetation, can have dramatic effects on shoreline microclimate and ecology, changing average and extreme conditions and increasing variation in the physical environment, thus creating a harsher environment for life. In contrast, unmodified shorelines are naturally buffered against such harsh physical conditions, and, for example, are presumably more taxonomically diverse (e.g., Connell 1978) and productive (e.g., Webb et al. 1978) as a result. While empirical causal relationships between anthropogenic alterations of beach microclimate and biological condition have not been established, studies of shoreline alteration and ecology support this proposition (Attrill et al. 1999; Dugan et al. 2003), including those in Puget Sound (Penttila 2001; Romanuk and Levings 2003; Sobocinski 2003). Most important, however, is that regardless of whether modified shorelines are less taxonomically diverse, productive, etc., they are certainly less "natural;" that is, shifted away from the conditions under which life evolved and thrived for millennia. This is an important distinction, because it is normal, natural ecological condition (i.e., biological integrity [Karr 1991]) that should be the typical baseline in environmental assessment, and it is normal, natural ecological condition that the law often says we should protect (Angermeier and Karr 1994).

Combined with site level evaluations such as this study and others in the Puget Sound region (Penttila 2001; Romanuk and Levings 2003; Sobocinski 2003), information on the nature and extent of anthropogenic shoreline modification in Puget Sound (at least one third of the shoreline is armored, for example [PSWQAT

2002]) suggests the potential for cumulative ecological effects of altered shoreline microclimate at the landscape scale. Assessing and effectively managing for such impacts will require examination of natural and anthropogenic influences of shoreline microclimate and associated biological effects at multiple spatial scales (Levin 1992; Chen et al. 1999). Management implications of such effects include minimizing further shoreline development, or rehabilitating altered beaches by revegetation, bulkhead removal, etc. (Williams and Thom 2001; Brennan and Culverwell 2004).

Beach Microclimate and Surf Smelt Embryos

The striking difference between the beaches in terms of the proportion of smelt eggs containing live embryos (on the altered beach approximately half of what they were on the natural beach) indicates that shoreline modification has adverse effects on surf smelt embryos. The similar difference between beaches in total smelt egg density is less conclusive because it was not statistically significant, and more importantly, it is impossible to know whether reduced total density, if it is in fact real, was the result of increased egg mortality or preferential use of the natural beach by spawning adults.

Numerous biotic and abiotic factors can influence development and survival of intertidal embryos but temperature is among the most important (Frank and Leggett 1981a; DeMartini 1999). Two potential temperature related causes of the observed differences in surf smelt embryo mortality in this study are higher developmental rates in a warmer incubating environment, or higher mortality resulting from thermal stress and desiccation. Because hatching is dependent on immersion by the tides, embryo development would ideally coincide with favorable tides. Similar to other upper-intertidal beach spawners (Middaugh et al. 1983; DeMartini 1999) the typical incubation time for surf smelt (approximately two weeks) does correspond with the timing of "spring" tide cycles. Thermal slowing of developmental rates might render embryos unprepared for the tidal inundation appropriate for hatching. Accelerated

development through increased temperature could cause smelt embryos to mature early and compromise their ability to await hatching opportunities or survive after hatching. Laboratory studies from another beach spawning species, the California grunion (*Leurestes tenuis*), found that the optimum temperature range for hatching is between 16°C and 27°C, and that hatching success rapidly declines outside this range (Ehrlich and Farris 1971). California grunion also develop more rapidly and are less able to extend incubation and delay hatching at elevated incubation temperatures (Smyder and Martin 2002). Beach spawning Capelin (*Mallotus villosus*) showed deterioration of larval condition with increased beach residence time of embryos (Frank and Leggett 1981b). If surf smelt embryos are similarly affected by conditions in the incubating environment, shoreline modification could have adverse effects on hatching success. Perhaps more likely to be a significant effect of shoreline modification on smelt embryos (including the lower proportions of live smelt embryos on the altered beach in this study) is acute lethality through extreme thermal stress and desiccation. No published information on the specific thermal tolerance of surf smelt embryos exists, but a laboratory study of desiccation showed significant mortality at environmentally relevant levels of low humidity (Chris Lee personal communication).

While this study does document significant differences in environmental conditions between a modified and a natural beach, and suggests that these differences affect surf smelt embryos, more detailed information on the specific environmental tolerances of smelt embryos would be useful. Expanded, systematic field studies (e.g., Frank and Leggett 1981a; Frank and Leggett 1981b) combined with controlled laboratory experiments could provide a mechanistic understanding of the effects of shoreline alterations on surf smelt embryo survival. Such understanding could better inform site level assessments of surf smelt spawning habitat, and could also provide the basis for a broader evaluation of the effects of shoreline alteration and surf smelt populations in Puget Sound as a whole.

Anthropogenic increases in low-quality spawning habitat throughout Puget Sound could shift the overall balance of surf smelt reproduction and mortality, possibly contributing to population declines. No rigorous stock assessments of surf smelt are conducted in Puget Sound, but historical beach surveys of spawning activity (Penttila 1995), and recent adult smelt catch data from nearshore surface trawls (C. Rice, NOAA Fisheries, unpublished data) show uneven distribution of spawning activity and adults in the Puget Sound landscape. Whether these patterns are at all explained by anthropogenic degradation of spawning habitat is impossible to know with existing information, although the results presented here and elsewhere (Penttila 2001; Chris Lee personal communication) suggest several important mechanisms by which reproductive success could be reduced. Explaining landscape, population- level patterns (e.g., Donovan and Thompson 2001; Feist et al. 2003) would likely require the collection of extensive demographic information on surf smelt such as spawning site fidelity or selection by adults, and habitat- and age-specific survival. It would also require comprehensive information on the quality and distribution of potential spawning habitat in the landscape, as well as an understanding of the distribution and dynamics of surf smelt predators and prey.

Conclusions

Results of this study demonstrate that anthropogenic shoreline alterations can result in significant changes in microclimate on affected beaches. Such changes are associated with biological consequences in nearshore Puget Sound, including reduced reproductive capacity of surf smelt, one of the major pelagic fish species in Puget Sound. These results suggest that preservation of natural shorelines and rehabilitation of altered shorelines could help protect and improve the ecological health of Puget Sound. Future research should evaluate relationships between natural and anthropogenic physical disturbance of the shoreline and its biological consequences at multiple spatial and temporal scales.

CHAPTER 3

Seasonal and Geographic Patterns of Density and Length in Marked and Unmarked Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) in the Neritic Waters of Greater Puget Sound

Summary

Two potentially important factors in the century-long decline of wild Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*) are the artificial propagation of salmon by hatcheries, and the anthropogenic degradation of estuarine environments. Determining the significance of these influences and mitigating adverse effects requires an understanding of the estuarine ecology of wild and hatchery Chinook in Puget Sound. This study focused on the rarely investigated neritic environment (surface waters overlaying the sublittoral zone), inhabited by juvenile Chinook during their late estuarine and early marine transition to the critically important first year at sea. I documented seasonal density and size in marked (known hatchery) and unmarked (majority naturally spawned) juvenile Chinook throughout much of greater Puget Sound. Monthly surface trawl sampling was conducted at twelve sites in the Skagit River estuary in 2002, and expanded to 52 sites covering five additional river mouth estuaries in 2003. Juvenile Chinook salmon were present in all months sampled. Unmarked Chinook in the northern portion of the study area showed broader seasonal distributions of density than both marked Chinook in all areas, and unmarked Chinook in central and southern Puget Sound. Lengths of unmarked fish tended to be smaller than in marked fish, and were more variable. These results suggest more extensive use of estuarine environments by wild than by hatchery Chinook, and differential use of various geographic regions of greater Puget Sound by juvenile Chinook in general. In addition, hatchery-generated timing, density, and length differences have implications for biological interactions between hatchery and wild Chinook throughout Puget Sound. Simple measurements of abundance and size in juvenile wild and hatchery Chinook in estuarine environments, particularly over broad spatial and temporal scales, are useful assessment and monitoring tools.

Introduction

Chinook salmon (*Oncorhynchus tshawytscha*) is among the most socioeconomically important fish species in western North America, historically providing major subsistence, cultural, commercial, and recreational fisheries (Healey 1991, NRC 1996, Stouder et al. 1997, Augerot et al. 2005). After a century of decline Chinook is also one of the most troubled species, with multiple wild populations, including Puget Sound Chinook, listed as threatened or endangered under the Endangered Species Act (Nehlsen et al. 1991, Myers et al. 1998). The decline continues despite major expenditures on hatcheries, fish passage engineering, and habitat "restoration" (Lichatowich 1999, Taylor 1999, Montgomery 2003).

Extensive rearing in estuaries by wild Chinook (Healey 1982, Simenstad et al. 1982, Thorpe 1994, Aitkin 1998), and the massive alteration of estuarine environments by humans (Bortleson et al. 1980, Beach 2002, Collins et al. 2003, Collins and Sheikh 2005) have made estuaries an increasingly important focus of Chinook salmon conservation and recovery efforts (Bottom et al. 2005b). Poor understanding of juvenile Chinook use (e.g., migration and rearing) of estuarine environments limits our ability to evaluate the condition of wild Chinook and develop effective plans for their recovery. Consequently, estuarine field studies are needed to provide essential natural history information—empirical facts that "must be reckoned with one way or another" (Slobodkin 1994) as we make inferences about the ecological requirements of wild Chinook salmon, the effects of hatcheries, and the condition of estuarine ecosystems, including Puget Sound.

This study focused on the poorly understood neritic estuarine habitats (surface waters overlying the sublittoral zone) to explore patterns of density and size in juvenile wild and hatchery Chinook in Puget Sound, and posed three primary research questions:

1) What are the seasonal and geographic distributions of density and length in marked (known hatchery) and unmarked (majority naturally spawned) juvenile Chinook salmon in neritic Puget Sound?

2)   How strongly related to sampling date, capture location, and origin (wild or hatchery) are densities, relative abundances, and lengths of juvenile Chinook salmon in neritic Puget Sound?

3)   What simple attributes of juvenile Chinook salmon in neritic environments might be useful in biological assessment and monitoring?

Answering these questions will improve our basic understanding of Chinook use of estuarine environments (from site-specific to estuary-wide spatial scales), including the potential for hatchery-wild interactions, and the role of various geographic areas of Puget Sound as juvenile habitat. It will also assist future research and monitoring efforts for Chinook salmon and other species of the Puget Sound ecosystem.

Chinook Salmon and Estuaries

Like all anadromous salmonids, Chinook rely on estuaries for migration, physiological and trophic transition, rearing, and refuge (Healey 1982, Simenstad et al. 1982, Healey 1991, Thorpe 1994). Adults can migrate through estuaries during much of the year en route to their freshwater spawning grounds, but most of this migration is concentrated in one to three seasonal peaks, or "runs," between early spring and late fall. Spawning occurs several weeks or months later, depending on the distance to, and environmental conditions in, the spawning habitats. After emergence, "stream-type" juveniles rear in freshwater for over a year, and "ocean-type" juveniles migrate to sea in their first year. Juvenile ocean-type Chinook use estuaries more extensively than any other salmonid species and life stage in terms of residence time, the diversity of life histories expressed, and the variety of habitats and prey used (Healey 1982, Simenstad et al. 1982, Healey 1991, Thorpe 1994, Wissmar and Simenstad 1998). Juveniles enter estuaries as early as winter and are often present into fall (Rich 1920, Reimers 1973, Myers and Horton 1982). Estimates of estuarine residence time range from several days to several months, depending on

the life history type of the fish, and, presumably, on the environmental characteristics of the estuary. Individual fish grow as they move downstream and offshore (Myers and Horton 1982, Healey 1991), and some of the highest growth rates of any life stage have come from juveniles in the estuary (Healey 1991). Juvenile Chinook feed on a diverse estuarine prey base that includes epibenthic and pelagic crustaceans, terrestrial and freshwater drift insects, and fish, but become more piscivorous with increasing size (Fresh et al. 1981, Healey 1991, Duffy 2003, Brennan et al. 2004).

Seasonal and geographic patterns of habitat use, as well as residence time, growth, and survival of juvenile Chinook in estuaries, strongly suggest that estuarine environments are critically important to overall survival (Rich 1920, Reimers 1973, Healey 1982, Simenstad et al. 1982, Healey 1991, Magnusson and Hilborn 2003). Recent quantitative modeling identifies estuarine performance by juveniles as a particularly important factor in determining survival to adulthood (Karieva et al. 2000, Greene and Beechie 2004, Greene et al. 2005). But while much is known about the basic estuarine ecology of Chinook, significant data gaps remain, particularly in the estuarine neritic waters, "downstream" of tidal freshwater and brackish estuarine wetlands, and offshore of brackish and euhaline intertidal shorelines. Topics in need of study include movement patterns and residence time, growth, survival, food web relationships, life history diversity, health and physiology, and hatchery-wild interactions (Brodeur et al. 2000, Bottom et al. 2005b, Fresh 2006).

Human transformations of estuaries in the Pacific Northwest during the last century and a half include physical, chemical, and biological changes that resulted in an ecological conditions very different from the ones in which wild Chinook evolved and thrived (Lichatowich 1999, Montgomery 2003, Pess et al. 2003). For example, filling and diking of estuaries for agricultural and industrial development (Bortleson et al. 1980, Dahl 2000, Collins et al. 2003, Collins and Sheikh 2005) severely reduced the extent and quality of estuarine wetlands—shallow, structurally complex,

and extremely productive environments used extensively by juveniles. Hydrologic alterations from dams, river diversions, and land use blocked migration routes and changed the magnitude, timing, and physical and chemical characteristics of water, sediment, and organic debris inputs to estuaries (Simenstad et al. 1992, Maser and Sedell 1994). This in turn affected physical habitat and probably food web structure (Simenstad et al. 1992), and consequently, physiological processes and migratory behavior of all salmonids (Pess et al. 2003, Bottom et al. 2005b, Quinn 2005). Industrial, urban, and agricultural activities generated a multitude of new chemical inputs, many of which are either acutely toxic to fish or cause sublethal disruption of biological processes such as reproduction, growth, homing, and immune function (Arkoosh et al. 1998, Scholz et al. 2000, Johnson et al. 2007). Hatcheries (and to a lesser degree fish farms) produced high numbers of potential competitors, predators, and disease vectors for wild fish to contend with, and also introduced genetic changes via straying or escaped adults (NRC 1996, Fresh 1997, Myers et al. 1998, HSRG 2004, Mobrand et al. 2005).

Some fundamental effects of these influences on Chinook salmon have been observed in a variety of life stages and habitats, including estuarine juveniles. In addition to severe declines in abundance and extinction of multiple local population segments among wild populations (Nehlsen et al. 1991, Myers et al. 1998), Chinook salmon throughout the Pacific Northwest probably underwent major changes in the number and relative abundance of life history types. For example, reconstruction of historical abundance patterns of juvenile Chinook migrating through and rearing in the Columbia River estuary indicates much greater life history diversity and more protracted seasonal distributions in historical versus contemporary populations (Figure 3.1, Burke 2004, Bottom et al. 2005b). Pulses of homogeneous, transient, and predominantly hatchery fish have replaced the diverse suite of life histories and broad seasonal habitat use characteristic of healthy wild populations. Such contrasts can provide the basis for assessment and monitoring: reference conditions against which we should measure, and toward which we hope our wild populations progress.
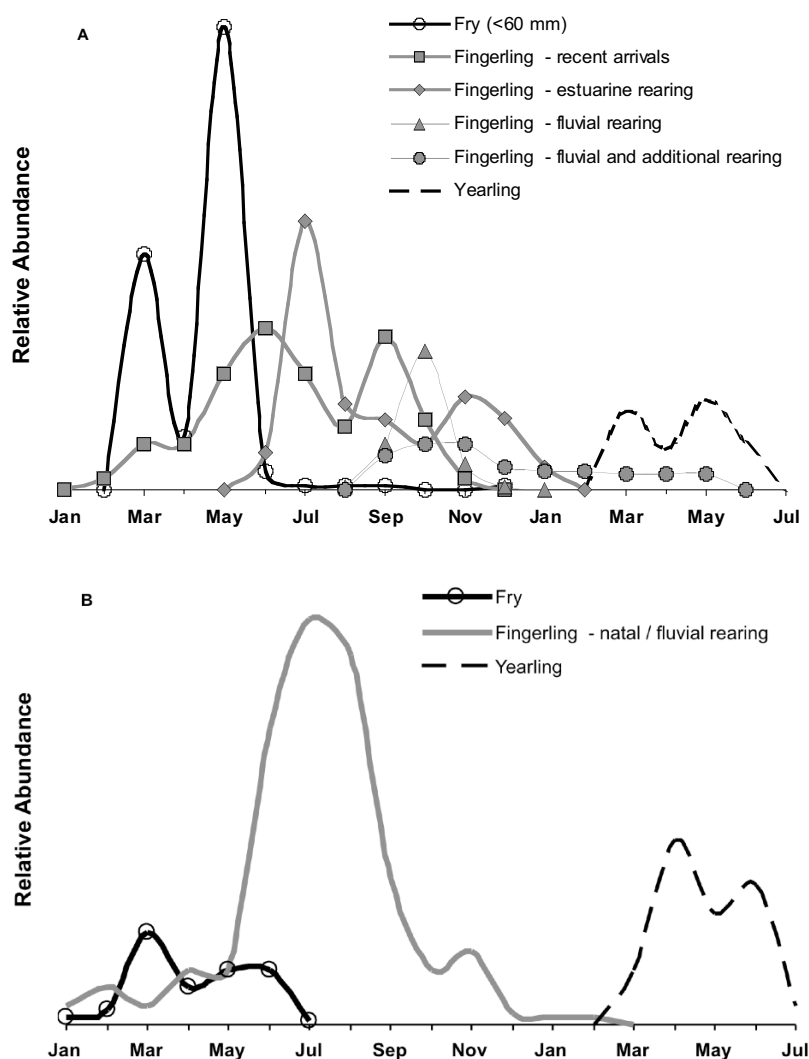
Figure 3.1. Reconstructed historical (top) and contemporary (bottom) relative
abundances of life history types for one brood year in the Columbia River
estuary (based on data from Rich 1920, and Dawley et al. 1985).
Compared to today, historical populations possessed more life hishory
diversity and showed broader seasonal distributions in the estuary.
(Source: Jennifer Burke, School of Aquatic and Fishery Sciences,
University of Washington.)

Puget Sound Chinook Salmon

The Puget Sound/Georgia Basin is a fjord-estuary complex used by anadromous salmonids from thousands of natal streams and rivers in the surrounding watersheds. This includes the 22 extant (of 31 historical) population segments identified in the Puget Sound Chinook salmon evolutionarily significant unit (ESU) (Myers et al. 1998, Ruckelshaus et al. 2006). While a minority of Puget Sound Chinook exhibit "stream-type" juvenile life histories and rear in freshwater for a year before migrating to sea, the vast majority are "ocean-type" and leave freshwater in their first year (Simenstad et al. 1982, Healey 1991). At least three basic life history tendencies are known among the ocean-type Chinook in Puget Sound (Beamer et al. 2005):

1) Fry migrants begin moving downstream rapidly after hatching, usually in February and March at approximately 30-50 mm, and pass quickly through delta environments and rear in a variety of estuarine shoreline areas outside of the natal delta.

2) Tidal delta rearing fry migrants show similar downstream migration timing and short freshwater residence as fry migrants but rear in natal delta environments for weeks to months reaching lengths of approximately 50-125 mm before moving into more exposed estuarine shoreline habitats.

3) Parr migrants rear in freshwater for several months before migrating to the estuary, usually in late May or June, at approximately 60-100 mm, and tend to move quickly through the natal delta and out into more exposed and saline environments.

Unlike coastal plain estuaries such as those of the Columbia and Sacramento Rivers, Puget Sound and other fjord-estuary systems include a portion of Chinook that never go to the ocean, spending the entire marine phase of their life cycle in these "inland seas." Such resident fish have supported significant recreational

fisheries at times of year outside the runs of adults returning from the ocean to spawn (Buckley 1969).

In the second half of the Nineteenth Century, intensive harvest and habitat destruction initiated the long decline of Puget Sound Chinook (Lichatowich 1999, Taylor 1999, Montgomery 2003). Total wild Puget Sound Chinook adult run size in 1908 (the year of the peak recorded Chinook harvest in Puget Sound) was estimated at 690,000 and today is typically less than 10 % of that (Myers et al. 1998). Declining abundance was recognized as early as the late 1800's, and hatchery production began in the early 1900's to compensate, assuming near limitless carrying capacity in estuarine and marine environments (Beamish et al. 2003, HSRG 2004). Eventually, nearly fifty state, tribal, and federal facilities were regularly releasing artificially propagated Chinook into Puget Sound tributaries, and another fifty or so public facilities did so intermittently (Myers et al. 1998). Transfer of eggs among watersheds inside and outside of the Puget Sound basin was common, and the most widely used hatchery stock in Puget Sound is from the Green River in central Puget Sound. Nearly two billion juvenile hatchery Chinook have been released into Puget Sound tributaries since the 1950's. Annual juvenile hatchery Chinook releases peaked at 76 million fish in 1990 (WDFW and PSTT 2004), and are currently approximately 30 million fish (NOAA Fisheries, unpublished data). No rigorous estimates of the historical wild Chinook juvenile production have been made. In recent decades, returns to hatcheries have accounted for nearly 60% of total spawning escapement, but the actual contribution of hatchery fish is probably much higher because of straying onto spawning grounds (Myers et al. 1998).

While massive hatchery production in Puget Sound continued throughout the Twentieth Century, so did the destruction of salmon habitat, including estuarine wetlands, over half of which have been lost (Bortleson et al. 1980, Collins and Sheikh 2005). In urban estuaries such as the Duwamish River (city of Seattle) and the Puyallup River (city of Tacoma) less than 3% of historical estuarine wetland area remains (Bortleson et al. 1980, Collins et al. 2003, Collins and Sheikh 2005).

Determining the significance of hatchery production and anthropogenic degradation of estuarine environments in the decline of Puget Sound Chinook requires an understanding of the interactions among wild fish, hatchery fish, and the full array of estuarine habitats they use. Yet, research and monitoring of juvenile salmonids in the Puget Sound region overwhelmingly focus on freshwater environments and often do not distinguish between hatchery and wild fish. The few estuarine studies have mostly been conducted at few or single sites and emphasized maximum abundances around the "peak outmigration" alongshore; that is, the perceived peak as seen primarily in beach seine catches. Few have attempted the comprehensive habitat and temporal coverage of Puget Sound that is necessary to adequately characterize the estuarine ecology of Chinook in Puget Sound. Sampling neritic environments adjacent to shore provides insights on the "next step" of the late estuarine and early marine phase of the life cycle, a transition into the first year at sea that is widely regarded as the most critical period in determining survival to adulthood (Pearcy 1992).

Study Sites

This study used surface trawl, or "townet" sampling to examine seasonal and geographic patterns in abundance and size in wild and hatchery Chinook salmon in neritic environments of greater Puget Sound, focusing first on the Skagit River estuary in 2002 (Figure 3.2). The Skagit is the largest river in Puget Sound and in several ways provides the best opportunity to study the estuarine ecology of Puget Sound Chinook. It is home to the healthiest remaining wild Chinook populations in the Puget Sound ESU (Myers et al. 1998, Ruckelshaus 2006), has significant remaining estuarine habitat (only approximately 75% of the estuarine wetland area has been lost, for example [Collins et al. 2003]), and has a relatively small hatchery component (about 20% of the total average outmigrant population) nearly all of which is marked with fin clips or coded wire tags (CWT) and therefore easily recognized. In 2003, the study expanded to a larger area extending from Bellingham

Bay in the north to Nisqually Reach, approximately 185 km away at the southern extent of Puget Sound (Figure 3). Sampling this larger area allowed comparisons among the Skagit and several other river-mouth estuaries and marine areas with different oceanographic characteristics and different degrees of human influence.

Neritic waters in river mouth estuaries were the primary focus of the study, but more marine areas in between river systems also were sampled. Objectives for site selection were extensive spatial coverage alongshore (at depths sufficient for the fishing gear to be deployed without hitting bottom), accessibility by boats and fishing gear at any tidal stage, proximity to historical townet sampling sites (e. g., Stober and Salo 1973, Fresh 1979), and, in the Skagit, proximity to existing beach seine monitoring sites sampled by the Skagit River System Cooperative. Sites in the river mouth estuaries were selected to sample at least the approximate center of the delta front and the two adjacent shorelines with the intent of sampling likely outmigration and rearing areas just seaward of the habitats typically sampled by beach seining. In 2002, 12 sites were sampled in Skagit Bay (Figure 3.2). In 2003, 52 sites were sampled (Figure 3.3), including the same 12 Skagit Bay sites. The 2003 sites covered an area from Bellingham Bay at the northern extremity, to Nisqually Reach, approximately 185 km to the south, and included six river mouth estuaries and several more marine areas. Sites were assigned to "basins" commonly used to describe different oceanographic regions of Puget Sound (Burns 1985). Areas outside of Puget Sound proper (Padilla and Bellingham Bays) were assigned to a "Rosario Basin" created for this study. Except for July 2002, sampling was conducted during neap tide series to reduce the influence of tides on fish distribution. In 2003, the neap tide series were allocated alternately to north and south regions, resulting in monthly sampling trips in the north and south separated by two weeks.
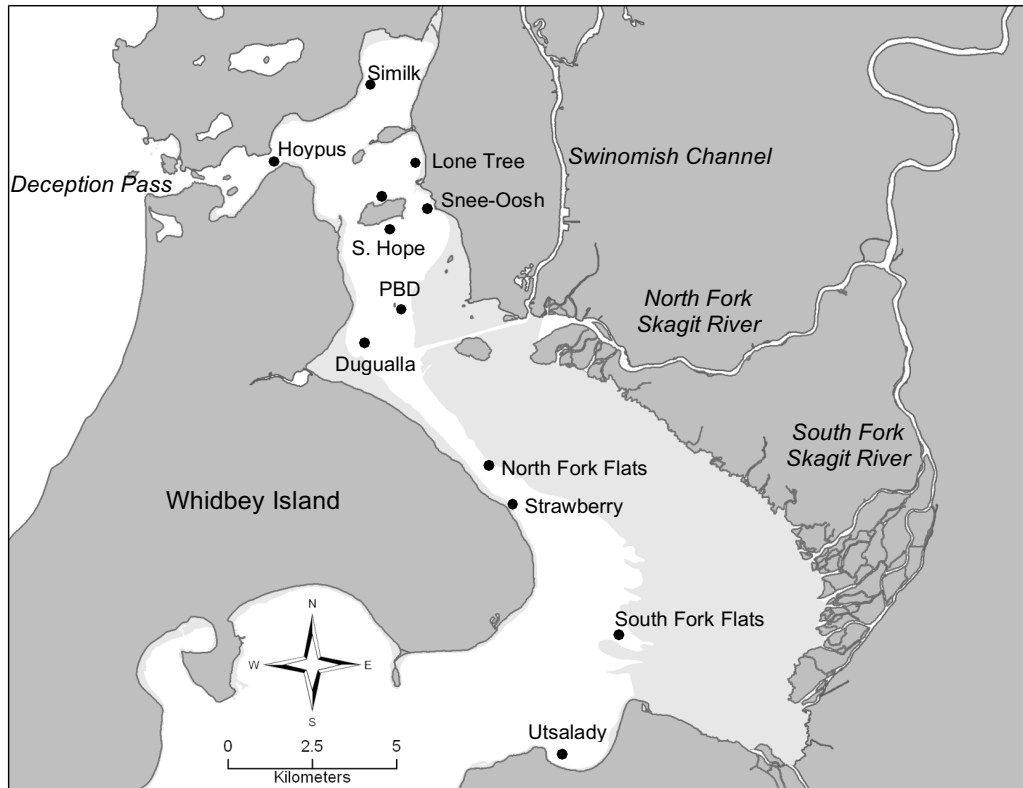
Figure 3.2. Twelve neritic sites sampled in Skagit Bay in 2002 and 2003. Intertidal areas colored lighter grey.
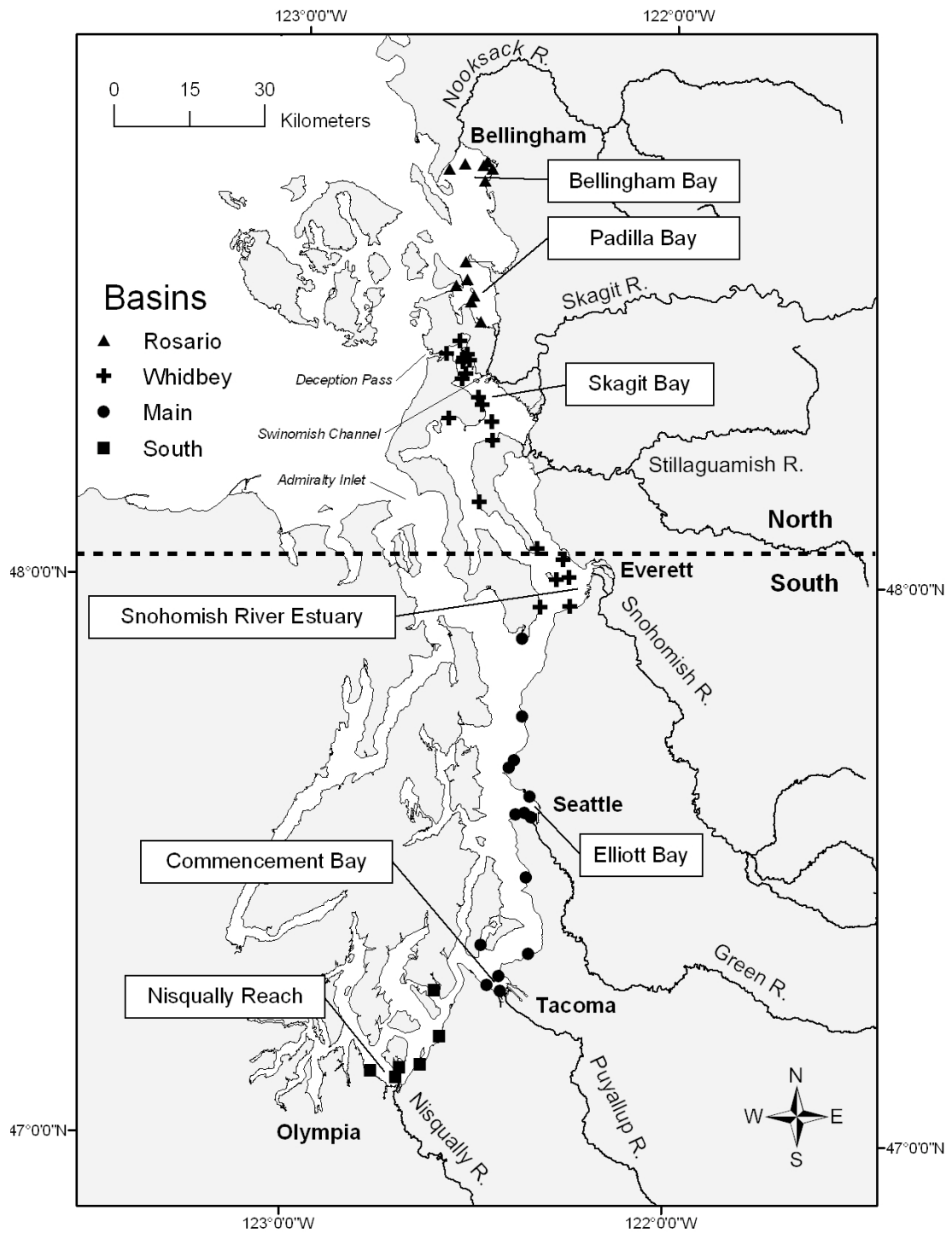
Figure 3.3. Sites sampled in 2003. Basins indicated by symbols; embayments marked by labels.

Methods

Fish Collection and Processing

Sites were sampled monthly from February to November in 2002, and from May through October 2003. Fishing employed a 3.1 m high x 6.1 m wide Kodiak surface trawl, or "townet," deployed between two boats, each with a 15.2 m tow line connected to a bridle on the net. Mesh sizes in the net range from 7.6 cm stretch in the forward section, and progressively smaller to 3.8 cm then 1.9 cm in the middle sections, and 0.6 cm in the codend. The primary vessel (13.7 m long, 174 hp inboard diesel) towed the left side of the net while trawling, and the second vessel (5.5 m long, 225 hp gasoline outboard) towed the right side. The net was towed at the surface for 10 minutes per tow, at 900-1000 rotations per minute (RPM) on the engine of the primary vessel and a typical towing speed of 2-3 knots through the water. Distance through the water was recorded with a mechanical flow meter (General Oceanics model 2030) deployed by the smaller vessel. Area swept was calculated as the distance traveled through the water x width of the net opening. In 2002 up to three tows were made per site. In 2003 this was reduced to up to two tows per site in order to accomodate the much broader geographic coverage. A total of 359 successful tows were conducted in 2002, and a total of 668 were completed in 2003.

At the end of each tow, the net was closed via a purse line running along the top of the net opening, the boats came together side by side and the net was hauled onboard the larger vessel. The entire catch was then placed in tanks supplied with flowing water from the site, and the fish were identified, counted, and weighed by species, and gelatinous zooplankton were weighed. Individual Chinook were measured for length and weight, visually examined for clipped adipose fins, and checked for coded wire tags (CWT) using a handheld detector wand (Northwest Marine Technologies, Inc.). Surface water temperature and salinity measurements were taken during each tow using an electronic meter (YSI model 30) on water

drawn continuously by the primary vessel's deck hose from a depth of approximately 1.2 m.

Statistical Analysis

Statistical analysis focused on the influence of season and location on densities, relative abundances, and lengths of marked and unmarked Chinook. Additional independent environmental variables (latitude, temperature, salinity, and depth) were analyzed after the initial models to refine the interpretation of the density and relative abundance data. Analyses were primarily done using Akaike's Information Criterion (AIC), an "information-theoretic" approach that allows a weight of evidence evaluation of multiple models without requiring the binary accept/reject decisions typical of conventional hypothesis testing (Akaike 1973, Anderson et al. 2000, Burnham and Anderson 2002, Dayton 2003). AIC scores are improved (made smaller) by better model fits, and penalized for additional parameters. A generalized linear model (GLM) univariate analysis of variance (ANOVA) was used to generate residual sum of squares (RSS) for use in the AIC analyses. Proportion of unmarked Chinook (# unmarked Chinook / total Chinook) in each tow was used as an integrative measure of marked vs. unmarked abundance. To reduce the effects of non-normal data distribution and unequal variance, fish lengths were $\log_{10}(x)$ transformed, fish densities were $\log_{10}(x+1)$ transformed, and proportions of unmarked Chinook were transformed using a modified arcsine square root transformation for proportional data (Zar 1996):

$$\arcsin \sqrt{\frac{X + \frac{3}{8}}{n + \frac{3}{4}}}$$

where X is the number of unmarked Chinook in the tow and n is the total number of Chinook in the tow. Sampling month and site were used as nominal independent variables in the initial density and proportion unmarked models for 2002. To evaluate the importance of site level environmental attributes (latitude, temperature,

salinity, and depth), subsequent AIC analysis of numerous linear regression models was performed using residuals from quadratic regressions of density and proportion unmarked versus sampling month (to remove the seasonal influence), and all single and additive combinations of the four environmental variables. For 2003 density and proportion unmarked models, sampling month, embayment, basin, and region (northern and southern portions of the study area) were used as nominal factors. As in the 2002 analysis, subsequent AIC analysis was performed on numerous regression models of the residuals from quadratic regressions of 2003 density and proportion unmarked against month, and latitude, temperature, salinity, and depth. Month and location were also used as independent variables in the length models, and presence or absence of marks was added as a nominal factor. Because sampling of north and south regions of the 2003 study area occurred on alternating 2-week intervals, and fish lengths change rapidly, north and south regions were not combined in the ANOVA models for length. Linear regression analysis was used to compare slopes of marked and unmarked fish length over a season. When present, yearling fish would likely have a large influence over mean length but some yearling fish could not be conclusively identified in this study. Consequently, the relatively few obvious yearlings in my samples were removed from the length regression analysis by excluding fish above 120 mm in June, and above 200 mm from July onward. Interactions between month and location were included in the initial density, proportion unmarked, and length models because of the migratory behavior and ontogenetic habitat changes characteristic of juvenile salmon. Fish densities were calculated by dividing the raw catch by the area swept. Origin of fish with coded wire tags (CWT) was determined using the Regional Mark Information System (RMIS) of the Pacific States Marine Fisheries Commission's Regional Mark Processing Center.

Graphical analysis of residuals from the candidate models generally confirmed normal distributions and equal variance among groups, with no extreme departures from assumptions of ANOVA. In addition to AIC scores, I calculated the difference

between each model's AIC score and that of the best model of the group. The larger this difference, the less likely that a given model was actually the best of those evaluated. In general, a distance of two or fewer AIC units between models is considered to have substantial empirical support, models with distances of four to seven from the best model are still plausible but much less likely, and models with distances greater than ten from the best model have essentially no support (Burnham and Anderson 2002). AIC "weights" and, in selected cases, relative variable importance (RVI) were also calculated. The AIC weights for a set of models sum to 1 and are interpreted as the probability that each model is in fact the best of the models considered, given the data used. Relative variable importance is the sum of the AIC weights from models that contain a given variable.

Results

Environmental Variables

Temperature and salinity showed clear seasonal patterns. Mean water temperature in Skagit Bay in 2002 ranged from 6.5°C to 13.1°C with a steady increase from March into June followed by stable mean temperatures of approximately 12-13°C until a drop to 10.0°C in early November (Figure 3.4). Mean salinity ranged from 13.0 in June to 27.2 ppt in November, decreasing through spring, followed by a large drop in June corresponding to peak Skagit River discharge, and a gradual rise into fall (Figure 3.5). In 2003, mean temperature showed a similar seasonal pattern among basins, starting at approximately 9°C in April, increasing to a summer peak of 16-17°C, then declining to approximately 10°C in November (Figure 3.6). Salinity in 2003 also showed seasonal patterns but differed substantially among basins (Figure 3.7). The Rosario and Whidbey basins in the north, with their much higher freshwater input from rain and snowmelt, had lower salinity than the Main Basin and South Sound overall and a wider range, from 16 in April to over 25 in September and October. The Main Basin and South Sound always averaged between 25 and 30, with a general increase from spring to fall (Figure 3.7).

Water depth at sampling sites in Skagit Bay in 2002 ranged from 5.8 m to 15.6

m with an overall mean of 8.6 m. In 2003, sampling occurred over a broader range of

depths, primarily because of steeper shoreline slopes and the presence of more large,

anthropogenic obstacles (e.g., piers) in central Puget Sound. Water depth at sampling

sites in 2003 ranged from 4.7 m to 46.7 m with an overall mean of 11.9 m.
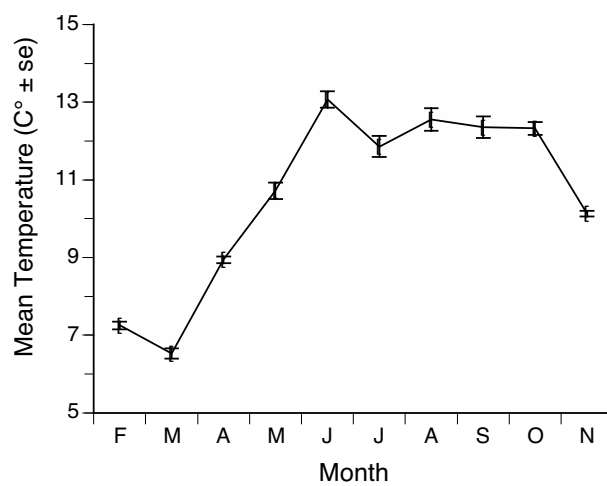
Figure 3.4. Mean monthly water temperature (± se) at twelve Skagit Bay sites
in 2002.
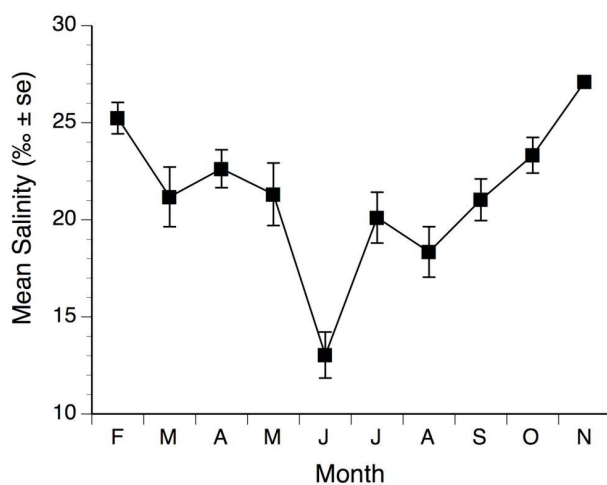
Figure 3.5. Mean monthly salinity (± se) at twelve Skagit Bay sites in 2002.
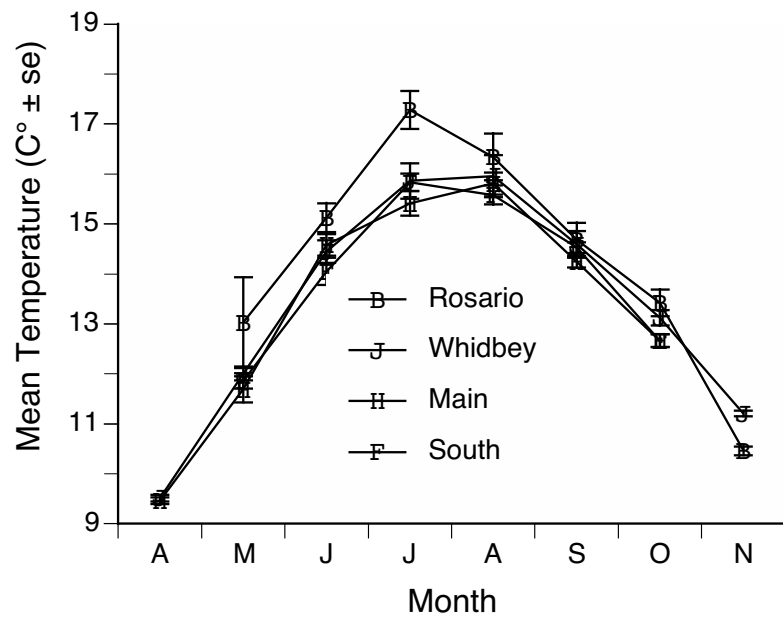
Figure 3.6. Mean monthly water temperature (± se) in the four basins in 2003.
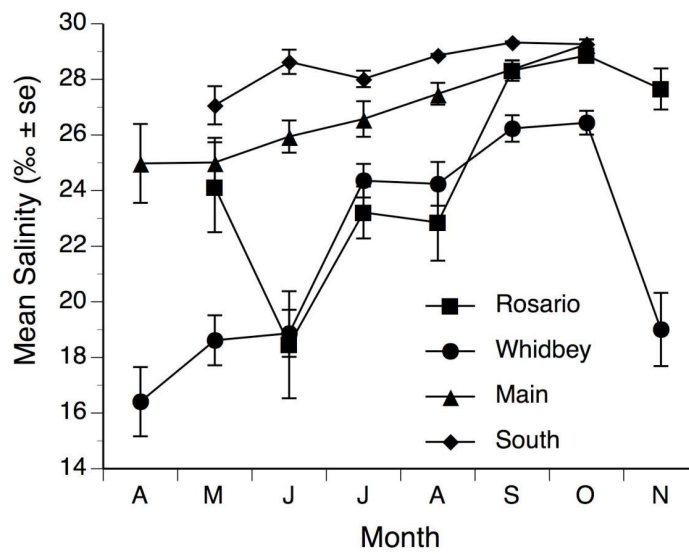
Figure 3.7. Mean monthly salinity (± se) in the four basins in 2003.

Densities of Marked and Unmarked Chinook Salmon

Skagit Bay

Juvenile Chinook salmon were caught in all months sampled in Skagit Bay in 2002 but marked fish were not present in February, March, or April because of the lack of hatchery releases before May. Unmarked fish were 67% of all Chinook caught and showed a more protracted seasonal distribution than did marked fish (Figure 3.8). Chinook densities ranged from approximately 0.1 fish/hectare in winter and early spring when small numbers of unmarked fry migrants were caught, to 1.0 fish/hectare in May corresponding to the arrival of yearling fish, followed by a steep increase to a peak of approximately 17 fish/hectare for both marked and unmarked fish in July. A rapid decline in marked fish and gradual decline in unmarked fish occurred through summer into fall. The spike of Chinook catch in July may in part be the result of tidal conditions during that sampling trip; scheduling problems forced the July 2002 sampling to occur during extreme, or "spring" tides, possibly influencing the distribution of fish (e.g., concentrating fish at low tide) differently than during the rest of the sampling trips.
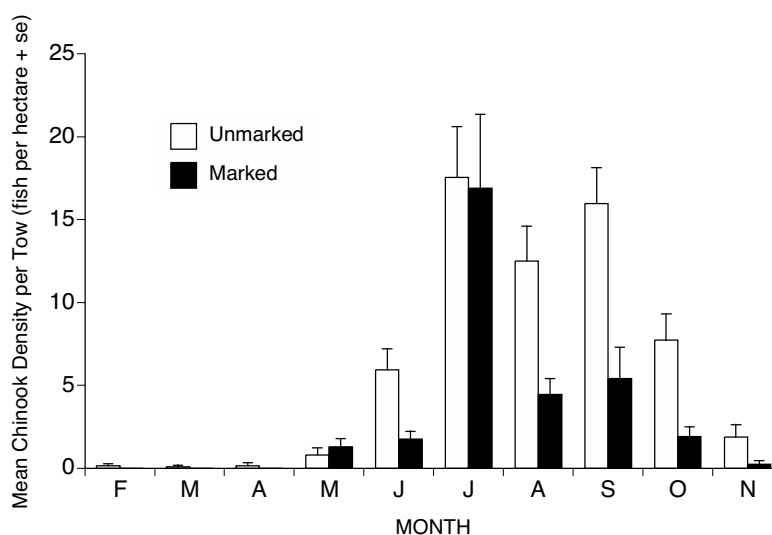
Figure 3.8. Mean density (+ se) of marked and unmarked Chinook in Skagit Bay in 2002. Twelve sites combined.

Temporal patterns were variable among sites in Skagit Bay in 2002 (Figure 3.9), and the proportion of unmarked fish also differed among sites. For example, the proportion of unmarked fish at Hoypus was relatively constant throughout the year, but at South Hope and most of the other sites it was not. This may reflect the differential use of Hoypus by the generally larger and presumably more transient hatchery fish migrating out through Deception Pass, likely the primary migration route for juveniles leaving, but occasionally entering, Skagit Bay. Density of all Chinook tended to be higher at sites in the northern part of the Bay. For example, four sites (Hoypus, Similk, Lone Tree, and South Hope) of the twelve accounted for over 60% of the total Chinook catch. The percentage of umarked Chinook was lower in the north (62%) than in the south (78%).
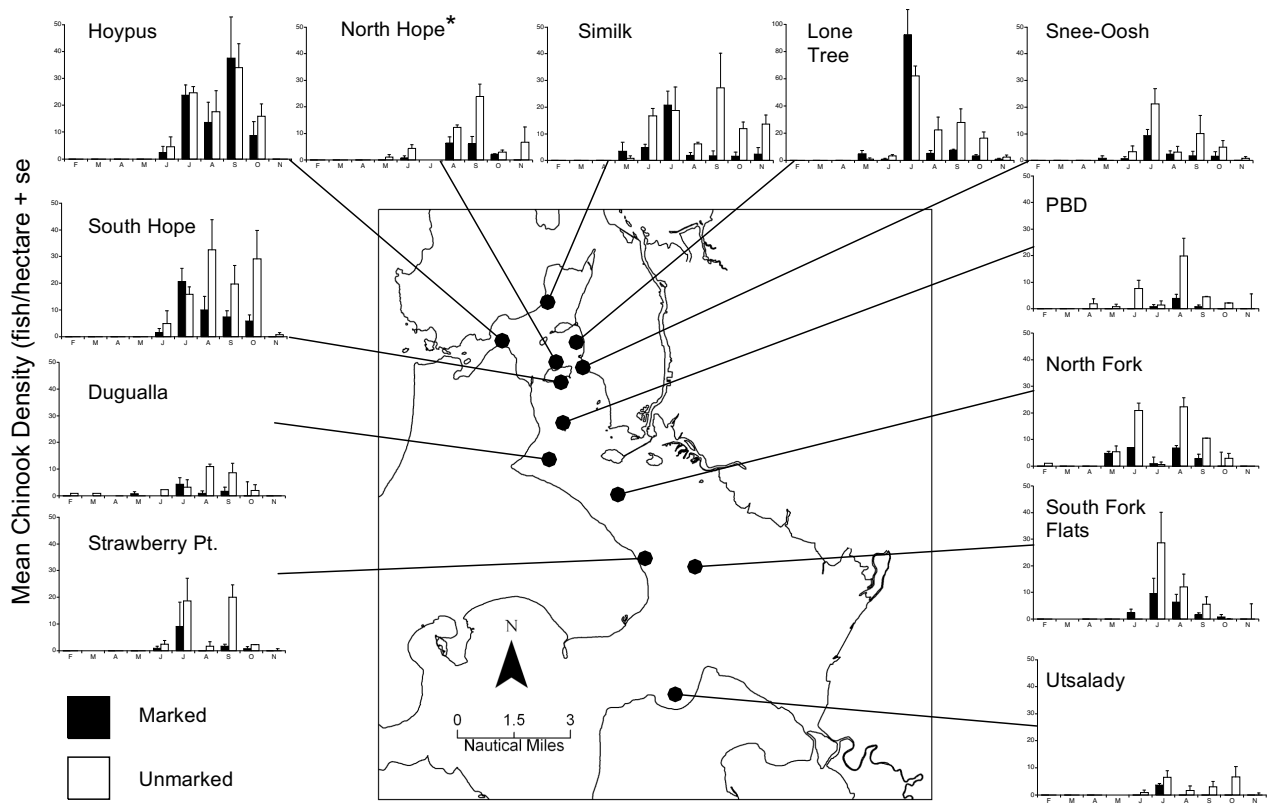
Figure 3.9. Mean densities (+se) of marked (black bars) and unmarked (white bars) juvenile Chinook by site and month in Skagit Bay in 2002. Note the change in scale on Lone Tree graph. * North Hope was not sampled in July.

AIC analysis of four basic ANOVA models for 2002 marked Chinook density, unmarked Chinook density, and proportion of unmarked fish, indicated that both sampling month and site were related to the densities and proportions of marked and unmarked fish, but relationships with month were strongest (Table 3.1). Interactions between month and site were very important in the density models, as the top models in both marked and unmarked fish include the interaction term, were approximately 100 AIC units apart from the second best models, and had an AIC weight of 1. The month by site interaction is much less important in the proportion unmarked models, ranking the model that includes the interaction third among the four models and 12 AIC units away from the top model of month plus site. Furthermore, the AIC distance between the additive model of month and site was only one AIC unit better than the model including only month (which has an AIC weight on its own of 0.36), suggesting that while site level differences in marked and unmarked density were significant, seasonal differences dominated.

Table 3.1. Summary of AIC ranking of four ANOVA models of month and site vs. densities of unmarked and marked Chinook, and proportion of unmarked Chinook in Skagit Bay, 2002 (M = month, S = site). See text for explanation.

Unmarked Chinook Density				Marked Chinook Density				Proportion Chinook Unmarked			
Model	AIC	Δ_i	AIC Weight	Model	AIC	Δ_i	AIC Weight	Model	AIC	Δ_i	AIC Weight
M + S + M * S	-782	0	1.000	M + S + M * S	-907	0	1.000	M + S	-520	0	0.641
M + S	-684	98	< 0.001	M + S	-798	109	< 0.001	M	-519	1	0.357
M	-650	132	< 0.001	M	-744	163	< 0.001	M + S + M * S	-508	12	0.002
S	-397	385	< 0.001	S	-594	313	< 0.001	S	-491	29	< 0.001

Looking further at the effect of environmental variables on the observed densities and proportion of unmarked fish, AIC analysis of linear regression models fitting seasonal residuals (from quadratic fits of densities and relative abundance to month) to all combinations of latitude, salinity, temperature, and depth suggests that latitude and salinity were most strongly associated with patterns of density (Figure 3.10).
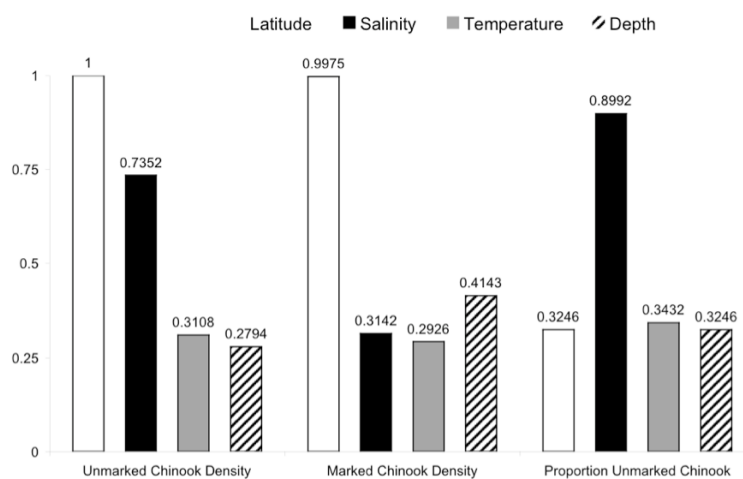
Figure 3.10. AIC-determined relative variable importance (RVI) for environmental attributes of sites in regression models of residuals from seasonal density and proportion unmarked models (quadratic regression); Skagit Bay, 2002. See text for explanation.

Latitude ranked high in both marked and unmarked density models but not in proportion unmarked models. Salinity also ranked high in unmarked density models, and was the dominant variable in the proportion unmarked models. The importance of latitude and salinity could be the result of oceanographic differences between north and south Skagit Bay that give rise to different habitat characteristics, and consequently, differential use of the regions by juvenile Chinook. While temperature and depth of sampling sites are fairly uniform across latitude, salinity is not (Figure 3.11). North Skagit Bay is generally deeper and more marine than south Skagit Bay,

which is dominated by the river delta and more directly influenced by fresh water. Such differences could result in more favorable habitat conditions (e.g., prey base and environmental conditions) (Levings 1994) for extended rearing in north Skagit Bay before migration to sea via Deception Pass and the Strait of Juan de Fuca. But the importance of salinity in the unmarked density and proportion unmarked models may be seasonal effects (i.e., those not captured by the curve fit of density and relative abundance on month) since salinity differences across the season (Figures 3.5 and 3.7) would be differentially associated with wild fish simply because of their broader seasonal distribution and more extensive use of shallow habitats (Beamer et al. in prep), regardless of any direct effects of salinity on density.
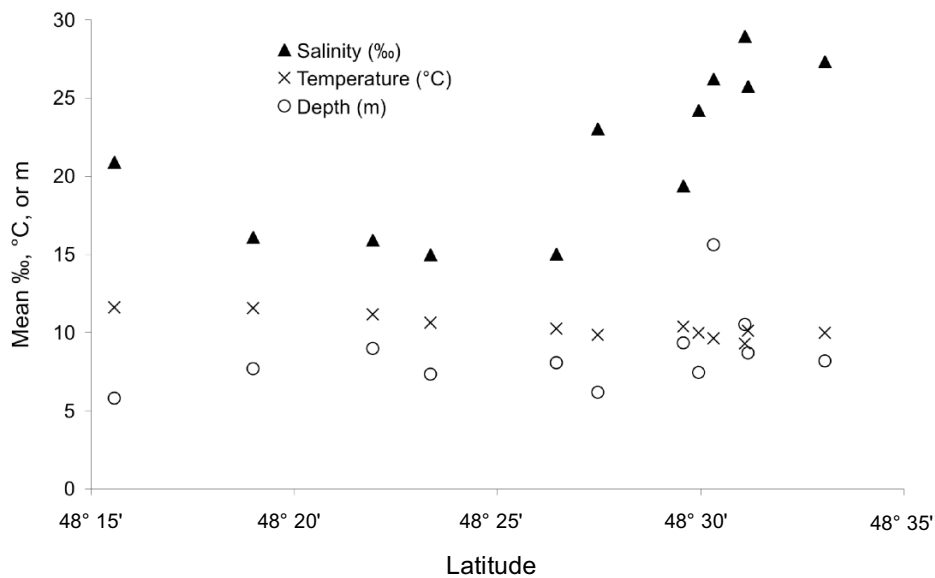
Figure 3.11. Mean salinity, temperature, and depth by latitude at twelve Skagit Bay sites in 2002

Greater Puget Sound

In 2003, general patterns of Chinook density were the same in the Skagit, and the Whidbey (dominated by the Skagit) and Rosario Basins showed the same overall pattern of different seasonal distributions in marked versus unmarked fish (Figure 3.12). In contrast, the Main Basin and South Sound were characterized by less distinguishable patterns between marked and unmarked fish, with both groups resembling the marked fish pattern in the north (abrupt increase in spring and early summer followed by rapid decline), but with higher peak mean densities, especially in South Sound where peak mean density was three to four times that of the other basins (Figure 3.12). These patterns held after the catch was broken down by embayment within basins, except that the Snohomish River estuary had seasonal distributions more like areas to the south but with lower peak densities (Figure 3.13). Thus, the broad seasonal patterns of marked and unmarked Chinook density suggest that while there were geographic differences at the level of embayments and conventional oceanographic basin designations, clear differences were also apparent at a still larger spatial scale. Combining the data into northern and southern regions (with the Snohomish included in the south because of its similarity with central and southern Puget Sound) illustrates the regional differences (Figure 3.14). Similarly, plotted as a cumulative density, marked and unmarked fish in the south showed the same pattern as marked fish in the north but shifted earlier (Figure 3.15). The geographic differences in the overall presence of marked and unmarked fish across the study area in 2003 are further illustrated by summing the mean densities over the sampling period (Figure 3.16).
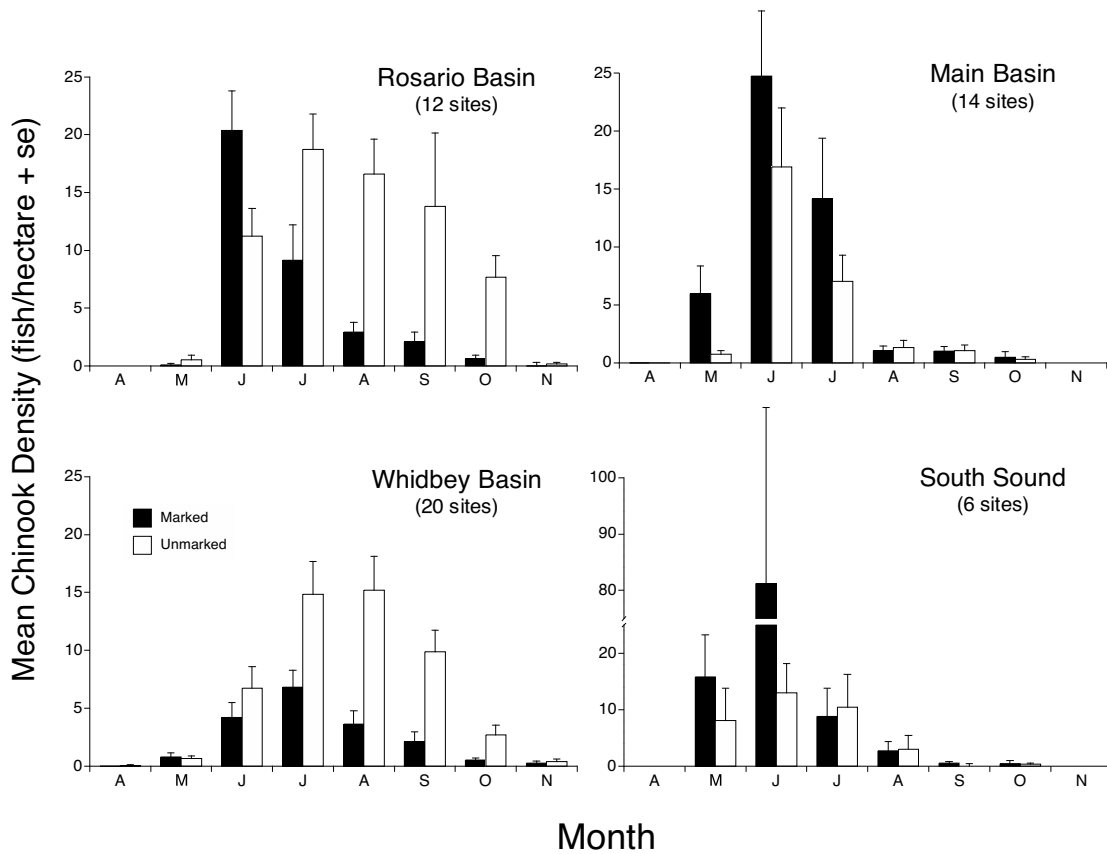
Figure 3.12. Mean density (+ se) of marked (black bars) and unmarked (white bars) Chinook in the Whidbey, Rosario, Main, and South Sound basins in 2003. Note the different density scale for South Sound.
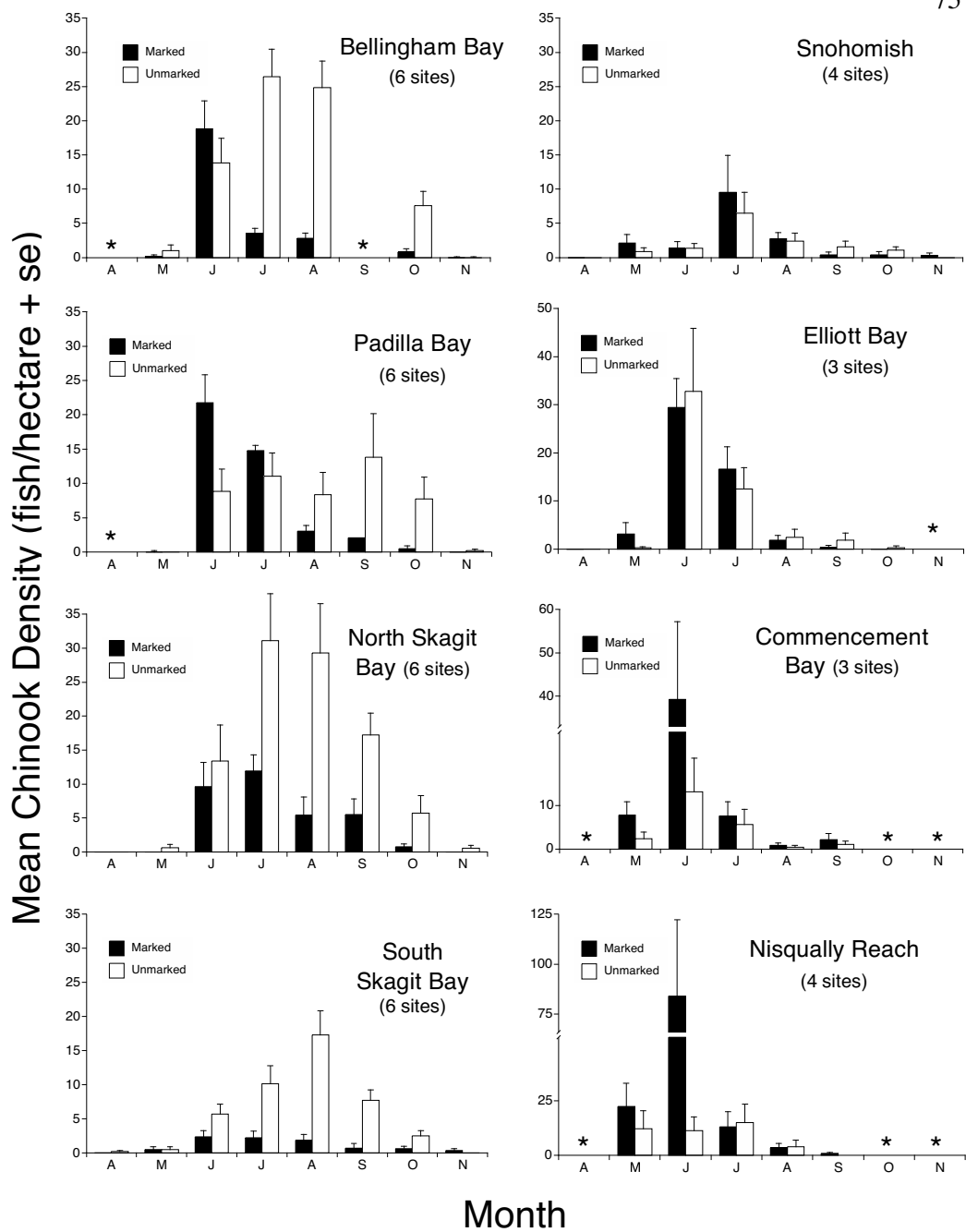
Figure 3.13. Mean densities (+ se) of marked and unmarked juvenile Chinook at six embayments in 2003. Note higher density scales for Elliott Bay, Commencement Bay, and Nisqually Reach. Asterisks (*) indicate months not sampled.
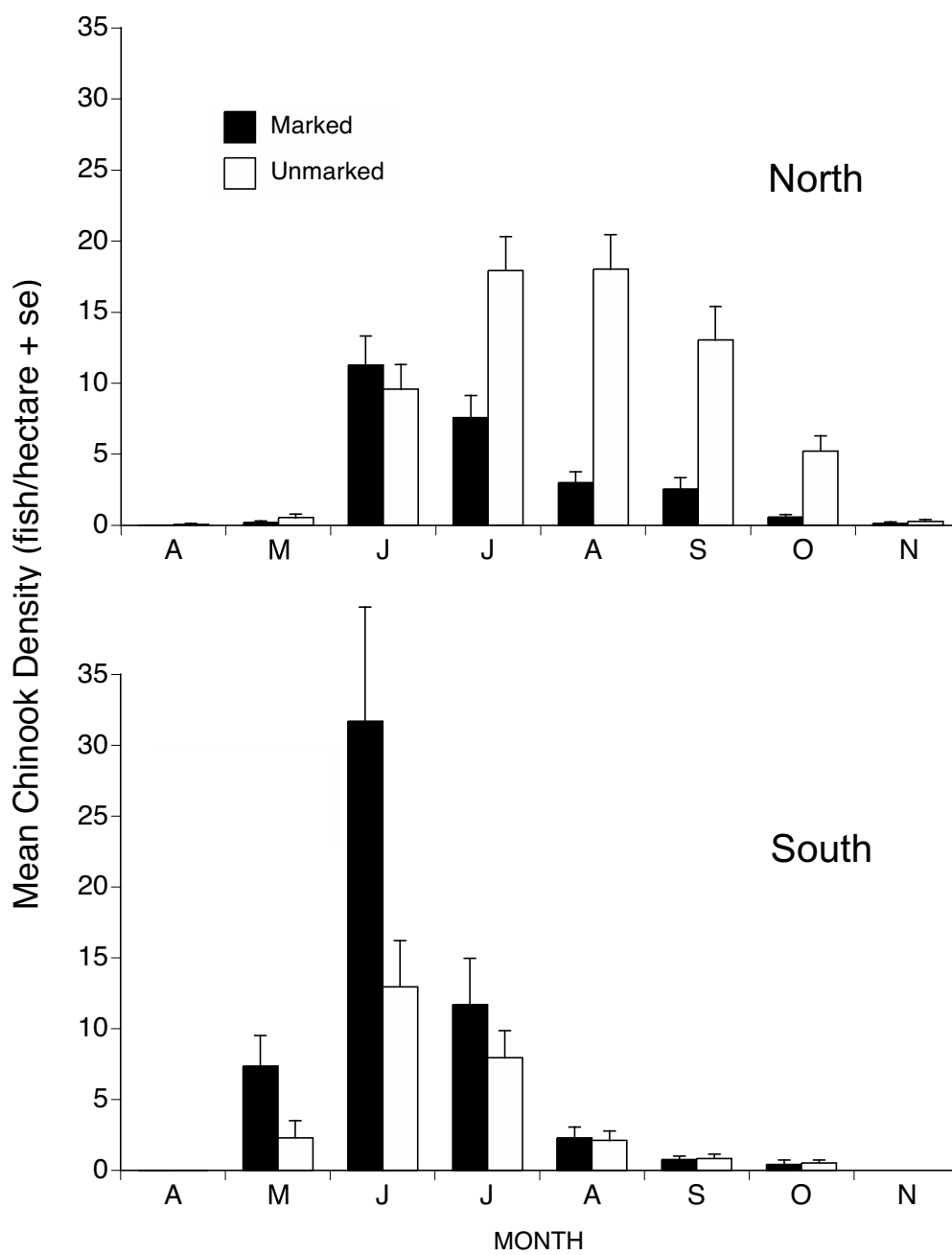
Figure 3.14. Mean density (+ se) of marked (black bars) and unmarked (white bars) Chinook by month in the northern (27 sites; 365 tows) and southern (25 sites; 303 tows) areas in 2003.
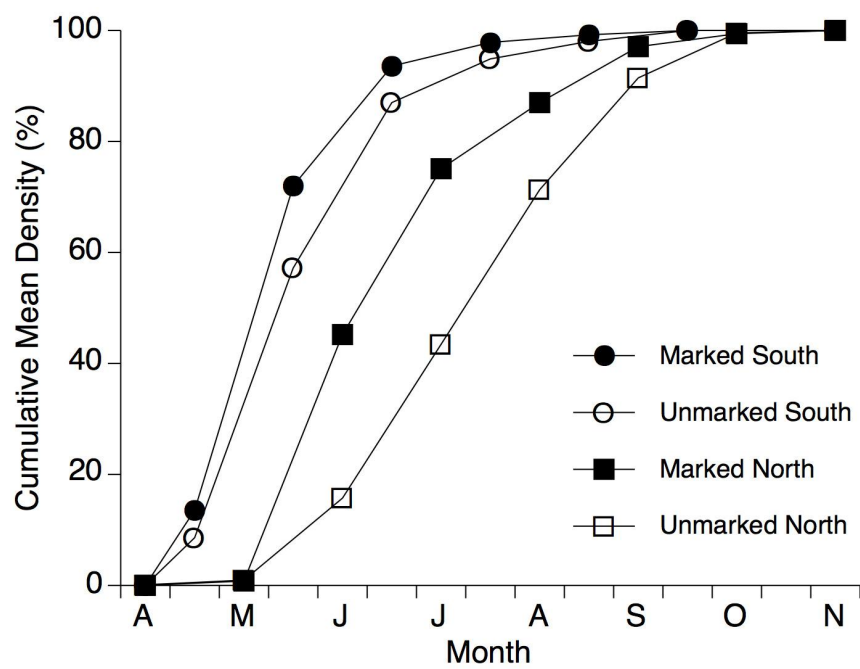
Figure 3.15. Cumulative mean density of marked and unmarked Chinook during 2003 in northern and southern areas of the Puget Sound region.
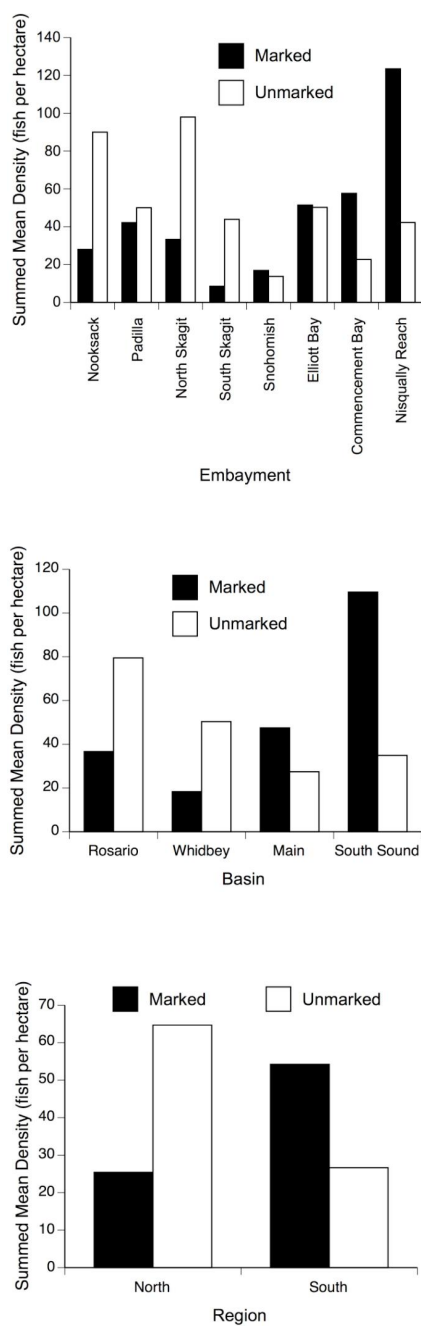
Figure 3.16. Summed mean density of marked (black bars) and unmarked (white bars) juvenile Chinook from monthly sampling in eight embayments (top), four basins (middle), and two regions (bottom) from April through November 2003. Densities for Commencement Bay and Nisqually were assumed to be zero in November.

Total percentage of unmarked Chinook captured in 2003 was 53%, with 72% and 33% unmarked in the northern and southern regions, respectively, consistent with more wild fish in the north and larger hatchery inputs in the central and southern Puget Sound (Table 3.2). Percentage unmarked by basin was: Rosario (66%), Whidbey (73%), Main (36%), and South Sound (25%). Percentage unmarked by the seven major embayments was: Bellingham Bay (76%), Padilla Bay (54%), Skagit Bay (77%), Snohomish River estuary (44%), Elliott Bay (29%), Commencement Bay (28%), and Nisqually Reach (25%). Because of incomplete and variable marking rates in much of Puget Sound these figures are likely overestimates of the percentage of wild fish. Information in the RMIS database suggests that at least 9 % of hatchery Chinook released into Puget Sound in 2003 were not marked with fin clips or CWTs.

Table 3.2. Estimates of hatchery releases and natural production of juvenile Chinook salmon into the four sampling basins in 2003 (E. Beamer SRSC and NOAA Fisheries, unpublished data).

Basin	Hatchery Chinook Released (2003)	Wild Juvenile Production
Rosario	4,822,147	Data unavailable
Whidbey	2,488,397	6,634,094
Main	12,564,172	Data unavailable
South	8,486,809	Data unavailable

Similar to the 2002 results from Skagit Bay, AIC analysis of ten basic ANOVA models applied to 2003 data demonstrated that month and location were related to unmarked density, marked density, and proportion unmarked, and that densities were strongly related to interactions between time and place (Table 3.3). However, the spatial scale most strongly related to density was different for unmarked versus marked fish. In unmarked fish, embayment was the spatial variable that produced the best model by a relatively wide margin (AIC weight = 1; a model with region is second by 32 AIC units). While embayment also ranked high in density models for marked fish, the larger spatial grouping of basin produced the best model, though by a less wide margin (AIC weight = 0.85; model using embayment is second by 3.4 AIC units). Also, month as a single factor ranked much higher for marked fish (third in marked density models versus seventh in the unmarked density models), suggesting more uniformity in seasonal density distributions in hatchery fish throughout the study area.

In contrast to Skagit Bay in 2002, the 2003 data, collected at a much larger spatial scale, showed strong geographic patterns in proportion of unmarked fish (Table 3.3). Geographic factors were in the top six of ten models considered, and month as a single factor ranked lowest of all. Embayment was particularly important, and was present in the top two models (the additive month + embayment model is best, separated from the model with a month by embayment interaction term by 2.6 AIC units). Models that include region (north and south portions of the study area [Figure 3.3]) were next, but by a relatively large margin of over 23 AIC units. The strength of embayment as a factor in the models is consistent with expectations that seasonal densities and relative abundances of marked and unmarked fish would be a function of the wild population status, habitat conditions (quality and quantity from headwaters to estuary), and hatchery practices in a given river system.

Table 3.3. Summary of AIC ranking of ten ANOVA models of month and site vs. densities of unmarked and marked Chinook, and proportion of unmarked Chinook in greater Puget Sound, 2003 (M = Month, E = embayment, B = basin, R = region). See text for explanation.

Unmarked Chinook Density				Marked Chinook Density				Proportion Chinook Unmarked			
Model	AIC	Δ_i	AIC Weight	Model	AIC	Δ_i	AIC Weight	Model	AIC	Δ_i	AIC Weight
M + E + M * E	-756.15	0.00	1.00	M + B + M * B	-695.69	0.00	0.85	M + E	-887.51	0.00	0.78
M + R + M * R	-724.19	31.96	< 0.01	M + E + M * E	-692.26	3.43	0.15	M + E + M * E	-885.43	2.57	0.22
M + E	-696.70	59.45	< 0.01	M	-655.73	39.96	< 0.01	M + R	-860.96	27.04	< 0.01
M + B + M * B	-680.26	75.89	< 0.01	M + E	-653.40	42.29	< 0.01	M + R + M * R	-859.64	28.36	< 0.01
M + R	-659.10	97.05	< 0.01	M + R + M * R	-648.65	47.04	< 0.01	M + B + M * B	-856.83	31.17	< 0.01
M + B	-628.83	127.32	< 0.01	M + B	-643.71	51.98	< 0.01	M + B	-846.63	41.37	< 0.01
M	-595.63	160.52	< 0.01	M + R	-636.13	59.56	< 0.01	E	-837.12	50.88	< 0.01
E	-573.50	182.65	< 0.01	E	-558.79	136.90	< 0.01	R	-815.41	72.59	< 0.01
R	-562.26	193.89	< 0.01	B	-554.20	141.49	< 0.01	B	-800.84	87.16	< 0.01
B	-536.43	219.72	< 0.01	R	-548.74	146.95	< 0.01	M	-774.59	113.41	< 0.01

To look further at the effect of location on the observed densities and relative abundances in 2003, 45 linear regression models (15 each for marked density, unmarked density, and proportion unmarked) were analyzed by AIC. Seasonal residuals from quadratic fits of densities and relative abundances to month were fit to all combinations of latitude, salinity, temperature, and depth. With dominant seasonal effects removed, latitude was overwhelmingly important in models of densities of both marked and umarked fish (Figure 3.17). Depth had a minor association, and salinity and temperature were negligible. However, salinity ranked just as high as latitude in proportion unmarked models, and temperature also had a strong association with residuals from the regression models (Figure 3.17).
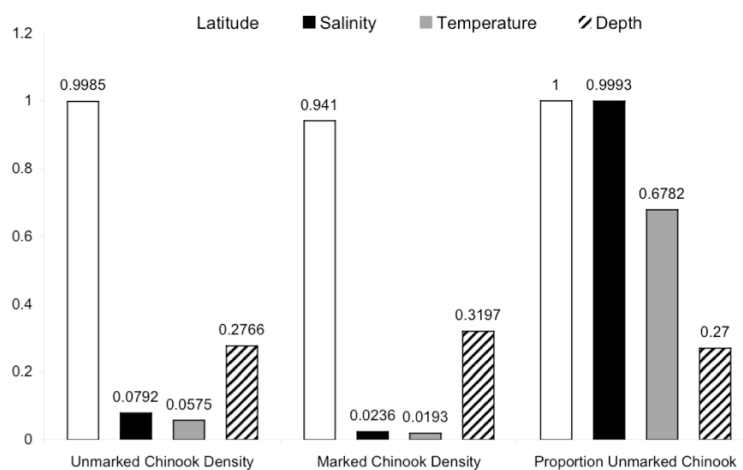
Figure 3.17. AIC-determined relative variable importance (RVI) for environmental attributes of sites in regression models of residuals from seasonal density and proportion unmarked models (quadratic regression); Greater Puget Sound, 2003. See text for explanation.

The importance of latitude and salinity could be the result of local oceanographic differences among areas that give rise to different environmental characteristics, and consequently, differential use of the regions by juvenile Chinook. While temperature and depth of sampling sites were fairly uniform across latitude, salinity was not (Figure 3.18). The northern region of the study area (including both the Skagit and Nooksack river estuaries) receives much more fresh water than central and southern Puget Sound. Such differences could result in more favorable habitat conditions (e.g., environmental conditions, prey base) (Levings 1994) for extended rearing before migration to sea via the Strait of Juan de Fuca or the Strait of Georgia. As noted previously for Skagit Bay in 2002, the importance of salinity (but in the opposite direction) in the proportion unmarked models could also be remaining seasonal effects (i.e., those not captured by the curve fit of density and proportion unmarked on month) since salinity and temperature differences across the season (Figures 3.6 and 3.7) would be differentially associated with wild fish simply because of their broader seasonal distribution and more extensive use of shallow habitats (Beamer and Rice, unpublished data), regardless of any direct effects of salinity or temperature on density. Geographic differences in hatchery releases may also be influencing the environmental variable results but could not be associated with individual sites for analysis.

Overall, the AIC results from 2002 and 2003 juvenile Chinook density data demonstrated that marked and unmarked fish had some similarities in seasonal and geographic patterns of habitat use, illustrated by the weaker relationship between the month by location interaction terms and proportion unmarked compared with absolute densities). However, the dominance of the month effect in all models, especially the proportion unmarked models, indicated that while seasonal patterns were strong in both marked and unmarked fish, the seasonal density distributions of marked and unmarked fish were quite different. Although not as dramatic as these seasonal patterns, densities and proportion unmarked were also related to geographic

location, and such relationships were much more apparent at a larger spatial scale that included multiple river systems.
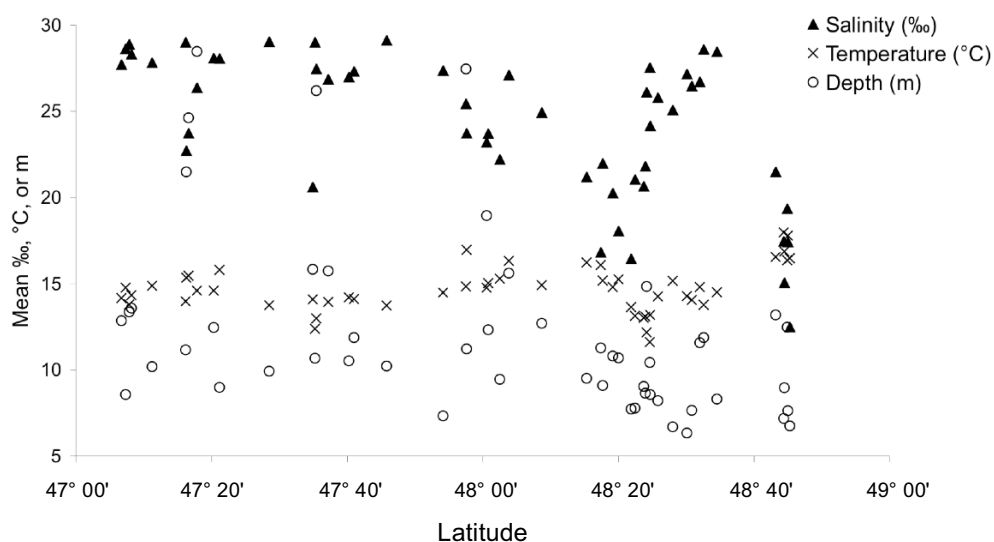
Figure 3.18. Mean salinity, temperature, and depth by latitude at 52 Greater Puget Sound sites in 2003.

Chinook Length

Chinook length generally increased over the season, in Skagit Bay during 2002, ranging from approximately 40 mm in the few fish captured in winter and early spring to over 120 mm in the fall (Figure 3.19). One obvious exception to this pattern was the abrupt increase and subsequent decrease in spring presumably as a result of the influence of yearling, or "stream type" fish that rear in freshwater for a year before migrating to sea, probably spending little time in the estuary (Simenstad et al. 1982, Healey 1991). Similar seasonal patterns were observed in greater Puget Sound during 2003 (Figure 3.20). At nearly all sampling locations and times, marked fish tended to be larger than unmarked fish, with more narrow size distributions and a higher rate of increase in mean size of captured fish over the season (Figures 3.19-3.20).
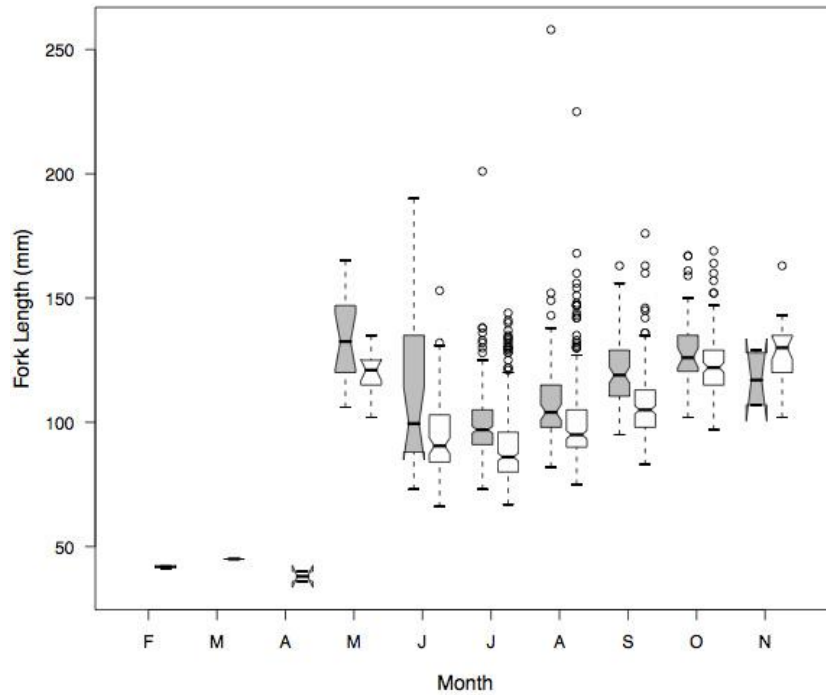
Figure 3.19. Fork lengths of all juvenile marked (grey boxes) and unmarked (white boxes) Chinook captured in Skagit Bay in 2002. Boxes and whiskers indicate 5th, 25th, 50th (median), 75th, and 95th percentiles; dots indicate points outside the 5th and 95th percentiles; notches indicate 95% confidence intervals around the median. Where boxes are turned back, confidence intervals are outside the interquartile range. The few fish in February, March, and April are unmarked fry migrants. The abrupt increase in May and subsequent decline by July is the result of relatively few (and presumably transient) yearling fish.
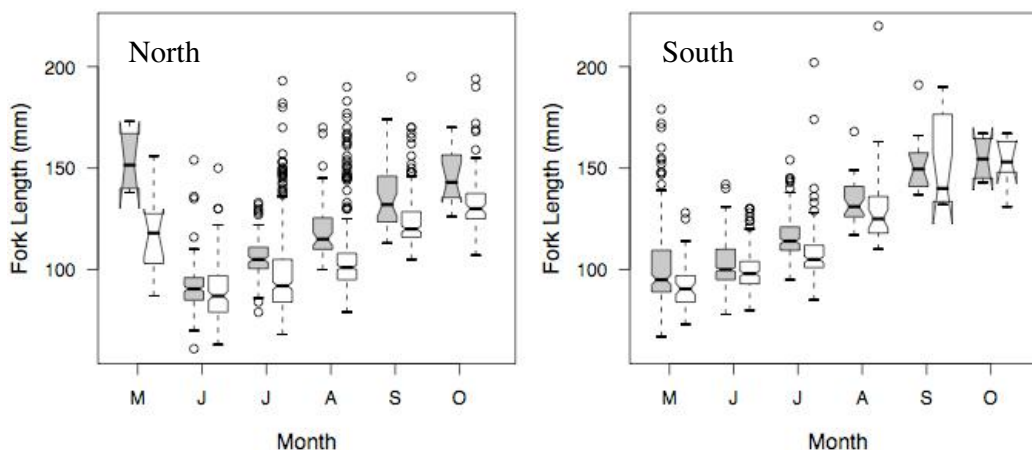
Figure 3.20. Fork lengths of all juvenile marked (grey boxes) and unmarked (white boxes) captured in the northern (left) and southern (right) regions of the study area in 2003. Boxes and whiskers indicate 5th, 25th, 50th (median), 75th, and 95th percentiles; dots indicate points outside the 5th and 95th percentiles; notches indicate 95% confidence intervals around the median. Where boxes are turned back, confidence intervals are outside the interquartile range.

Analysis of 2002 length data from Skagit Bay showed that none of the three individual factors of month, site, and mark were dominant individually, but all were related to length (the top three models evaluated by AIC include all three) (Table 3.4). The importance of the interaction between site and month in the top model (an AIC distance of 49 units from the next best model) was likely a consequence of changing habitat use by migrating and growing fish over the season. In 2003, AIC analysis of eleven ANOVA models using length data from north and south regions of the study area (Table 3.5) also showed that month, geographic location, and mark were all important in combination, but the relative influence of mark as a factor appeared different between north and south regions. In the north, mark was in the top five models (the best model included a month by mark interaction term), and the AIC

distance of 15 among these models was relatively small. Mark was less important in the south, and the model with a month by embayment interaction term was the best by a wide margin (46 AIC units). The second best model included no mark factor at all. These results coincide with more wild fish in the northern areas such as the Skagit, more hatchery dominated systems in the south, and more uniformity in hatchery vs. wild fish, although incomplete marking of hatchery fish outside the Skagit may have obscured some actual differences between hatchery and wild fish.

Table 3.4. Summary of AIC ranking of eleven ANOVA models of month, mark, and site vs. lengths of Chinook in Skagit Bay, 2002 (lower AIC score is better). M = month, X = mark, S = site. See text for explanation.

Model	AIC	Δ_i	AIC Weight
M + S + X + M*S	-8078.16	0.00	1.000
M + S + X + M*X	-8029.24	48.92	< 0.001
M + S + X	-8026.31	51.85	< 0.001
M + S + M*S	-7986.33	91.83	< 0.001
M + X + M*X	-7983.84	94.32	< 0.001
M + X	-7981.34	96.82	< 0.001
M + S	-7924.46	153.70	< 0.001
M	-7861.38	216.78	< 0.001
X + S	-7285.05	793.11	< 0.001
S	-7267.89	810.27	< 0.001
X	-7237.80	840.36	< 0.001

Regression analysis of 2002 length data from probable subyearlings (obvious yearlings removed) showed that the change in size over time in marked fish was greater than that in unmarked fish (Figure 3.21). In the larger area in 2003, data showed a similar pattern between marked and unmarked fish in the north, but marked and unmarked fish in the south were barely distinguishable from each other, or from marked fish in the north (Figure 3.22). The difference in these slopes could be the result of differences in growth rate, but also because of differences in the demographics of wild and hatchery fish. Hatchery fish are more uniform in life history type and size, and the initiation of their outmigration is more compressed in time than for wild fish. Consequently, wild fish probably enter the lower end of the neritic size distribution (after transitioning from shallower and "upstream" habitats) over a longer period of time, bringing down the overall mean length. Predation that differentially removes fish from the lower end of the size distribution (e.g., Parker 1971) could also result in differences in mean length between marked and unmarked fish.
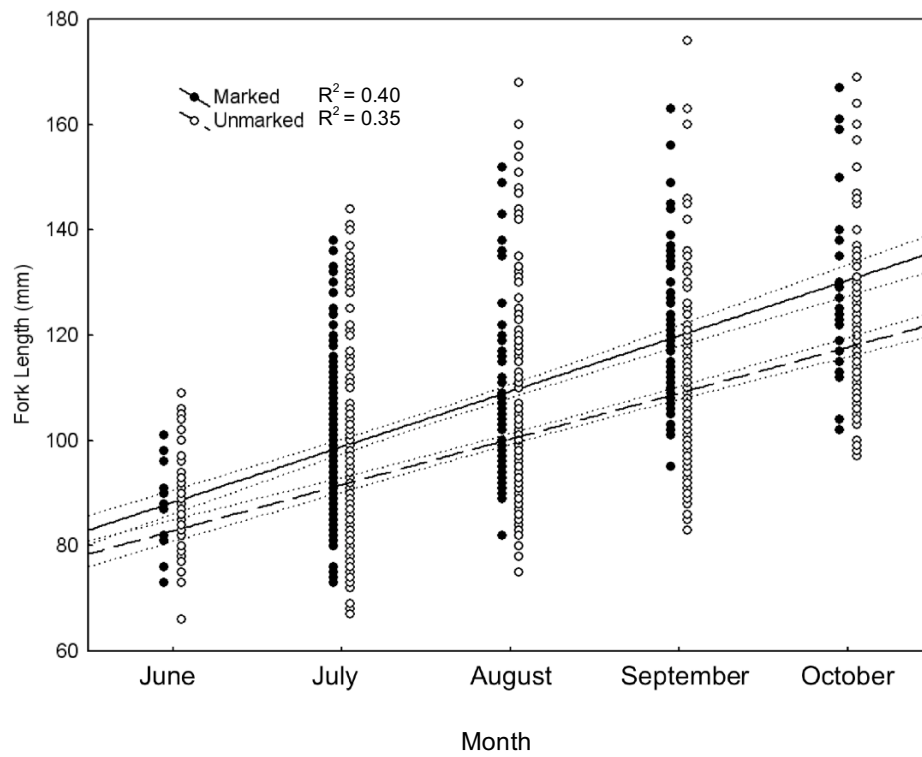
Figure 3.21. Linear regressions (with 95% confidence intervals) of fork lengths over time for juvenile marked (filled circles and solid line) and unmarked (open circles and dashed line) captured in Skagit Bay from June to Ocober in 2002 (obvious yearlings removed). Slopes statistically different (p = 0.020).
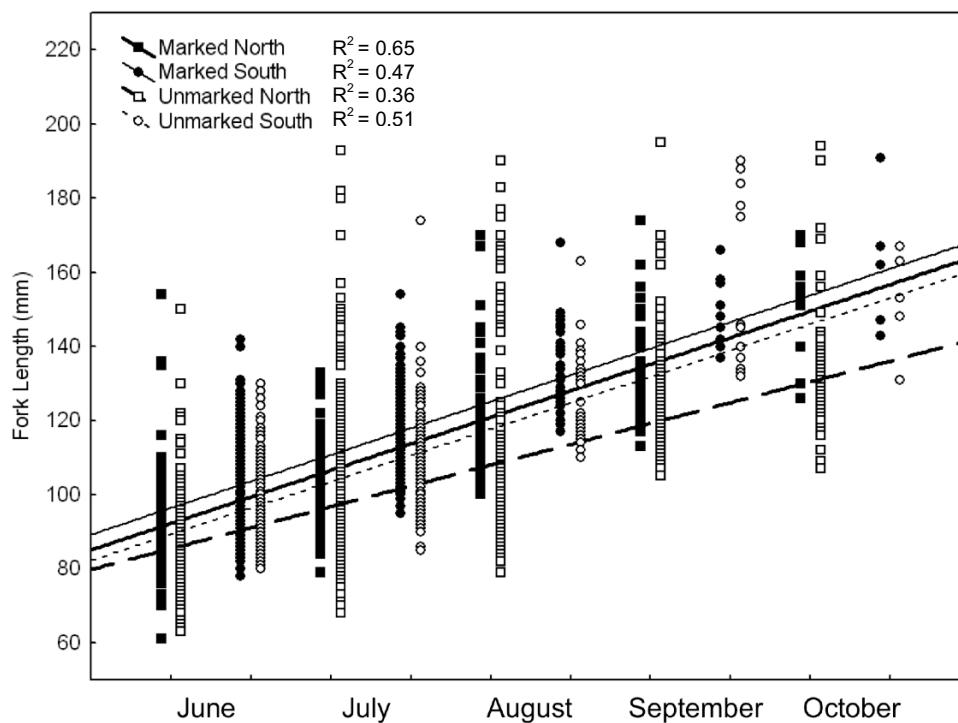
Figure 3.22. Fork lengths of juvenile marked (filled symbols and solid lines) and unmarked (open symbols and dashed lines) Chinook captured in the northern (circles and thick lines) and southern (squares and thin lines) regions of the study area in 2003 (obvious yearlings removed). No statistically significant differences exist among slopes except that unmarked north was different from all three other groups ($p \leq 0.003$).

Table 3.5. Summary of AIC ranking of eleven ANOVA models of month, mark, and site vs. lengths of Chinook in greater Puget Sound, 2003 (lower AIC score is better). M = month, X = mark, E = embayment. See text for explanation.

North (Bellingham Bay, Padilla Bay, Skagit Bay)				South (Snohomish River estuary, Elliott Bay, Commencement Bay, Nisqually Reach)			
Model	AIC	Δ_i	AIC Weight	Model	AIC	Δ_i	AIC Weight
M + E + X + M*X	-8902.27	0.00	0.53	M + E + X + M*E	-6745.82	0.00	1.00
M + E + X + M*E	-8901.52	0.75	0.37	M + E + M*E	-6699.86	45.96	< 0.01
M + X + M*X	-8898.82	3.45	0.10	M + E + X	-6521.70	224.12	< 0.01
M + E + X	-8891.68	10.59	< 0.01	M + E + X + M*X	-6516.58	229.24	< 0.01
M + X	-8887.25	15.02	< 0.01	M + X	-6493.73	252.09	< 0.01
M + E + M*E	-8805.71	96.56	< 0.01	M + X + M*X	-6490.92	254.90	< 0.01
M + E	-8799.45	102.82	< 0.01	M + E	-6464.93	280.89	< 0.01
M	-8784.55	117.72	< 0.01	M	-6452.50	293.32	< 0.01
E	-7972.81	929.46	< 0.01	X + E	-6099.70	646.12	< 0.01
X + E	-7970.96	931.31	< 0.01	E	-6087.96	657.86	< 0.01
X	-7944.44	957.83	< 0.01	X	-6015.25	730.57	< 0.01

Coded Wire Tagged Chinook

Density of CWT-marked Chinook in 2002 followed the same seasonal distribution of marked fish overall, and represented at least eleven different hatchery stocks, including one from the Chilliwack River in British Columbia and one from the Dungeness River on the Olympic Peninsula in Washington State. The majority of individuals were from the Skagit River system, but proportions of CWT fish from outside the Skagit generally increased over the season (Figure 3.23). In 2003, abundance of CWT-marked Chinook also followed the same seasonal pattern as marked fish overall, and fish from a total of nineteen different hatcheries were captured, again including one fish from the Chilliwack River in British Columbia and one from the Dungeness River on the Olympic peninsula. All basins contained juveniles from numerous source populations, but mixing was greatest in the middle portion of the study area (Main and Whidbey Basins), with little exchange of fish between the most northern (Rosario) and southern (South Sound) areas (Figure 3.24). Patterns of CWT fish distribution by capture location (Table 3.6) and time since release (Table 3.7) suggest that fish are overall most likely to be captured close to their origin but less so over the season, and that time since release increases with distance from origin. Average time since release varied by life history type and origin. The few yearling fish captured averaged 24 days since release, about half that for subyearlings (54 days). Fish from central and southern Puget Sound had less time since release, possibly reflecting different rearing and migratory patterns that may have moved these fish out of the study area sooner than in the north. Time since release in these neritic fish was generally longer than for those captured in beach seines in other studies (Brennan et al. 2004, Duffy et al. 2005).
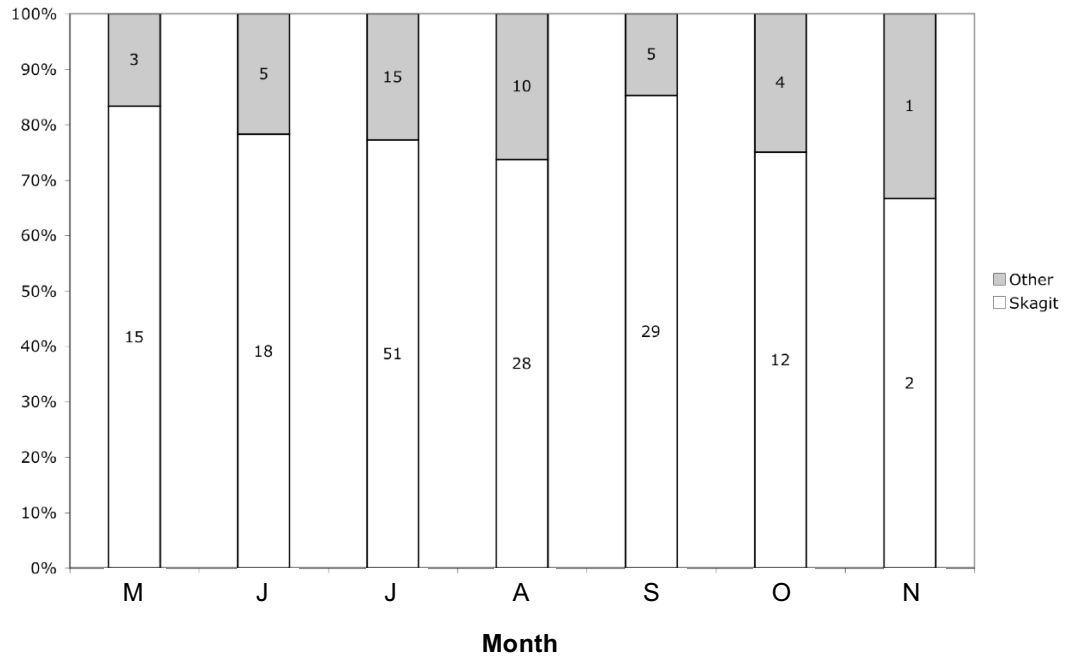
Figure 3.23. Percentage of coded wire tagged fish from the Skagit (white) and outside (white) the Skagit captured in Skagit Bay in 2002.
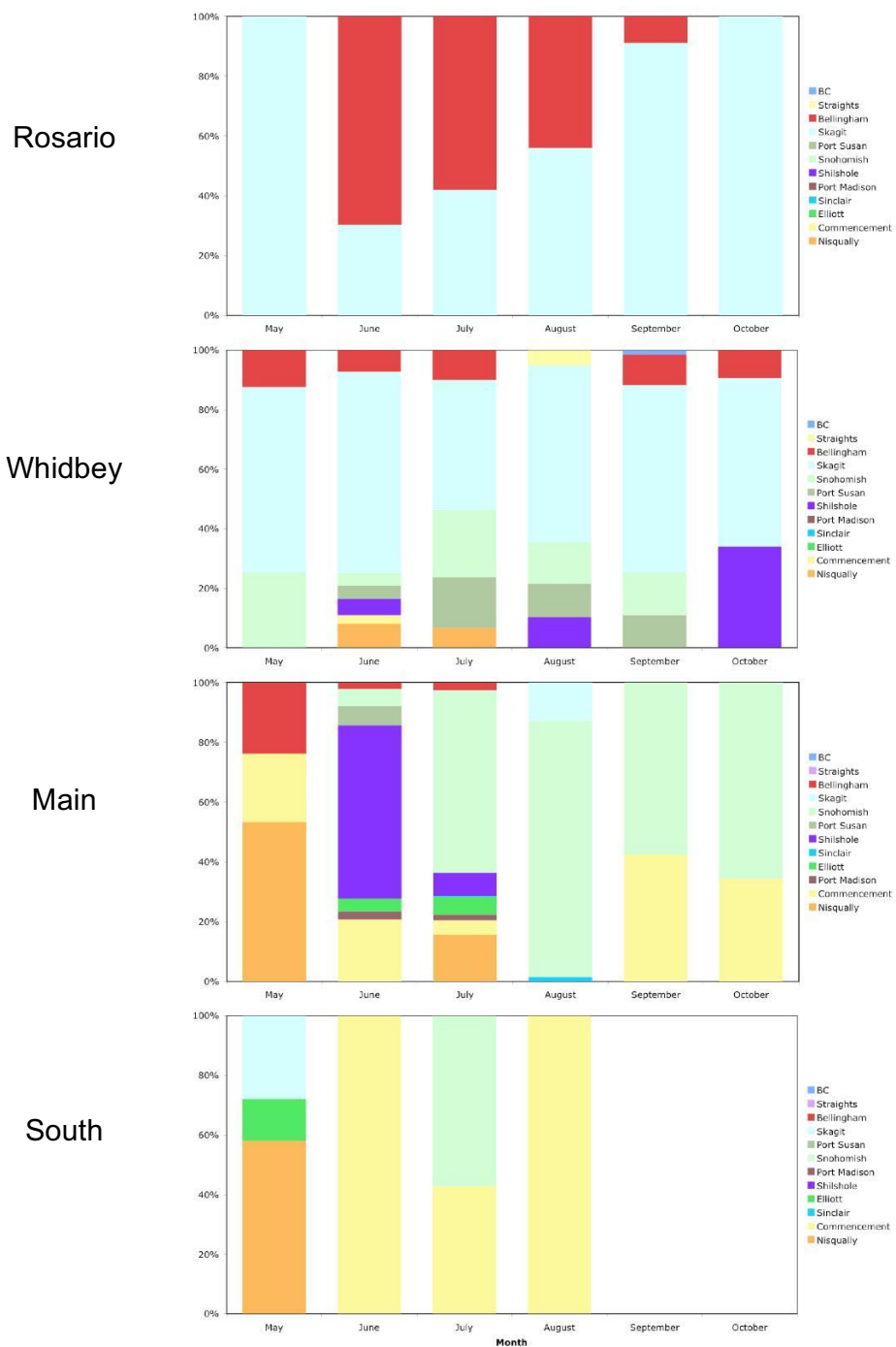
Figure 3.24. Percentages of coded wire tagged (CWT) Chinook from various embayments of origin captured in the four basins (origins [graphs] and capture basins [colors] arranged from north to south). Data adjusted for total release size of each CWT group.

Table 3.6. Percentage of coded wire tagged Chinook originating from each basin that were captured in the four basins in 2003. Values are adjusted for fishing effort. Bold values indicate where origin and capture areas were the same.

Capture Basin

Origin	Rosario	Whidbey	Main	South
Rosario	**68**	24	8	0
Whidbey	23	**56**	19	1
Main	0	11	**76**	14
South	0	11	67	**22**

Table 3.7. Average number of days since earliest possible release for CWT fish by basin of origin and basin of capture in 2003. Bold values indicate where origin and capture location were the same; dashes indicate combinations of fish origin and capture location not encountered.

Capture Basin

Origin	Rosario	Whidbey	Main	South	Total	
					Subyearling (273 fish)	Yearling (6 fish)
Rosario (62 fish)	**54**	72	43	—	58	—
Whidbey (163 fish)	60	**57**	49	55	56	24
Main (37 fish)	—	80	**47**	35	49	—
South (16 fish)	—	40	31	**27**	31	—

Discussion

Juvenile Chinook salmon are found in neritic environments of greater Puget Sound during much of the year, especially in spring and summer, but seasonal density and fish size differ considerably by location and whether fish are marked (known hatchery) or unmarked (majority naturally spawned). The coded wire tag (CWT) results show considerable mixing of populations and movement by individuals within the study area, similar to other recent results from central Puget Sound (Brennan et al. 2004, Duffy et al. 2005, Fresh et al. 2006).

Most striking are differences between the seasonal density distributions of marked and unmarked Chinook in areas such as the Skagit, where substantial populations of wild fish exist, hatchery inputs are relatively low, and nearly complete marking of hatchery fish allows more conclusive field identification of wild and hatchery fish. These contrasts suggest more extensive use of estuarine environments by wild fish than by hatchery fish. My results are consistent with those from two other studies. In the Campbell River estuary, British Columbia (Levings et al. 1986) estuarine residence times of wild Chinook (typically 40–60 days) were approximately double those of hatchery fish. In the Columbia River estuary, reconstruction of historical and contemporary abundances of juveniles (Figure 3.1; Burke 2004) showed more protracted seasonal distributions in historic populations than in contemporary, hatchery-dominated populations.

Differences in the seasonal density of juvenile Chinook salmon among geographic areas suggest that overall, these fish use neritic waters in central and southern Puget Sound for a shorter period of time than in northern Puget Sound. Beach seine catches in 2002 from southern (Nisqually) and more northern (the Snohomish River estuary) areas of Puget Sound also showed relatively high peak catches and more rapid declines in catch in the south compared to the north (Duffy et al. 2005). The large-scale geographic contrasts in the 2002 study were not as strong as those reported here, probably because my results spanned a larger spatial scale (both of the areas in

the Duffy et al. [2005] study would be in the southern half of this one), and I employed offshore sampling and more sampling effort (more sites and time points).

The most likely explanation for these patterns is differential migration or mortality by location (e.g., north vs. south) or by fish origin (hatchery or wild), and could be the result of a variety of factors, including:

1) Hatchery practices. Hatchery rearing environments and release practices, as well as genetics in hatchery fish, often result in larger, more uniform fish, and higher local densities, all of which could contribute to more rapid migration. In fact, rapid migration to sea is a stated goal of many hatchery programs (WDFW and PSTT 2004). Also, because of artificially lowered mortality at early life stages in the hatchery and little, if any, experience with predators, these fish probably experience higher mortality rates than wild fish in their estuarine and early marine phase. Central and south Puget Sound received more hatchery fish and have fewer wild fish than the northern half of the study area so either of these phenomena, if present, would be more apparent in those areas.

2) Geography and hydrology. Because of the configuration of Puget Sound all Chinook from central and southern Puget Sound and going to sea must migrate north and pass through Admiralty Inlet, Deception Pass, or Swinomish Channel. Of these migration routes, Admiralty Inlet is the most direct to the sea. Admiralty Inlet is also the conduit for the vast majority of water exchange between Puget Sound and the Straits of Juan de Fuca and Georgia, with a net seaward transport of surface waters, possibly facilitating seaward movement of fish. Deeper surface trawl surveys in more offshore waters in the Strait of Georgia and central Puget Sound indicate that Admiralty Inlet is the primary migration route for Chinook and coho salmon from central and southern Puget Sound (Sweeting et al. 2003a). Also, the further south in Puget Sound, the further the migration route to sea, and the more time or speed required to complete it. These circumstances may

require fish to leave their basin of origin sooner and migrate north. Either of these influences could move fish out of the southern half of the study area sooner than in the northern region.

3)   Habitat extent and quality. The more that habitat is limited in amount or quality (e.g., accessibility, sufficient food supply, high predator densities, physiological suitability, etc.) for juvenile Chinook, they are more likely to migrate rapidly or die. Many natural and anthropogenic influences affect habitat conditions. For example, Central and southern Puget Sound differ oceanographically from the Whidbey basin, receiving far less freshwater, and are also more hatchery influenced and urbanized.

The observed differences in densities of juvenile Chinook salmon among regions, and the differences in proportion of unmarked fish, most likely result from a combination of these factors, as indicated by the statistical importance of interaction between month and capture location. One implication of these patterns is that perhaps we should have different ecological expectations for different areas of Puget Sound, and consequently, different approaches and priorities for conservation and restoration among areas. Most important, though, is that these differences raise questions about the effects of human activity on the Puget Sound ecosystem, and argue for thorough evaluation of its ecological consequences. Prolonged, large-scale hatchery production is associated with declining populations of wild Chinook in the Columbia River system (Levin et al. 2001) and replacement (rather than augmentation) of wild salmon by hatchery salmon in British Columbia (Hilborn and Winton 1993, Sweeting et al. 2003b) and Alaska (Hilborn and Eggers 2000) (But see Wertheimer et al 2001, Hilborn and Eggers 2001, and Beamish and Sweeting 2007). Anthropogenic degradation of estuarine environments is associated with reduced survival to maturity in Chinook salmon in coastal Oregon (Magnusson and Hilborn 2003). Given these results, and the similar hatchery practices, environmental changes, and wild Chinook declines in Puget Sound, understanding the estuarine

ecology of wild and hatchery Chinook in the context of human alterations of the entire Puget Sound ecosystem should be a high priority for Puget Sound science and management. Although the importance of such an understanding is increasingly acknowledged (WDFW and PSTT 2004, Mobrand et al. 2005, SSPS 2005), little field and laboratory work is occurring to develop it.

The work that has been done provides some possible explanations for the causes and consequences of the patterns observed in this study, and also suggests some likely changes in these patterns that might result from management actions such as habitat restoration and changes in hatchery practices. Loss and degradation of estuarine wetlands in Puget Sound appear to have greatly restricted the rearing capacity for wild Chinook. In the delta of the Skagit River estuary, for example, evidence of density-dependent movement and size has been reported (Beamer et al. 2005, Beamer and Greene in prep). This observation raises the question of whether we would expect to see changes in existing patterns of estuarine habitat use by juvenile Chinook in response to restoration of estuaries. Some evidence suggests that we will. After extensive dike removal in salt marshes of the Salmon River in Oregon, seasonal distributions of juvenile Chinook broadened, and overall presence of juvenile Chinook in the estuary increased (Bottom et al. 2005a).

While estuarine restoration is a relatively recent and modest attempt to address declining salmon populations, the primary management tool—hatcheries—have been operating at industrial scale for a century with scant consideration of their effects on estuarine receiving environments (Beamish et al. 2003, HSRG 2004, Bottom et al. 2005b). Size, density, and temporal and spatial distribution of larval and juvenile fishes are fundamental factors in food web dynamics (Cushing 1990, Houde 1997, Rice et al. 1997, Cowan et al. 2000). Hatchery practices change all of these basic biological characteristics in juvenile salmon populations (White et al. 1995, Flagg et al. 2000, Weber and Fausch 2003), often intentionally (WDFW and PSTT 2004). As a result, hatchery fish have direct ecological implications for wild Chinook and other biota, especially during periods of unfavorable environmental conditions (Cooney

and Brodeur 1998, Pearcy et al. 1999, Levin et al. 2001). For example, in contrast to the protracted diffusion of wild fish of multiple life history types into the estuary, hatchery practices typically result in the abrupt appearance of larger, more uniform fish in artificially high densities. This could alter food webs by hatchery fish either feeding at different times or trophic levels than historical and contemporary wild populations because of migration timing and size-related prey preferences, or by "grazing down" available food because of high densities. Such scenarios have been suggested by several authors (Reimers 1973, Healey 1991, Beamish et al. 2003, Duffy 2003, Ruggerone and Nielsen 2004), but no rigorous evaluation has been done. Some potential for prey limitation for juvenile chum and pink salmon has been shown in Pacific Northwest estuaries (Sibert 1979, Godin 1981, Simenstad and Salo 1982, Wissmar and Simenstad 1988), and apparent reductions of growth and prey consumption by juvenile wild Chinook did coincide with high densities of hatchery Chinook in the Duwamish estuary in Central Puget Sound (Nelson et al. 2004, Ruggerone et al. 2006). However, no rigorous studies have been done, and no conclusive evidence exists on estuarine prey limitation in juvenile Chinook, or prey resource competition between juvenile hatchery and wild Chinook. Besides competitive pressures, high densities of hatchery fish can attract predators, and may facilitate disease transmission, putting added pressure on wild fish.

In addition to potential adverse effects of high densities of hatchery fish, larger average length in hatchery fish could affect competitive interactions with wild fish in the estuary. Larger size is often associated with lower mortality in fishes (Peterson and Wroblewski 1984, Lorenzen 1996, Sogard 1997), especially in larval and juvenile life stages (Houde 1997, Cowan et al. 2000). In juvenile fishes, larger size is generally advantageous in avoiding predators (although avian predators often select larger individuals), resisting starvation, tolerating environmental extremes, and competing for food and space (Sogard 1997). Positive relationships between individual outmigration size and survival in salmon have been documented (Bilton 1984, Holtby et al. 1990, Beamish and Mahnken 2001, Beamish et al. 2004,

Ruggerone and Goetz 2004, Quinn et al. 2005). For outmigrant juvenile Chinook in the Columbia river, differences in length of approximately 5–10 mm (3–6%) in both hatchery and wild fish was related to survival to adulthood (Zabel and Williams 2002). The importance of differences in absolute, as opposed to relative, within-year, length is not as strong across years (Zabel and Achord 2004, Quinn et al. 2005), suggesting that intraspecific competition or predation vulnerability are important factors in determining survival. Higher growth rate in juvenile hatchery salmon of the same size at outmigration is also related to increased survival (Beckman et al. 1999).

During the ocean phase of the life cycle, reductions in growth and survival among Pacific salmon caused by density and ocean conditions, particularly during the first year at sea, are better demonstrated (Peterman 1984, Emlen et al. 1990, Pearcy 1992, Beamish and Bouillon 1993, Mantua et al. 1997, Cooney and Brodeur 1998, Hare et al. 1999, Pearcy et al. 1999, Ruggerone and Nielsen 2004). Hatchery Chinook from Puget Sound, for example, show reduced growth and survival during the alternating years of high pink salmon abundance (Ruggerone and Goetz 2004), although this pattern does not appear to hold for wild Chinook in the Skagit River system (Greene et al. 2005).

If the lower average size and rate of change over the season in unmarked Chinook observed in this study reflect competition with hatchery fish and reduced access to, and extent and quality of habitat, these factors could be having negative, long-term effects on wild Chinook salmon. Such effects would argue in support of habitat restoration and changes in hatchery practices to reduce adverse effects on wild Chinook in estuaries. However, these simple differences in average length and apparent growth (change in size of captured fish over time) between marked and unmarked fish should be interpreted cautiously, as the two groups are not necessarily directly comparable. The broader outmigration period in wild fish results in continued recruitment at the lower end of the length distribution over the season in wild fish. Also, hatchery salmonids may be competitively inferior to wild fish of the

same size (Flagg et al. 2000, Weber and Fausch 2003). Detailed analysis of individual otoliths in both juveniles and returning adults, preferably across a range of local environmental conditions and ocean regimes, is needed to provide a more rigorous assessment of differences in length and growth rates between hatchery and wild fish across Puget Sound.

Confounding Factors

Evaluation of the results from this study is complicated by two potentially important problems: uncertainty regarding fish origin (hatchery or wild) and sampling bias. Incomplete and variable marking of hatchery fish makes it impossible to conclusively identify hatchery individuals, and could mask or distort differences between hatchery and wild fish. For example, if the proportion of unmarked hatchery fish in Skagit Bay changes over time as a result of migratory fish from hatcheries with lower marking rates, some of the apparent differences between hatchery and wild fish could actually be the result of geographic differences in habitat use by different hatchery populations. Conversely, in hatchery-dominated regions such as southern Puget Sound, low density of wild fish and incomplete marking of hatchery fish could have obscured seasonal differences in abundance and length between hatchery and wild fish as I strongly suspect. Considering the potential importance of hatchery influences, and the extra cost and time imposed by marking techniques that do not allow for easy, non-lethal field identification (e.g., thermal or chemical marking of otoliths), marking by fin clips or coded wire tags of all hatchery fish released into Puget Sound should be mandatory.

Sampling bias in this study came from both site selection and fishing gear. Because the opportunistic, exploratory sampling design was nonrandom and involved repeated sampling of the same sites and river mouth estuaries, it is possible that areas not sampled (e.g., protected embayments in central Puget Sound) could be desirable rearing habitat for juvenile Chinook after they leave the river mouth estuaries and might be occupied for more of the season than the sites sampled.

However, 20 of the 52 sites sampled were outside of the estuaries, and each tow typically samples approximately 0.4 hectares (a substantial amount of area totaling 156 hectares in 2002, and 252 hectares in 2003), increasing the likelihood that these data are representative of broad patterns of neritic habitat use by juvenile Chinook across Puget Sound. Ideally, a more probabilistic sampling design that proportionally covered all the neritic waters of the Puget Sound/Georgia Basin system would provide a more complete and reliable assessment of the use of these environments by juvenile Chinook.

Changes in capture efficiency of the fishing gear with fish size, for example, could also have biased the results. Concurrent summer sampling with the same kind of townet used in this study and purse seines in Skagit Bay in 1972 (Stober and Salo 1973) and 2006 (C. Greene, NOAA Fisheries, unpublished data), and in Sinclair Inlet in 2002 (Fresh et al. 2006) showed that in catches of Chinook ranging from 70 – 150 mm average lengths in townet were approximately 10 mm smaller than in purse seines, and had a distribution skewed more toward the lower end of the size range. The effect of this bias against larger fish would presumably be greater for the relatively large, yearling fish in spring, and would increase on subyearlings as the season progresses and fish grow. Limited seasonal coverage in these other studies precludes evaluation of seasonal effects.

Implications for Assessment and Monitoring

Effective management of Puget Sound Chinook salmon requires the definition and detection of meaningful changes in the productivity and likelihood of long-term persistence of wild Puget Sound Chinook regardless of cause—human or natural, local or global. Measuring appropriate biological and environmental attributes will allow the monitoring of changes in the status of Puget Sound Chinook populations and, as our understanding improves, further explain the causes of those changes and better inform management choices. Ideally, monitoring and assessment would regularly check for adverse ecological effects from a growing human population, and

evaluate the efficacy of management actions, including habitat restoration and changes in hatchery and harvest practices. In the case of an anadromous metapopulation such as Puget Sound Chinook, which uses many different environments over a large geographic area, this will require sampling across life stages and habitat types at a variety of spatial and temporal scales. Such comprehensive monitoring is seen as a key element for improved salmon management (Walters 1997, Mantua and Francis 2004).

Sampling juvenile Chinook salmon in neritic environments provides unique information on the potentially critical late estuarine and early marine portion of the life cycle, and compliments information collected from other more commonly sampled environments. For example, results from this study combined with data from other habitat and gear types in the Skagit River estuary in 2002, illustrate similarities and differences in the transitions of juvenile wild and hatchery fish through delta, marine shoreline, and neritic environments (Figures 3.25 & 3.26; Beamer et al. in prep.). This differential use of habitat types (Figure 3.27; Beamer et al. in prep.), suggests that the potential for estuarine interactions between hatchery and wild fish increases as they move downstream and offshore. Thus, studies of neritic environments may be particularly informative in evaluating hatchery-wild interactions in the estuary, where a management goal has been to minimize interactions between hatchery and wild fish through temporal isolation by manipulating size and release timing of hatchery fish (WDFW and PSTT 2004). Another benefit from townetting comes from the fact that ecological influences on one life stage may not be apparent until later in the life cycle. Hence, the presence of juvenile Chinook in neritic environments through summer and into fall provides an opportunity to more fully evaluate the effects of "upstream" influences, such as habitat restoration in estuarine wetlands and freshwater habitats.
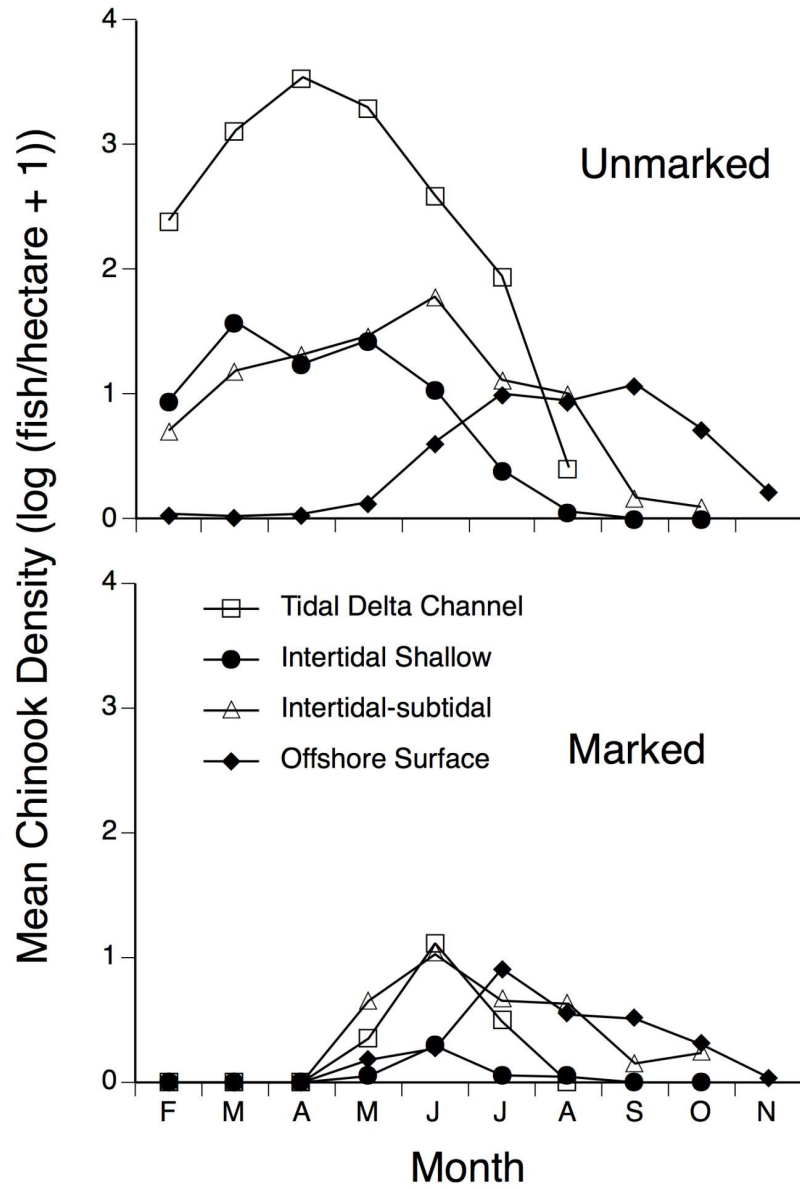
Figure 3.25. Mean density of unmarked (top) and unmarked (bottom) juvenile
Chinook in four different estuarine habitats in Skagit Bay, 2002 (data
from Beamer et al. in prep.).
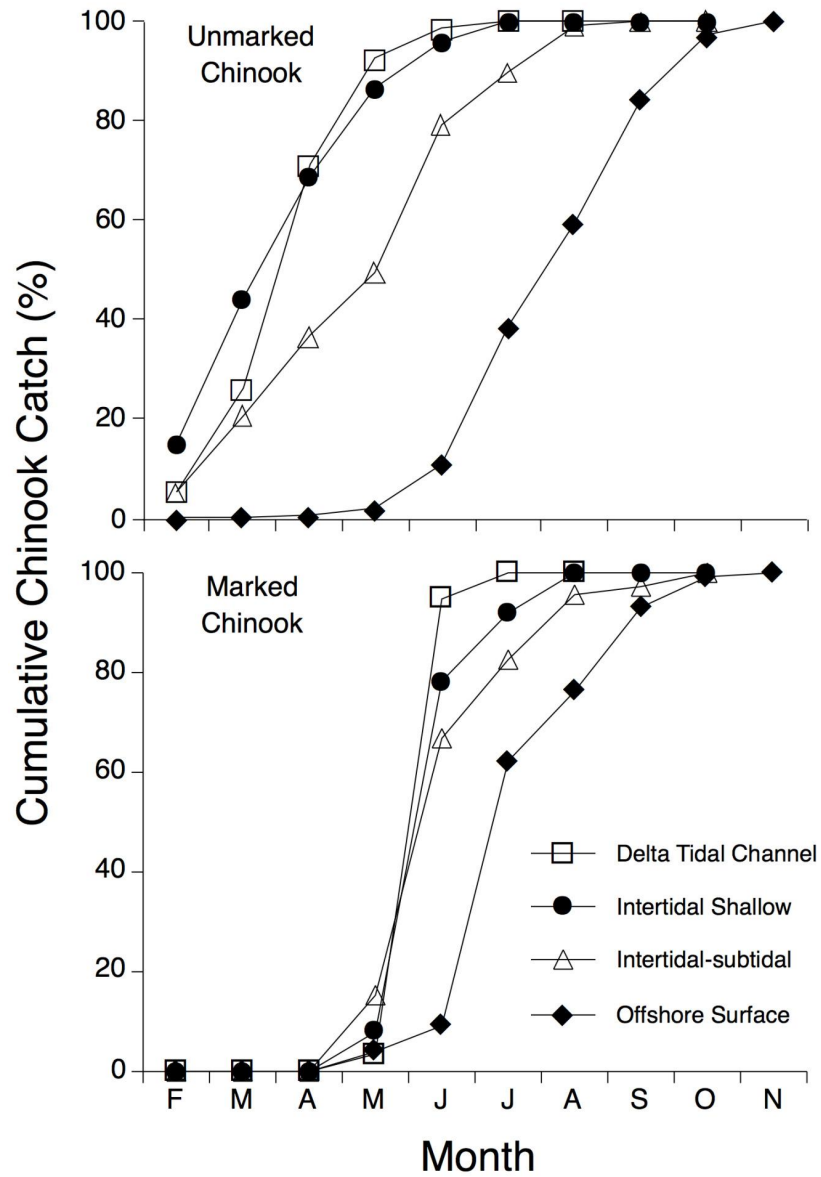
Figure 3.26. Cumulative mean density of unmarked (top) and unmarked (bottom) juvenile Chinook in four different estuarine habitats in Skagit Bay, 2002 (data from Beamer et al. in prep.).
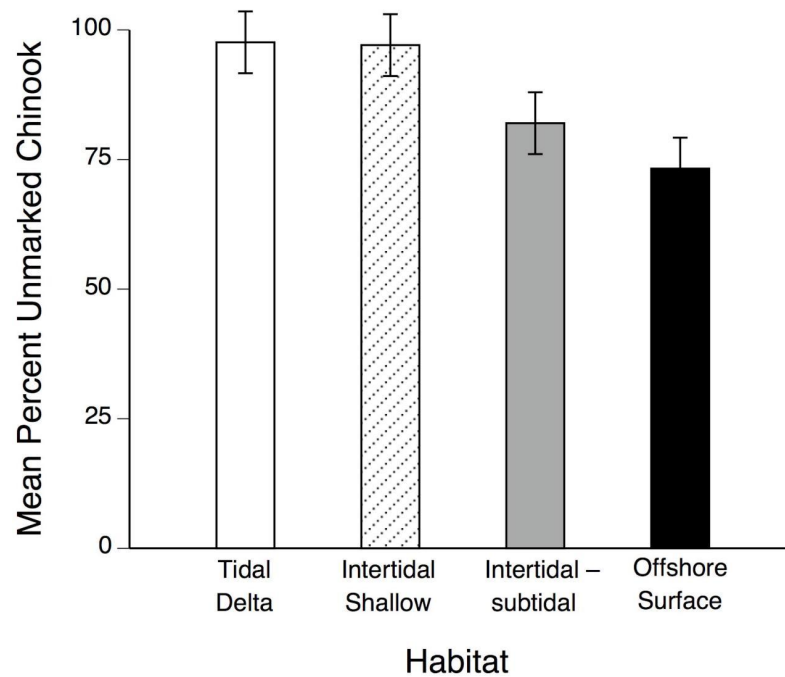
Figure 3.27. Mean percentage (± se) of unmarked juvenile Chinook in total catch in four different estuarine habitats in Skagit Bay, 2002 (Beamer et al. in prep.).

Appropriate biological response metrics are critically important, and ought to be both biologically meaningful, and have statistical qualities that maximize the power to detect differences. Cumulative seasonal density of juvenile Chinook is an integrative expression of both the abundance and life history diversity of salmon populations, and the productive capacity of the environment. By calculating such area- under-the-curve estimators, or "fish days," this annual presence can be consistently captured in a single number (Figure 3.16), and compared across years and among the various habitats and regions of Puget Sound (Beamer et al. 2005, Skalski 2005). Sampling across broad spatial and temporal scales is not only advisable on ecological grounds (Puget Sound Chinook is a large, anadromous metapopulation; Puget Sound is a large, complex ecosystem) but also on statistical grounds since increasing extensive (e.g., between site), as opposed to intensive

(e.g.,within site) effort, is typically more effective in optimizing sampling designs (Cochran 1977). The consistency of the distributions and variability of these seasonal density measures of marked and unmarked Chinook in Skagit Bay across years (Figures 3.8 and 3.13; Beamer et al. 2005), demonstrates the value of sampling at broad spatial and temporal scales in controlling for site-level variation and within-season shifts in peak abundance. The field sampling efforts to collect such information also provide the opportunity to gather valuable information on many more detailed aspects of juvenile Chinook ecology, including individual life history, size, growth, health (Rhodes et al. 2006) , physiology, and trophic relationships.

Concluding Thoughts

Physical and biological heterogeneity are fundamental in maintaining the integrity and continuity of ecosystems (Hutchings et al. 2000). In Puget Sound, human activity has reduced these traits in many ways. Managers have poured ever more uniform hatchery salmon into an ever more uniform (and shrinking) habitat landscape for over a century, paying little attention to the consequences. If wild Puget Sound Chinook salmon and the ecosystems on which they depend are to be protected and recovered, society must rigorously evaluate those consequences, and heed the results. This study takes a small step forward by providing current natural history information for a potentially critical part of the Chinook life cycle, and uses that information to suggest simple assessment and monitoring methods.

The results reported here are not surprising but are informative:

1) Juveniles use neritic estuarine environments for much of the year but the various areas of Puget Sound appear to differ in their role as habitat for juveniles.

2) Although wild and hatchery fish are similar enough to commingle, they are different enough that conclusions regarding the estuarine ecology of wild Chinook drawn solely from hatchery fish may be incorrect.

3)   Studying small, migratory fish in a large, dynamic ecosystem presents many challenges, but this study demonstrates that productive Sound-wide research and monitoring can be done with a reasonable amount of effort.

We cannot expect to recover wild Puget Sound Chinook if a) we don't know what's broken, and b) we don't check our work. Future efforts should build on this and other estuarine studies of juvenile Chinook to better understand the estuarine ecology of wild and hatchery Chinook throughout Puget Sound and the Georgia Basin, and to implement and refine estuarine monitoring and assessment methods. Particularly pressing is the need to evaluate when and how density-dependent processes in estuarine environments are influencing juvenile performance, especially with respect to hatchery practices and ocean conditions. Finally, while the need for monitoring is explicitly recognized in all prominent management plans for both wild and hatchery Chinook, support for estuarine monitoring of juveniles is limited at the project scale, and absent at the ecosystem scale. This is particularly unwise in situations where wild populations are at risk, and where considerable resources have been invested (in hatcheries and habitat restoration, for example) to ensure healthy and sustainable fisheries resources.

CHAPTER 4

Fish and Jellies in Puget Sound Surface Waters: Geographic and Seasonal Patterns of Assemblage Composition

Summary

Small pelagic fishes and gelatinous zooplankton, or "jellies," exert major top-down and bottom-up control in pelagic food webs. Increased abundance of jellies has been reported in marine and estuarine ecosystems around the world, often coincident with declines in small pelagic fishes, and sometimes associated with local human activity (e.g., fisheries, eutrophication, species introductions, substrate hardening) and climate change. Consequently, abundances of jellies and pelagic fishes may be useful indicators of coastal ecosystem condition. In Puget Sound, an oceanographically diverse and urbanized fjord estuary, understanding of pelagic ecology is poor, and biological monitoring of many key ecosystem components, including plankton and pelagic fishes, is negligible. To explore patterns of jelly and fish abundance in surface waters across four oceanographic sub-basins of greater Puget Sound, I used surface trawl data from 52 sites ranging from Bellingham Bay in the north, to Nisqually Reach, 185 km to the south. Sites were sampled opportunistically as part of a juvenile salmon study (Chapter 3), monthly during daylight, from May to September 2003. Taxonomic composition differed seasonally and geographically but geographic differences were more distinct. Biomass of jellies, and biomass and counts of five of thirty-six fish species encountered (Pacific herring [*Clupea harengus pallasi*], surf smelt [*Hypomesus pretiosus*], three-spine stickleback [*Gasterosteus aculeatus*], juvenile Chinook salmon [*Oncorhynchus tshawytscha*], and chum salmon [*Oncorhynchus keta*]) most heavily influenced assemblage similarities within basins and months. The environmental factors with the strongest relationship to biotic composition were latitude and water clarity, followed by weak relationships with salinity, depth, and temperature, suggesting that observed patterns were the result of unmeasured characteristics (e.g., water column structure, oxygen and nutrient levels, phytoplankton and zooplankton abundance and size distributions)

resulting from underlying oceanographic features (e.g., bathymetry, connectivity, wind and wave exposure) or other factors, including human activity. Absolute and relative abundance of jellies, hatchery Chinook salmon, and chum salmon decreased with latitude, whereas absolute and relative abundance of most fish species (including herring, surf smelt, and wild Chinook salmon) increased with latitude. Mean fish species richness also declined with latitude. These results document geographic and seasonal heterogeneity in the taxonomic composition in Puget Sound surface waters, and demonstrate the need for better understanding of natural and anthropogenic influences on pelagic ecology.

Introduction

Calls for more holistic approaches to coastal ecosystem management (Christensen et al. 1996, POC 2003, Pikitch et al. 2004) highlight the need for comprehensive ecological understanding and monitoring of estuaries. This requires sampling across taxa at a appropriate spatial and temporal scales (Weins et al. 1986, Levin 1992), and identifying and monitoring attributes of the biota that are responsive to natural and anthropogenic influences (NRC 1990, Hughs et al. 2005, Karr 2006). Such efforts are lacking in most coastal ecosystems, including Puget Sound, a fjord-estuary of high ecological and socioeconomic value, and affected in many ways by growing regional and global human populations.

Historical changes in Puget Sound biota include population extinction or decline in marine and anadromous fishes, marine birds and waterfowl, and marine mammals (West 1997, PSAT 2007b). Many of these animals occupy the pelagic zone, yet our understanding of the character and interrelationships of pelagic ecosystem parts (e.g., biota and water masses) and processes (water circulation, nutrient cycling, production of biomass, predation, migration, etc.) in Puget Sound is poor, particularly with respect to how human activity affects them.

Direct biological observations provide the most useful information in understanding ecosystems (Slobodkin 1994). Compositional attributes (e.g., number

of taxa, relative abundance of certain species or species groups) are especially valuable in diagnosing ecological condition (Karr 1991, Karr and Chu 1999, Hughs et al. 2005, Karr 2006) but are rarely used in Puget Sound monitoring and research. To explore patterns of macrofaunal assemblage composition in Puget Sound's neritic environment (pelagic surface layer of waters overlying the sublittoral zone), I used data collected opportunistically as part of an estuarine natural history study of juvenile Chinook salmon (*Oncorhynchus tshawytscha*) (Chapter 3). Such "bycatch" information is often overlooked in biological field surveys (when it is collected at all) but may provide valuable insights into the ecology and health of Puget Sound.

Research Questions

The specific research questions for this study were:

4) What are the geographic and seasonal patterns of macrofaunal composition in neritic Puget Sound?

5) How does macrofaunal composition differ among months and geographic areas of Puget Sound?

6) What macrofauna taxa make different months and areas similar or distinct?

7) Are measured environmental variables associated with the composition?

8) What biological attributes of pelagic macrofauna in neritic environments might be useful in monitoring and assessment of Puget Sound?

Answering these questions will improve our understanding of pelagic environments in Puget Sound from embayment to Sound-wide spatial scales, and assist in the development of future research and monitoring efforts.

Fishes and Jellies in Coastal Ecosystems

Small pelagic fishes and gelatinous zooplankton (pelagic cnidarians and ctenophores; hereafter referred to as jellies) are major components of pelagic food webs as secondary consumers preying on zooplankton (and occasionally, in the case

of some fishes, as primary consumers on phytoplankton), and as prey for many species (Mills 1995, Alaska Sea Grant 1997, Cury et al. 2000, Arai 2005). Consequently, these species exert crucial top-down and bottom-up control in pelagic food webs. Because jellies have far fewer predators than fishes, jellies are considered by some to represent alternate trophic pathways that prevent the flow of energy to higher trophic levels such as predatory fishes, birds, and mammals (Greve and Parsons 1977, Parsons and Lalli 2002). Interactions between fishes and jellies include competition for food, predation (primarily by jellies on fishes, especially eggs and larvae, but some fish species such as chum salmon [*Oncorhynchus keta*] prey on jellies), and commensalism (Purcell and Arai 2001, Lynam and Brierley 2007, Brodeur et al. in review).

Increased abundance of jellies, including the frequency and magnitude of seasonal "blooms," have been associated with oceanographic or climatic changes, and with anthropogenic disturbances such as over-fishing, eutrophication, species introductions, and increases in hard substrates (Mills 1995, Arai 2001, Mills 2001, Kideys 2002, Purcell in review). Inverse relationships between abundance of jellies and pelagic fishes have been recorded throughout the world (Mills 1995, CIESM 2001, Brodeur et al. 2002). Several attributes of jelly biology (e.g., short life span, sexual and asexual reproduction, passive feeding, low metabolic requirements) enable them to reproduce rapidly when resources become available, and also to tolerate certain environmental stresses better than fishes. In a low oxygen environment with low prey abundance, for example, jellies may simply shrink and drift where fish would starve and die. Jellies are also quite conspicuous in the environment, often dominating the catch in some commercial fisheries and research surveys. These characteristics, and the apparent increase in frequency and severity of jelly "blooms" around the world, is generating more interest in jellies as indicators of ecosystem condition (Brodeur et al. 2002, Hay 2006, Attrill et al. 2007, Purcell in review).

Puget Sound

Puget Sound is a fjord-estuary complex (many sub-estuaries exist within the larger Puget Sound estuary) with a productive pelagic environment (Strickland 1983), typical of coastal upwelling ecosystems. The ecology of pelagic environments in Puget Sound is presumably tied closely to physical forcing from the Pacific Ocean and freshwater runoff from the surrounding watersheds (Strickland 1983, Gargett 1997, Pinnix 1999), both of which influence hydrology, and the delivery of nutrients, organic matter, and biota. Pelagic ecology is also likely to be locally affected by many forms of human activity (Verity et al. 2002) including fishing, addition of nutrients and chemical pollutants, physical and hydrologic alterations in the surrounding watersheds (e.g., Simenstad et al. 1992), physical disturbance at the land-water interface (Chapter 2), species introductions (Cohen et al. 1998), the release of artificially propagated salmonids (see Chapter 3), as well as climate change at the global scale (Snover et al. 2005).

Although Puget Sound remains a biologically rich and productive ecosystem (PSP 2006, PSAT 2007b, Ruckelshaus and McClure 2007), dozens of Puget Sound fish and wildlife species have declined during recent decades (West 1997, PSAT 2007b), and pressure on the ecosystem from growing regional and global human populations is steadily increasing. While documentation of many of these declines is sufficient to signal problems and elicit serious concern on the part of resource managers and occasionally the public, understanding and documentation of the mechanisms, relative importance of, and interactions among, the causes is incomplete at best. Further, the estuarine ecology of many Puget Sound species is poorly understood, efforts to assess and monitor their condition are often small or nonexistent, and the efficacy of various management options is largely unknown (see Chapter 1). Together these problems undermine efforts to protect and recover the living systems of Puget Sound.

The condition of the pelagic environment is crucial to many Puget Sound species. Among the most dependant on, and important in, the pelagic environment are the

small pelagic fishes, including juvenile salmonids (*Oncorhynchus spp*), Pacific herring (*Clupea harengus pallasi*), surf smelt (*Hypomesus pretiosus*), and sand lance (*Ammodytes hexapterus*). Knowledge of the autecology of all these species in Puget Sound is limited, especially for herring, surf smelt, and sand lance. Environmental tolerances, biogeography, and demographics of Puget Sound populations—fundamental in understanding the role of these species in the Puget Sound ecosystem—are poorly understood in herring, and barely known at all in surf smelt, and sand lance. Instead, only spawning aggregations of adult herring are targeted in acoustic surveys, and estimates of spawning biomass are calculated from roe surveys (Stick 2005). No information is collected on adults (or other life stages) during the rest of the year. Surf smelt and sand lance are not monitored at all in Puget Sound, except for presence/absence of surf smelt and sand lance spawn on intertidal beaches (Penttila 1995), and some recreational catches of surf smelt (Lemberg et al. 1997). Further, research and monitoring on the structure (e.g., composition of plankton and fish assemblages) and dynamics (e.g., seasonal productivity, migration patterns, demographics) of pelagic ecosystems in Puget Sound, and their natural and anthropogenic influences, is nearly absent. Documenting seasonal and geographic patterns in the assemblage composition of pelagic macrofauna will improve our ecological understanding of Puget Sound, and may help develop useful monitoring and assessment tools.

This study used a Kodiak surface trawl, or townet, to measure biomass of jellies, and biomass and individual counts of fishes across much of Puget Sound. Although rarely used in Puget Sound since the early 1980's, neritic sampling with townets was used extensively to characterize pelagic fish assemblages, although much of the "Puget Sound" work was actually outside of Puget Sound proper (Stober and Salo 1973, Bax et al. 1978, Fresh 1979). These studies documented seasonal patterns of fish assemblage composition, migration, and food web relationships for multiple species, but were limited in geographic coverage, occurred over two decades ago, and did not collect data on jellies. Bringing townetting back into use today allows us

to take a contemporary look at some aspects of Puget Sound's pelagic zone, an environment widely regarded as imperiled but rarely explored and not well understood. Sampling a large geographic area allowed comparisons across a range of oceanographic characteristics and degrees of human influence.

Methods

Study Sites

Data for this study were collected opportunistically during estuarine natural history research on juvenile Chinook salmon (*Oncorhynchus tshawytscha)* that focused on neritic waters in river mouth estuaries (see Chapter 3). Secondarily, more marine areas in between river systems were sampled. Objectives for site selection were extensive geographic coverage alongshore (at depths sufficient for the fishing gear to be deployed without hitting bottom), accessibility by boats and fishing gear at any tidal stage, proximity to historical townet sampling sites (e. g., Stober and Salo 1973, Fresh 1979), and, in Skagit Bay, proximity to existing beach seine monitoring sites sampled by the Skagit River System Cooperative. Sites in the river mouth estuaries were selected to, at a minimum, sample the approximate center of the delta front and the two adjacent shorelines with the intent of sampling likely salmon outmigration and rearing areas just seaward of the habitats typically sampled by beach seining. Fifty-two sites were sampled covering an area from Bellingham Bay in the north, to Nisqually Reach, approximately 185 km to the south, and included six river mouth estuaries and several areas in between (Figure 4.1). Sites were assigned to "basins" commonly used to describe different oceanographic regions of Puget Sound (Burns 1985). Areas outside of Puget Sound proper (Padilla and Bellingham Bays) were assigned to a "Rosario Basin" for this study. Sampling was conducted during neap tide series to reduce the influence of tides on the spatial distribution of the biota. The neap tide series were allocated alternately to north and south halves of the study area, resulting in monthly sampling trips in the north and south separated by two weeks.
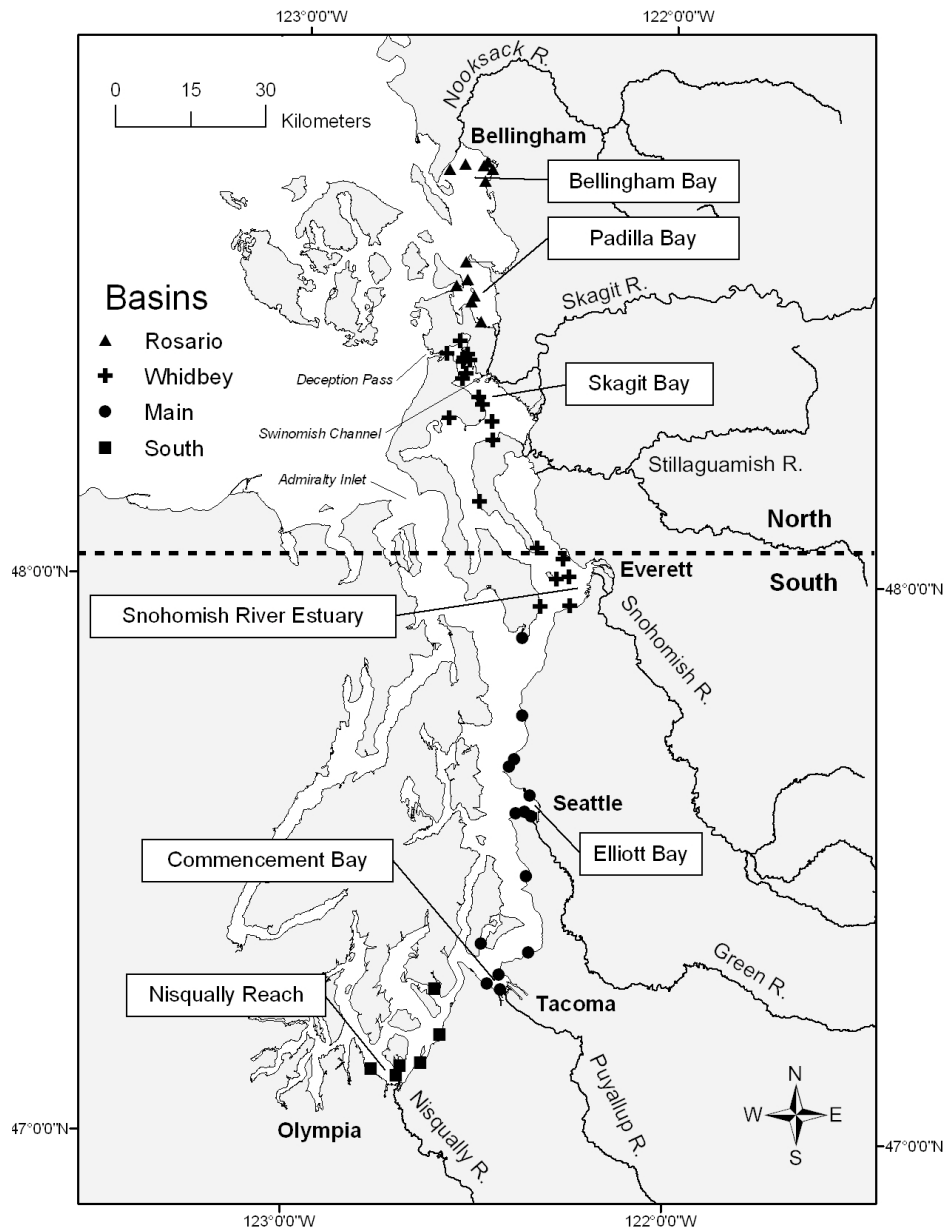
Figure 4.1. Sites sampled monthly from May to September 2003. Basins indicated by symbols; embayments marked by labels. Northern and southern areas were sampled on alternating neap tide series two weeks apart. Bellingham Bay was not sampled in September, and South Sound was not sampled in October.

Sample Collection and Processing

Monthly sampling from May through September in 2003, employed a 3.1 m high x 6.1 m wide Kodiak surface trawl, or "townet," deployed between two boats, each with a 15.2 m towline connected to a bridle on the net. Mesh sizes in the net range from 7.6 cm stretch in the forward section, and progressively smaller to 3.8 cm then 1.9 cm in the middle sections, and 0.6 cm in the codend. The primary vessel (13.7 m long, 174 hp inboard diesel) towed the left side of the net while trawling, and the second vessel (5.5 m long, 225 hp gasoline outboard) towed the right side. The net was towed at the surface for 10 minutes per tow, at 900-1000 rotations per minute (RPM) on the engine of the primary vessel and a typical towing speed of 2-3 knots through the water. Distance through the water was recorded with a mechanical flow meter (General Oceanics model 2030) deployed by the smaller vessel. Area swept was calculated as the distance traveled through the water multiplied by the width of the net opening. Two tows (in opposite directions) were made per site, and a total of 502 successful tows were completed in 251 site/month visits.

At the end of each tow, the net was closed via a purse line running along the top of the net opening, the boats came together side by side and the net was hauled onboard the larger vessel. The entire catch was then placed in tanks supplied with flowing water from the site, and the fish were identified, counted, and weighed by species, and all gelatinous zooplankton was combined and weighed. Surface water temperature and salinity measurements were taken during each tow using an electronic meter (YSI model 30) on water drawn continuously by the primary vessel's deck hose from a depth of approximately 1.2 m. Secchi depth was recorded once for each site/month visit using a 20 cm black and white disk.

Statistical Analysis

Statistical analysis focused on influences of month and oceanographic basin on total biomass and individual counts of fish species, and biomass of all gelatinous zooplankton (medusae and ctenophores combined). Secondarily, relationships

between biotic composition at each site/month combination and latitude, salinity, temperature, water depth, and secchi depth were considered. Longitude was not included because of the north-south orientation and narrow width of the study area. Biomass was the primary abundance measure because it was the only information collected on both jellies and fishes. Fish counts were also analyzed to compare the results based on the two measures of abundance.

Because Bellingham Bay was not sampled in September due to mechanical problems, only May, June, July, and August were used in the statistical analyses. Biomass and individual density (kg or individuals per hectare) for each taxon was calculated by dividing the biomass or count of each taxon in each tow into the area swept by each tow. Marked (adipose fin clip or coded wire tag) and unmarked (majority naturally spawned) Chinook salmon were treated as separate species in the analysis in order to evaluate similarities and differences between hatchery and wild fish. Other hatchery salmonids were either not distinguishable (no detectable marks) or were rarely caught.

Multivariate statistical treatments were used to evaluate biomass and count differences among, and similarities within, months and basins; and relationships between taxonomic composition at individual sites and latitude, salinity, temperature, water depth, and secchi depth. All multivariate analyses were conducted following the general approach of Clarke and Warwick (Clark 1993, Clark and Warwick 2001) using Primer statistical software version 6 (Clarke and Gorley 2006) and consisted of the following steps.

1. Biomass of Jellies and each fish species was averaged by site/month combination (two tows). The data matrix was then transformed to down-weight the effect of abundant taxa using a square-root transformation.
2. A resemblance matrix of all pairwise similarities between sites, based upon the taxa present and their biomass, was calculated individually for each month using Bray-Curtis dissimilarity measure.

3. A non-metric multidimensional scaling (MDS) ordination was created based upon the rank similarities within the resemblance matrix from each month to graphically display relationships among sites from the four basins in 2-dimensions. A dimensionless "stress" value provides an estimate of reliability of the ordination. It is a nonparametric regression based procedure that compares the distances between sites in the MDS plot with the dissimilarity values in the resemblance matrix. Values < 0.2 give a useful and interpretable representation of the relationships among sites.

4. Using the square root transformed biomass table created in step 1, a Bray-Curtis resemblance matrix was created for all months and sites.

5. A two-way analysis of similarity procedure (ANOSIM) was applied to the resemblance matrix to evaluate *differences* in the biomass composition based on month and basin. The ANOSIM procedure calculates an R statistic based upon the difference between average within group rank similarities and average among group rank similarities. Values of R range between 1(all replicates within areas or months are more similar to each other than any replicates from different areas or months) and 0 (rank similarities between and within areas or months are the same, on average) and significance level is computed using a permutation procedure.

6. A two-way similarity percentages procedure (SIMPER) was applied to the same resemblance matrix to evaluate the contribution of various taxa to *similarities* in the biomass composition by month and oceanographic basin.

7. Temperature, salinity, and water depth were averaged for each site/month combination (two tows) to create a table of environmental information by site and month. Latitude and secchi depth (one measurement per site/month visit) was added to this table as an environmental variable, and water depth data were log (x) transformed to remove high skew. All four variables were then normalized to a common scale by subtracting the mean and dividing by the standard deviation over all values within each variable. A resemblance matrix

of site environmental characteristics was calculated based upon Euclidean distance for each month.

8. A rank correlation procedure (BEST: Bio-Env) was applied to biological and environmental resemblance matrices from each month to evaluate associations between overall patterns in the biomass composition and the environmental variables. The BEST procedure uses a step-wise approach to calculate multiple environmental resemblance matrices, in an effort to determine those environmental variables that have the strongest association with the biological assemblage matrix.

9. All of the above steps were repeated for fish biomass, and fish counts.

Results

Taxonomic Composition

Composition varied by oceanographic basin and by season. Jellies comprised over 60 % of the total wet biomass for all sites and months combined and was nearly 90% of the total biomass in the Main Basin and South Sound, but less than 45 % in the Rosario and Whidbey basins (Figure 4.2). Species richness generally peaked in June and July, and except for May, showed a positive trend with latitude (Figure 4.3). In total, thirty-six fish species were encountered but fish assemblages at each site were typically composed of fewer than ten species per tow, averaging between four and seven species in the Rosario and Whidbey basins, and one to four species in the Main Basin and South Sound (Figure 4.3). Herring, surf smelt, sand lance, threespined stickleback, and juvenile salmonids dominated the fish biomass (Figure 4.4). The most striking differences in fish biomass among basins was the high percentage of chum and marked Chinook salmon, and low percentage of herring and "other" species in South Sound compared with the rest of the basins. Compositional patterns were similar based on counts (data not shown), except that the relative

abundance of smaller species (e.g. threespine stickleback [*Gasterosteus aculeatus*]) was greater based on counts than on biomass.

Total biomass in the Rosario and Whidbey basins was at its lowest in May, increased into the summer then dropped in July, but in the Whidbey Basin increased in September (Figure 4.5). In contrast, total biomass in the Main Basin and South Sound was already at its highest of the study period when sampling started in May and remained there until a steep drop in August. The seasonal pattern in the fish portion of the biomass was generally similar to that in the total biomass (Figure 4.6). Cumulative mean biomass of jellies and fish (Figure 4.7) illustrates the overall patterns by basin and season, including an apparent inverse relationship between fishes and jellies.
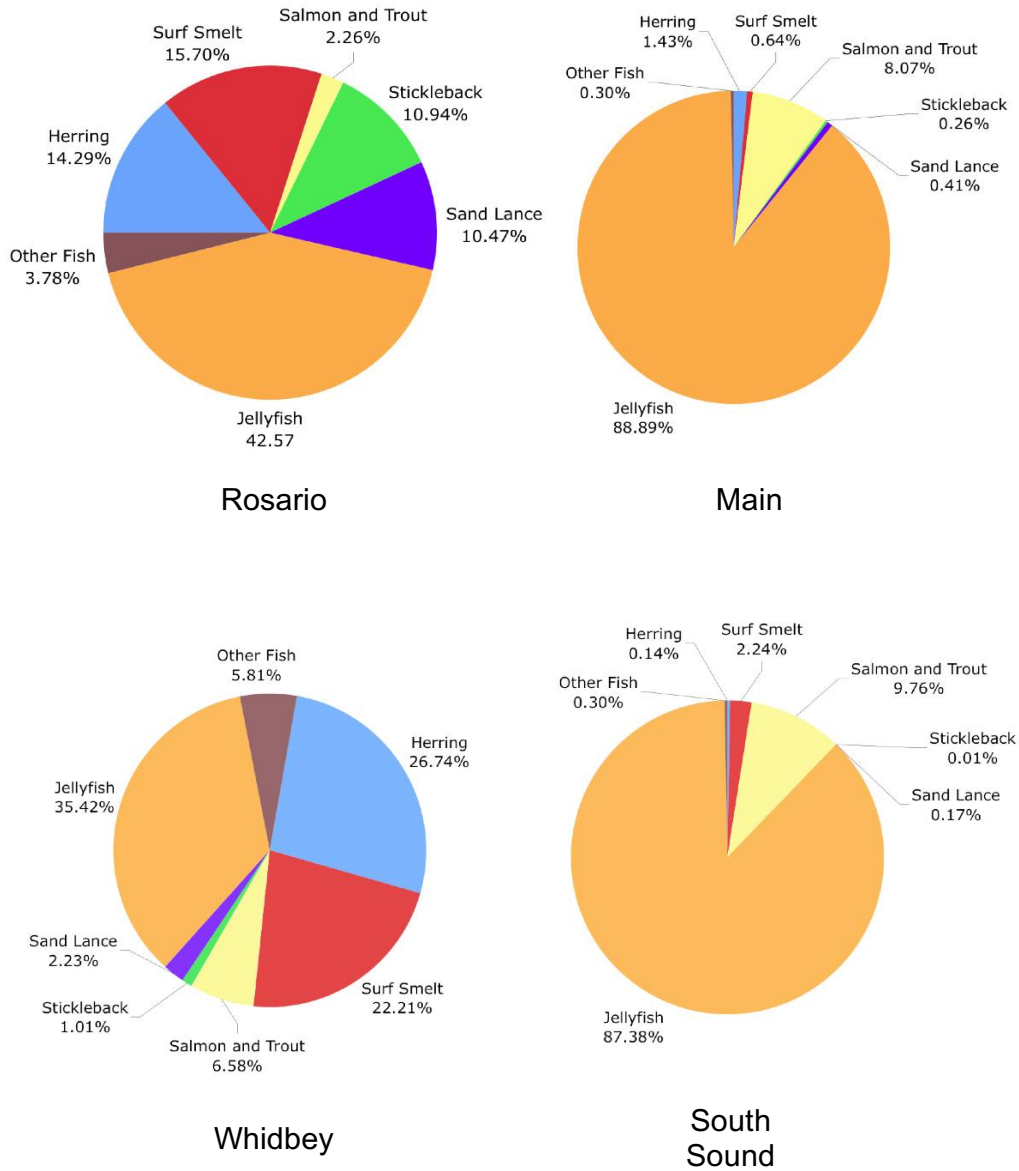
Figure 4.2. Biomass composition in the Rosario (top left), Whidbey (bottom left), Main (top right), and South Sound (bottom right) basins of Puget Sound in 2003. All months combined.
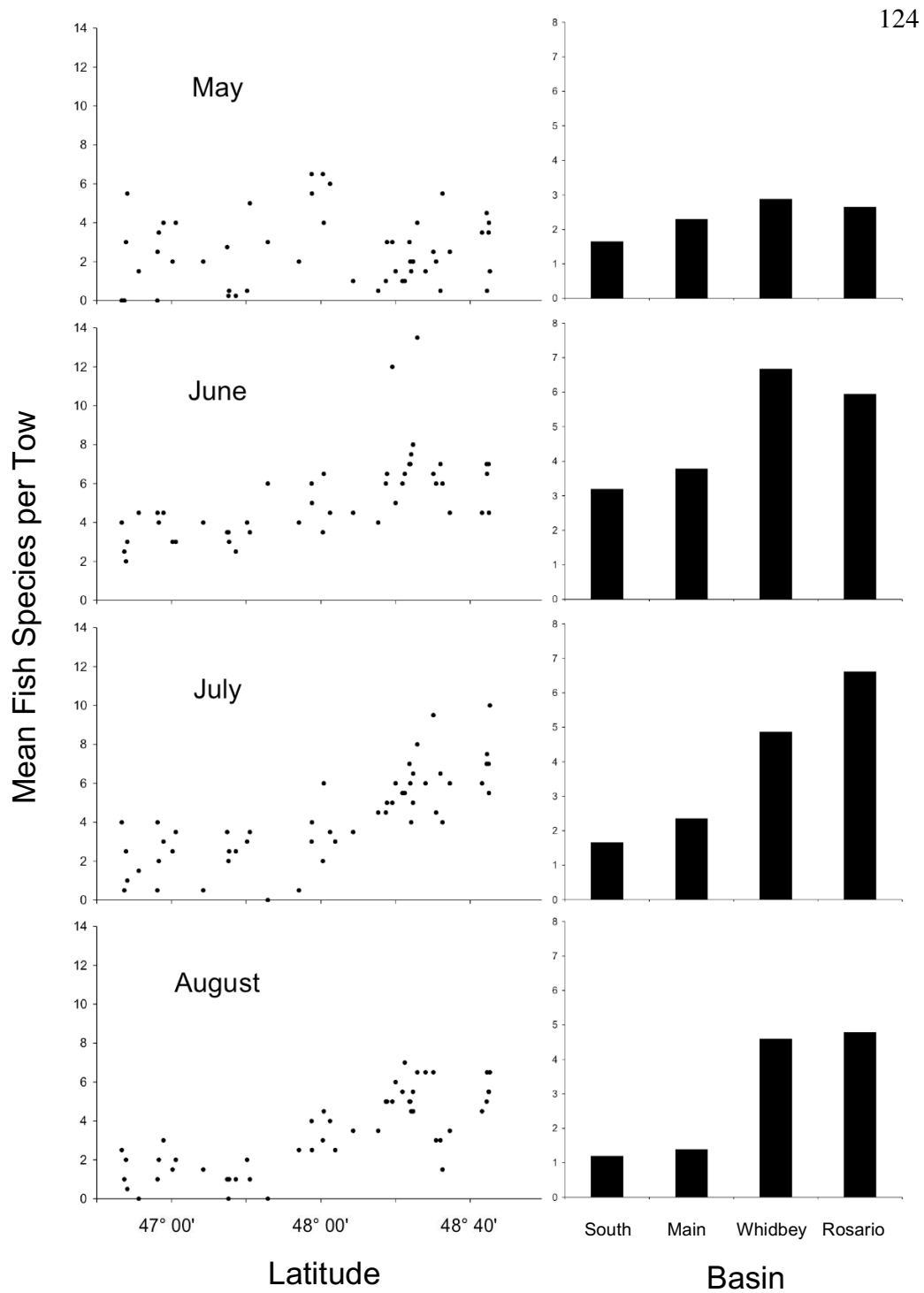
Figure 4.3. Mean fish species richness per tow at each site by month, latitude (left), and basin (right).
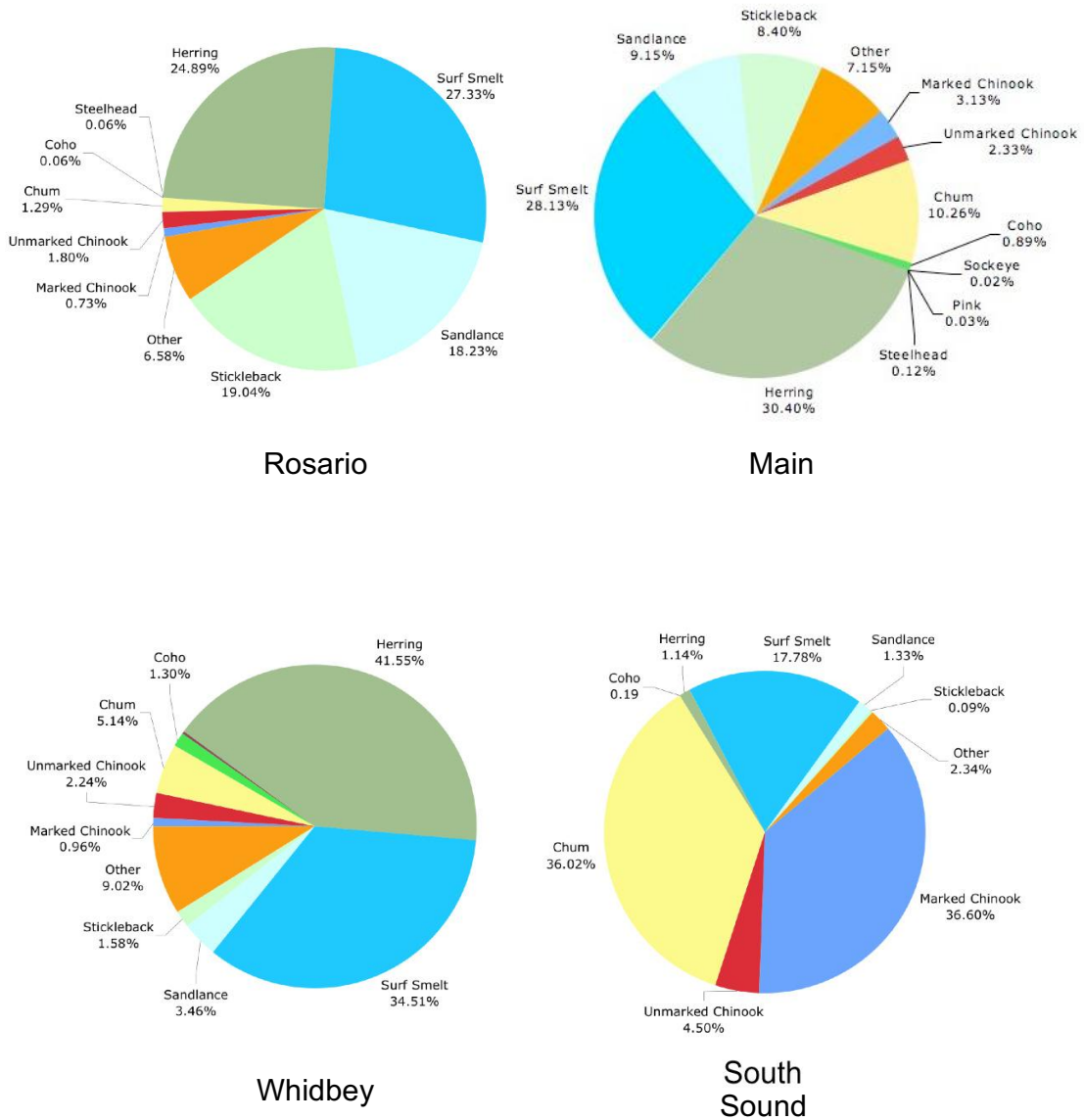
Figure 4.4. Fish biomass composition in the Rosario (top left), Whidbey (bottom left), Main (top right), and South Sound (bottom right) basins of Puget Sound in 2003. All months combined.
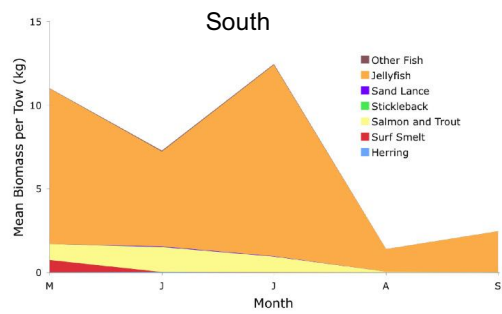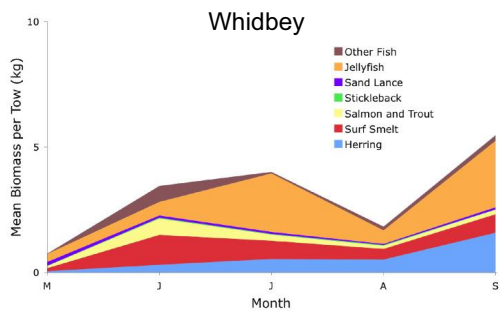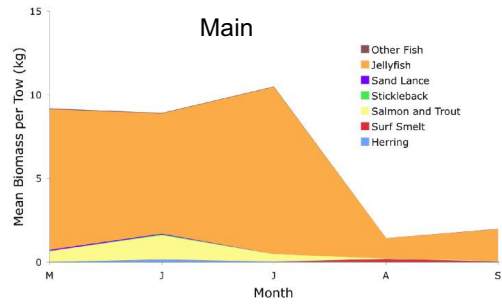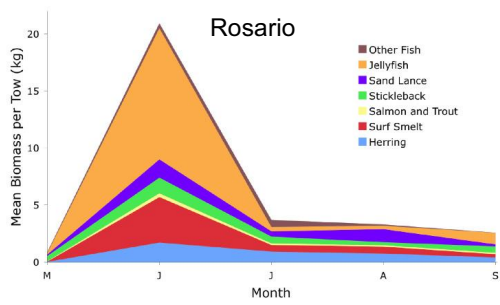
Figure 4.5. Total biomass composition by basin and month (note different scales on y axis).
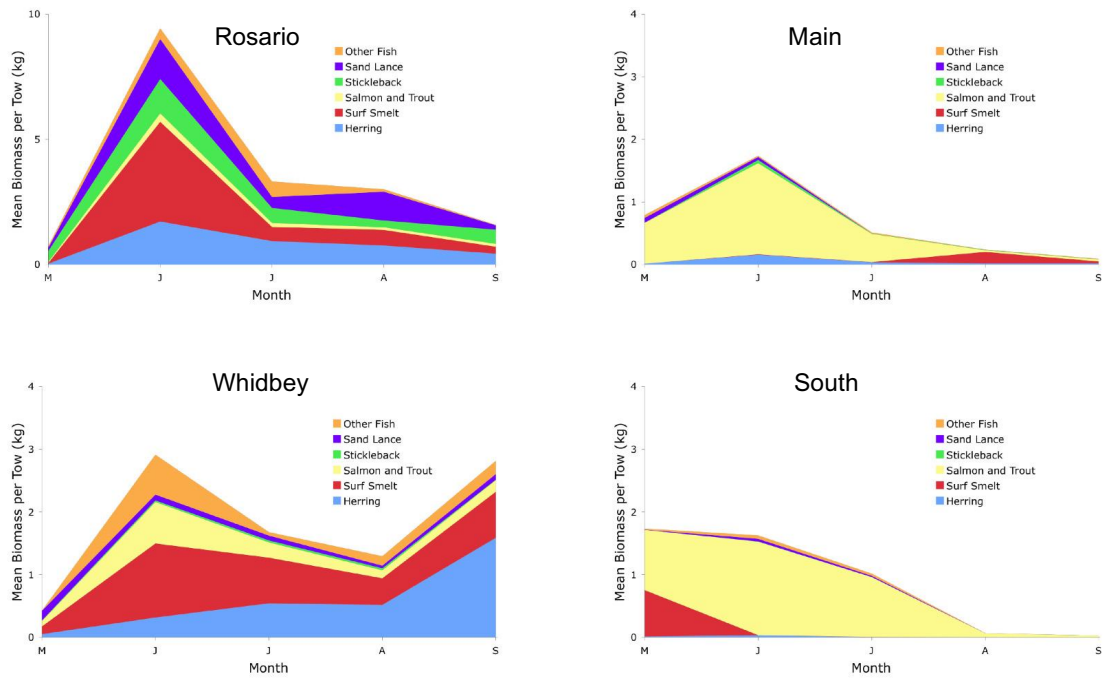
Figure 4.6. Fish biomass composition by basin and month (note different scale on y axis for Rosario Basin).
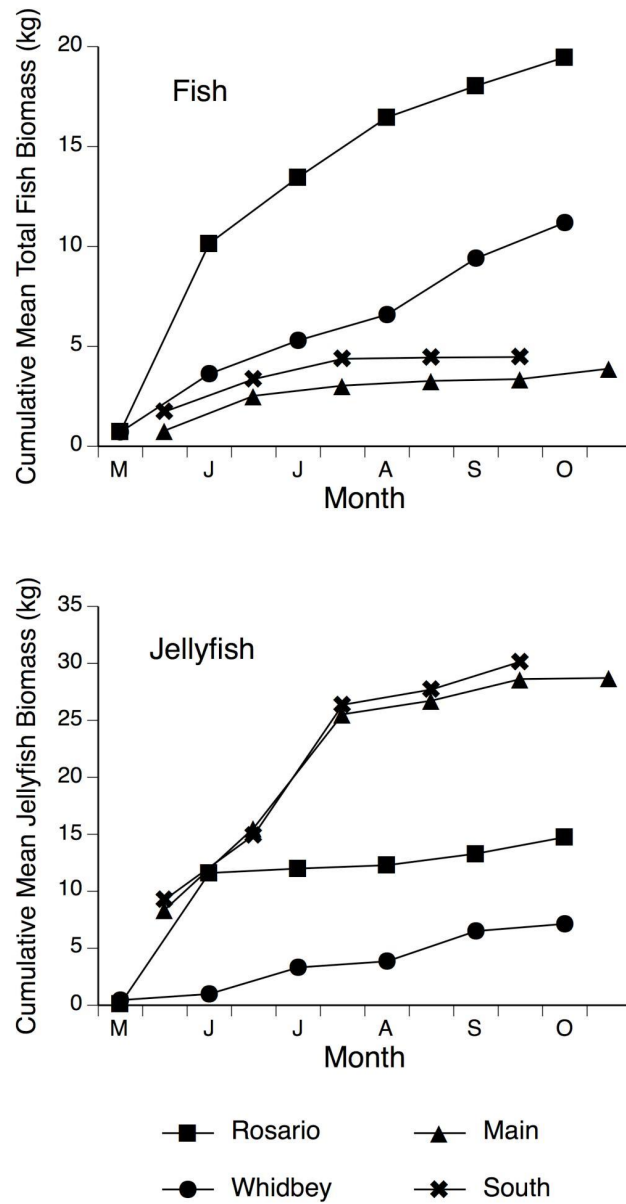
Figure 4.7. Cumulative mean fish biomass (top) and jelly biomass (bottom) per tow in four basins in 2003.

MDS ordinations of sites based on taxonomic assemblage biomass (Figure 4.8) showed moderately strong multivariate structure (2-D "stress" values between 0.16 and 0.19) but clear and consistent graphical grouping of sites within basins across months. The Rosario and Whidbey Basins had more scatter among sites than the Main Basin and South Sound. ANOSIM tests for differences among basins (Table 4.1) and months (Table 4.2) both produced clear differences among groups, but patterns by basin were more distinct (higher R values) than those by month. The greatest basin differences generally corresponded to geographic distances and connectivity between basins, although the ranks based on composition of total biomass, fish biomass, and fish counts differed slightly (Table 4.1). Assemblages in all basins were statistically different from one another by all three abundance measures, except for in the Main Basin and South Sound based on total biomass (Table 4.1). Assemblages in all months differed for all three abundance measures, except for June and July based on total biomass (Table 4.2).

Analysis of within group similarity of basins (Tables 4.3-4.5) and months (Tables 4.6-4.8) using the SIMPER procedure also showed clear differences; geographic patterns were more distinct than temporal ones, but site similarities within basins and months changed somewhat with the three abundance measures. Biomass of jellies accounted for most within-group similarity across basins and months but was most dominant in the Main Basin and South Sound (74 % and 86 %, respectively), where only two fish species (chum salmon [*Oncorhynchus keta*] and hatchery Chinook salmon) contributed to the top 90 % of group similarity. In contrast, jellies dominated less in the Rosario and Whidbey basins (32 % and 30 %, respectively), and six fish species (Pacific herring [*Clupea pallasi*], surf smelt [*Hypomesus pretiosus*], threespine stickleback [*Gasterosteus aculeatus*], juvenile Chinook salmon, chum salmon, and river lamprey [*Lampetra ayresi*]) were among the taxa contributing to the top 90 % of the similarity. Except for contributions from chum, coho (*Oncorhynchus kisutch*), and hatchery Chinook salmon (which contributed

most in June), jellies, surf smelt, herring, and threespine stickleback accounted for most similarities (typically 70 - 80 %) across months.

When jellies were removed and only fish biomass was considered (Tables 4.4 and 4.7), the species ranks were similar, but other species entered the groups contributing to the top 90 % of the similarity, especially in the Main Basin and South Sound, where the within site similarities dropped markedly, from 51 and 57, to 31 and 21 for the Main Basin and South Sound, respectively. When counts of individual fish were used as the abundance measure (Tables 4.5 and 4.8), the species, contributions, and ranks were essentially the same as in the fish biomass results, except that threespine stickleback and river lamprey (species with relatively small body size) moved up in contribution and rank.
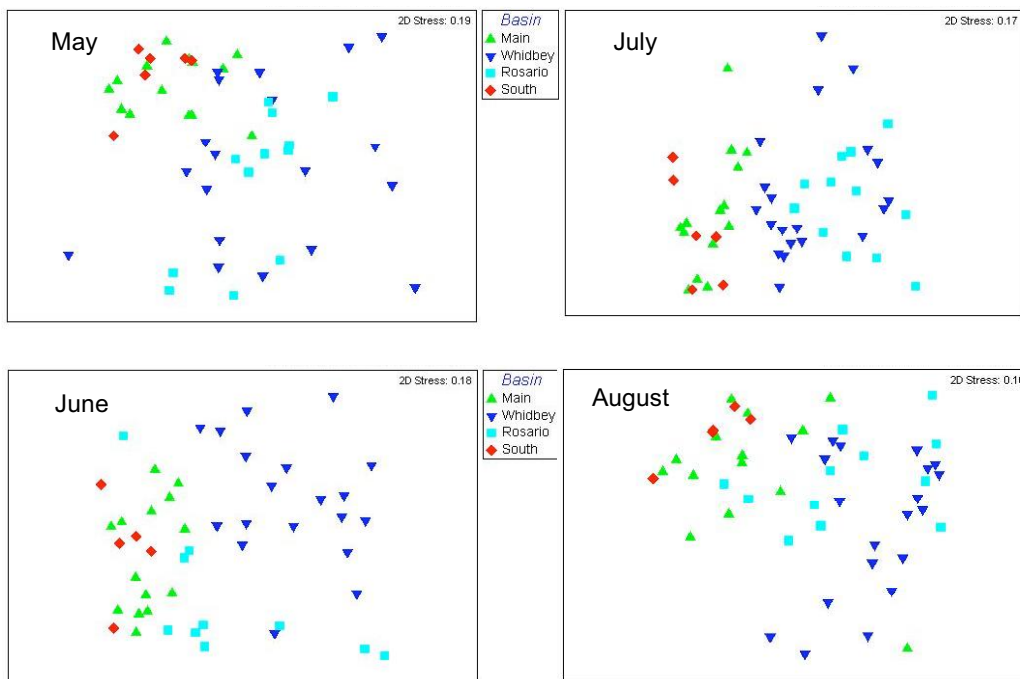
Figure 4.8. MDS ordination plots for biomass at all sites by month. Symbols indicate basin designation of individual sites. Position of points relative to one another is what is plotted. Rotation of plots as a whole does not matter.

Table 4.1. ANOSIM R statistics for comparisons between basins. Two-way tests for differences in total biomass composition, fish biomass composition, and counts of individual fish (across all month groups; 999 permutations). Ranked from most to least different based on total biomass; ranks of comparisons in bold.

Between Group Comparison	Total Biomass $R^a = 0.32$ (p = 0.001)	Fish Biomass $R^a = 0.35$ (p = 0.001)	Fish Counts $R^a = 0.38$ (p = 0.001)
Rosario vs. South	0.57 (p = 0.001) **1**	0.63 (p = 0.001) **2**	0.63 (p = 0.001) **2**
Rosario vs. Main	0.53 (p = 0.001) **2**	0.37 (p = 0.001) **4**	0.42 (p = 0.001) **3**
Whidbey vs. South	0.48 (p = 0.001) **3**	0.66 (p = 0.001) **1**	0.68 (p = 0.001) **1**
Whidbey vs. Main	0.43 (p = 0.001) **4**	0.40 (p = 0.001) **3**	0.42 (p = 0.001) **3**
Rosario vs. Whidbey	0.16 (p = 0.001) **5**	0.13 (p = 0.001) **6**	0.20 (p = 0.001) **4**
Main vs. South	-0.05 (p = 0.84) **6**	0.15 (p = 0.03) **5**	0.13 (p = 0.03) **5**

[a]Global R statistic for overall differences

Table 4.2. ANOSIM R statistics for comparisons between months. Two-way tests for differences in total biomass composition, fish biomass composition, and counts of individual fish (across all month groups; 999 permutations). Ranked from most to least different based on total biomass; ranks of comparisons in bold.

Between Group Comparison	Total Biomass $R^a = 0.21$ (p = 0.001)	Fish Biomass $R^a = 0.16$ (p = 0.001)	Fish Counts $R^a = 0.18$ (p = 0.001)
May vs. August	0.29 (p = 0.001) **1**	0.22 (p = 0.001) **2**	0.27 (p = 0.001) **1**
June vs. August	0.28 (p = 0.001) **2**	0.18 (p = 0.001) **4**	0.17 (p = 0.001) **4**
May vs. June	0.24 (p = 0.001) **3**	0.19 (p = 0.001) **3**	0.22 (p = 0.001) **3**
May vs. July	0.20 (p = 0.001) **4**	0.22 (p = 0.001) **1**	0.26 (p = 0.001) **2**
July vs. August	0.12 (p = 0.001) **5**	0.05 (p = 0.03) **6**	0.07 (p = 0.001) **6**
June vs. July	0.12 (p = 0.84) **6**	0.08 (p = 0.008) **5**	0.08 (p = 0.03) **5**

[a]Global R statistic for overall differences

Table 4.3. Average similarity of total biomass composition among sites within each basin, and ranked similarity percentages of taxa contributing 90 % to the similarity within each basin (two-way SIMPER test; adjusted for month effect). UM Chin = unmarked Chinook salmon; M Chin = marked Chinook salmon.

Rosario Average similarity = 41			Whidbey Average similarity = 38			Main Average similarity = 51			South Average similarity = 57		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Jelly	32	32	Smelt	30	30	Jelly	74	74	Jelly	86	86
Herring	19	50	Jelly	20	50	Chum	11	85	M Chin	6	92
Stickle	17	68	Herring	18	68	M Chin	5	91			
Smelt	12	80	UM Chin	11	79						
UM Chin	8	88	Stickle	5	84						
M Chin	4	92	M Chin	5	89						
			Lamprey	3	92						

Table 4.4. Average similarity of fish biomass composition among sites within each basin, and ranked similarity percentages of taxa contributing 90 % to the similarity within each basin (two-way SIMPER test; adjusted for month effect). UM Chin = unmarked Chinook salmon; M Chin = marked Chinook salmon.

Rosario Average similarity = 37			Whidbey Average similarity = 38			Main Average similarity = 31			South Average similarity = 21		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Herring	28	28	Smelt	37	37	Chum	40	40	M Chin	44	44
Stickle	26	54	Herring	23	60	M Chin	23	63	Chum	27	71
Smelt	17	71	UM Chin	14	74	Herring	13	76	UM Chin	18	89
UM Chin	11	83	Stickle	7	80	UM Chin	13	89	Shiner	8	97
M Chin	7	89	M Chin	6	86	Smelt	4	93			
Sandl	4	93	Lamprey	4	90						

Table 4.5. Average similarity of fish count composition among sites within each basin, and ranked similarity percentages of taxa contributing 90 % to the similarity within each basin (two-way SIMPER test; adjusted for month effect). UM Chin = unmarked Chinook salmon; M Chin = marked Chinook salmon.

Rosario Average similarity = 34			Whidbey Average similarity = 38			Main Average similarity = 31			South Average similarity = 21		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Stickle	33	33	Smelt	37	37	Chum	43	43	M Chin	38	38
Herring	28	62	Herring	21	58	M Chin	19	62	Chum	34	73
Smelt	14	76	Stickle	11	69	Herring	12	74	UM Chin	17	90
UM Chin	7	83	UM Chin	10	79	UM Chin	11	85	Shiner	6	96
Sandl	6	89	Lamprey	5	84	Stickle	6	91			
M Chin	5	94	M Chin	5	89						
			Sandl	3	92						

Table 4.6. Average similarity of total biomass composition among sites within each month, and ranked similarity percentages for taxa contributing 90 % of the similarity within each month (two-way SIMPER test, adjusted for basin effect). Perfect similarity is 100, no similarity is 0.

May Average similarity = 33.42			June Average similarity = 45.42			July Average similarity = 46.81			August Average similarity = 43.99		
Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %
Jelly	67.43	67.43	Jelly	33.38	33.38	Jelly	40.84	40.84	Jelly	33.79	33.79
Smelt	7.93	75.36	Smelt	18.67	52.05	Smelt	16.03	56.87	Smelt	20.14	53.93
Herring	5.52	80.88	Herring	15.16	67.21	Herring	12.79	69.66	Herring	16.04	69.97
Chum	5.30	86.18	Chum	9.41	76.61	UM Chin	8.59	78.26	UM Chin	13.19	83.16
Stickle	5.23	91.41	M Chin	6.12	82.74	M Chin	6.67	84.92	Stickle	7.80	90.96
			UM Chin	5.36	88.10	Stickle	5.79	90.71			
			Coho	3.30	91.40						

Table 4.7. Average similarity of fish biomass composition among sites within each month, and ranked similarity percentages for taxa contributing 90 % of the similarity within each month (two-way SIMPER test, adjusted for basin effect). Perfect similarity is 100, no similarity is 0.

May Average similarity = 18			June Average similarity = 43			July Average similarity = 43			August Average similarity = 37		
Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %
Smelt	20	20	Smelt	24	24	Smelt	24	24	Smelt	30	30
Chum	19	39	Herring	22	46	Herring	20	44	Herring	24	54
Stickle	17	56	Chum	17	63	UM Chin	14	59	UM Chin	20	74
Herring	15	71	M Chin	12	75	M Chin	13	72	Stickle	12	86
Sandl	8	79	UM Chin	9	84	Chum	12	84	M Chin	8	94
M Chin	7	86	Stickle	5	89	Stickle	8	92			
UM Chin	6	92	Coho	4	93						

Table 4.8. Average similarity of fish count composition among sites within each month, and ranked similarity percentages for taxa contributing 90 % of the similarity within each month (two-way SIMPER test, adjusted for basin effect). Perfect similarity is 100, no similarity is 0.

May Average similarity = 19			June Average similarity = 43			July Average similarity = 41			August Average similarity = 35		
Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %	Species	Contrib. %	Cumul. %
Chum	23	23	Smelt	24	24	Smelt	24	24	Smelt	31	31
Smelt	21	44	Chum	19	42	Herring	20	44	Herring	26	57
Stickle	20	64	Herring	18	60	Chum	13	57	Stickle	18	75
Herring	9	73	M Chin	11	71	UM Chin	12	69	UM Chin	13	88
Sandl	9	82	UM Chin	9	80	Stickle	12	81	M Chin	6	94
M Chin	5	87	Stickle	7	87	M Chin	11	92			
UM Chin	3	90	Lamprey	6	93						

Environmental Variables

Water temperature showed clear seasonal patterns but was fairly uniform across basins (Figures 4.9 and 4.11). Mean water temperature in ranged from 11.7°C to 17.3°C with a steady increase from May to a summer peak in July, then a steady decline into October (Figure 4.10). Salinity varied seasonally and among basins (Figures 4.10 and 4.11). The Rosario and Whidbey basins in the north, with their much higher freshwater input from rain and snowmelt, had lower salinity than the Main Basin and South Sound overall and a wider range, from 16 in April to over 25 in September and October. The Main Basin and South Sound always averaged between 25 and 30, with a general increase from spring to fall (Figure 4.10).

Water depth at sampling occurred over a broader range of depths than preferred, primarily because of steeper shoreline slopes and the presence of more large, anthropogenic obstacles (e.g., piers) in central Puget Sound. Water depth at sampling sites ranged from 4.7 m to 46.7 m with an overall mean of 11.9 m (Figure 4.11).

Secchi depth varied seasonally but most distinctly geographically (Figure 4.12), ranging from over 10 m (the length of the line on the instrument) in South Sound to less that 1 m at the mouth of the Puyallup River. The Main Basin and South Sound secchi depths were typically around 6 m whereas sites in the Whidbey and Rosario were typically near 3 m. As the season progressed into summer, secchi depth in central and south sites increased, and secchi depth at sites in the northern basins decreased.

Relationships between resemblance matrices from the abundance data and the environmental conditions at each site differed by month but more so by the kind of abundance data used (Table 4.9). Correlations were weakest in May and strongest in July and August, especially when using fish only abundance measures (Table 4.9). Latitude was the variable most strongly related to total biomass composition. While latitude was also important in models using only fish abundance, secchi depth was more so.

The relative abundance and total biomass of the dominant taxonomic groups revealed in the graphical and statistical analyses changed with latitude. Jellies were more abundant in central and southern Puget Sound, whereas fishes were more abundant in the north (Figure 4.13). Within the fish assemblages, hatchery Chinook and chum salmon dominated at central and southern sites, but surf smelt, herring, threespine stickleback, and sand lance dominated northern sites (Figure 4.14).
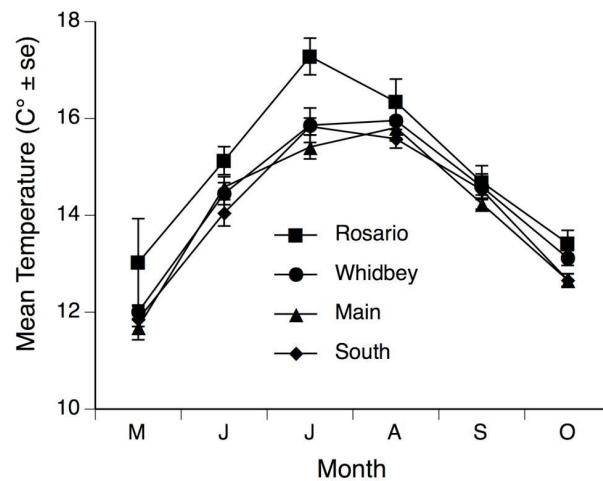
Figure 4.9. Mean monthly water temperature (± se) in the four basins.
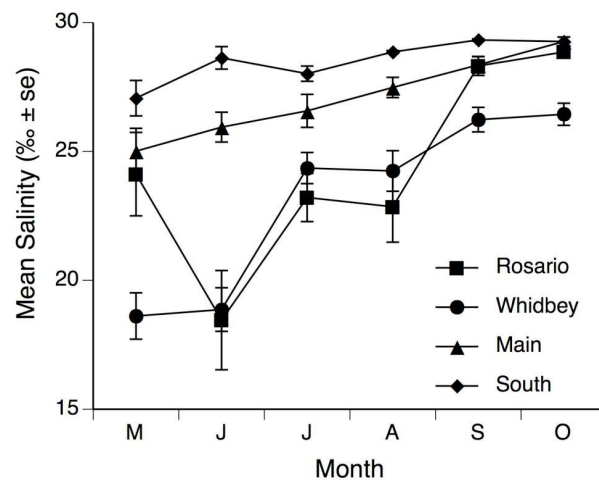
Figure 4.10. Mean monthly salinity (± se) in the four basins.
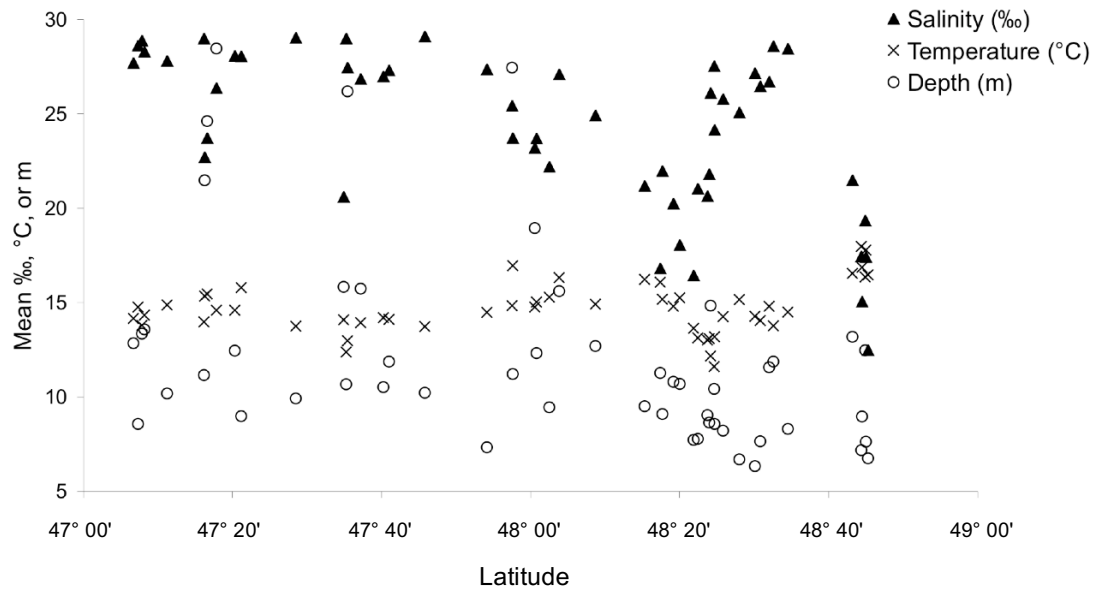
Figure 4.11. Mean salinity, temperature, and depth by latitude at 52 Greater Puget Sound sites in 2003.
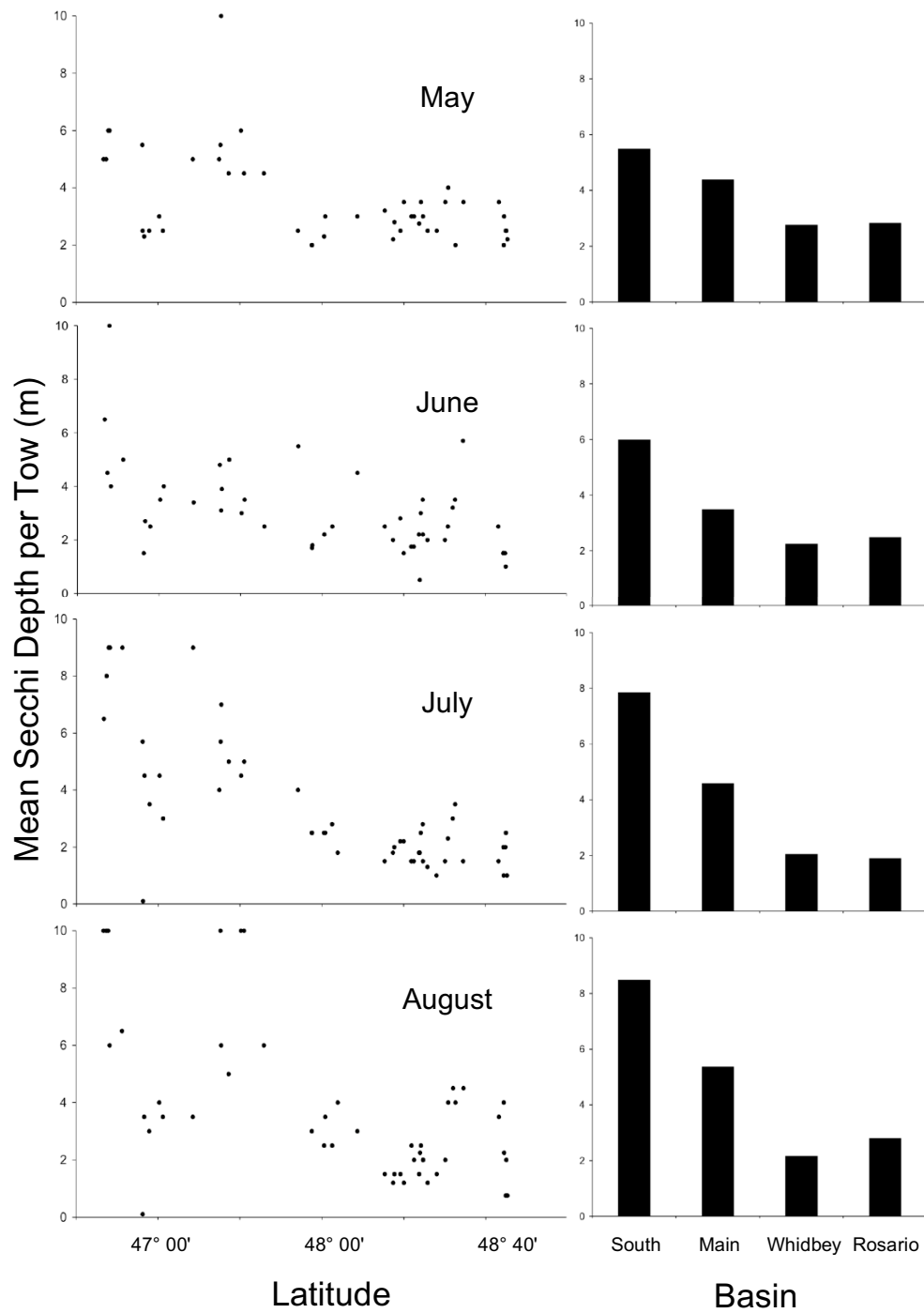
Figure 4.12. Mean secchi depth per tow at each site by month, latitude (left), and basin (right).

Table 4.9. Spearman rank correlations by month for five best models based on similarity matrices from jelly and fish biomass (top), fish biomass (middle), and fish count (bottom) composition and environmental variables for all sites (BIOENV test; 99 permutations). SD = secchi depth, L = latitude, S = salinity, T = temperature, D = depth. Models ranked from best to worst.

Jelly and Fish Biomass Composition

May R = 0.28; p = 0.01		June R = 0.30; p = 0.01		July R = 0.45; p = 0.01		August R = 0.39; p = 0.01	
Corr.	Var.	Corr.	Var.	Corr.	Var.	Corr.	Var.
0.28	L, S	0.30	L, S	0.45	L	0.39	L
0.28	L	0.27	SD, L, S	0.41	L, D	0.35	SD, L, D, T
0.26	L, T	0.27	L, T, S	0.39	SD, L, D	0.34	SD, L, D
0.26	L, T, S	0.26	L	0.37	SD, L	0.34	SD, L
0.25	L, D, S	0.26	SD, L, T, S	0.37	SD, T	0.24	L, D

Fish Biomass Composition

May R = 0.29; p = 0.01		June R = 0.36; p = 0.01		July R = 0.56; p = 0.01		August R = 0.52; p = 0.01	
Corr.	Var.	Corr.	Var.	Corr.	Var.	Corr.	Var.
0.29	SD	0.36	SD, L	0.56	SD, L	0.52	SD, L
0.25	SD, T	0.35	L	0.51	L	0.48	SD, L, D
0.22	SD, L, T	0.34	SD, L, S	0.50	SD, L, D	0.43	SD
0.21	SD, L	0.33	SD, L, T	0.50	SD	0.42	SD, L, D, T
0.18	SD, L, T, S	0.33	L, S	0.48	SD, L, S	0.41	SD, L, T

Fish Count Composition

May R = 0.34; p = 0.01		June R = 0.33; p = 0.01		July R = 0.58; p = 0.01		August R = 0.52; p = 0.01	
Corr.	Var.	Corr.	Var.	Corr.	Var.	Corr.	Var.
0.34	SD	0.33	SD, L, S	0.58	SD, L	0.52	SD, L
0.29	SD, T	0.33	SD, L	0.53	SD	0.45	SD, L, D
0.29	SD, L	0.32	L	0.52	L	0.45	L,
0.28	SD, L, T	0.32	SD	0.51	SD, L, D	0.42	SD, L, T
0.23	SD, L, T, S	0.32	L, S	0.50	SD, L, S	0.40	SD
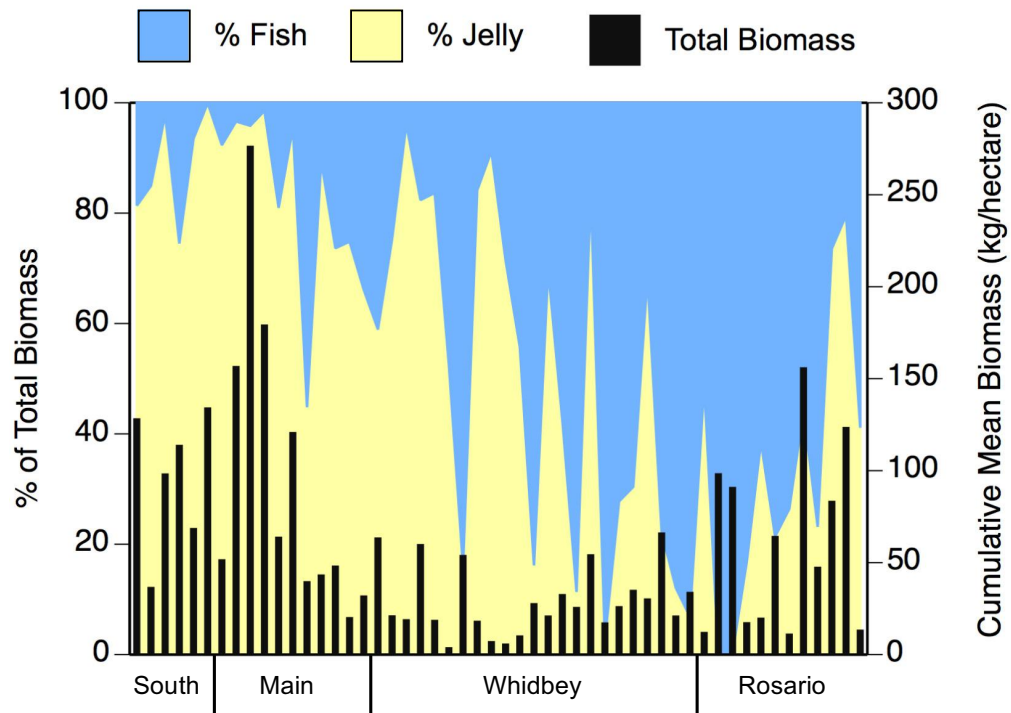
Figure 4.13. Percentage fish (blue area) and jelly (yellow area) in the total biomass (black bars) for sites within each region. Each bar is the sum of the four monthly means from May to August for each site.
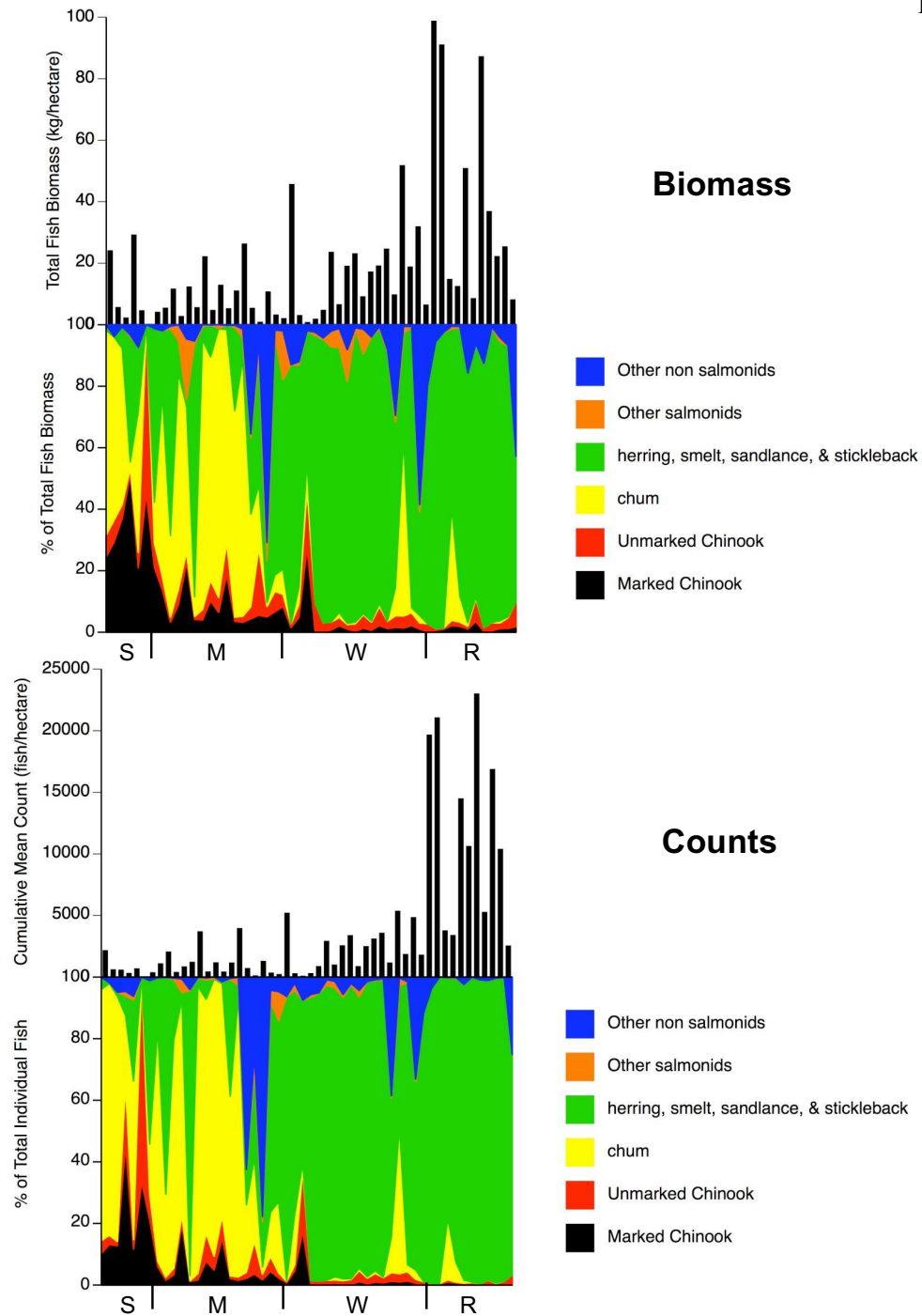
Figure 4.14. Percentage selected fish groups in the total biomass (top), and in the total counts across the study area (S = South Sound, M = Main Basin, W = Whidbey Basin, R = Rosario Basin. Each bar is the sum of the four monthly means from May to August for each site.

Discussion

Composition of macrofaunal assemblages during late spring and summer in neritic Puget Sound differed somewhat with month but more distinctly among geographic areas. Jellies and several small pelagic fish species dominated the biomass, and jelly biomass was inversely related to both biomass and the diversity of fish assemblages. These patterns suggest differences in pelagic food web structure among different sub basins of Puget Sound. Multiple natural and anthropogenic influences may be behind these patterns, and here I review some of them. I also consider the implications for Puget Sound science and management.

The strong geographic patterns and statistical importance of latitude and water clarity suggest that the patterns in macrofaunal composition are caused by unmeasured characteristics (e.g., phytoplankton and zooplankton abundance and size distributions, water column structure, oxygen and nutrient levels) resulting from underlying oceanographic features (e.g., bathymetry, connectivity, wind and wave exposure) or other factors, including human activity. The various basins of Puget Sound are quite different in terms of bathymetry, connectivity to ocean water, freshwater input, and tidal regime, all of which can influence physical, chemical, and biological oceanography, and consequently, pelagic ecology, including the structuring of food webs (Syvitski et al. 1987, Mann 2000). The observed biological patterns, then, are somewhat consistent with expectations, especially in the South Sound, were the shallow bathymetry, relatively low freshwater input, and position at the head of the larger Puget Sound system overall, result in longer residence time and less mixing of the water column, and, at times, nutrient depletion (Strickland 1983, Albertson et al. 2002). Latitudinal gradients exist in intertidal invertebrates in central and southern Puget Sound (Dethier and Schoch 2005) and are related to gradients in the physical environment (tidal range, exposure, etc.) that result from the position in the landscape.

The relationships between taxonomic composition, and latitude and water clarity suggest that production of suitable prey resources for small pelagic fishes may also

correspond to position in the landscape. Water clarity is influenced by the amount of river sediment in the water column but sediments in river inputs would presumably be declining into the summer and early fall, thus contributing less to turbidity. The July and August increases in the strength of statistical relationships between water clarity and fish abundance and assemblage composition (Table 4.9) could be the result of seasonal decline in hatchery fish density and redistribution of hatchery fish over time after release. That is, if water clarity values in this study generally correspond to fish food abundance in the environment, and hatchery releases are independent of prey distribution in the landscape, we would expect the relationship between water clarity and fish to be stronger as hatchery fish leave the system, or redistribute to areas of higher prey density.

Apparent inverse relationships between jellies and both fish biomass and diversity is consistent with hypotheses regarding bifurcated pelagic food webs (Greve and Parsons 1977, Parsons and Lalli 2002), one dominated by jellies and the other by small pelagic fishes. In South Sound, for example, the vast majority of the fauna is jellies, hatchery subyearling Chinook salmon that rapidly disappear from the area after release (see Chapter 3), and chum salmon, one of the few fish species that seem to be adapted to eat jellies (Welch 1997, Arai 2005), and thriving in South Puget Sound in recent years. Stable isotope studies of chum and other salmon in estuarine (Romanuk and Levings 2005) and oceanic (Welch and Parsons 1993) environments strongly suggest that chum prey on carnivorous jellies, and diets in chum captured in one ongoing Georgia Basin and Puget Sound study (offshore and deeper than collections in this study) were often dominated by ctenophores (Sweeting et al. 2007). Recent declines of some South Sound recreational fisheries, and poor survival in South Sound yearling hatchery coho and Chinook salmon (both piscivorous life stages; hatchery Chinook subyearling survival has remained relatively stable) have raised concerns about the ecological health of South Sound (Preikshot and Beattie 2001). A recent modeling effort to explore these perceived problems was plagued by

insufficient data for many taxa, including jellies and small pelagic fishes (Preikshot and Beattie 2001).

Human Influences

Human activities that may structure pelagic food webs and favor jellies include removal of fishes by harvest and habitat destruction, nutrient addition, substrate hardening, and introduction of invasive species (Purcell in review, Parsons and Lalli 2002). Although significant fisheries on small pelagic fish existed in the past in Puget Sound, they have been greatly restricted since the 1980s, and while few surf smelt, herring, and Pacific sand lance were captured in the Main Basin and South Sound in this study, all three species still spawn there (Penttila 1997, Lemberg et al. 1997). Anthropogenic eutrophication has not historically been identified as a serious problem in Puget Sound, although this has changed in recent years with apparent increases in the incidence of low oxygen events (and associated fish kills), particularly in Hood Canal (Newton 2002, PSAT 2007b). Major declines in salmonids and other pelagic fishes (including three gadoid species now virtually absent from Puget Sound) are partially the result of harvest. Hatcheries have "replaced" much of the juvenile production, dramatically (and often intentionally [WDFW and PSTT 2004]) altering the individual size, timing, density, and species and life history composition of juvenile populations (White et al. 1995, Flagg et al. 2000, Weber and Fausch 2003). These attributes in larval and juvenile fishes are all fundamental factors in food web dynamics (Cushing 1990, Houde 1997, Rice et al. 1997, Cowan et al. 2000). As a result, hatchery releases have direct ecological implications, especially during periods of unfavorable environmental conditions (Cooney and Brodeur 1998, Pearcy et al. 1999, Levin et al. 2001, Beamish et al. 2003, Beamish et al. 2004).

Puget Sound's basins differ in the nature and magnitude of human activity. For example, the Main Basin (including the Duwamish and Puyallup River estuaries) is heavily urbanized, the Whidbey Basin is altered more by agriculture, and South

Sound receives a disproportionately high density of hatchery salmonids (see Chapter 3). All basins have varying degrees of sewage and storm water management, and differing physical modification of the shoreline (this is relevant to the pelagic zone in part because surf smelt, herring, and sand lance are all obligate beach spawners, and because the polyp stages of many jellies require hard substrates). Thus, the contrasting neritic fauna in the different basins are not simply a matter of local urbanization, since Skagit Bay and Nisqually Reach are two of the most different areas biologically, but are two of the least disturbed estuaries in Puget Sound. However, human activity may play a role in several ways; locally, for example, through nutrient loading and manipulation of the pelagic fish fauna by hatchery supplementation, and globally through influences on climate that affect physical forcing and nutrient delivery from the Pacific Ocean (Snover et al. 2005). Efforts to understand the many effects of human activity on the pelagic environment in Puget Sound have been negligible.

Historical Information

Although no historical data on jellies at the sites sampled are available, the fish composition in Padilla (Fresh 1979) and Skagit (Stober and Salo 1973) Bays recorded in this study is qualitatively similar to that recorded in the 1960s and 1970s. In South Sound, data collected in 1977 and 1978 from night tows at sites in the Nisqually Reach area at or close to those sampled in this study (Fresh et al. 1979), suggest some similarities, especially with chum catches, but also that more fish were encountered back then. Herring in particular were common in the historical catches, comprising approximately 25 % of the individuals (herring were approximately 1 % of the individuals captured in South Sound in this study). Unfortunately, most of the tows in the earlier study were conducted at night (see confounding factors section below) and the data from daytime catches are not available so a direct comparison is not possible.

Seasonal patterns of the fish species composition in this study are also qualitatively similar to the scarce historical information available. Salmonids are most heavily studied, and contributions to within month similarity observed here is consistent with known patterns of juvenile salmon use of estuaries (Simenstad et al. 1982, Groote and Margolis 1991). The statistical importance of chum salmon in May, followed by chum, Chinook, and coho in June, and only Chinook in July and August corresponds to the typical outmigration timing and degree of estuarine use of the three species. The difference between marked (known hatchery) and unmarked (majority natural spawn) Chinook demonstrates the contrasting seasonal abundance distributions of hatchery and wild Chinook salmon. Wild fish tend to have a more protracted seasonal presence (Chapter 3), and longer individual residence time (Levings et al. 1986), than hatchery fish. Depending on the abundance measure used, marked Chinook account for approximately 6-8% of the within month similarity in June, July, and August, whereas unmarked Chinook begin at 5 % in June and steadily increase to 13% by August, reflecting their extensive estuarine residence, and probably, their increasing piscivory over the season.

Composition of assemblages observed here gives clues to what determines pelagic food web structure in Puget Sound and raises some interesting questions, including:

- Have hatchery releases (and other human activities) over the last century played a significant role in structuring pelagic food webs in Puget Sound?

- Does the relative abundance of juvenile chum in South Sound and the health of these populations reflect their adaptations to eat jellies?

- Are the different spawning seasons of surf smelt in different areas of Puget Sound an adaptation to coincide with seasonal productivity?

- Why do surf smelt vary widely in spawning season within Puget Sound but herring and sand lance do not?

- Are anchovies (broadcast pelagic spawners that may have different prey preferences) well suited to particular conditions that favor them over the other three species?

Implications

The size and complexity of the Puget Sound ecosystem presents daunting challenges for science and management. Natural physical and biological components and processes are not evenly distributed across the landscape, and neither are the many human activities that affect them. Thus, different areas vary, for example, as locations for rearing and spawning by pelagic fishes, but also in their vulnerability to, and recoverability from, human stressors. Consequently, we should have different ecological expectations for different areas of Puget Sound, and different approaches to, and priorities for, science and management.

The task, then, is to understand the character of the biota across all of Puget Sound, and to identify, characterize, and separate the natural influences on the biota from the human ones. This requires the identification of comparable spatial and temporal units, and responsive and measurable biological attributes. Results of this study demonstrate fundamental differences in the pelagic biota across Puget Sound that would not be apparent looking at one area and one species, and suggest that conventional designations for the various basins of Puget Sound are ecologically meaningful units that can be sampled with a reasonable amount of effort, and so may be a useful basis for designing monitoring and research programs. This study also indicates that simple attributes of pelagic macrofauna—specifically, biomass and counts of jellies and small pelagic fishes—may be valuable both in understanding the basic ecology of Puget Sound, and in detecting significant changes in biological condition, including those resulting from local and global human influences. But because of the spatial heterogeneity of the ecosystem and the temporal dynamics of both the biota and the environment (e.g., interdecadal fluctuations in both small pelagic fish populations [Baumgartner et al. 1992] and ocean conditions [Francis et al. 1998]), studying relationships between pelagic macrofauna and monitoring change will often require sampling across taxa, preferably at large spatial and temporal scales.

Confounding Factors

Evaluation of the results from this study is complicated by four potentially important sources of sampling bias: site selection, fishing gear, time of day, and seasonal coverage. Because the opportunistic, exploratory sampling design (focused on juvenile salmon) was nonrandom and involved repeated sampling of the same sites and river mouth estuaries, it is possible that many of the areas not sampled (e.g., protected embayments in central Puget Sound) could be desirable habitat for small pelagic fishes, and consequently, occupied more by fishes and less by jellies than the sites sampled. But 20 of the 52 sites sampled were outside of the river mouths, and each tow typically sampled approximately 0.4 hectares (a substantial amount of area totaling approximately 250 hectares), increasing the likelihood that these data are representative of broad patterns of neritic biomass composition across Puget Sound. Ideally, a more probabilistic sampling design that proportionally covered all the neritic waters of the Puget Sound/Georgia Basin system, would provide a more complete and reliable assessment of the use of these environments by fishes and jellies. Stratification of such a design to explicitly include environmental characteristics (e.g., the degree of water column stratification, connectivity to ocean water) that could influence the biology would also be essential, including coverage of the full range of human influences (from little to massive) and inclusion of the different dimensions of human influence (physical alteration of wetlands and shorelines, nutrient and toxic chemical addition, hatchery releases, etc).

Changes in capture efficiency of the sampling gear with organism size, for example, also bias the results. Concurrent summer sampling with the same kind of townet used in this study and purse seines in Skagit Bay in 1972 (Stober and Salo 1973) and 2006 (C. Greene, NOAA Fisheries, unpublished data), and in Sinclair Inlet in 2002 (Fresh et al. 2006) showed that purse seines tended to capture more fish at the larger end of the size distribution. This size bias, then, may mischaracterize individual size distributions, biomass, and life history attributes of different taxa, for

example, progressively missing more of the larger individual juvenile Chinook salmon as they grow and become more piscivorous, or entirely missing some species such as large, predatory anadromous trout. Consequently, some of more potentially useful indicators of biological condition (Karr and Chu 1999) are not measured with the townet. Body shape can also influence capture efficiency. Sand lance, for example, have narrow, tube-shaped bodies and slip through some of the townet mesh more than other fish of similar length. For jellies, the variable mesh sizes in the net certainly bias the sampling, for example, toward the capture of large medusae (which can reach nearly a meter in diameter), and against the capture of ctenophores (approximately 2 cm in diameter). True spatial or temporal differences in the relative abundances of small and large bodied jellies, then, would confound the interpretation of the data.

Diel vertical migration is common in both jellies and pelagic fishes, and migration of many species toward the surface occurs at night. Consequently, much of the biota of interest may have been out of reach of the sampling gear. However, all sites were sampled consistently during daytime so only major geographic differences in vertical migration could explain the observed patterns in resident assemblages. Finally, the limited seasonal coverage of this study gives an incomplete view of the biological character of the different basins. Sampling South Sound in March and April would have been especially informative since annual productivity probably begins earlier there than in any of the other basins (Strickland 1983) which may, for example, support more small pelagic fishes earlier in the year than what was observed later in the season by this study.

Conclusion

Considering the vulnerability of Puget Sound's pelagic zone to human stressors such as fisheries, eutrophication, and climate change, science and management should focus more attention on it. Most pressing is the need to characterize seasonal and geographic patterns of biotic composition and identify natural and human

influences on them. Based on that knowledge, pelagic attributes should be monitored that are most effective at detecting and diagnosing problems, and evaluating the efficacy of management actions. Jellies and small pelagic fishes are likely to be very responsive to stressors on the pelagic environment, and, as this study demonstrates, are also readily observable. We do not know if jellies historically dominated pelagic environments in central and southern Puget Sound or whether the patterns observed in this study are the result of a significant change in ecosystem condition. Such changes do occur in coastal ecosystems, sometimes as a result of human activity, and anecdotal information from Puget Sound hints that similar changes may be occurring there, yet no scientific study and monitoring of such phenomena occurs. Instead, most information on the character of pelagic environments in Puget Sound is on the chemical and physical properties of the water column (PSAT 2007b). Since the biota is the primary concern to society and is the foundation of nearly all environmental legislation and policy, effective stewardship of Puget Sound requires that these and other abiotic and biotic properties be related to biological responses, and the resulting relationships used to guide monitoring and management of the ecosystem.

CHAPTER 5

Marine Bird and Waterfowl Assemblage Composition Along Urbanization Gradients in Greater Puget Sound

Summary

Abundance of several marine bird and waterfowl species in Puget Sound and adjacent waters has declined markedly in recent decades. Causes of these declines are poorly understood but presumably include many local and remote influences. Although abundances of individual species and species groups are monitored in greater Puget Sound, few analyses of local natural and anthropogenic influences on taxonomic composition have been done. Studying these relationships will not only improve our understanding of birds, but also of the greater Puget Sound ecosystem, and may assist in the development of improved monitoring and assessment tools. I used aerial bird surveys and maps of physical shoreline structure and land cover to explore changes in marine bird and waterfowl assemblage composition across years, oceanographic sub-basins, estuaries, and simple urbanization gradients in greater Puget Sound. Ten years of annual winter surveys (1993-2003), and four years of annual summer surveys (1993-1996) were combined with maps of shoreline segments, and the 2002 winter survey was also combined with 2002 urban land cover within 2 km of the shoreline. Consistent with observed declines in individual population abundance during recent decades, mean taxa richness in summer and winter surveys declined across years in much of the study area. Assemblage composition differed by season and oceanographic sub-basin, but also between urban and non-urban areas, and the relative abundance of some taxa changed along gradients of urban land cover across the study area. Urbanization was associated with increases in the percent frequency of opportunistic and tolerant taxa (e.g., large gulls and cormorants), and with declines in the percent frequency of wading and shallow bottom feeding taxa (e.g. dabbling ducks, herons, and shorebirds). Percent frequency of diving ducks (the most abundant taxon in the winter surveys) alongshore also declined as urban land cover along shore increased. These results document

declining diversity in marine bird and waterfowl assemblages across greater Puget Sound, and demonstrate that local human activity influences assemblage composition. In addition, oceanographic sub-basins, estuaries, and individual shoreline segments are effective sampling units, and simple taxonomic, trophic, and natural history attributes of assemblage composition are informative metrics for studying marine birds and waterfowl.

Introduction

Effective ecosystem monitoring and assessment requires sampling across taxa at appropriate spatial and temporal scales (Weins et al. 1986, Levin 1992), and identifying and monitoring attributes of the biota that are responsive to natural and anthropogenic influences (NRC 1990, Hughs et al. 2005, Karr 2006). Such efforts are lacking in most coastal ecosystems, including Puget Sound, a fjord-estuary of high ecological and socioeconomic value, and affected in many ways by growing regional and global human populations. Historical changes in Puget Sound biota include population extinction or decline in many species, including marine birds and waterfowl (West 1997, PSAT 2007b), yet documentation of the full extent, character, and causes of these changes is poor, particularly with respect to the role of human activity.

Direct biological observations provide the most useful information in understanding ecosystems (Slobodkin 1994). Compositional attributes (e.g., number of taxa, relative abundance of certain species or species groups) are especially valuable in diagnosing ecological condition (Karr 1991, Karr and Chu 1999, Hughs et al. 2005, Karr 2006) but are rarely used in Puget Sound research and monitoring programs. Bird assemblages are useful environmental indicators because of their ecological diversity, sensitivity to human activity, ease of observation, and public appeal (Furness and Greenwood 1993). To explore relationships of marine bird and waterfowl assemblage composition across years, geographic locations, and degrees of human influence in greater Puget Sound, I combined and reanalyzed data from

several separate environmental monitoring and assessment efforts: aerial bird surveys (Nyswander et al. 2002, PSAT 2007b) and maps of physical shoreline structure (WDNR 2001) and land cover (Hepinstall et al. in prep). My specific research objectives were to determine whether the taxonomic diversity of marine birds and waterfowl assemblages has declined over time, and whether urbanization affects marine bird and waterfowl assemblage structure. Also, I compared taxonomic assemblage structure among oceanographic sub-basins, estuaries, and shoreline segments to explore spatial and habitat patterns of bird assemblage composition, and to see if these categories might be effective sampling units for marine birds and waterfowl. These analyses allowed me to identify some biological attributes and analytical methods that could be useful in research and monitoring in greater Puget Sound.

Greater Puget Sound

Puget Sound is a diverse and productive fjord-estuary complex (many sub-estuaries exist within the larger Puget Sound estuary) that is part of an inland sea that includes the Strait of Georgia and Strait of Juan de Fuca (see Chapter 1). Areas adjacent to Puget Sound proper (the Strait of Juan de Fuca, San Juan Islands, and Bellingham and Padilla Bays) are often included in the definition of Puget Sound. Here, I collectively call them greater Puget Sound to refer to the whole study area while recognizing the distinct oceanographic boundaries of Puget Sound proper.

Although greater Puget Sound remains a biologically rich and productive ecosystem (PSP 2006, PSAT 2007b, Ruckelshaus and McClure 2007), dozens of fish and wildlife species have declined during recent decades (West 1997, PSAT 2007b), and pressure on the ecosystem from growing regional and global human populations is steadily increasing. The ecology of greater Puget Sound is presumably tied closely to many natural factors such as the geologic composition and morphology, configuration and connectivity of water masses (Strickland 1983), physical forcing and nutrient delivery from the Pacific Ocean, and freshwater runoff from the

surrounding watersheds (Strickland 1983, Gargett 1997, Pinnix 1999), but also by many forms of human activity including fishing, addition of nutrients and chemical pollutants, physical and hydrologic alterations in the surrounding watersheds (e.g., Simenstad et al. 1992), physical disturbance at the land-water interface (Williams and Thom 2001, Sobocinski 2003, Toft et al. 2004, Rice 2006), and species introductions (Carlton 2001), as well as climate change at the global scale (Snover et al. 2005).

This diversity of natural and human factors results in a complex mosaic of ecosystem parts and processes that presents many challenges for ecological science (e.g., selecting appropriate scales, sampling units, and measurements for research and monitoring) and management (e.g., identifying priorities for conservation and restoration). Biological data on ecologically diverse taxa over extensive spatial and temporal scales large enough to capture the full range of natural and human influences can help meet these challenges by defining spatial and temporal units of distinct biotic character, and identifying attributes of the biota that are responsive to natural and, particularly important in monitoring and assessment, human influences.

One potentially useful source of Sound-wide biological information is bird surveys. Under the auspices of the Puget Sound Assessment and Monitoring Program (PSAMP) the Washington Department of Fish and Wildlife (WDFW) has conducted aerial marine bird and waterfowl surveys along the entire shoreline of greater Puget Sound since 1992, at least once per year in winter and also during summer through 1998 (Nyswander et al. 2002). These surveys were based on surveys done in the Strait of Juan de Fuca during late 1970s as part of the Marine Ecosystem Analysis project (Wahl et al. 1981, Long 1982), and expanded the geographic coverage to all of greater Puget Sound. Maps of individual species distributions and population estimates for selected species of concern have been produced from these data, and severe declines have been detected in a number of species (PSAT 2007b). Recent Western Washington University ground and boat surveys at several historical MESA sites in eastern Strait of Juan de Fuca indicate

that 23 of the 35 most abundant species in the MESA studies have declined by at least 20 % (John Bower, Western Washington University, unpublished data).

To date, however, few multispecies analyses have been done on marine birds and waterfowl in greater Puget Sound, and no attempt has been made to relate changes in bird populations and assemblages to human activity. By applying GIS analysis to the PSAMP bird data supplemented with environmental and life history information, I explored the utility of marine bird and waterfowl fauna as a tool for monitoring and research on the biological condition of Puget Sound. In addition to summarizing the overall patterns of taxonomic assemblage structure at several spatial scales (oceanographic sub-basin, estuary, beach segment), I used analysis of land cover and shoreline structure data to characterize various taxa in terms of their degree of association with urbanization. Sampling a large geographic area allowed comparisons across a range of oceanographic conditions and degrees of human influence.

Methods

All analyses were done with existing data manipulated by geographic information
system (GIS), spreadsheet, and statistical analysis software. A summary of the data
processing steps is presented in Figure 5.1.

```
┌─────────────────────────────────────────────────────────────────┐
│          Acquire and edit existing GIS data layers              │
│  (bird counts, Shorezone Inventory, land cover, bird life        │
│                  history information)                            │
└─────────────────────────────────────────────────────────────────┘
                              │
                              ▼
┌─────────────────────────────────────────────────────────────────┐
│               Create new GIS data layers                        │
│  (sub-basin, river mouth estuaries, urbanized land cover         │
│                     gradient)                                    │
└─────────────────────────────────────────────────────────────────┘
                              │
                              ▼
┌─────────────────────────────────────────────────────────────────┐
│          Merge selected features from GIS data layers           │
│  (bird points, sub-basin polygons, shore segment arcs, urban     │
│                    land cover)                                   │
└─────────────────────────────────────────────────────────────────┘
                              │
                              ▼
┌─────────────────────────────────────────────────────────────────┐
│      Export tabular data for graphical and statistical analysis │
└─────────────────────────────────────────────────────────────────┘
```

Figure 5.1. Data processing stages for analysis.

Data Sources and Processing

Bird data were obtained through the Wildlife Resources Data Systems (WRDS) of
the Washington Department of Fish and Wildlife (WDFW) and are point
observations from aerial surveys conducted from 1992 to 2004 as part of the Puget
Sound Assessment and Monitoring Program (PSAMP) (Nyswander et al. 2002,
PSAT 2007b). The surveyors fly along shore and also do transects across open water
recording the occurrence, count, and taxonomic identification of birds. Summer
surveys were done from 1992 to 1998 in July, and winter surveys were done in all
years from December to February. Because of budget cuts, 1997 and 1998 summer

surveys did not cover central and southern Puget Sound, and summer surveys were terminated entirely in 1999. Only years with complete coverage of the study area (summer 1993-1996; winter 1993-2003) were used in the analysis. Land cover data was produced by the University of Washington's Urban Ecology Research Lab for the Puget Sound Regional Synthesis Model (Hepinstall et al. in prep). The data are from a classified 2002 LANDSAT image and are 30 x 30 m grid cells each assigned to one of seventeen classes. Shoreline structure information was taken from the ShoreZone inventory produced by the Washington Department of Natural Resources (WDNR 2001).

I used GIS software (ArcGIS versions 8 and 9 by ESRI, Inc.) to combine bird, land cover, and ShoreZone data. Oceanographic sub-basins generally followed conventional designations for greater Puget Sound (Burns 1985) but were subdivided in the Main Basin to better capture heterogeneity in oceanography and urban development (Figure 5.2). Polygons were also created around seven of the larger river mouth estuaries in Puget Sound proper (Figure 5.2). Bird points (Figure 5.3) were intersected with sub-basin and estuary polygons to assign sub-basin and estuary identifiers to bird points. Because of changes over years in the identification of several taxa (e.g., gull species) taxonomic designations were created to avoid this bias across years (Appendix A). Life history information was assigned to each taxon based on information summarized in several sources (Angell and Balcomb 1982, Cornell Lab of Ornithology 2005). Additional taxonomic and life history information was then added to all bird point observations using a table join. A 500 m buffer was applied to the entire shoreline of the study area to select bird points alongshore for use in analysis of relationships between assemblage composition and environmental attributes (sub-basin and urbanization) of the adjacent littoral and upland areas.
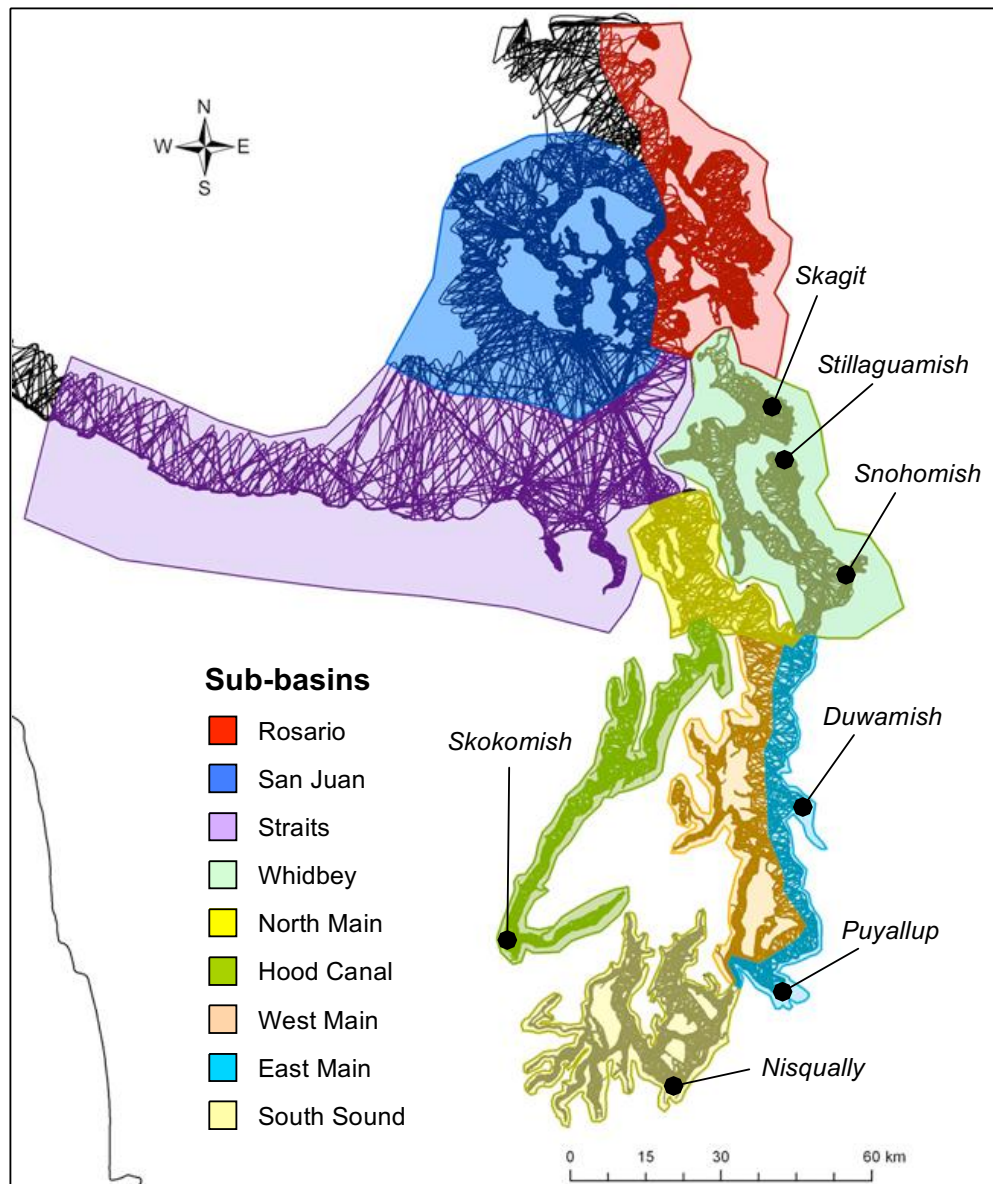
Figure 5.2. Map of the greater Puget Sound study area showing sub-basin designations and river mouth estuaries used in the analysis. The area traditionally designated the Main Basin was subdivided into northern, eastern, and western sub-basins to better capture effects of urbanization and oceanographic heterogeneity. Colored areas are polygons used to select features (bird points, flight lines, etc.) in the GIS analysis. Lines are the actual flight lines traveled during the collection of the data used in the study (1993-2003).
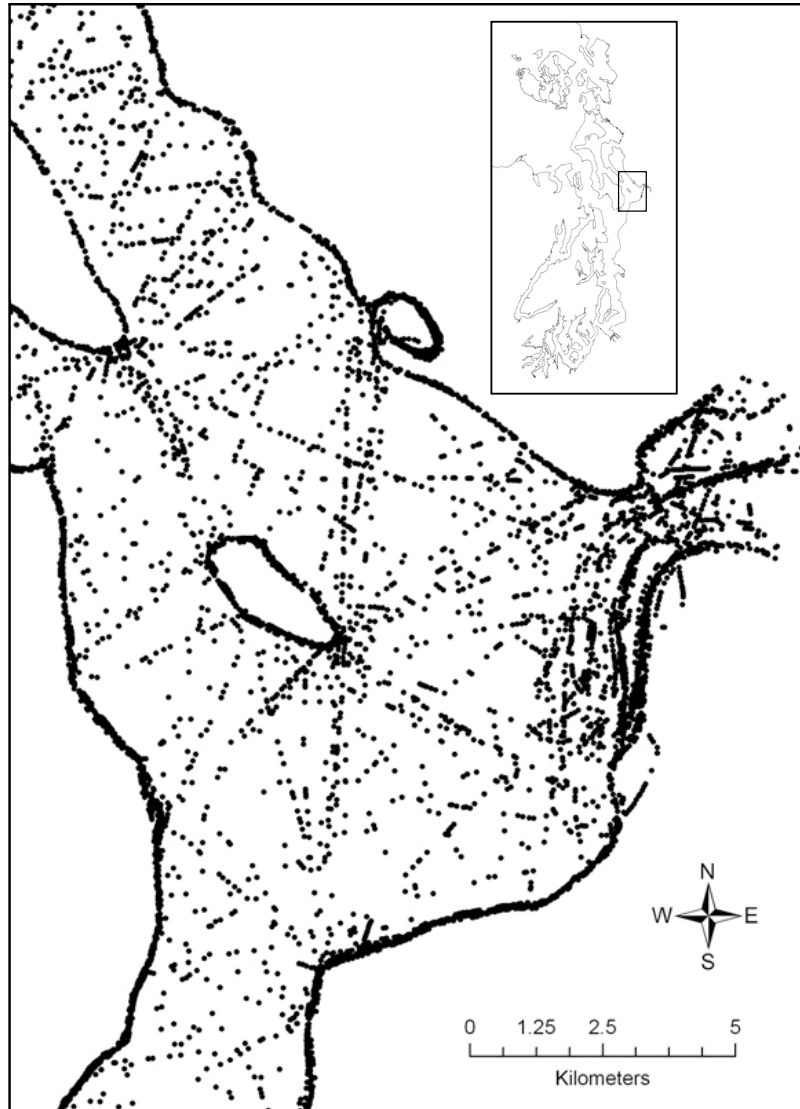
Figure 5.3. Bird points in the Snohomish River estuary and Port Susan area. Winter point observations from 1993 to 2003 are shown. Aerial surveys fly along all shorelines, then fly transects over open water. Inset map shows Puget Sound and location of detail.

Using the Spatial Analyst tool of ArcGIS, land cover was reclassified from 17 classes to three. To create a simple urbanization gradient, the two most urban classes were classified as "1," all other land cover was classified as "0," and water, clouds, or missing data were classified as "no data," yielding an urban/non-urban land cover map (Figure 5.4). To associate the urban land cover information with shoreline segments, each individual cell of the land cover grid was assigned to ShoreZone segments using a Euclidean allocation tool in three iterations at 0.5 km, 1 km, and 2 km. This function assigned each cell an identifier based on its nearest shoreline segment without relating each cell to more than one segment. Cells with the same identifier were then combined and converted to polygons using Spatial Analyst. These polygons were then joined with individual ShoreZone segments. The "zonal statistics as table" tool was used to calculate the percentage of urban land area for each polygon. Thus, each shoreline polygon acquired the attributes of the ShoreZone segments, and a percent urban value. The 2 km land cover areas produced the strongest gradient in urbanization (data not shown) and thus was used in the analysis. Using a spatial join, each individual bird point alongshore acquired all of the environmental attributes of the shoreline polygons (Figure 5.5).

Sums of point observations and total abundance of each taxon were calculated for each combination of year and sub-basin, estuary, and shorleline segment. Relative frequencies and relative abundances were calculated by dividing the values for individual taxa by the total point observation counts or total individual abundances for all taxa.
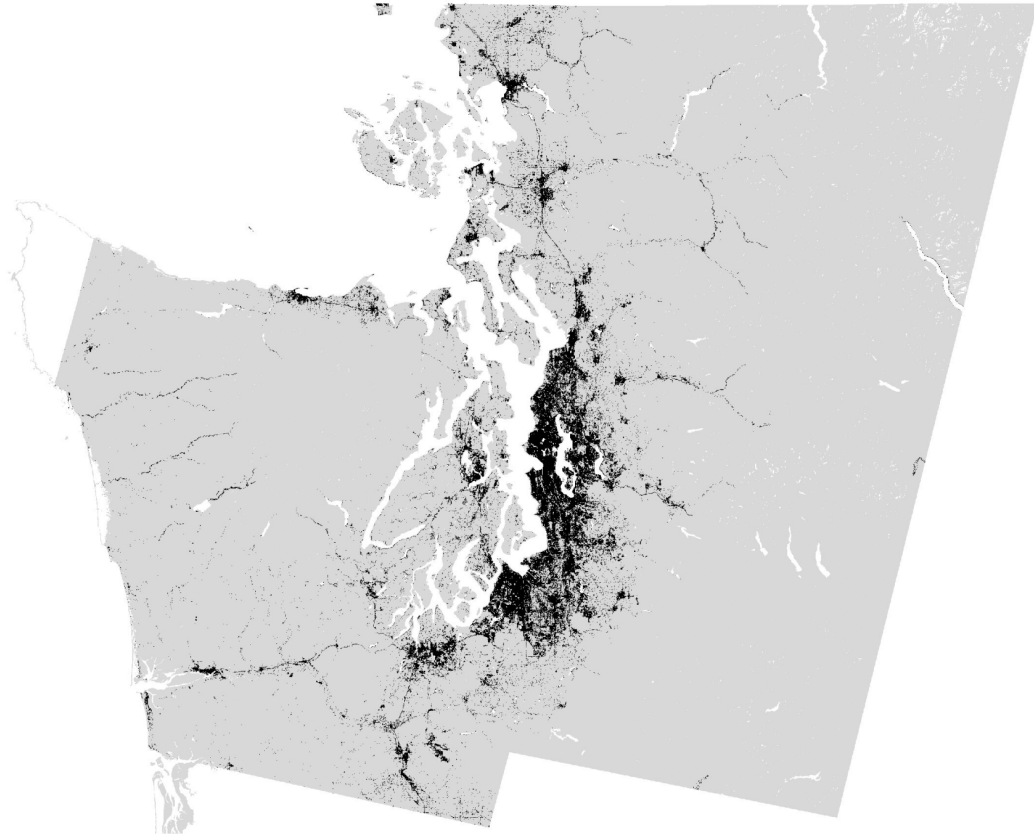
Figure 5.4. Land cover of the greater Puget Sound study area converted to urban (black areas), nonurban (light grey areas), and other (white). Seventeen land cover classes were collapsed to these three (data from Hepinstall et al. in preparation).
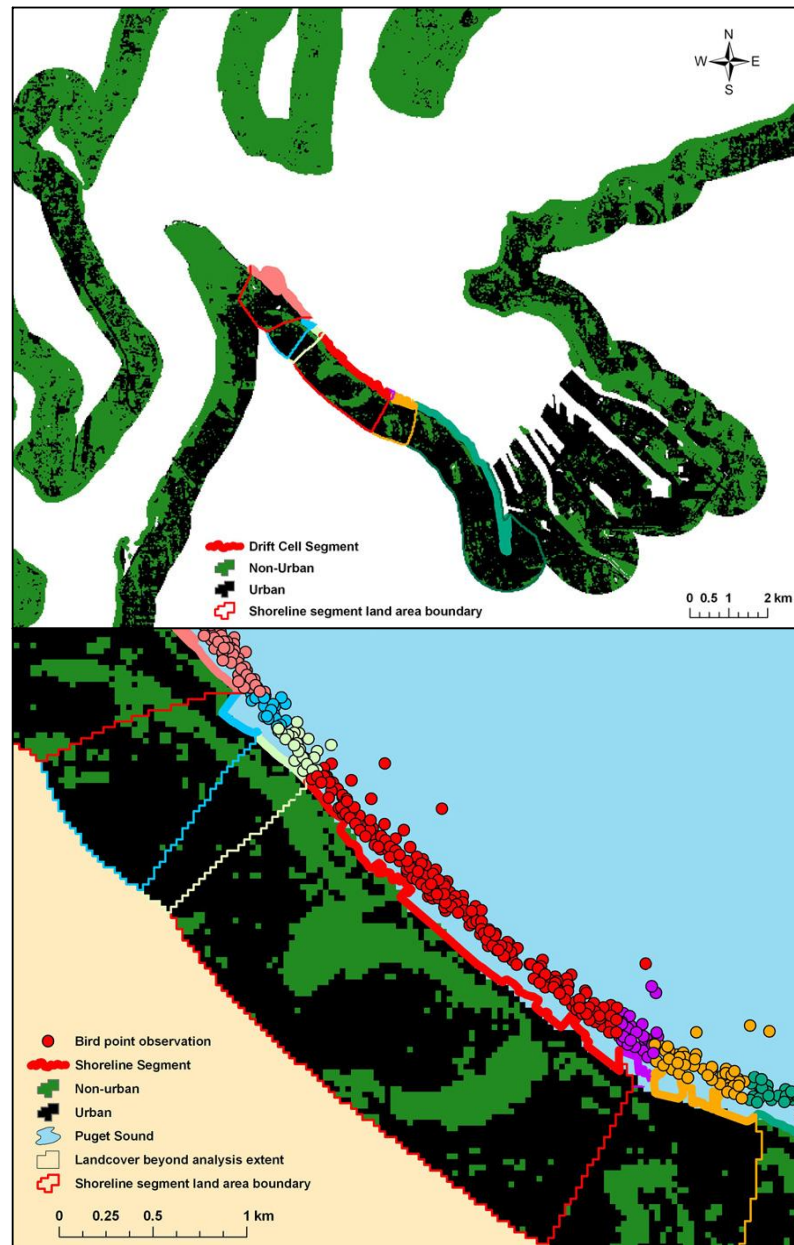
Figure 5.5. Shorelines in and around Commencement Bay (top panel) showing shoreline segments (thick colored lines) and urban (black) and nonurban (green) land areas in 1 km buffers alongshore (2 km buffer was used in the analysis). Bottom panel is a detail of colored shoreline segments in top panel showing alongshore bird points (colored dots), shoreline segment (thick colored lines), and urban (black) and nonurban (green) land cover.

Statistical Analysis

Statistical analysis focused on changes in number of taxa encountered in each shoreline segment across years, and the influence of season, geographic location, and urbanization on relative frequency and relative abundance of individual taxa. Multivariate statistical treatments were used to evaluate bird assemblage composition differences among, and similarities within, years, sub-basins, and estuaries; and graphs were used to evaluate relationships between taxonomic composition and urbanization along shore. All multivariate analyses were conducted following the approach of Clarke and Warwick (Clark 1993, Clark and Warwick 2001) using Primer statistical software version 6 (Clarke and Gorley 2006).

Relative frequencies and relative abundances were calculated for each combination of year, season, and sub-basin; and year, season, and estuary. The data matrices were then transformed to down-weight the effect of highly abundant taxa using a square-root transformation, and non-metric multidimensional scaling (MDS) ordinations were performed on Bray-Curtis dissimilarity matrices of the data. Graphs of the resulting MDS ordinations were used to evaluate relationships among years from the four basins in 2 and 3 dimensions. A dimensionless "stress" value provides a non-parametric regression estimate of reliability of the ordination. Values < 0.2 are considered to give a useful and interpretable representation of the relationships among samples (Clark and Warwick 2001, McCune and Grace 2002).

A two-way analysis of similarity procedure (ANOSIM) was applied to the resemblance matrices to evaluate *differences* in the assemblage composition based on year, season, and sub-basin; and also for year, season, and estuary. The ANOSIM procedure calculates an R statistic based upon the difference between average within group rank similarities and average among group rank similarities. Values of R range between 0 (rank similarities between and within areas or months are the same, on average) and 1 (all replicates within areas or months are more similar to each other than any replicates from different areas or months), and significance level is computed using a permutation procedure. A two-way similarity percentages

procedure (SIMPER) was applied to the same resemblance matrices to evaluate the contribution of various taxa to *similarities* in the assemblage composition by year, season, and oceanographic basin or estuary.

To evaluate relationships between taxonomic composition of bird assemblages and urbanization along shore, shoreline segments were aggregated into ten groups (0 – 100% urbanization in 10% quantiles) and mean number of taxa and relative frequency of selected taxa analyzed by graphs of means across the urbanization gradient.

Results

Beach Characteristics

A total of 3918 individual shoreline segments occurred in the designated study area, and segments between basins differed considerably in length and urbanization (Table 5.1). Segment lengths averaged 2.2 km overall and were longest (4.8 km) in the northern sub-basin of the Straits, and shortest in the West Main and Sand Juan areas (1.6 km). Urbanization within 2 km of the shoreline averaged 12 %, and the only area above 13 % was the East Main sub-basin, which averaged 53 %.

Table 5.1. Shoreline counts, lengths, and percent urban land cover (2002) for each
sub-basin in the study area.

Sub-basin	Segment Count	Mean Length (km)	Median Length (km)	Mean Urbanization (%)	Median Urbanization (%)
Rosario	419	2.5	1.8	13	3
San Juan	745	1.6	1.3	8	2
Straights	200	4.8	3.1	11	5
Whidbey	481	1.9	1.3	10	6
North Main	196	2.5	2.1	10	5
East Main	190	2.1	1.6	53	56
West Main	573	1.6	1.3	13	6
Hood Canal	397	2.7	1.9	5	2
South Sound	717	2.4	1.9	8	4
Total	3918	2.2	1.6	12	4

Overall Assemblage Composition

   Numbers of taxa, point observations, and total individual birds included in the
analysis are listed in Table 5.2. Assemblage composition based on relative
abundance and relative frequency both showed clear statistical differences among
sub-basins, individual estuaries, and urban and non-urban estuaries, but differences
among years were either weak (by relative frequency) or not present (by relative
abundance) (Table 5.3). Relative frequency generally showed the strongest statistical
differences, and the strongest overall difference was between urban and non-urban
estuaries (Table 5.3). Analyses were performed using both relative frequency and
relative abundance. Because of the similarity in overall results from the two
measures, and the generally stronger statistical performance of relative frequency,
only relative frequency and taxa richness analyses are subsequently shown.

Table 5.2. Counts of taxa, point observations, and individual birds included in the analysis.

| | Summer | | Winter | | | |
| | All | Shore | All | | Shore | |
	1993-1996	1993-1996	1993-1996	1993-2003	1993-1996	1993-2003
Taxa	23	21	21	21	21	21
Points	47,636	31,829	146,424	349,434	109,743	266,287
Individuals	350,981	248,975	1,399,003	2,543,880	978,673	1,856,898

Table 5.3. ANOSIM global R statistics for comparisons for assemblage *differences* among years, seasons, sub-basins, river mouth estuaries, and estuaries classified as urban or non-urban. Two-way tests for differences based on relative abundance and relative frequency (999 permutations in each test).

| Comparison Type | Winter | | Summer | |
Oceanographic Sub-basins	Relative Abundance	Relative Frequency	Relative Abundance	Relative Frequency
Year (across basins)	0.08 (p = 0.13)	0.34 (p = 0.03)	0 (p = 0.88)	0.07 (p = 0.17)
Sub-basin (across years)	0.63 (p = 0.01)	0.69 (p = 0.01)	0.59 (p = 0.01)	0.74 (p = 0.01)
Estuaries				
Year (across estuaries)	0.03 (p = 0.33)	0.24 (p = 0.03)	0 (p = 0.90)	0 (p = 0.84)
Estuary (across years)	0.53 (p = 0.01)	0.71 (p = 0.01)	0.36 (p = 0.01)	0.49 (p = 0.01)
Urban/Non-urban (across years)	0.32 (p = 0.02)	0.82 (p = 0.01)	0.51 (p = 0.01)	0.43 (p = 0.01)

Assemblage Composition in Oceanographic Sub-basins

Despite the generally weak taxonomic signals in the multivariate analysis of geographic areas based on year, clear differences were seen over time when shoreline segment was the sampling unit. Mean number of taxa per shoreline segment in both summer (Figure 5.6) and winter (Figure 5.7) was between 1 and 5, varied among sub-basins, and showed downward trends in all sub-basins except for South Sound in summer. Patterns were similar after normalizing number of taxa for length of shoreline segment (data not shown).
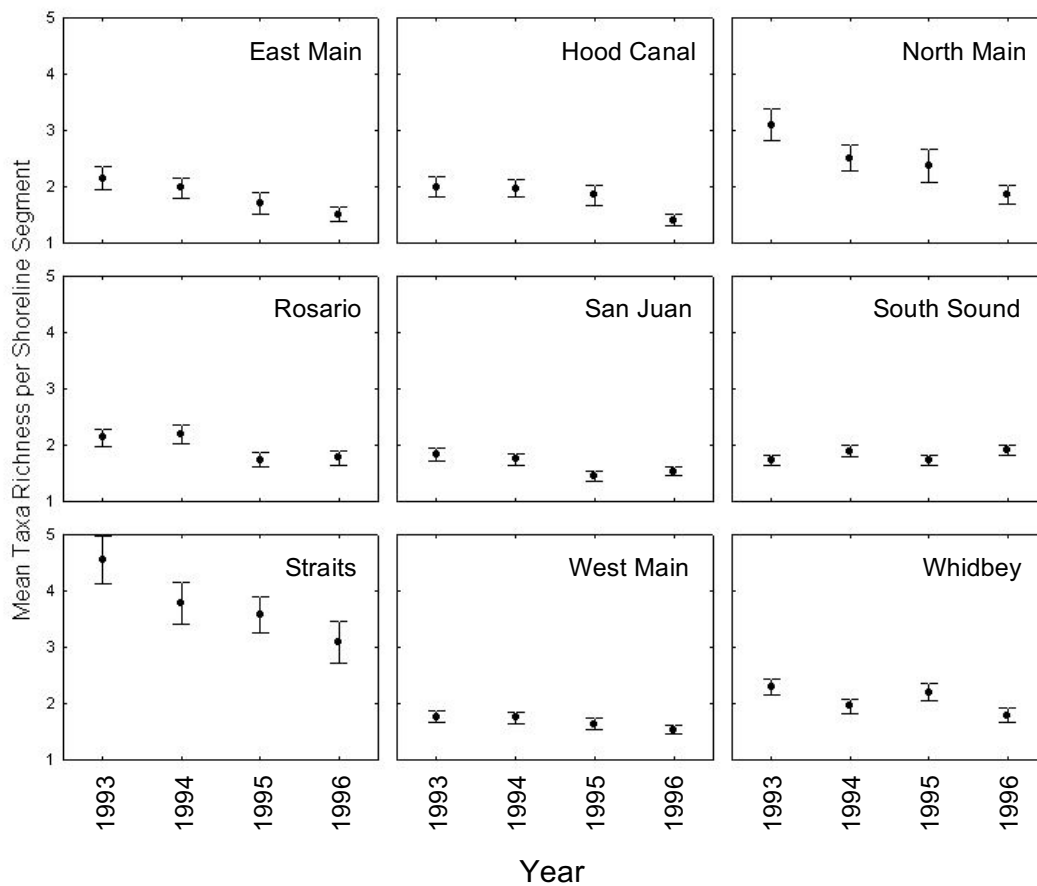
Figure 5.6. Mean number of taxa (± 95% CI) per shoreline segment during summer in all nine sub-basins from 1993 to 1996.
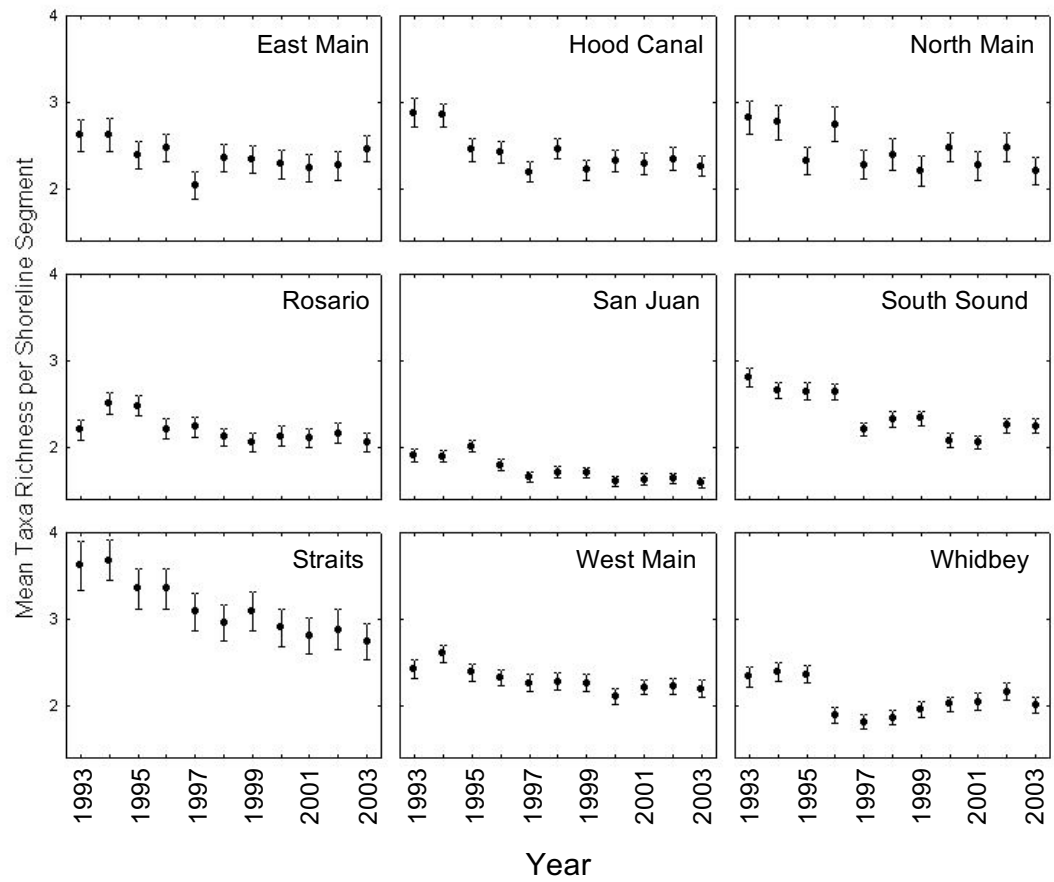
Figure 5.7. Mean number of taxa (± 95% CI) per shoreline segment during winter in all nine sub-basins from 1993 to 2003.

Seasonal differences in assemblage composition were statistically very strong (global R = 1 across all sub-basins using year as the sampling unit), reflecting the change in composition associated with seasonal migrations, particularly among winter migrants such as diving ducks and grebes (Table 5.4). Comparisons between individual sub-basins (Table 5.5) also showed strong differences with a few exceptions (e.g., between Hood Canal and West Main and South Sound in summer and winter; and, in summer, between Hood Canal and Whidbey; South Sound, West Main, and East Main; and North Main and the Straits). MDS ordinations of assemblage composition based on sub-basin designations showed moderately strong statistical structure and clear graphical grouping among most sub-basins in summer and winter (Figure 5.8) including the subdivisions of what is traditionally called the Main Basin. Contributions of various taxa to within sub-basin similarity in summer and winter (Appendix B) show that large gulls dominate in all areas (especially the East Main) in summer and are second to diving ducks in winter. Taxonomic similarity of sub-basins with extensive estuarine wetlands or protected embayments also tended to be influenced more by waders (e.g., herons and shorebirds) and shallow bottom feeders (e.g., dabbling ducks) (Appendix B).

Table 5.4. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter across all sub-basins (SIMPER test; 1993-1996).

Summer			Winter		
Average similarity: 87			Average similarity: 90		
Taxon	Contrib.	Cumul.	Taxon	Contrib.	Cumul.
	%	%		%	%
Large Gulls	31	31	Diving Ducks	26	26
Auks	15	46	Large Gulls	19	45
Herons	10	56	Grebes	9	54
Crows	7	63	Cormorants	8	62
Cormorants	6	69	Auks	8	70
Diving Ducks	6	75	Dabbling Ducks	7	77
Terns	4	79	Loons	6	83
Small Gulls	4	83	Herons	3	86
Shorebirds	3	86	Crows	2	88
Kingfishers	3	89	Geese	2	90
Raptors	3	92			

Table 5.5. ANOSIM comparisons of assemblage differences among sub-basins in summer and winter (1993-1996) ranked in order of greatest difference.

Summer Global R = 0.79 (p = 0.01)			Winter Global R = 0.74 (p = 0.01)		
Comparison	R	p	Comparison	R	p
East Main, Straights	1.00	0.03	Hood Canal, San Juan	1.00	0.03
North Main, Rosario	1.00	0.03	Rosario, San Juan	1.00	0.03
North Main, South Sound	1.00	0.03	Rosario, South Sound	1.00	0.03
North Main, West Main	1.00	0.03	San Juan, South Sound	1.00	0.03
North Main, Whidbey	1.00	0.03	San Juan, West Main	1.00	0.03
Rosario, Straights	1.00	0.03	San Juan, Whidbey	1.00	0.03
San Juan, South Sound	1.00	0.03	East Main, San Juan	0.96	0.03
San Juan, West Main	1.00	0.03	East Main, Rosario	0.95	0.03
San Juan, Whidbey	1.00	0.03	Straights, Whidbey	0.94	0.03
South Sound, Straights	1.00	0.03	Rosario, West Main	0.90	0.03
Straights, West Main	1.00	0.03	Hood Canal, Straights	0.89	0.03
Straights, Whidbey	1.00	0.03	Straights, West Main	0.88	0.03
West Main, Whidbey	1.00	0.03	North Main, San Juan	0.87	0.03
South Sound, Whidbey	0.99	0.03	East Main, Straights	0.85	0.03
Rosario, San Juan	0.98	0.03	Hood Canal, Rosario	0.84	0.03
East Main, San Juan	0.97	0.03	East Main, Hood Canal	0.83	0.03
Rosario, Whidbey	0.97	0.03	East Main, Whidbey	0.81	0.03
North Main, Straights	0.96	0.03	North Main, Rosario	0.81	0.03
East Main, North Main	0.95	0.03	Rosario, Straights	0.81	0.03
Hood Canal, Straights	0.91	0.03	East Main, South Sound	0.80	0.03
Rosario, West Main	0.90	0.03	East Main, North Main	0.79	0.03
Rosario, South Sound	0.89	0.03	Rosario, Whidbey	0.79	0.03
East Main, Whidbey	0.87	0.03	South Sound, Straights	0.79	0.03
Hood Canal, San Juan	0.82	0.03	South Sound, Whidbey	0.78	0.03
East Main, South Sound	0.80	0.03	North Main, Whidbey	0.68	0.03
East Main, Rosario	0.78	0.03	North Main, West Main	0.63	0.03
Hood Canal, North Main	0.78	0.03	West Main, Whidbey	0.57	0.03
North Main, San Juan	0.76	0.03	North Main, South Sound	0.56	0.03
East Main, Hood Canal	0.74	0.03	Hood Canal, North Main	0.55	0.03
East Main, West Main	0.65	0.03	San Juan, Straights	0.48	0.03
Hood Canal, Whidbey	0.64	0.03	Hood Canal, South Sound	0.43	0.03
South Sound, West Main	0.53	0.03	Hood Canal, West Main	0.40	0.09
San Juan, Straights	0.50	0.03	South Sound, West Main	0.34	0.06
Hood Canal, Rosario	0.26	0.06	Hood Canal, Whidbey	0.24	0.14
Hood Canal, West Main	0.21	0.14	East Main, West Main	0.21	0.14
Hood Canal, South Sound	0.07	0.29	North Main, Straights	0.05	0.40
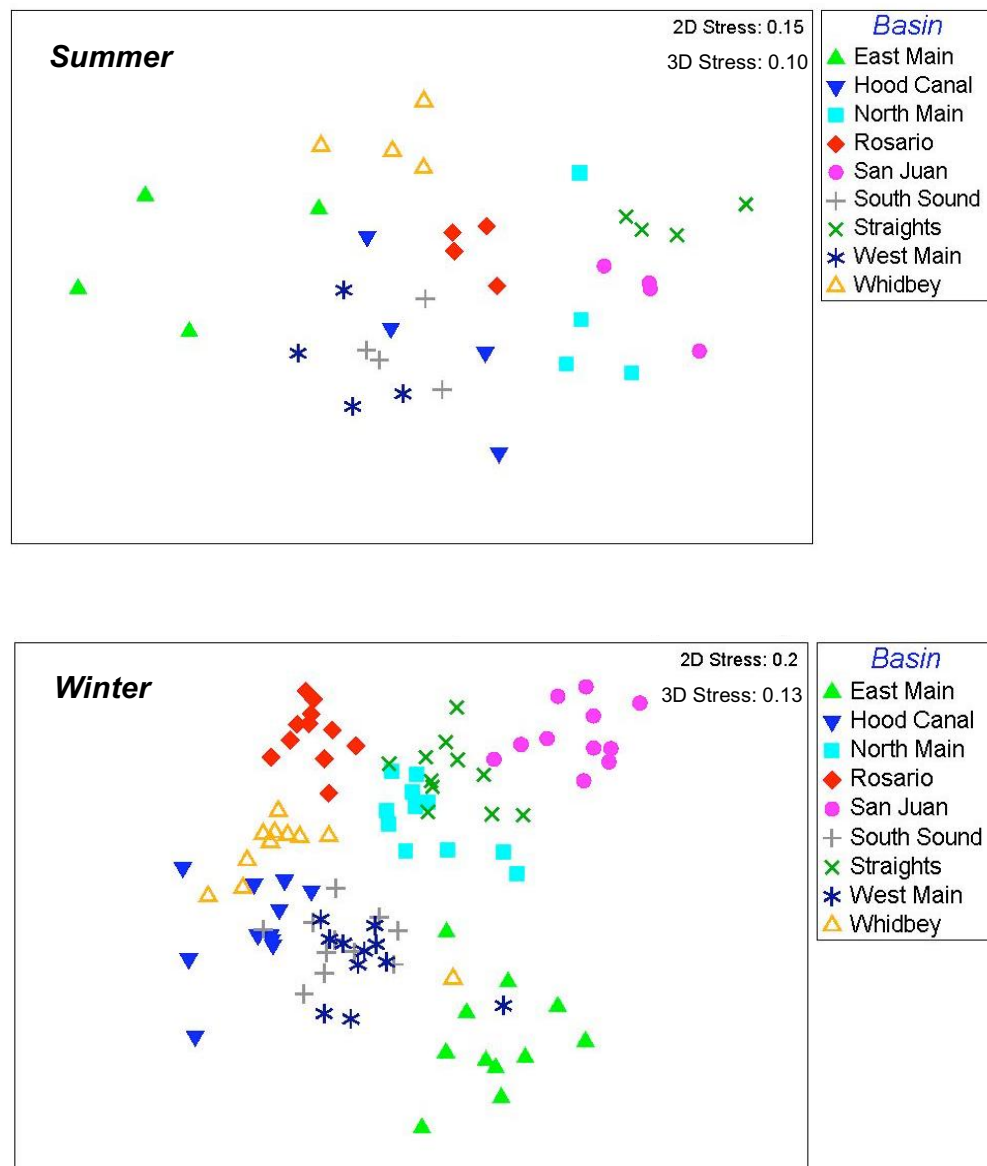
Figure 5.8. MDS ordination solution for summer (top) and winter (bottom) bird assemblage composition in sub-basins based on relative frequency of occurrence. Each point is one year (1993-2003).

Assemblage Composition in Estuaries

Comparisons between individual estuaries (Table 5.6) also showed strong differences especially between the heavily urbanized estuaries of the Duwamish, Puyallup, and (to a lesser degree) Snohomish rivers, and all others. MDS ordinations of assemblage composition based on estuary designations showed moderately strong statistical structure, but clear graphical grouping among years within estuaries, and the urban and non-urban estuaries grouping separately (Figure 5.9). Contributions of various taxa to within estuary similarity in summer and winter (Appendix C) show that large gulls dominate in all areas (especially the Duwamish where they contribute 96 %) in summer but are second to diving ducks in winter in some estuaries. Composition similarity of non-urban estuaries was influenced much more by waders (e.g., herons and shorebirds) and shallow bottom feeders (e.g., dabbling ducks) (Table 5.7; Appendix C).

Table 5.6. ANOSIM comparisons of assemblage differences among river mouth estuaries in summer and winter (1993-1996) ranked in order of greatest difference.

Summer			Winter		
Global R = 0.68 (p = 0.01)			Global R = 0.78 (p = 0.01)		
Comparison	R	p	Comparison	R	p
Duwamish, Nisqually	1.00	0.03	Duwamish, Skagit	1.00	0.03
Duwamish, Skagit	1.00	0.03	Duwamish, Skokomish	1.00	0.03
Duwamish, Snohomish	1.00	0.03	Duwamish, Stillaguamish	1.00	0.03
Puyallup, Skagit	1.00	0.03	Nisqually, Skagit	1.00	0.03
Duwamish, Stillaguamish	0.98	0.03	Puyallup, Skagit	1.00	0.03
Nisqually, Snohomish	0.96	0.03	Puyallup, Skokomish	1.00	0.03
Skagit, Snohomish	0.96	0.03	Puyallup, Stillaguamish	1.00	0.03
Nisqually, Puyallup	0.93	0.03	Snohomish, Stillaguamish	1.00	0.03
Nisqually, Skagit	0.89	0.03	Nisqually, Puyallup	0.98	0.03
Puyallup, Snohomish	0.78	0.03	Skokomish, Snohomish	0.98	0.03
Duwamish, Puyallup	0.76	0.03	Duwamish, Nisqually	0.97	0.03
Puyallup, Stillaguamish	0.69	0.03	Nisqually, Stillaguamish	0.96	0.03
Puyallup, Skokomish	0.66	0.03	Skagit, Snohomish	0.94	0.03
Skokomish, Snohomish	0.66	0.03	Skagit, Skokomish	0.84	0.03
Duwamish, Skokomish	0.63	0.03	Nisqually, Snohomish	0.79	0.03
Skagit, Skokomish	0.43	0.03	Skokomish, Stillaguamish	0.75	0.03
Nisqually, Stillaguamish	0.41	0.03	Nisqually, Skokomish	0.67	0.03
Nisqually, Skokomish	0.34	0.09	Puyallup, Snohomish	0.62	0.06
Snohomish, Stillaguamish	0.34	0.03	Duwamish, Snohomish	0.35	0.06
Skokomish, Stillaguamish	0.15	0.26	Duwamish, Puyallup	0.13	0.26
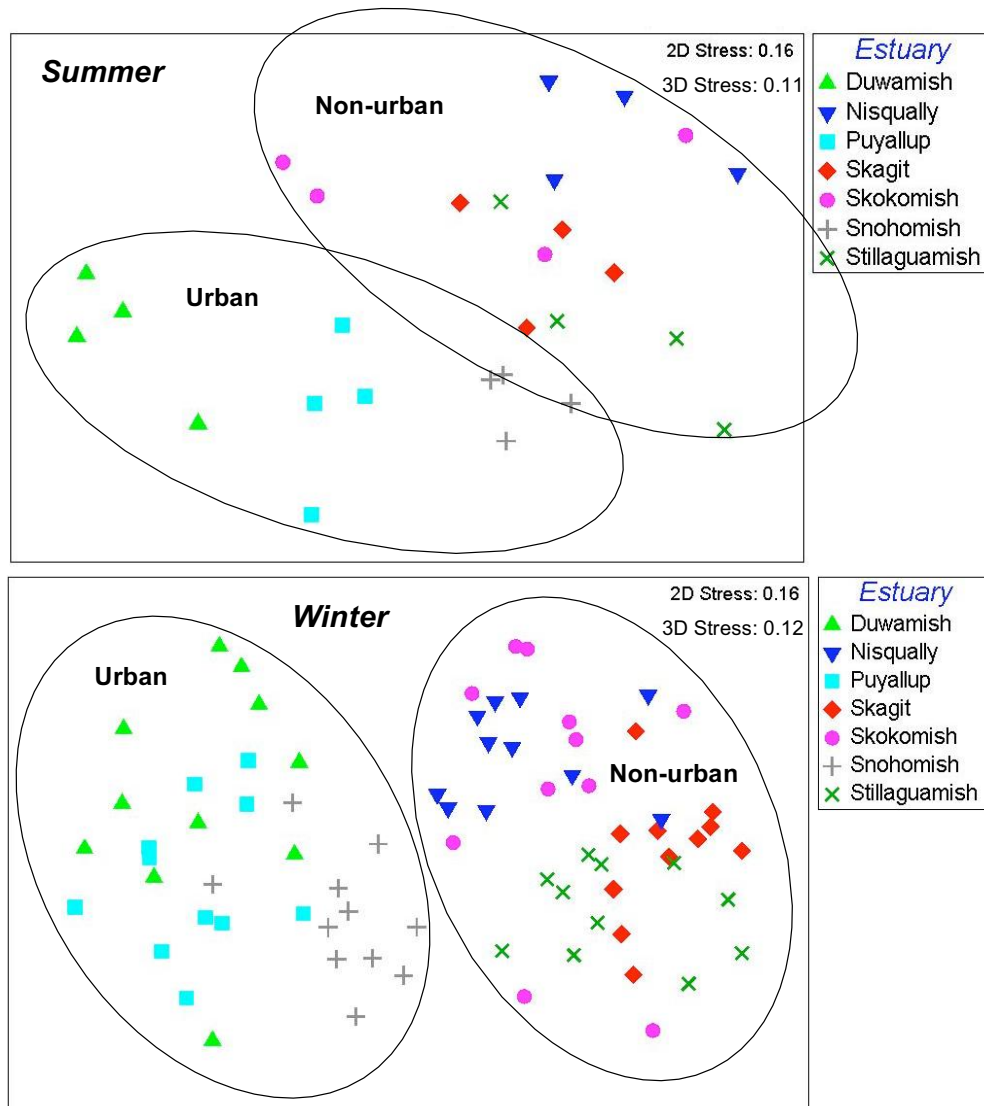
Figure 5.9. MDS ordination solution of summer (top) and winter (bottom) bird assemblage composition in river mouth estuaries based on relative frequency of occurrence. Each point is one year (1993-2003).

Table 5.7. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in urban (Duwamish, Puyallup, and Snohomish), and non-urban (Skagit, Stillaguamish, Skokomish, and Nisqually) estuaries (SIMPER test; 1993-1996).

Summer

Urban			Non-urban		
Average similarity across years: 82			Average similarity across years: 88		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	69	69	Large Gulls	43	43
Terns	9	78	Herons	15	58
Geese	8	86	Terns	11	70
Crows	6	92	Diving Ducks	7	77
			Small Gulls	5	82
			Crows	4	85
			Dabbling Ducks	4	89
			Raptors	3	92

Winter

Urban			Non-urban		
Average similarity across years: 76			Average similarity across years: 76		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	32	32	Diving Ducks	29	29
Diving Ducks	23	54	Large Gulls	21	50
Grebes	20	74	Dabbling Ducks	15	65
Cormorants	16	90	Grebes	7	72
			Loons	6	78
			Cormorants	5	83
			Herons	4	88
			Shorebirds	4	91

Assemblage Composition Along the Urban Gradient

Scatterplots of the number of taxa in each shoreline segment across the study area in 2002 showed an apparent decline with urbanization (Figure 5.10). The overall form of the relationship appears to be a wedge-shaped "factor ceiling" distribution where the measured variable (urbanization) captures the overall limit, and points are scattered below the limit (Thomson et al. 1996, Scharf et al. 1998). But plotted as means (Figure 5.11) the number of taxa increases from approximately 2 to 2.2 at near 30 % urban, then declines to just under 2, with increasing variability as urbanization increased, at least in part a consequence of the sample distribution of beaches (Figure 5.12).
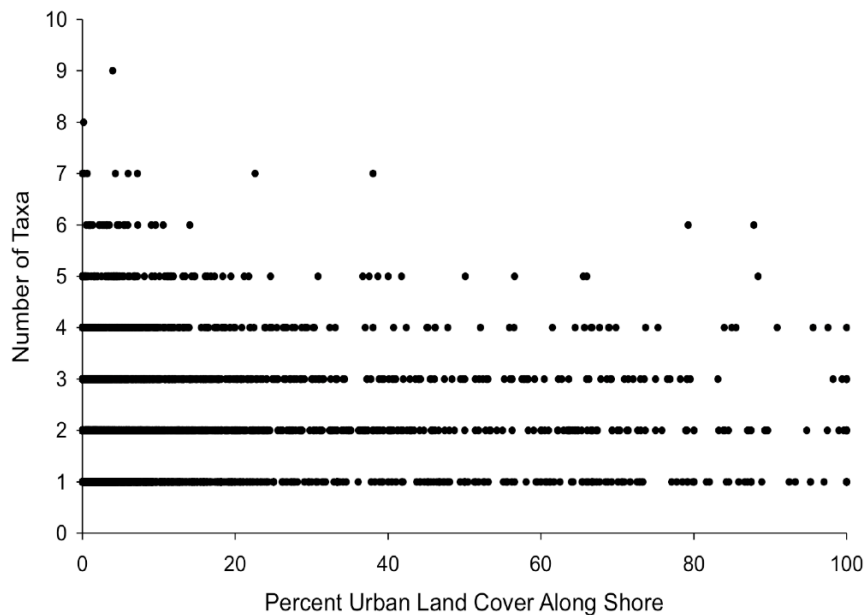
Figure 5.10. Number of taxa and percent urban land cover along shore in each of 3918 shoreline segments throughout the study area in winter 2002.
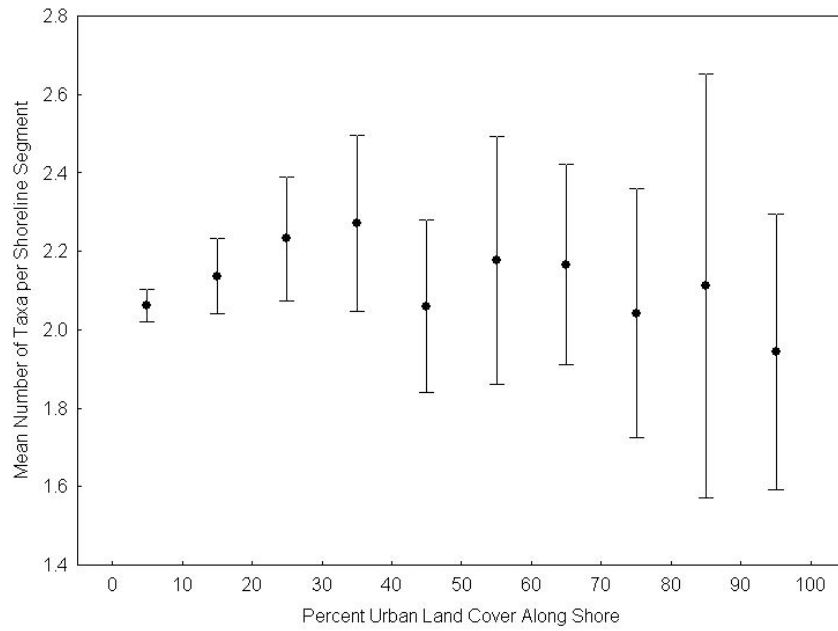
Figure 5.11. Mean number of taxa and percent urban land cover along shore in each of 3918 shoreline segments throughout the study area in 2002.
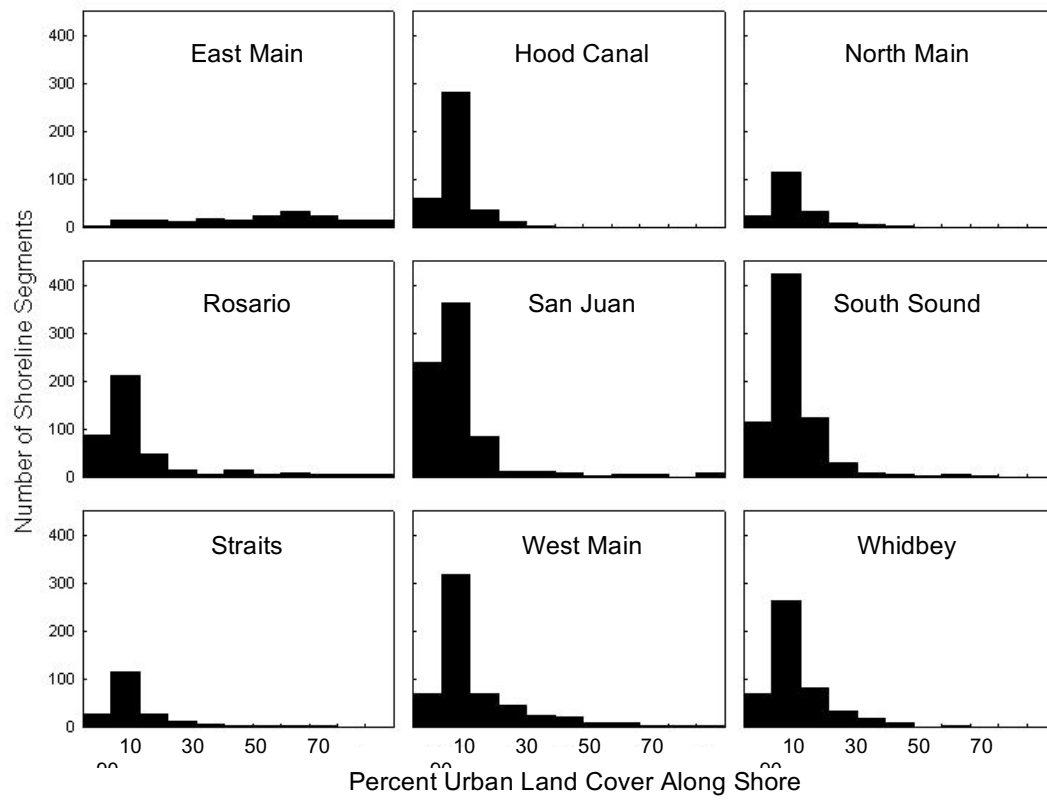
Figure 5.12. Frequency distributions of shorelines in each sub-basin based on percentage of urban land cover within 2 km of the shoreline.
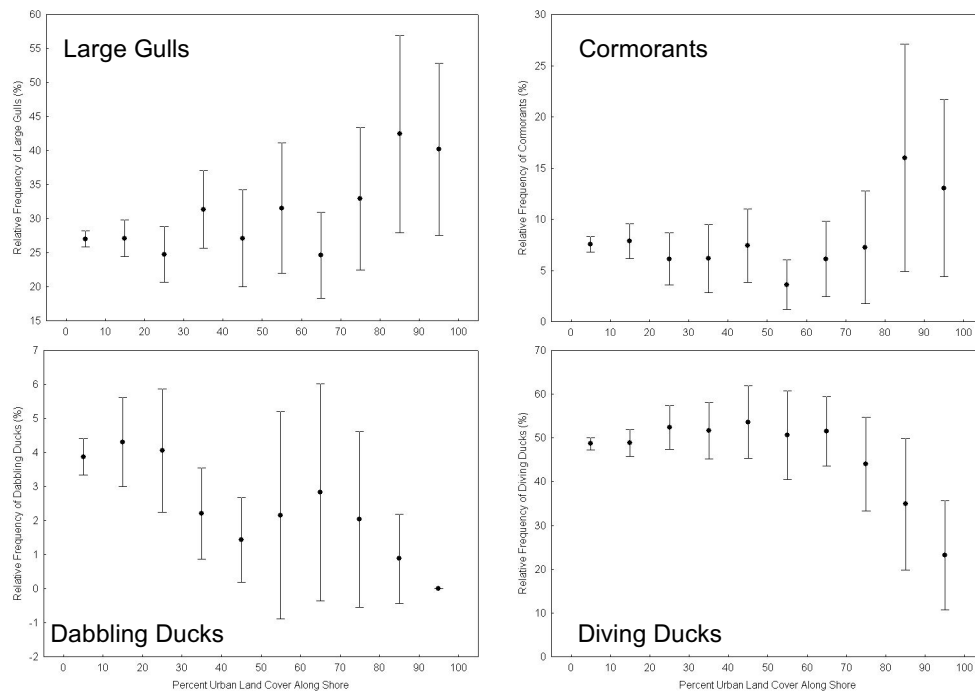
Figure 5.13. Mean relative frequency (± 95 % CI) of large gulls (top left), cormorants (top right), dabbling ducks (bottom left), and diving ducks (bottom right) along the land cover urbanization gradient across all sub-basins (3918 beach segments) in winter 2002.
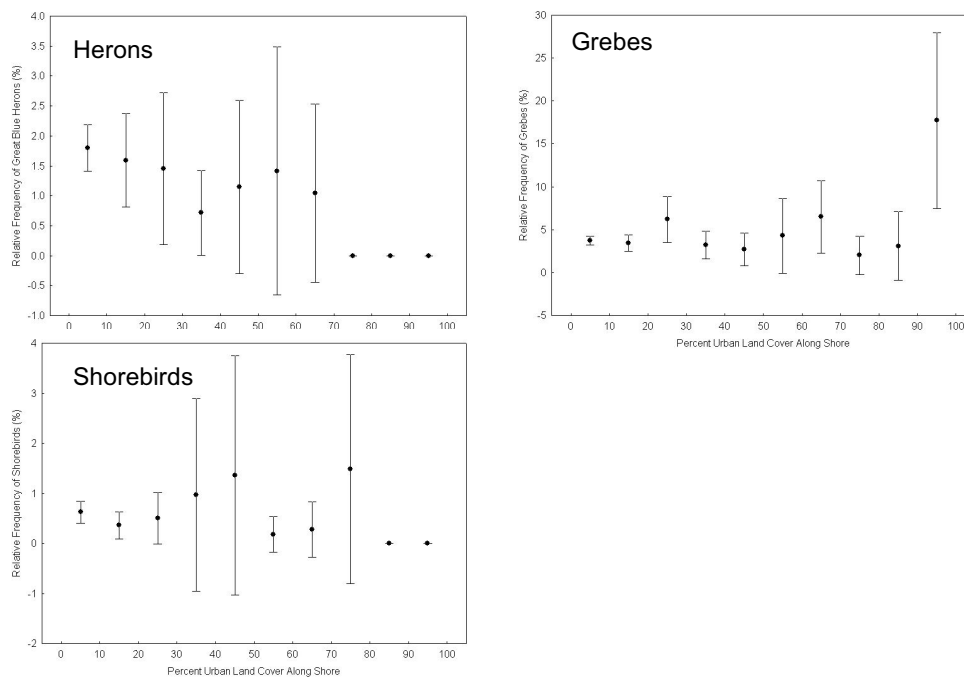
Figure 5.14. Mean relative frequency (± 95 % CI) of great blue herons (top left), Grebes (top right), and shorebirds (bottom left) along the land cover urbanization gradient across all sub-basins (3918 beach segments) in winter 2002.

Discussion

Taxonomic diversity in marine bird and waterfowl assemblages generally declined throughout greater Puget Sound between 1993 and 1996 in summer, and between 1993 and 2003 in winter. Oceanographic sub-basins and estuaries showed distinct assemblage composition among areas, including between urban and non-urban areas, which showed some of the strongest differences among groups. Despite considerable environmental and biotic heterogeneity, differences in assemblage composition were apparent along an urban shoreline gradient throughout the study area. These results indicate that human activity affects the taxonomic composition of marine bird and waterfowl assemblages across greater Puget Sound, and that such changes can be detected at a variety of spatial scales with simple measures of taxonomic composition and urbanization using geomorphic and oceanographic features as sampling units.

The qualitative changes in assemblage composition I observed are consistent with other findings. Global (Valiela and Martinetto 2007) and local (PSAT 2007b) patterns of declining bird abundance would likely coincide with changing taxonomic composition simply as a result of differences among taxa in abundance trends. Changes in bird assemblages along urban and other human influence gradients in terrestrial environments (Blair 1996, O'Connell et al. 2000, Bryce et al. 2002, Marzluff 2005, Bryce 2006) include increases in tolerant taxa, decreases in sensitive taxa, but peak diversity at intermediate levels of disturbance, all of which are apparent in my results. Gulls often thrive in human dominated ecosystems (Belant 1997), and the relative frequency of large gulls in this study was greatest in the most urban oceanographic sub-basin (East Main), and in the urban estuaries (Duwamish, Puyallup, and Snohomish), and increased with urbanization in individual shoreline segments. Conversely, relative frequency of diving ducks declined with urbanization along shore, and relative frequency of wading and shallow bottom-feeding taxa such as dabbling ducks and herons was lower in urban areas and also declined along the urban shoreline gradient. The relative abundance of diving ducks and grebes in urban

areas is not necessarily a contradiction, as this reflects more an absence of the wetland taxa than a preference for urban areas by diving ducks and grebes.

That clear patterns were observed despite coarse taxonomic lumping and a limited urban disturbance gradient demonstrates that the approach used here is successful in identifying changes in marine bird and waterfowl assemblage composition over time, and more important, changes associated with natural (habitat type) and anthropogenic (urbanization) influences. Refining these techniques by using more detailed environmental information (e.g., shore form typology and change analysis now being completed in greater Puget Sound [McBride and Beamer 2007, Collins and Sheikh 2005, Fung and Davis 2005]), and more comprehensive human influence gradients, will further develop our understanding of the responses of marine birds and waterfowl to natural and anthropogenic influences. This could include the development of quantitative monitoring and assessment tools to evaluate biological condition at a variety of spatial scales, from individual beaches to all of greater Puget Sound. Such tools will be useful in tracking biological condition as pressure on the ecosystem increases from local and global human influences, and as we attempt to rehabilitate degraded parts of the landscape.

CHAPTER 6

Puget Sound Science at the Crossroads

Introduction

Puget Sound presents daunting challenges for ecosystem science and management. The large, heterogeneous, and dynamic natural system is affected in many ways by the activities of a large and growing human population with often conflicting socioeconomic interests. As in most ecosystems throughout the world (Dayton et al. 1998, Jackson 2001, Steneck and Carlton 2001), environmental assessment in Puget Sound began well after the system was heavily altered by human activity, limiting our understanding of historical conditions and the specific timeline and character of anthropogenic biological change. Early efforts at environmental assessment were narrowly conceived and focused, and thus have had limited success (see Chapter 1). And despite the growth of modern environmental awareness, environmental assessment has generally been a low priority for society.

All of these problems are important, but developing a useful understanding of relationships among human and natural elements in Puget Sound is made even more difficult when science fails to focus on and measure significant biological responses to the full range of human and natural influences; does not sufficiently consider the whole ecosystem; and rarely synthesizes and communicates scientific information to the broader scientific community, policy makers, and the public. Because such problems are persistent in Puget Sound science, we will not be able to protect and recover the living systems of Puget Sound until we have solved them. Relatively favorable political winds are now pushing society to better address the continuing signs of biological decline (PSP 2006). This task cannot be successful without new scientific information and analytical approaches. Chapters 2 through 4 contribute new data and analyses on the biological character of Puget Sound and how it is affected by human activity, while Chapter 1 provides historical context that is missing from the public discussion of Puget Sound science and management (e.g.,

PSP 2006, Ruckelshaus and McClure 2007). In this concluding chapter, I briefly discuss lessons from other ecosystems, and then revisit key results from the four research chapters in the context of basic elements of monitoring and assessment, and historical problems in Puget Sound science identified in Chapter 1. Finally, I make recommendations for future monitoring and research in Puget Sound.

Lessons from Other Ecosystems

Efforts to understand and manage other ecosystems have much in common with those in Puget Sound, but the unique ecological, historical, and management contexts among systems yielded important differences. Common to most systems are historical transitions in conceptual frameworks and analytical methods that correspond to major developments in the fields of ecology, resource management, and environmental assessment (Larkin 1996, Elmgren 2001). Notions of closed and static ecosystems, and of single limiting factors and sustainable yields operating on single species in isolation gave way to concepts of open, dynamic systems with multiple drivers and alternate states. Advances in analytical methods (in chemistry and computing, for example) continually opened up new avenues for research and assessment.

Some systems progressed more rapidly than others in both scientific understanding and the development of monitoring and assessment programs. The Chesapeake and San Francisco Bays, for example, are two of the most extensively studied and monitored coastal systems in the world. An enormous amount of scientific information exists on them—across taxa and habitats, over considerable spatial and temporal extent (Conomos 1979, Hollibaugh 1996, TBI 2003, CBFEAP 2006, CBP 2007). In contrast, scientific information on the ecology of Puget Sound is far less extensive and not as well integrated. Biological oceanography, for example, is so poorly understood that we cannot describe the seasons (e.g., seasonal patterns of plankton and fish abundance and assemblage composition across the

various sub-basins) with much quantitative or even qualitative precision. Biological monitoring of the pelagic zone is nearly absent (see Chapters 1–4).

But recognition of the complexity and interconnectedness of coastal ecosystems (including the many dimensions and cumulative effects of human activity), the development of new analytical tools, and the collection of enormous quantities of data have not often translated into successful management. Decline of coastal ecosystems may have been slowed and partially reversed in some cases (Kideys 2002, TBI 2003, CBP 2007), but generally continues throughout the world, in part due to poor monitoring and assessment. Two common problems are focusing on single or very limited number of human influences and not measuring biological condition directly and comprehensively (see Chapter 1). In the Chesapeake, for example, eutrophication has been the overwhelming focus of monitoring and management efforts, yet the system continues to decline for reasons beyond eutrophication. Further, instead of field measurements, modeling results were sometimes used to evaluate the efficacy of management actions to control nutrient inputs, overestimating regulatory effectiveness (GAO 2005). Too often those models depend on overestimates in our ability to parameterize them, and unrealistic, even false, assumptions about typically deterministic models (Pilkey and Pilkey-Jarvis 2007); as a result, regulatory effectiveness is overestimated and worse, rarely validated. These mistakes represent lost opportunities to better understand and manage the Chesapeake. Monitoring and management is improving somewhat in the Chesapeake through more comprehensive approaches, including fisheries management (CBFEAP 2006).

Essential Elements of Monitoring and Assessment and the Role of Exploratory Science:
Key Results and Conclusions of Research Chapters

The primary consideration in environmental monitoring and assessment should be the measurement of biologically significant responses to human activity (see Chapter 1). This requires the definition of: 1) homogenous sets of sampling units that are similar in biotic and abiotic character; 2) desired biological condition (the "reference" against which we measure and toward which we hope our degraded systems progress); and 3) biological variables that are responsive to human influences (Karr and Chu 1999). All of this is dependent on comprehensive natural history (knowledge of living systems in the field) and understanding of biological responses to diverse human influences. When that information does not exist, one obvious first step is to observe and document biological patterns across natural and human influence gradients to develop the regional natural history narrative, document responses of the biota to human influences, formulate further hypotheses for monitoring and research, and provide pilot data to plan more focused monitoring and research.

For the most part, the studies presented in this dissertation are not the experimental-predictive type that is sometimes, and incorrectly (Diamond 1986, 1997; Francis and Hare 1994; Cleland 2001), viewed as the only serious approach to science. Except for Chapter 2 (a specific, hypothesis-driven study) these studies are primarily descriptive and historical, intended to explore broad biological patterns (taxonomic, geographic, seasonal, interannual), and help to define—Sound-wide—the essential elements mentioned above: sampling units, reference conditions, and responsive biological attributes. The results allow us to draw clearer inferences about the ecology of greater Puget Sound and how it is affected by human activity, to generate useful hypotheses, and to assist in the development of improved monitoring and research metrics for Puget Sound (Tables 6.1 – 6.4).

Table 6.1. Key results from Chapter 2, and related inferences, hypotheses, and potential monitoring and research metrics.

Key Results	• Modified beach had higher light levels, higher substrate and air temperature, and lower humidity than natural beach
	• Modified beach had lower embryo density and lower percentage of live smelt embryos
Inferences	• Shoreline modification adversely affects biology across Puget Sound through changes in physical structure and microclimate
	• Shoreline modification may adversely affect populations of beach-spawning pelagic fishes
	• Preservation and restoration of natural shoreline conditions will maintain and improve the biological condition of shorelines and the larger Puget Sound ecosystem
Hypotheses	• Biological communities in areas of Puget Sound with higher shoreline modification have altered taxonomic composition, and higher relative abundance of tolerant taxa
	• Areas of Puget Sound with higher shoreline modification have higher embryo mortality in beach spawning pelagic fishes (particularly in summer spawning populations because of increased thermal stress and desiccation)
	• Preservation and restoration of natural shoreline conditions will maintain and increase embryo survival and expand geographic and seasonal distribution of spawning activity by beach spawning pelagic fishes
Potential Monitoring and Research Metrics	• Biological: percentage of live smelt embryos; geographic and seasonal distribution of smelt and sand lance embryos
	• Human Influence: degree of armoring; land cover

Table 6.2. Key results from Chapter 3, and related inferences, hypotheses, and potential monitoring and research metrics.

Key Results	• Unmarked (majority naturally spawned) juvenile Chinook salmon had lower peak densities than marked (known hatchery) Chinook throughout greater Puget Sound, and more protracted seasonal distributions than marked Chinook in areas where extensive wild populations remain
	• Peak seasonal densities of juvenile Chinook were three to four times higher in hatchery dominated areas of greater Puget Sound than in less hatchery dominated areas
	• Unmarked fish are smaller (mean length) but vary more in individual size at the population level than marked fish
	• Proportion of marked fish is higher in neritic waters than in more shallow estuarine habitats
	• Juveniles from many source populations intermingle in neritic waters
Inferences	• Human influences (e.g., hatchery practices and habitat destruction) have altered patterns of estuarine habitat use by juvenile Chinook salmon across greater Puget Sound
	• Wild juvenile Chinook use estuarine habitats more extensively (more habitat types; broader seasonal and geographic distributions) than hatchery fish
	• Sub-basins and estuaries of greater Puget Sound differ in their use by different species, life history types, and developmental stages of juvenile salmon
	• Hatcheries may influence density dependent processes (growth, movement, disease) among juvenile Chinook and other pelagic fishes in estuarine environments
	• Hatchery salmon may not be suitable information analogs for wild fish (i.e., indicator stock concept may be less valid)
	• Conservation and restoration of estuarine environments and improvements in hatchery practices (e.g., reduced and less concentrated releases, disease prevention) will improve estuarine conditions for wild fish
Hypotheses	• Conservation and restoration of salmon habitats will expand seasonal and geographic distributions of wild fish, improve growth and survival of wild fish, and increase expression of life history diversity in wild fish
	• Improvements in hatchery practices (e.g., reduced and less concentrated releases, disease prevention) will improve growth and survival in wild fish, and reduce disease in all Chinook
Potential Monitoring and Research Metrics	• Biological: cumulative mean density of wild and hatchery fish; relative abundance of hatchery and wild fish; seasonal and geographic distribution of wild and hatchery fish; number and relative abundance of life history types; individual growth; disease prevalence
	• Human Influence: degree of habitat loss and degradation (watershed and estuary); number of hatchery fish released; extent of disease control in hatcheries

Table 6.3. Key results from Chapter 4, and related inferences, hypotheses, and potential monitoring and research metrics.

Key Results	• Taxonomic composition of neritic macrofauna differs markedly among sub-basins of greater Puget Sound during daytime in spring and summer
	• Percentage biomass of gelatinous zooplankton (jellies) and small pelagic fishes are inversely related
	• Fish species richness, percentage of fish in total biomass, and relative abundance of wild Chinook salmon increased with latitude; relative abundance of jellies, hatchery Chinook salmon, and chum salmon decreased with latitude
Inferences	• Oceanographic heterogeneity in greater Puget Sound results in different pelagic fish and zooplankton assemblage composition among sub-basins
	• Human activities (local, regional, and global) may interact with natural oceanographic features to restructure pelagic fish and zooplankton assemblages
	• Changes from fish dominated to jelly dominated food webs have negative implications for upper trophic levels
	• Jelly dominated pelagic macrofauna may be diagnostic of degraded biological condition
Hypotheses	• Oceanographic features (circulation, stratification, nutrients) of individual sub-basins favor either fish or jelly dominated pelagic food webs
	• Local human influences (substrate hardening, nutrient and contaminant loading, fishing) increase abundance of jellies
	• Ocean and climate related changes in water column circulation, stratification, and nutrient delivery will favor either jelly or fish dominated pelagic food webs
Potential Monitoring and Research Metrics	• Biological: relative abundance (biomass) of fish and gelatinous zooplankton; fish species richness; relative abundance of hatchery and wild salmonids
	• Human Influence: timing and magnitude of nutrient and chemical contaminant addition; degree of substrate hardening; number of hatchery fish released

Table 6.4. Key results from Chapter 5, and related inferences, hypotheses, and potential monitoring and research metrics.

Key Results	• Taxonomic composition of marine birds and waterfowl differs among geographic areas of greater Puget Sound, and between winter and summer
	• Consistent with precipitous declines in abundance of certain species during recent decades, taxonomic diversity of marine birds and waterfowl has generally declined across greater Puget Sound over the time period considered (1993-2003)
	• More urbanized sub-basins, estuaries, and shoreline segments are dominated by fewer taxa and by more opportunistic and tolerant taxa.
Inferences	• Human activity affects marine bird and waterfowl assemblage composition throughout greater Puget Sound
	• Dramatic declines of individual bird population abundances recorded in recent decades are likely related to local influences
	• Relationships between human activity and bird assemblage condition can be characterized with simple measures of disturbance (e.g., land cover) and biology (e.g., relative frequency of taxa) at site, estuary, and sub-basin scales
	• Continued urbanization and other human alterations of Puget Sound will result in declining condition of marine bird and waterfowl assemblages
	• Conservation or restoration of estuarine environments will improve the condition of marine bird and waterfowl assemblages
Hypotheses	• Historical estuarine wetlands in Puget Sound converted to human uses today have lower bird taxa richness, higher relative abundance of large gulls, and lower relative abundance of waders, than do remaining more natural wetlands
	• Restoring estuarine wetlands and natural shoreline features in heavily altered estuaries will increase taxa richness and decrease relative abundance of large gulls
Potential Monitoring and Research Metrics	• Biological: bird taxa richness; relative abundance or frequency of opportunistic, tolerant, wading, or sensitive taxa
	• Human Influence: urban land cover along shore; wetland loss; shoreline modification; boat and foot traffic

Research Chapter Conclusions

Consideration of biological, human influence, and ecosystem contexts is a requirement for successful management of Puget Sound (see Chapter 1), and much of the value of the four data chapters comes from the fact that they all, to varying degrees, emphasize these three areas. The beach study (Chapter 2, and see Rice 2006) addressed the important topic of the ecological effects of anthropogenic shoreline modification (a disturbance that affects approximately one third of the entire Puget Sound shoreline (PSAT 2002)) and documented severe changes in microclimate, a strong determinant of biological character (e.g., Karr and Freemark 1983, Chen et al. 1999). The measured biological response was embryo density and mortality in surf smelt (*Hypomesus pretiosus*), one of three beach spawning pelagic fishes that are a major component of pelagic fish fauna in Puget Sound. The study is very limited in spatial and temporal scale, and in part simply a confirmation of observations made by others over many decades (Shaefer 1936, Penttila 1973, Levings and Jamieson 2001, Penttila 2001, Brennan and Culverwell 2004). But unlike so much of Puget Sound science, the beach study synthesizes relevant information on the topic, adds new data that explicitly address human activity and biological response, and puts that information into the public domain via the peer reviewed literature. One of the few publications on ecological consequences of anthropogenic shoreline modification in the Pacific Northwest (Romanuk and Levings 2003, Rice 2006, Toft et al. 2007), it is the first in the region to document anthropogenic changes in shoreline microclimate and increased mortality in fish embryos.

The results of the beach study (Chapter 2) could be extended in a number of useful directions for monitoring and assessment (Table 6.1). Presence, density, and condition of beach spawning fish embryos; alone or in combination with diverse attributes of other assemblages known to respond to shoreline modification (e.g., birds [see Chapter 5]; and supratidal invertebrates [Romanuk and Levings 2003, Sobocinski 2003]) could be used to evaluate the biological condition of shorelines

across Puget Sound, including historically degraded beaches now being manipulated to improve ecological health. Relationships between beach modification and embryo abundance and mortality could be combined with environmental data (e.g., WDNR 2001, McBride and Beamer 2007) to estimate the spatial distribution and condition of potential spawning habitats across Puget Sound. This information could then be used to develop probabilistic field sampling designs, or as part of population models, to better evaluate the true biological significance of shoreline degradation (and subsequent restoration) on populations of surf smelt and other beach spawning pelagic fishes.

The poor status of wild Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*) populations is one of the most pressing environmental management problems in Puget Sound (SSPS 2005, PSP 2006), yet basic information is lacking on juvenile Chinook use of the severely degraded but presumed critical estuarine habitats, and the ecological effects of hatchery fish (historically the primary management response to wild fish declines) in estuaries. The study in Chapter 3 informs the issue by providing basic, Sound-wide natural history information on neritic estuarine rearing by juveniles, likely a crucial influence on survival to adult (Simenstad et al. 1982, Healey 1991, Bottom et al. 2005b, Greene et al. 2005). This information improves our understanding of how juvenile Chinook use Puget Sound and is a useful basis for future monitoring and research (Table 6.2). The direct effects of human activity on wild Chinook were not evaluated, but information on human influences (specifically, hatchery practices and habitat destruction) was included through the reporting of data on hatchery fish and the broad geographic and seasonal coverage of the sampling, including heavily urbanized and hatchery-influenced estuaries, from spring into fall. Marked (known hatchery) juvenile Chinook had higher peak densities, larger individual size and narrower size distributions, and skewed and compressed seasonal and geographic density distributions compared to unmarked (majority naturally spawned) Chinook. These contrasts highlight the different biological characteristics of human-dominated and

natural salmon populations, and thereby illustrate some of the attributes of healthy wild salmon systems—the reference conditions—that are the goal of wild salmon and Puget Sound ecosystem protection and recovery planning (SSPS 2005, PSP 2006, Ruckelshaus and McClure 2007).

The observed differences between marked and unmarked fish, and among different geographic areas of greater Puget Sound must in part be the result of influences outside of the estuarine rearing environments (e.g., destruction of freshwater habitat, hatchery manipulation of size and release timing, ocean harvest of adults), but also raise questions about potentially adverse interactions between hatchery and wild fish in the estuary, and about the suitability of various regions of Puget Sound as juvenile Chinook rearing habitat. The densities and seasonal distributions of Chinook in central and southern Puget Sound do not, in and of themselves, demonstrate that these areas are unsuitable for extended rearing by juveniles, but they do demonstrate that, for whatever reason, some estuarine attributes documented in healthy wild Chinook populations (from historical populations [Burke 2004] or less disturbed areas [Beamer et al. 2005, Chapter 3]) are not being expressed in those areas. Research and monitoring should document, diagnose, and track those patterns to evaluate the performance of management actions (e.g. habitat restoration and hatchery reform). Follow-up studies of otoliths (for growth and life history information), diet, and disease (Rhodes et al. 2006, Rhodes et al. in prep) are being conducted on the fish captured in 2002 and 2003 to advance such efforts. For example, total Chinook density (affected by hatchery practices [see Chapter 3 and references therein] and by habitat loss [Beamer et al. 2005]) is a significant factor in the prevalence of bacterial kidney disease (Rhodes et al. in prep).

But even based solely on the density results in Chapter 3, estuarine monitoring of juvenile Chinook in Puget Sound could today realistically pose this simple and useful hypothesis: As recovery plans are implemented in watersheds and estuaries, seasonal distributions of juvenile Chinook in neritic waters of individual estuaries

and Puget Sound as a whole, will become flatter, broader, and less skewed. Neritic sampling is not just relevant in evaluating neritic environments, but also in detecting responses to upstream effects from, for example, restoration actions in streams, rivers, and estuarine wetlands.

That clear patterns were observed at all in the Chinook study demonstrates that productive Sound-wide monitoring and research on juvenile Chinook in neritic environments can be done. That the strongest patterns cannot be seen without broad seasonal coverage, argues for monitoring metrics such as cumulative density across seasons, habitats, and years (Beamer et al. 2005), that best capture the full expression of wild Chinook life history diversity and productivity over time.

By exploring basic patterns of density and size in wild and hatchery Chinook across seasons and at large spatial scales, Chapter 3 establishes several empirical facts that must be reconciled as we make inferences about the condition of wild Chinook salmon and the Puget Sound ecosystem on which they depend. For example, the extensive commingling of hatchery and wild fish from many source populations, but clear and consistent size and density differences between hatchery and wild fish, draw into question two common but unchecked assumptions in salmon management (WDFW and PSTT 2004): 1) that hatchery practices effectively minimize ecological interactions between juvenile hatchery and wild fish in the estuary through temporal isolation, and 2) that hatchery fish are suitable information analogues for wild fish (the "indicator stock" concept). Evaluating such assumptions is crucial in ensuring that management actions (supported by public funds to act in the public interest) use scientifically valid methods and are effective without harming public natural resources.

Despite the clear and informative patterns produced in the juvenile Chinook salmon study, its inferential power was limited by its exploratory and nonrandom sampling scheme. Building on that design, the data from Chapter 3 provided an excellent basis for a rigorous, probabilistic sampling design (Skalski 2005) for the neritic component of a long-term estuarine restoration monitoring program in Skagit

Bay (Figure 6.1). This program is among the most comprehensive estuarine fish monitoring efforts in the Pacific Northwest, and is uniquely positioned to detect biological response, at the scale of a whole estuary, to restoration actions. In addition, samples collected during the original study and the ongoing monitoring are producing extensive information on abundance, size, diet, genetics, and disease in juvenile salmon and other small pelagic fishes (e.g., Rhodes et al. 2006, Hershberger 2007).
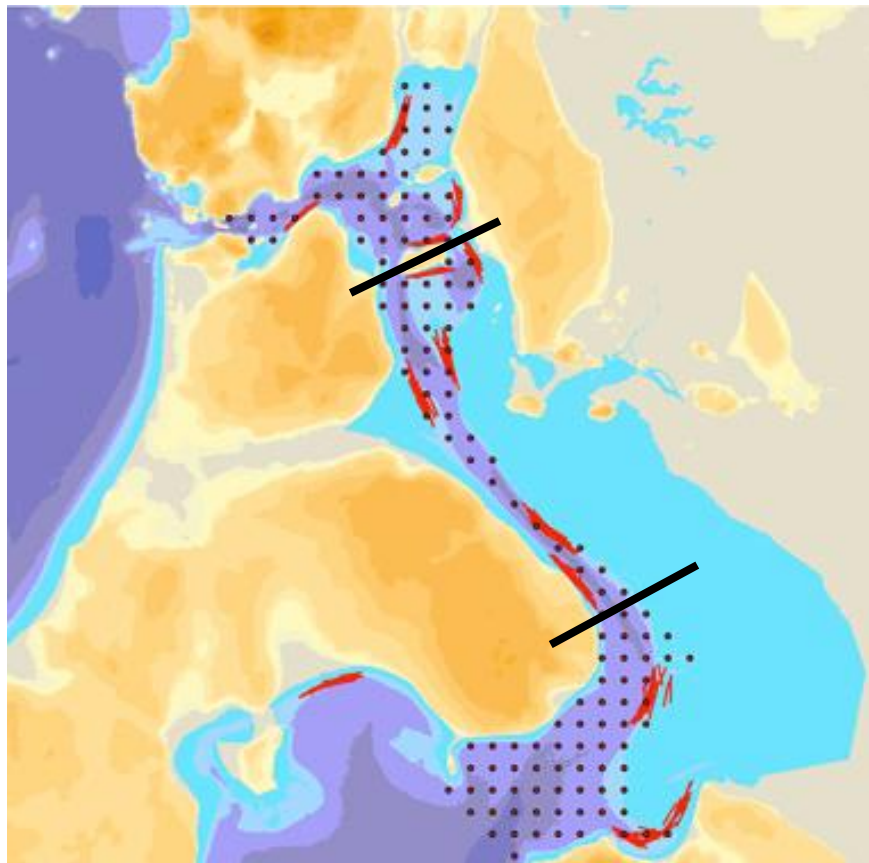
Figure 6.1. Map of Skagit Bay, Washington showing surface trawl paths (red lines) from juvenile Chinook salmon study in Chapter 3, and stratified random sample point grid (strata separated by black lines) of monitoring design developed from the data in Chapter 3. Base map by Jason Hall, NOAA Fisheries.

Reactive management to single species in crisis is a persistent problem in environmental affairs, and studying the attributes of Chinook salmon alone will not be sufficient to protect and recover them or any other Puget Sound species—all are embedded in the broader context of the whole ecosystem. The "by-catch" study in Chapter 4 takes advantage of a single species effort (Chapter 3) to go beyond the single species focus and provide some of that context. The opportunistic collection and analysis of fish and gelatinous zooplankton (jellies) demonstrates the value of sampling multiple taxa at large spatial scales in describing the overall biological character of Puget Sound, and the data reveal compelling patterns that could provide useful background for future research and monitoring of the pelagic zone (Table 6.3). Although the results do not explicitly relate human influences to biological responses, they do raise fundamental questions about the nature of pelagic environments across Puget Sound. Specifically, what are the roles of natural oceanographic features and human activity in structuring pelagic assemblages and food webs, and what biological attributes of the pelagic zone need to be monitored? This study is the first to report landscape scale jelly data and inverse relationships between pelagic fishes and jellies in Puget Sound. Chapter 4 also identifies attributes of pelagic macrofauna (e.g., biomass of fishes and jellies) that are relatively simple to measure and might be useful in evaluating the effects of local (e.g., eutrophication, substrate hardening) and global (e.g., climate change) human influences on Puget Sound.

In addition to collecting new data, we can develop our understanding of the ecological consequences of human activity by applying new analytical approaches to existing data. Combination and reanalysis (Chapter 5) of data from several efforts (WDNR 2001, Nyswander et al. 2002, Hepinstall et al. in prep), with explicit focus on assemblage composition (as opposed to absolute abundance of single taxa) and urbanization, revealed biological responses (e.g., increases in gulls and decreases in waders, diving ducks, and dabbling ducks; greater dominance by tolerant taxa) to human influence across the landscape. The use of urban land cover gradients at the

level of oceanographic sub-basins, estuaries, and individual shoreline segments was successful in demonstrating consistent urban signals across spatial scales and environment types, and demonstrated the utility of various geomorphic units as a way to stratify and sample marine birds and waterfowl. While monitoring programs have reported declining abundance in single species and species groups (Nyswander et al. 2003, PSAT 2007, John Bower unpublished data), Chapter 5 is the first study to document corresponding declines in the diversity of marine bird and waterfowl assemblages, or to relate any changes in these assemblages to local environmental features, including human activity. Simply by emphasizing diverse attributes of the biota and human influences within an ecosystem context (a well-established formula for success in environmental monitoring [Karr and Chu 1999, Karr 2006]), the analysis in Chapter 5 introduces a promising approach to evaluating the condition of marine birds and waterfowl across greater Puget Sound.

Recommendations for Puget Sound

Two major themes emerge from the preceding chapters. First, Puget Sound science has not sufficiently focused on biology and human activity in an ecosystem context, and consequently, has produced neither a comprehensive regional natural history narrative, nor a comprehensive diagnosis of ecosystem condition. Second, simple, exploratory studies that do explicitly consider biology, human influence, and ecosystem context (across taxa, sub-basins, seasons, etc.) quickly reveal (most of the key results in chapters 2–5 come from one year of data or less) clear and compelling patterns that improve our understanding of Puget Sound and how it is, or may be, affected by human activity. These themes suggest future directions for environmental science in Puget Sound, and are the basis for the following recommendations.

1) *Improve the conceptual framework*

Existing conceptual models (e.g., Newton et al. 2000) should be reviewed and revised to develop a more comprehensive framework for Puget Sound monitoring and assessment that 1) clearly articulates appropriate goals; 2) better delineates ecological zones and spatial and temporal scales for sampling that maximize our ability to differentiate anthropogenic and natural variation across the whole ecosystem; 3) better identifies the most important human stressors and likely biological consequences in various ecological contexts (e.g., Whidbey Basin vs. Main Basin; benthic vs. pelagic); and 4) identifies a comprehensive suite of biological attributes that are likely to most effectively measure anthropogenic biological change.

The primary stated goal of the Puget Sound Assessment and Monitoring Program (PSAMP) is to evaluate "health" (PSAMP Management Committee 2005, PSAT 2007b) but this is not clearly defined, and most monitoring focuses on changes over time in physical, chemical, and biological attributes, often without explicit consideration of human influences on biological condition (see Chapter 1). Health is reasonable as a goal for ecosystem management but is meaningless unless defined and explicitly included in monitoring designs as the desired, or reference, against which the biological condition of a place is evaluated. Biological integrity is an effective and well-established concept of ecosystem health (Karr 1981, 1991, Karr and Chu 1999) originally defined as "a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Karr and Dudley 1981a). Implicit in biological integrity, are other attributes (e.g., biodiversity and resilience) often proposed as goals for ecosystem management; not just biodiversity and resilience per se, but appropriate biodiversity and resilience for a given place (Angermeier and Karr 1994). The integrity, or reference condition, of a place is simply the biological character before disturbance by modern humans, and can be characterized by measuring diverse attributes (e.g., taxonomic and trophic

composition, size structure, individual condition) of the biota in undisturbed or minimally disturbed places.

Evaluating the condition of places in this way is more effective (especially in diagnostic and effectiveness monitoring) than, for example, measures of absolute abundance of single species over time with little or no power to identify the causes of change (see Chapter 5). Understanding of biological integrity is not well developed in Puget Sound. What, for example, should healthy phytoplankton, zooplankton, macrophyte, bird, fish, and marine mammal assemblages look like in the various sub-basins and ecosystem types of Puget Sound? Considerable information does exist on community-environment associations in Puget Sound (e.g., Miller et al. 1980, Simenstad 1983, Phillips 1984, Dethier 1990, Simenstad et al. 1991) but little of it has been employed to define reference condition for monitoring and assessment purposes.

Detecting meaningful change in the many aspects of Puget Sound's biological condition requires covering the whole system, and dividing it into zones and sampling strata to reduce natural variation and better relate human stressors directly to biological response. The existing conceptual framework for PSAMP (Newton et al. 2000) suggests dividing the Sound into nearshore, bays and inlets, and open basin groups but this does not correspond to what are likely to be the strongest environmental gradients in Puget Sound (see Ruckelshaus and McClure 2007 for an overview of biotic character of different environments). An alternative would be to partition the system into nearshore, pelagic, and demersal zones, and stratify within each of those by spatial and geomorphic characteristics, as well as by time period depending on the nature of the biological response (Figure 6.2).
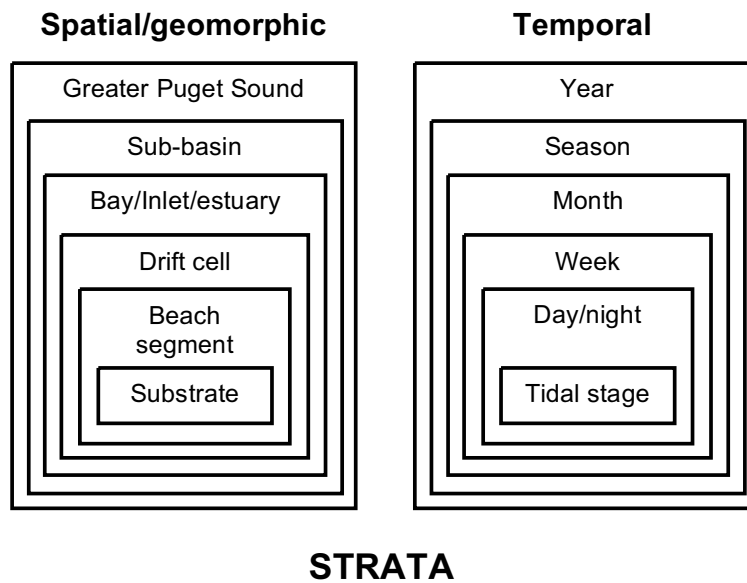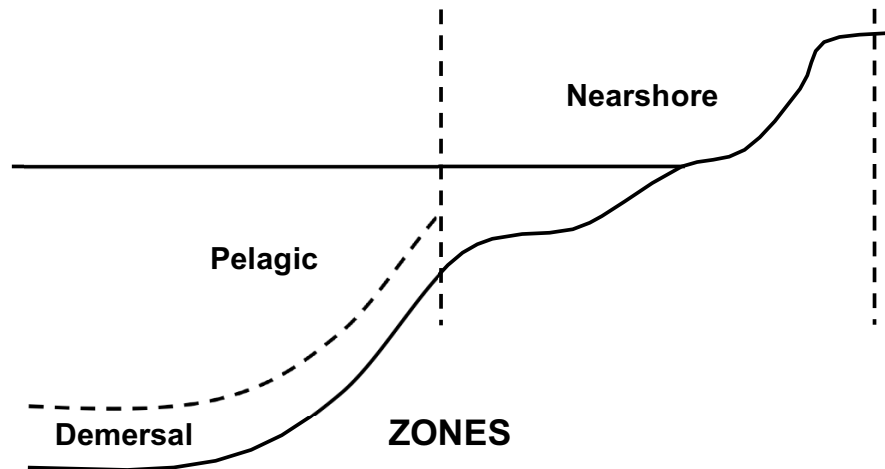
Figure 6.2. Suggested ecosystem zones and basic sampling strata for consideration in monitoring and assessment in Puget Sound.

The zonal designations reflect both unique ecological character (i.e., species composition, sources of energy, etc.), and different sensitivity to various anthropogenic stressors. For example, fish faunas in the three zones differ in the species composition and life stages present. Nearshore fish assemblages have dozens of species and many juveniles (Wingert and Miller 1979), pelagic assemblages are dominated by adults of a few small, schooling species but also have planktonic larvae of many species (see Chapter 4), and demersal assemblages have dozens of species but are more characterized by adult flatfishes (*Pleuronectiformes*) and adults of long lived species such as the rockfishes (*Sebastes spp.*) (see references in Miller and Borton 1980). In terms of anthropogenic stressors, the nearshore zone, for example, is more directly affected by physical disturbance of the shoreline than is the pelagic or demersal zones, the pelagic zone is presumably more directly affected by eutrophication and changes in circulation than are the nearshore and demersal zones, and the demersal and pelagic zones are more directly affected by fishing than is the nearshore. Conceptual work done in a nearshore restoration context (Simenstad et al. 2006) could be applied to the other ecological zones to assist in building a more comprehensive basic categorical structure for monitoring and assessment program for Puget Sound.

While many different human influences are identified in conceptual models of Puget Sound (Newton et al. 2000, Gelfenbaum et al. 2006, Simenstad et al. 2006), chemical contaminants and water quality have received overwhelming emphasis in actual monitoring activities, despite the fact that they are not likely the most important cause of biological decline (West 1997, Lotze et al. 2006). Further, measurements of chemical and physical characteristics are often not related directly to biological response in Puget Sound monitoring, and data from biological monitoring are rarely related to environmental data, including anthropogenic influences (see Chapter 1). This disconnect between environmental and biological information must be corrected (see Chapter 5 for an example), along with the development of a thorough list of human stressors and expected biological responses,

the most important of which should be taken beyond conceptual models and translated into actual monitoring and assessment programs.

Much guidance exists on ecological indicators (Gibson et al. 2000, NRC 2000, Heinz Center 2002, Bortone 2005, Jorgensen et al. 2005, USGAO 2005), but the reality in coastal ecosystems is that effective, comprehensive indicators of ecosystem health are not well developed. Efforts to do so, including adaptation of multimetric indexes (e.g., Deegan et al. 1997, Jameson et al. 2001) have produced promising but decidedly mixed results (Rice 2003, TBI 2003, Bortone 2005, CBP 2007, PSAT 2007b).

Building better ecological indicators for Puget Sound will be an iterative process, and the first step is to do foundational work to better define sampling units and reference conditions for selected components of the biota, and identify diverse sets of biological attributes that are responsive to a wide range of human influence. Continuing to pursue lists of isolated physical, chemical, and biological attributes just because we can, or have been, measuring them, regardless of their ability to inform management, will continue to result in failure. Chemical measurements and "habitat" maps, for example, often tell us little about biology (Slobodkin 1994, Karr 1995, Dayton et al. 1998).

Determining suitable biological response variables involves choice of taxa and specific metrics within them that are most responsive to human activity. Ecologically diverse taxa with large individuals have informational and practical advantages, i.e., they are likely to be responsive to multiple human activities and easy to sample and identify (Karr 1981, Furness and Greenwood 1993, Simon 1999). Aquatic macrophytes are often monitored because of their role as animal habitat but it must be kept in mind that the same vegetation map can have very different animal assemblages associated with it, possibly masking significant anthropogenic changes of the biota (Dayton et al. 1998). Middle and upper trophic levels that depend on and "integrate" diverse ecosystem services are also commonly favored (Karr 1981, Boyd et al. 2006). These attributes argue for fishes, birds, and marine mammals as subjects

for biological monitoring in Puget Sound and all are currently monitored to varying degrees but with limited effect (Puget Sound Action Team 2007b, Chapter 1). Some existing efforts, then, could provide a useful basis for improved monitoring and assessment programs (see Recommendation 2 and Chapter 5).

One critical omission in historical research and monitoring in Puget Sound is the plankton (see Chapters 1 and 4). Although they do not share many of the above attributes, as first level biological integrators of, for example, hydroclimatic conditions, plankton can provide particularly useful "early warning" information on productivity and trophic status of pelagic systems (Beaugrand 2005, Hooff and Peterson 2006). Such information will likely be increasingly important over the coming decades given expected climate induced changes in Puget Sound circulation (Mote et al. 2003, Snover et al. 2005). Plankton can also be responsive to local human influences (Parsons and Lalli 2002, Purcell in review, and see Chapter 4), but none of these potentially useful relationships have been explored in Puget Sound.

2) *Evaluate the rationale, approaches, and effectiveness of current efforts*

A comprehensive review of existing environmental monitoring efforts should be conducted to evaluate and optimize their ability to detect meaningful biological responses to human influences across all of Puget Sound. The following questions could be used to guide the process:

- How does this program advance our understanding of the biological effects of human activity?
- How does this program inform management choices and evaluate management actions?
- How could this program be improved with respect to the above tasks?
- What is its value compared with other existing or possible efforts?

Programs should be revised or eliminated if they do not contribute to our understanding of anthropogenic effects. Sampling designs of eelgrass, bird, and fish components of PSAMP monitoring, for example, should be revisited with the goal of explicitly incorporating human disturbance gradients. Designs of oceanographic and water quality sampling (in PRISM cruises and PSAMP programs, for example) should be similarly revised but in addition should seek ways to incorporate useful biological measures (e.g., abundance, and taxonomic and size composition of plankton and pelagic fishes). New programs and indicators should be created if they do not exist for at least minimal representation of key biotic components in nearshore, pelagic, and demersal zones across greater Puget Sound.

The spatial and temporal extent and resolution of sampling in all programs should be rigorously optimized to efficiently detect biologically relevant change and discriminate between human and natural influences. Biological sampling at large spatial scales is often seen as too "variable" or costly, yet more spatially extensive designs (more sites as opposed to more samples within sites) are often essential in understanding biological character across the full range of natural and human influences (see Chapters 3–5), and are typically more efficient statistically (Cochran 1977b).

3) *Check assumptions about the ecological effects of human activity*

Confirmatory and exploratory field (e.g., Chapter 2) and historical (e.g., Chapter 5) studies documenting relationships between major anthropogenic stressors and biological response should be aggressively pursued. Understanding such relationships is critical in monitoring and assessment, yet the ecological consequences of some of the major human alterations of the environment (e.g., filling of estuarine wetlands, shoreline armoring) and supposed solutions (e.g., hatcheries and restoration projects) are poorly documented. Existing environmental data sets (e.g., WDNR 2001) and conceptual models (Newton et al. 2000, Simenstad et al. 2006) could be revised and used to help generate hypotheses and design

probabilistic field sampling designs to rigorously document these relationships throughout greater Puget Sound. As Chapter 5 demonstrates, designing studies to characterize diverse biological attributes across a range of human influence can provide immediate insights without having to wait for the accumulation of a time series. The importance of this kind of foundation work has already been established in the monitoring and assessment of freshwater systems throughout North America (Fore 2003).

Also demonstrated in the bird analysis (Chapter 5), is the fact that historical datasets contain useful information, and many Puget Sound sources have not been fully explored or are not readily available (see Chapter 1). Worse, some have undoubtedly been lost forever. Only a fraction of the demersal fish data (Puget Sound Action Team 2007b) collected by WDFW for PSAMP, for example, has ever been reported and even less made available to other researchers. From the little we do know, what does it say about the condition of rockfish assemblages that the more long-lived and predatory species have declined and the smaller, shorter-lived species have increased (W. Palsson WDFW personal communication)? These and many other details should be explored in historical data sets to build the regional natural history narrative, including the effects of human activity, and improve monitoring and assessment programs. Other data sets from academic institutions and government agencies should be collected, preserved, digitized, made available, and reanalyzed. Some of this is being done (Jon Reum, University of Washington, personal communication) but not in a comprehensive and systematic way.

Paleoecology (e.g., Baumgartner et al. 1992, Hairston et al. 2005) can provide valuable insights into the long-term and pre-human characteristics of ecosystems, and has not been pursued much at all in Puget Sound, especially from an anthropogenic effects standpoint. What, for example, can diatom remains in Puget Sound sediments, isotopes in tree rings throughout the watersheds, or fish and invertebrate remains in shell middens across the Puget Sound basin tell us about its

ecological character and history, and how did it change with the activity of modern human society?

4) *Fill basic gaps in ecological understanding of Puget Sound*

Identify critical ecological knowledge gaps in Puget Sound (e.g., spatial and temporal patterns in the distribution of the biota and their relationship to oceanographic characteristics of the various sub-basins) and support research to fill them. Academic institutions (especially publicly funded ones such as the University of Washington) that have not historically emphasized the ecology of Puget Sound in teaching and research should be encouraged to do so. Study of the pelagic zone, including the plankton and small pelagic fishes, is one particularly notable weakness.

5) *Integrate and collaborate*

Integrative, interdisciplinary science is always desirable in environmental assessment and especially important in understanding and managing the large and extremely complex ecosystems of the coastal zone (Peterson and Estes 2001, Hughs et al. 2005, Peterson et al. 2006). This requires data from across taxa and trophic levels at large spatial and temporal extent, and sampling designs and data management that facilitate easy access and combination for use in multiple analytical approaches. If crude analysis of disparate data not designed for integration can reveal clear and compelling patterns (see Chapter 5) imagine what could be done if we had such analyses in mind to begin with.

7) *Produce rigorous but accessible syntheses of existing scientific information*

Shared access to and understanding of scientific information is essential in environmental assessment but existing scientific information on Puget Sound is rarely published and poorly synthesized (see Chapter 1). Recent efforts are a start (PSP 2006, Ruckelshaus and McClure 2007), but much more rigorous and comprehensive syntheses should be done, possibly by following the original concept

of the Puget Sound Books (but covering more topics in greater depth) and making the information available in a variety of media formats, including books.

Conclusion

Puget Sound science is indeed at a crossroads. Faced with current political circumstances that are focusing more attention on the condition of the Sound and ostensibly reorganizing its management (PSP 2006, Ruckelshaus and McClure 2007), Puget Sound science can continue the long history of fragmented, uneven, and narrowly focused efforts, or it can take this opportunity to change course toward a more informative and efficient model. The central focus of that alternative model is improved understanding and measurement of the biological consequences of human activity in an ecosystem context, and the information presented here provides some useful background, data, and analysis toward that end.

The outlook so far is very similar to the past; most resources are slated to go to the perceived threats of chemical contamination and poor water quality (PSAT 2007a), and public and political misperception (e.g., Cornwall 2007) of the true nature and full scope of the problem continues. Scientific fragmentation and patchiness persist, as existing programs are simply repackaged, and the various research and monitoring entities compete for scarce funds mostly to do what they have always done. Several entities are simultaneously positioning themselves as the "go-to" source for Puget Sound science, we have two (USEPA 2006, PSAT 2007c) inadequate "report cards" for the Sound, and little critical, public review of historical and ongoing monitoring and research efforts is occurring.

But there is some cause for optimism. The inadequacy of historical scientific efforts becomes more apparent as ecological problems persist and satisfactory answers are not forthcoming. Accumulating knowledge from other systems is informing the current discussion, and a wider circle of scientists is engaged. Perhaps alternative approaches and new programs starting with better designs will perform well and become the model.

References

Adair, J. B. 1909. Mineral resources of Washington. Northwest Mining Journal Special AYPE.

Aitkin, J. K. 1998. The importance of estuarine habitats to anadromous salmonids of the Pacific Northwest: a literature review. U.S. Fish and Wildlife Service, Lacey, WA.

Akaike, H. 1973. Information theory and an extension of the maximum liklihood principle.*in* B. N. Petrov and F. Csaki, editors. Proceedings of the Second International Symposium on Information Theory. Akademiai Kiado, Budapest, Hungary.

Alaska Sea Grant, editor. 1997. Forage Fishes in Marine Ecosystems: Proceedings of the International Symposium on the Role of Forage Fishes in Marine Ecosystems. Alaska Sea Grant College Program, University of Alaska, Fairbanks, Anchorage, AK.

Albertson, S.L., K. Erickson, J.A. Newton, G. Pelletier, R.A. Reynolds, and M. Roberts. 2002. South Puget Sound Water Quality Study Phase 1. Report No. 02-03-021. Washington State Department of Ecology, Olympia, Washington.

Alt, D. D., and D. W. Hyndman. 1995. Northwest Exposures: A Geologic Study of the Northwest. Mountain Press Publishing Company, Missoula, Montana.

Anderson, D. R., K. P. Burnham, and W. L. Thompson. 2000. Null hypothesis testing: problems, prevalence, and an alternative. Journal of wildlife management 64:912-923.

Angell, T., and K. C. Balcomb. 1982. Marine birds and mammals of Puget Sound. University of Washington Press, Seattle, WA.

Angermeier, P. L., and J. R. Karr. 1994. Biological integrity versus biological diversity as policy directives. Bioscience 44:690-697.

Arai, M. N. 2001. Pelagic coelenterates and eutrophication: a review. Hydrobiologia 451:69-87.

Arai, M. N. 2005. Predation on pelagic coelenterates: a review. Journal of the Marine Biological Association of the United Kingdom 85:523-536.

Arkoosh, M. A., E. Casillas, P. Huffman, E. Clemons, J. E. Stein, and U. Varanasi. 1998. Increased susceptibility of juvenile chinook salmon from a contaminated estuary to Vibrio anguillarum. Transactions of the American Fisheries Society 127:360–374.

Attrill, M. J., D. T. Bilton, A. A. Rowden, S. D. Rundle, and R. M. Thomas. 1999. The impact of encroachment and bankside development on the habitat complexity and supralittoral invertebrate communities of the Thames Estuary foreshore. Aquatic Conservation-Marine and Freshwater Ecosystems 9:237-247.

Attrill, M. J., J. Wright, and M. Edwards. 2007. Climate-related increases in jellyfish frequency suggest a more gelatinous future for the North Sea. Limnology and Oceanography 52:480-485.

Augerot, X., D. N. Foley, C. Steinbeck, N. Fobes, K. Spencer, and A. Fuller. 2005. Atlas of Pacific salmon. The first map-based assessment of salmon in the North Pacific. University of California Press, Berkely, CA.

Ayers, H. B. 1899. Washington forest reserve.*in* H. Gannett, editor. Nineteenth annual report of the United States Geological Survey, 1897-1898. Part V -- Forest reserves. Government Printing Office, Washington, D.C.

Baumgartner, T. R., A. Sutar, and V. Ferreira-Bartrina. 1992. Reconstruction of the history of Pacific sardine and northern anchovy populations over the past two millenia from sediments of the Santa Barbara basin, California. CalCOFI Rep., 33, 24-41.

Bax, N. J., E. O. Salo, B. P. Snyder, C. A. Simenstad, and W. J. Kinney. 1978. Salmonid outmigration studies in Hood Canal, Final Report Phase III, January to July 1977. FRI-UW-7819, University of Washington, Fish. Res. Inst., Seattle.

Beach, D. 2002. Coastal sprawl: The effects of urban design on aquatic ecosystems in the United States. Pew Oceans Commission, Arlington, VA.

Beamer, E., A. McBride, C. Greene, R. Henderson, G. Hood, K. Wolf, K. Larsen, C. Rice, and K. Fresh. 2005. Delta and Nearshore Restoration for the Recovery of Wild Skagit River Chinook Salmon: Linking Estuary Restoration to Wild Chinook Salmon Populations. Appendix D of the Skagit Chinook Recovery Plan, Skagit River System Cooperative, LaConner, Washington.

Beamer, E., and C. M. Greene. in prep. Biotic and abiotic factors controlling wild juvenile Chinook abundance in the Skagit River tidal delta.

Beamer, E., C.A. Rice, R. Henderson, and D. Lomax. In preparation. Juvenile Chinook salmon in the Skagit River estuary: seasonal patterns of density and size across multiple habitats.

Beamish, R. J., and D. R. Bouillon. 1993. Pacific Salmon Production Trends in Relation to Climate. Canadian Journal of Fisheries and Aquatic Sciences 50:1002-1016.

Beamish, R. J., and C. Mahnken. 2001. A critical size and period hypothesis to explain natural regulation of salmon abundance and linkage to climate and climate change. Progress in Oceanography 49:423-437.

Beamish, R. J., C. Mahnken, and C. M. Neville. 2004. Evidence that reduced early marine growth is associated with lower marine survival of coho salmon. Transactions of the American Fisheries Society 133:26-33.

Beamish, R. J., C. M. Neville, and R. Sweeting. 2003. Ecosystem-based management should include and assessment of the impacts of Pacific salmon escapements and enhancement on the juvenile salmon that rear in the Strait of Georgia. Puget Sound Research 2003, Proceedings.

Beamish, R. J., and R. Sweeting. 2007. Distribution information from coded-wire tag recoveries of juvenile coho and chinook salmon released into the Strait of Georgia and Puget Sound from 1997 to 2007. Georgia Basin Puget Sound Research 2007 Proceedings.

Beaugrand, G. 2005. Monitoring pelagic ecosystems using plankton indicators. Ices Journal of Marine Science 62:333-338.

Beckman, B. R., W. W. Dickhoff, W. S. Zaugg, C. Sharpe, S. Hirtzel, R. Schrock, D. A. Larsen, D. Ewing, A. Plamisano, C. B. Schreck, and C. V. W. Mahnken. 1999. Growth, smoltification, and smolt-to-adult return of spring chinook salmon (*Oncorhynchus tshawytshcha*) from hatcheries on the Deschutes River, Oregon. Transactions of the American Fisheries Society 128:1125-1150.

Beechie, T. J., E. Beamer, and L. Wasserman. 1994. Estimating coho salmon rearing habitat and smolt production losses in a large river basin, and implications for restoration. North American Journal of Fisheries Management 14:797-811.

Beechie, T. J., B. D. Collins, and G. R. Pess. 2001. Holocene and recent geomorphic processes, land use and salmonid habitat in two north Puget Sound river basins.*in* J. B. Dorava, D. R. Montgomery, F. Fitzpatrick, and B. Palcsak, editors. Geomorphic processes and riverine habitat, Water Science and Application Volume 4. American Geophysical Union, Washington, D.C.

Belant, J. L. 1997. Gulls in urban environments: landscape-level management to reduce conflict. Landscape and Urban Planning 38:245-258.

Bilton, H. T. 1984. Returns of chinook salmon in relation to juvenile size at release. Canadian Technical Report of Fisheries and Aquatic Sciences 1245:1-33.

Bish, R. L. 1982. Governing Puget Sound. University of Washington Press, Seattle, WA.

Blair, R. B. 1996. Land use and avian species diversity along an urban gradient. Ecological Applications 6:506-519.

Booth, D. B. 1987. Glaciofluvial infilling and scour of the Puget Lowland, Washington, during ice-steet glaciation. Geology 22:695-698.

Bortleson, G. C., M. J. Chrzastowski, and A. K. Helgerson. 1980. Historical changes of shoreline and wetland at eleven major deltas in the Puget Sound region, Washington. U.S. Geological Survey hydrological investigations atlas HA-617.

Bortone, S. A., editor. 2005. Estuarine Indicators. CRC Press, Boca Raton, FL.

Bottom, D. L., K. K. Jones, T. J. Cornwell, A. Gray, and C. A. Simenstad. 2005a. Patterns of Chinook salmon migration and residency in the Salmon River estuary (Oregon). Estuarine and Coastal Shelf Science 64:79-93.

Bottom, D. L., C. A. Simenstad, J. Burke, A. M. Baptista, D. A. Jay, K. K. Jones, E. Casillas, and M. Schiewe. 2005b. Salmon at river's end: the role of the estuary in the decline and recovery of Columbia River Salmon. NOAA Technical memorandum NMFS-NWFSC-68, U.S. Department of Commerce, Seattle, WA.

Boyd, I., S. Wanless, and C. J. Camphuysen, editors. 2006. Top Predators in Marine Ecosystems: Their Role in Monitoring and Management. Cambridge University Press, Cambridge, UK.

Boyd, R. 1999. The Coming of the Spirit of Pestilence: Introduced Infectious Diseases and Population Decline Among Northwest Coast Indians, 1774-1874. UBC Press; University of Washington Press, Vancouver; Seattle.

Brennan, J. S., and H. Culverwell. 2004. Marine riparian: An assessment of riparian functions in marine ecosystems. Washington Sea Grant, Seattle, Washington.

Brennan, J. S., K. F. Higgins, J. R. Cordell, and V. A. Stamatiou. 2004. Juvenile Salmon Composition, Timing, Distribution, and Diet in Marine Waters of Central Puget Sound in 2001-2002. King County Department of Natural Resources and Parks, Seattle, WA.

Brodeur, R. D., G. W. Boehlert, E. Casillas, M. B. Eldridge, J. H. Helle, W. T. Peterson, W. R. Heard, S. T. Lindley, and M. H. Schiewe. 2000. A coordinated research plan for estuarine and ocean research on Pacific salmon. Fisheries 25:7-16.

Brodeur, R. D., C. L. Shuchman, D. C. Reese, T. W. Miller, and E. A. Daly. in review. Spatial overlap and trophic interactions between pelagic fish and large jellyfish in the northern California Current.

Brodeur, R. D., H. Sugisaki, and G. L. Hunt. 2002. Increases in jellyfish biomass in the Bering Sea: implications for the ecosystem. Marine Ecology-Progress Series 233:89-103.

Brosofske, K. D., J. Chen, R. J. Naiman, and J. F. Franklin. 1997. Harvesting effects on microclimatic gradients from small streams to uplands in western Washington. Ecological Applications 7:1188-1200.

Bryce, S. A. 2006. Development of a bird integrity index: Measuring avian response to disturbance in the Blue Mountains of Oregon, USA. Environmental Management 38:470-486.

Bryce, S. A., R. M. Hughes, and P. R. Kaufmann. 2002. Development of a bird integrity index: Using bird assemblages as indicators of riparian condition. Environmental Management 30:294-310.

Buckley, R.M. 1969. Analysis of the Puget Sound sport fishery for resident coho salmon, *Oncorhynchus kisutch* (Walbaum). MS thesis. University of Washington.

Burke, J. L. 2004. Life histories of Columbia River juvenile Chinook salmon, 1914 to the present. MS thesis. Oregon State University, Corvallis, OR.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: a Practical Information-theoretic Approach. 2nd edition. Springer-Verlag, New York.

Burns, R. 1985. The Shape and Form of Puget Sound. University of Washington Press, Seattle, WA.

Butler, V. L., and S. K. Campbell. 2004. Resource intensification and resource depression in the Pacific Northwest of North America: A zooarchaeological review. Journal of World Prehistory 18:327-405.

Carlton, J. T. 2001. Introduced species in US Coastal waters: environmental impacts and management priorities. Pew Oceans Commission, Arlington, FL.

CBFEAP (Chesapeake Bay Fisheries Ecosystem Advisory Panel). 2006. Fisheries ecosystem planning for Chesapeake Bay. American Fisheries Society, Trends in Fisheries Science and Management 3, Bethesda, MD.

CBP (Chesapeake Bay Program). 2007. Chesapeake Bay 2006 Health and Restoration Assessment (Parts One and Two). CBP/TRS 283/07. EPA 903R-07001. Chesapeake Bay Program, Annapolis, MD.

Chasan, D. J. 1981. The Water Link: A History of Puget Sound as a Resource. University of Washington Press, Seattle, WA.

Chasan, D. J. 2000. The rusted shield: government's failure to enforce—or obey—our system of environmental law threatens the recovery of Puget Sound's wild salmon. Report commissioned by the Bullitt Foundation, Seattle, Washinton.

Chen, J. Q., S. C. Saunders, T. R. Crow, R. J. Naiman, K. D. Brosofske, G. D. Mroz, B. L. Brookshire, and J. F. Franklin. 1999. Microclimate in forest ecosystem and landscape ecology - Variations in local climate can be used to monitor and compare the effects of different management regimes. Bioscience 49:288-297.

Cheney, D. P., and T. Mumford. 1986. Shellfish and Seaweed Harvests of Puget Sound. University of Washington Press, Seattle, WA.

Christensen, N. L., A. M. Bartuska, J. H. Brown, S. Carpenter, C. Dantonio, R. Francis, J. F. Franklin, J. A. MacMahon, R. F. Noss, D. J. Parsons, C. H. Peterson, M. G. Turner, and R. G. Woodmansee. 1996. The report of the ecological society of America committee on the scientific basis for ecosystem management. Ecological Applications 6:665-691.

CIESM. 2001. Gelatinous zooplankton outbreaks: theory and practice. CIESM Workshop Series no 14. CIESM (Mediterranean Science Commission), Monaco.

Clark, K. R. 1993. Non-parametric multivariate analysis of changes in community structure. Australian Journal of Ecology 18:117-143.

Clark, K. R., and R. M. Warwick. 2001. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. Plymouth Marine Laboratory, Plymouth.

Clarke, K. R., and R. N. Gorley. 2006. Primer v6: user manual/tutorial, Plymouth, UK.

Cleland, C. E. 2001. Historical science, experimental science, and the scientific method. Geology 29:987-990.

Cochran, W. G. 1977. Sampling Techniques. Third edition. John Wiley and Sons, New York.

Cohen, A. N., and seventeen others. 1998. Puget Sound Expedition. A Rapid Assessment Survey of Non-indigenous Species in the Shallow waters of Puget Sound. Washington Department of Natural Resources, Olympia, WA.

Collias, E., and S. I. Andreeva. 1977. Puget Sound marine environment: an annotated bibliography. Washington Sea Grant, Seattle.

Collins, B. D., D. R. Montgomery, and A. J. Sheikh. 2003. Reconstructing the historical riverine landscape of the Puget Lowland. Pages 79-128 *in* D. R. Montgomery, S. Bolton, D. B. Booth, and L. Wall, editors. Restoration of Puget Sound Rivers. University of Washington, Seattle.

Collins, B. D., and A. J. Sheikh. 2005. Historical reconstruction, classification, and change analysis of Puget Sound tidal marshes. Project completion report to Washington Department of Natural Resources, Aquatic Resources Division. Olympia, WA.

Colombini, I., and L. Chelazzi. 2003. Influence of marine allochthonous input on sandy beach communities. Oceanography and Marine Biology 41:115-159.

Cone, J., and S. Ridlington, editors. 1996. The Northwest Salmon Crisis: A Documentary History. Oregon State University Press, Corvalis, Oregon.

Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs. Science 199:1302-1310.

Conomos, T. J., editor. 1979. San Francisco Bay: The Urbanized Estuary. Pacific Division of the American Association for the Advancement of Science, San Francisco.

Cooney, R. T., and R. D. Brodeur. 1998. Carrying capacity and North Pacific Salmon production: stock-enhancement implications. Bulletin of Marine Science 62:443-464.

Cornell Lab of Ornithology. 2005. Birds of North America Online. Cornell Lab of Ornithology and the American Ornithologists Union. http://bna.birds.cornell.edu/BNA

Cornwall, W. 2007. Huge task faces Puget Sound's anointed savior. Seattle Times, 5/20/2007. http://seattletimes.nwsource.com/html/localnews/2003714525_ruckelshaus20m.html

Cowan, J. H., K. A. Rose, and D. R. DeVries. 2000. Is density-dependent growth in young-of-the-year fishes a question of critical weight? Reviews in Fish Biology and Fisheries 10:61-89.

Cury, P., A. Bakun, R. J. M. Crawford, A. Jarre, R. A. Quinones, L. J. Shannon, and H. M. Verheye. 2000. Small pelagics in upwelling systems: patterns of interaction and structural changes in "wasp-waist" ecosystems. Ices Journal of Marine Science 57:603-618.

Cushing, D. H. 1990. Plankton production and year-class strength in fish populations: an update of the match/mismatch hypothesis. Advances in Marine Biology 26:250-293.

Dahl, T. E. 2000. Status and Trends of Wetlands in the Conterminous United States 1986 to 1987. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C.

Day, J. W., C. A. S. Hall, W. M. Kemp, and A. Yanez-Arancibia. 1989. Estuarine Ecology. John Wiley and Sons, New York, NY.

Dayton, C. M. 2003. Model comparisons using information measures Journal of applied statistical methods 2:281-292.

Dayton, P. K., M. J. Tegner, P. B. Edwards, and K. L. Riser. 1998. Sliding baselines, ghosts, and reduced expectations in kelp forest communities. Ecological Applications 8:309-322.

Deegan, L. A., J. T. Finn, S. G. Ayvazian, C. A. Ryder-Kieffer, and J. Buonaccorsi. 1997. Development and validation of an estuarine biotic integrity index. Estuaries 20:601-617.

DeLacy, A. C., B. S. Miller, and S. F. Borton. 1972. Checklist of Puget Sound fishes. Washington Sea Grant, Seattle, WA.

DeMartini, E. E. 1999. Intertidal spawning. Pages 143-164 in M. H. Horn, K. L. M. Martin, and M. A. Chotkowski, editors. Intertidal Fishes: Life in Two Worlds, 1st edition. Academic Press, San diego, California.

Dethier, M. N. 1990. A Marine and Estuarine Classification System for Washington State. Washington Natural Heritage Program. Washington Department of Natural Resources, Olympia, WA.

Dethier, M. N., and G. C. Schoch. 2005. The consequences of scale: assessing the distribution of benthic populations in a complex estuarine fjord. Estuarine and Coastal Shelf Science 62:253-270.

Diamond, J. 1986. Overview: Laboratory experiments, field experiments, and natural experiments. Pages 3-22 in J. Diamond and T. J. Case, editors. Community Ecology. Harper & Row, New York.

Diamond, J. 1997. Guns, Germs, and Steel. The Fates of Human Societies. W.W. Norton & Company, New York, NY.

Donovan, T. M., and F. R. Thompson. 2001. Modeling the ecological trap hypothesis: A habitat and demographic analysis for migrant songbirds. Ecological Applications 11:871-882.

Downes, B. J., L. A. Barmuta, P. G. Fairweather, D. P. Faith, M. J. Keough, P. S. Lake, B. D. Mapstone, and G. P. Quinn. 2002. Monitoring Ecological Impacts: Concepts and Practice in Flowing Waters. Cambridge University Press, Cambridge, UK.

Downing, J. 1983. The Coast of Puget Sound: its Processes and Development. Washington Sea Grant, Seattle, WA.

Duffy, E. J. 2003. Early marine distribution and trophic interactions of juvenile salmon in Puget Sound. Master's thesis. University of Washington, Seattle, Washington.

Duffy, E. J., D. A. Beauchamp, and R. M. Buckley. 2005. Early marine life history of juvenile Pacific salmon in two regions of Puget Sound. Estuarine and Coastal Shelf Science 64:94-107.

Dugan, J. E., D. M. Hubbard, M. D. McCrary, and M. O. Pierson. 2003. The response of macrofauna communities and shorebirds to macrophyte wrack subsidies on exposed sandy beaches of southern California. Estuarine Coastal and Shelf Science 58:25-40.

Ehrlich, K. F., and D. A. Farris. 1971. Some influences of temperature on the development of the grunion *leuresthes tenuis* (Ayres). California Fish and Game 57:58-68.

Elmgren, R. 2001. Understanding human impact on the Baltic ecosystem: Changing views in recent decades. Ambio 30:222-231.

Emlen, J. M., R. R. Reisenbichler, A. M. McGie, and T. E. Nickelson. 1990. Density-Dependence at Sea for Coho Salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 47:1765-1772.

English, T. S., and R. E. Thorne. 1977. Puget Sound interim studies. Acoustic and net surveys of fishes and zooplankton. Final report. M77-17, Municipality of Metropolitan Seattle, Seattle, Washington.

Feist, B. E., E. A. Steel, G. R. Pess, and R. E. Bilby. 2003. The influence of scale on salmon habitat restoration priorities. Animal Conservation 6:271-282.

Flagg, T. A., B. A. Berejikian, J. Colt, W. W. Dickhoff, L. W. Harrell, C. E. Maynard, C. E. Nash, M. S. Strom, R. N. Iwamoto, and C. V. W. Mahnken. 2000. Ecological and behavioral impacts of artificial production strategies on the abundance of wild salmon populations. NOAA Tech. Memo NMFS-NWFSC-41, NOAA Fisheries, Seattle.

Flannery, T. 2001. The eternal frontier: an ecological history of North America and its peoples. Atlantic Monthly Press, New York, NY.

Fore, L. 2003. Developing Biological Indicators: Lessons Learned from Mid-Atlantic Streams. Report prepared for EPA under Contract No. 50-CMAA-900065. EPA 903/R-003/003. U.S. Environmental Protection Agency, Office of Environmental Information and Mid-Atlantic Integrated Assessment Program, Region 3, Ft. Meade, MD.

Francis, R. C., and S. R. Hare. 1994. Decadal-scale regime shifts in the large marine ecosystems of the North-east Pacific: A case for historical science. Fisheries Oceanography 3:279-291.

Francis, R. C., S. R. Hare, A. B. Hollowed, and W. S. Wooster. 1998. Effects of interdecadal climate variability on he oceanic ecospstems of the NE Pacific. Fisheries Oceanography 7:1-21.

Frank, K. T., and W. C. Leggett. 1981a. Prediction of egg development and mortality rates in capelin (*Mallotus villosus*) from meteorological, hydrographic, and biological factors. Candian Journal of Fisheries and Aquatic Sciences 38:1327-1338.

Frank, K. T., and W. C. Leggett. 1981b. Wind regulation of emergence times and early larval survival in capelin (*Mallotus villosus*). Candian Journal of Fisheries and Aquatic Sciences 38:215-223.

Fresh, K. L. 1979. Distribution and abundance of fishes occuring in the nearshore surface waters of northern Puget Sound. M.S. Thesis. University of Washington, Seattle.

Fresh, K. L. 1997. The role of competition and predation in the decline of Pacific salmon and steelhead. Pages 245-275 *in* D. J. Stouder, P. A. Bisson, and R. J. Naiman, editors. Pacific salmon and their ecosystems: status and future options. Chapman and Hall, New York.

Fresh, K. L. 2006. Juvenile Pacific Salmon in the Nearshore Ecosystems of Washington State. Puget Sound Nearshore Partnership Report No. 2006-6, Seattle District, U.S. Army Corps of Engineers, Seattle, Washington.

Fresh, K. L., R. D. Cardwell, and R. R. Koons. 1981. Food habits of Pacific salmon, baitfish and their potential competitors and predators in the marine waters of Washington, August 1978 to September 1979. State of Washington Department of Fisheries progress report number 145, , Olympia, Washington.

Fresh, K. L., D. Rabin, C. A. Simenstad, E. O. Salo, K. Garrison, and L. Matheson. 1979. Fish ecology studies in the Nisqually Reach area of southern Puget Sound, Washington. Fisheries Research Institute, College of Fisheries, University of Washington, Seattle, Washington.

Fresh, K. L., D. J. Small, H. Kim, C. Waldbillig, M. Mizell, M. I. Carr, and L. Stamatiou. 2006. Juvenile salmon use of Sinclair Inlet, Washington, in 2001 and 2002. Technical Report No. FPT 05-08, Washington Department of Fish and Wildlife, Olympia, WA.

Fung, J., and C. Davis. 2005. Characterization of WRIA 9 Shoreline Landforms. Puget Sound Nearshore Partnership Report No. 2005-01. Washington Department of Fish and Wildlife, Olympia, Washington.

Furness, R. W., and J. J. D. Greenwood. 1993. Birds as Monitors of Environmental Change. Chapman and Hall, London.

Gaeckle, J., P. Dowty, B. Reeves, H. Berry, S. Wyllie-Echeverria, and T. Mumford. 2007. Puget Sound submerged vegetation monitoring project 2005 Monitoring report. Nearshore Habitat Program, Aquatic Resources Division, Washington State Department of Natural Resources Olympia, Washington.

Gannett, H. 1899. Nineteenth annual report of the United States Geological Survey, 1897-1898. Part V -- Forest Reserves. Government Printing Office, Washington, D.C.

GAO (Government Accounting Office). 2005. Chesapeake Bay Prgram: Improved Strategies Are Needed to Better Assess, Report, and Manage Restoration Progress. GAO-06-96. U.S. Government Accounting Office, Washington, DC.

Gargett, A. E. 1997. The optimum stability 'window': A mechanism underlying decadal fluctuations in North Pacific salmon stocks? Fisheries Oceanography 6:109-117.

Geiger, R. 1965. The Climate Near the Ground. Harvard University Press, Cambridge, MA.

Gelfenbaum, G. T., T. Mumford, J. Brennan, H. Case, M. Dethier, K. Fresh, F. Goetz, M. van Heeswijk, T. M. Leschine, M. Logsdon, D. Myers, J. Newton, H. Shipman, C. A. Simenstad, C. Tanner, and D. Woodson. 2006. Coastal Habitats in Puget Sound: A research plan in support of the Puget Sound Nearshore Partnership. Puget Sound Nearshore Partnership Report No. 2006-1. U.S. Geological Survey, Seattle, Washington.

Gibson, G. R., M. L. Bowman, J. Gerritsen, and B. D. Snyder. 2000. Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance. EPA 822-B-00-24, U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Godin, J.-G. J. 1981. Daily patterns of feeding behavior, daily rations, and diets of juvenile pink salmon (*Oncorhynchus gorbuscha*) in two marine bays of British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 38:10-15.

Greene, C., and T. J. Beechie. 2004. Consequences of potenial density-dependent mechanisms on recovery of ocean-type chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 61:590-602.

Greene, C. M., D. W. Jensen, E. Beamer, G. R. Pess, and E. A. Steel. 2005. Effects of environmental conditions during stream, estuary, and ocean residency on Chinook salmon return rates in the Skagit River, WA. Transactions of the American Fisheries Society 134:1562-1581.

Greve, W., and T. R. Parsons. 1977. Photosynthesis and fish production: hypothetical effects of climate change and pollution. Heig. wissen. Meeresunter 20:666-672.

Groote, C., and L. Margolis, editors. 1991. Pacific Salmon Life Histories. University of British Columbia Press, Vancouver.

Hairston, N. G., C. M. Kearns, L. P. Demma, and S. W. Effler. 2005. Species-specific Daphnia phenotypes: A history of industrial pollution and pelagic ecosystem response. Ecology 86:1669-1678.

Hare, S. R., N. J. Mantua, and R. C. Francis. 1999. Inverse production regimes: Alaska and West Coast Pacific salmon. Fisheries 24:6-14.

Hatchery Scientific Review Group (HSRG). 2004.

Hay, S. 2006. Marine ecology: Gelatinous bells may ring change in marine ecosystems. Current Biology 16:R679-R682.

Healey, M. C. 1982. Juvenile Pacific salmon in estuaries: the life support system. Pages 315-341 *in* V. S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

Healey, M. C. 1991. The life history of chinook salmon (*Oncorhynchus tshawytscha*). Pages 311-393 *in* C. Groote and L. Margolis, editors. Life history of Pacific salmon. University of B.C. Press, Vancouver, B.C.

Heinz Center. 2002. The state of the nation's ecosystems: measuring the lands, waters, and living resources of the United States. Cambridge University Press, Cambridge, UK.

Hensleigh, J. E. 1994. Puget Sound Groundfish: A Case of Bifurcated Fisheries Management. Master's Thesis. University of Washington, Seattle, Washington.

Hepinstall, J. A., M. Alberti, and S. Coe. in prep. Developing urban landscape trajectories for Puget Sound, Washington using multi-temporal landsat thematic mapper imagery.

Hershberger, P. 2007. Impacts of disease to wild fish populations with special reference to the Salish Sea region. Proceedings of the 2007 Georgia Basin Puget Sound Research Conference.

Hilborn, R., and J. Winton. 1993. Learning to Enhance Salmon Production - Lessons from the Salmonid Enhancement Program. Canadian Journal of Fisheries and Aquatic Sciences 50: 2043-2056.

Hilborn, R., and D. Eggers. 2000. A review of the hatchery programs for pink salmon in Prince William Sound and Kodiak Island, Alaska. Transactions of the American Fisheries Society 129:333-350.

Hilborn, R., and D. Eggers. 2001. A review of the hatchery programs for pink salmon in Prince William Sound and Kodiak Island, Alaska: Response to comment. Transactions of the American Fisheries Society 130:720-724.

Hollibaugh, J. T., editor. 1996. San Francisco Bay: The Ecosystem. Further Investigations into the Natural History of San Francisco Bay and Delta With Reference to the Influence of Man. Pacific Division of the American Association for the Advancement of Science, San Francisco.

Holtby, L. B., B. C. Anderson, and R. K. Kadowski. 1990. Importance of smolt size and early ocean growth to interannual variability in marine survival of coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 47:2181-2194.

Hooff, R. C., and W. T. Peterson. 2006. Copepod biodiversity as an indicator of changes in ocean and climate conditions of the northern California current ecosystem. Limnology and Oceanography 51:2607-2620.

Houde, E. D. 1997. Patterns and consequences of selective processes in teleost early life histories. Pages 173-196 *in* C. Chambers and E. Trippel, editors. Early life history and recruitment in fish populations. Chapman and Hall, London.

Hughs, T. P., D. R. Bellwood, C. Folke, R. S. Steneck, and J. Wilson. 2005. New paradigms for supporting the resilience of marine ecosystems. Trends in Ecology and Evolution 20:380-386.

Hutchings, M. J., E. A. John, and A. J. A. Stewart. 2000. The Ecological Consequences of Environmental Heterogeneity. Blackwell Science, Ltd., Oxford.

Jackson, J. B. C. 2001. What was natural in the coastal oceans? Proceedings of the National Academy of Sciences of the United States of America 98:5411-5418.

Jameson, S. C., M. V. Erdman, J. R. Karr, and K. W. Potts. 2001. Charting a course toward diagnostic monitoring: a continuing riview of coral reef attributes and a research strategy for creating coral reef indexes of biotic integrity. Bulletin of Marine Science 69:701-744.

Jedrzejczak, M. F. 2002. Spatio-temporal decay "hot spots" of stranded wrack in a Baltic sandy coastal system. Part I. Comparative study of the pattern: 1 type of wrack vs 3 beach sites. Oceanologia 44:491-512.

Johnson, L. L., G. M. Ylitalo, M. R. Arkoosh, A. N. Kagley, C. Stafford, J. L. Bolton, J. Buzitis, B. F. Anulacion, and T. K. Collier. 2007. Contaminant exposure in outmigrant juvenile salmon from Pacific Northwest estuaries of the United States. Environmental Monitoring and Assessment.

Johnson, O. W., W. S. Grant, R. G. Kope, K. Neely, F. W. Waknitz, and R. S. Waples. 1997. Status review of chum salmon from Washington, Oregon, and California. Tech. Memo. NMFS-NWFSC-32, U.S. Dept. Commer., NOAA, Seattle, WA.

Jorgensen, S. E., R. Costanza, and F.-L. Xu, editors. 2005. Handbook of Ecological Indicators for Assessment of Ecosystem Health. CRC Press, Boca Raton, Florida.

Karieva, P., M. Marvier, and M. Mcclure. 2000. Recovery and management options for spring-summer Chinook salmon in the Columbia River basin. Science 290:977-979.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21–27.

Karr, J. R. 1991. Biological integrity: A long–neglected aspect of water resource management. Ecological Applications 1:66–84.

Karr, J. R. 1995. Clean water is not enough. Illahee 11:51-59.

Karr, J. R. 2006. Seven foundations of biological monitoring and assessment. Biologia Amientale 20:7-18.

Karr, J. R., and E. W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington, D.C.

Karr, J. R., and D. R. Dudley. 1981a. Ecological perspective on water quality goals. Environmental Management 5:55-68.

Karr, J. R., and D. R. Dudley. 1981b. Ecological perspectives on water quality goals. Environmental Management 5:55-68.

Karr, J. R., and K. E. Freemark. 1983. Habitat Selection and Environmental Gradients - Dynamics in the Stable Tropics. Ecology 64:1481-1494.

Kelsey, K. A., and D. W. West. 1998. Riparian Wildlife. Pages 235-260 in R. J. Naiman and R. E. Bilby, editors. River Ecology and Mangement: Lessons From the Pacific Coastal Ecoregion. Springer-Verlag, New York.

Kideys, A. E. 2002. Fall and rise of the Black Sea ecosystem. Science 297:1482-1484.

Knox, G. A. 2001. The Ecology of Seashores. CRC Press, Boca Raton, FL.

Kozloff, E. N. 1983. Seashore life of the northern Pacific Coast. University of Washington Press, Seattle, WA.

Krahn, M. M., M. J. Ford, W. F. Perrin, P. R. Wade, R. P. Angliss, M. B. Hanson, B. L. Taylor, G. M. Ylitalo, M. E. Dahlheim, J. E. Stein, and R. S. Waples. 2004. Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-62.

Kruckeberg, A. R. 1991. The Natural History of the Puget Sound Country. University of Washington Press, Seattle, WA.

Larkin, P. A. 1996. Concepts and issues in marine ecosystem management. Reviews in Fish Biology and Fisheries 6:139-164.

Lemberg, N. A., M. F. O'Toole, D. E. Penttila, and K. C. Stick. 1997. 1996 forage fish stock status report. Washington Department of Fish and Wildlife, Olympia, Washington.

Levin, P. S., R. W. Zabel, and J. G. Williams. 2001. The road to extinction is paved with good intentions: negative association of fish hatcheries with threatened salmon. Proceedings Of The Royal Society Of London Series B-Biological Sciences 268:1153-1158.

Levin, S. A. 1992. The problem of pattern and scale in ecology. Ecology 73:1943-1967.

Levings, C. D. 1994. Feeding behavior of juvenile salmon and significance of habitat during estuary and early sea phase. Nordic Journal of Freshwater Research 69:7-16.

Levings, C. D., and G. Jamieson. 2001. Marine and estuarine riparian habitats and their role in coastal ecosystems, Pacific region. Research Document 2001/109, Canadian Science Advisory Secretariat, Ottowa, Canada.

Levings, C. D., C. D. McAllister, and B. D. Cheng. 1986. Differential use of the Campbell River estuary, British Columbia, by wild and hatchery-reared juvenile chinook salmon. Canadian Journal of Fisheries and Aquatic Sciences 43:1386-1397.

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

Lincoln, J. H. 1952. A tidal model of Puget Sound. Unpublished manuscript. University of Washington Department of Oceanography.

Lombard, J. 2006. Saving Puget Sound: a conservation strategy for the 21st century. American Fisheries Society, Bethesda, MD.

Long, E. R., editor. 1982. A Synthesis of Biological Data from the Strait of Juan de Fuca and Northern Puget Sound. EPA-600/7-82-004. US EPA, Washington, DC.

Loosanoff, V. L. 1937. The spawning run of the Pacific surf smelt, *Hypomesus pretiosus* (Girard). Internationale Revue der gesamter Hydrobiologie and Hydrographie 36:170-183.

Lorenzen, K. 1996. The relationship between body weight and natural mortality in juvenile and adult fish: a comparison of natural ecosystems and aquaculture. Journal of Fish Biology 49:627-647.

Lotze, H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, and C. H. Peterson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312:1806-1809.

Love, M. S., M. Yoklavich, and L. Thorsteinson. 2002. The Rockfishes of the Northeast Pacific. University of California Press, Berkely, California.

Lynam, C. P., and A. S. Brierley. 2007. Enhanced survival of 0-group gadoid fish under jellyfish umbrellas. Marine Biology 150:1397-1401.

Macdonald, K., D. Simpson, B. Paulson, J. Cox, and J. Gendron. 1994. Shoreline Armoring Effects on Physical Coastal Processes in Puget Sound, Washington. Report 94-78. Washington Department of Ecology. Olympia, WA.

Magnusson, A., and R. Hilborn. 2003. Estuarine influence on survival rates of Coho (Oncorhynchus kisutch) and Chinook salmon (*Oncorhynchus tshawytscha*) released from hatcheries on the US Pacific Coast. Estuaries 26:1094-1103.

Mann, K. H. 2000. Ecology of coastal water with implications for coastal management. 2nd edition. Blackwell Science, Inc., Malden, Mass.

Mantua, N. J., and R. C. Francis. 2004. Natural climate insurance for Pacific Northwest salmon and salmon fisheries: Finding our way through the entangled bank. Pages 127-140 *in* E. E. Knudsen and D. MacDonald, editors. Fish in our Future? Perspectives on Fisheries Sustainability. A special publication of the American Fisheries Society. American Fisheries Society, Bethesda, MD.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society 78:1069-1079.

Martin, K. L. M., and D. L. Swiderski. 2001. Beach spawning in fishes: Phylogenetic tests of hypotheses. American Zoologist 41:526-537.

Marzluff, J. M. 2005. Island biogeography for an urbanizing world: how extinction and colonization may determine biological diversity in human-dominated landscapes. Urban Ecosystems 8:157-177.

Maser, C., and J. R. Sedell. 1994. From forest to the sea: the ecology of wood in streams, rivers, estuaries, and oceans. St. Lucie Press, Delray Beach, FL.

McBride, A., and E. Beamer. 2007. A landscape process based restoration plan for North Fidalgo Island, Washington. Proceedings of the 2007 Georgia Basin Puget Sound Research Conference.

McCune, B. M., and J. B. Grace. 2002. Analysis of Ecological Communities. MjM Software Design, Glenedon Beach, Oregon.

McKee, B. 1972. Cascadia: the geologic evolution of the Pacific Northwest. McGraw-Hill, San Francisco, CA.

McLellan, P. M., C. A. Barnes, H. E. Babcock, E. E. Collias, D. R. Doyle, R. H. Fleming, W. Irvin, P. Kosche, M. Rattray, and J. Shaw. 1953-1954. Puget Sound and approaches; a literature survey. Volumes I-III. University of Washington, Seattle, Washington.

Middaugh, D. P., H. W. Kohl, and L. E. Burnett. 1983. Concurrent Measurement of Intertidal Environmental Variables and Embryo Survival for the California Grunion, Leuresthes-Tenuis, and Atlantic Silverside, Menidia-Menidia (Pisces, Atherinidae). California Fish and Game 69:89-96.

Miller, B. S., and S. F. Borton. 1980. Geographical distribution of Puget Sound fishes: maps and data source sheets. Fisheries Research Institute, College of Fisheries, University of Washington, Seattle, Washington.

Miller, B. S., C. A. Simenstad, J. N. Cross, K. L. Fresh, and S. N. Steinford. 1980. Nearshore fish and macroinvertebrate assemblages along the Strait of Juan de Fuca including food habits of the common nearshore fish: final report of three years' sampling, 1976-1979. EPA-600/7-80-E027, Environmental Protection Agency, Washington, D.C.

Mills, C. E. 1995. Medusae, Siphonophores, and Ctenophores as Planktivorous Predators in Changing Global Ecosystems. Ices Journal of Marine Science 52:575-581.

Mills, C. E. 2001. Jellyfish blooms: are populations increasing globally in response to changing ocean conditions? Hydrobiologia 451:55-68.

Mills, C. E., and C. O. Hermans. 2007. Historical centennial timeline for the University of Washington's Friday Harbor Laboratories 1903-2005. http://faculty.washington.edu/cemills/FHLTimeline.html.

Mobrand, L. E., J. Barr, L. Blankenship, D. E. Campton, T. T. P. Evelyn, T. A. Flagg, C. V. W. Mahnken, L. W. Seeb, P. R. Seidel, and W. W. Smoker. 2005. Hatchery reform in Washington State: principles and emerging issues. Fisheries 30:11-23.

Mofjeld, H. O. 1992. Subtidal sea level fluctuations in a large fjord system. Journal of Geophysical Research 97(C12):20191-20199.

Mofjeld, H. O., and L. H. Larsen. 1984. Tides and tidal currents of the inland waters of western Washington. Technical Report NOAA Technical Memorandum OAR PMEL-56. Pacific Marine Environmental Laboratory, Seattle, Washington.

Montgomery, D. R. 2003. King of fish: the thousand-year run of salmon. Westview Press, Cambridge, MA.

Montgomery, D. R., S. Bolton, D. B. Booth, and L. Wall, editors. 2003. Restoration of Puget Sound Rivers. University of Washington Press, Seattle.

Mote, P. W., E. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. Climatic Change 61:45-88.

Myers, J. M., R. G. Kope, G. J. Bryant, D. Teel, L. J. Lierheimer, T. C. Wainwright, W. S. Grant, F. W. Waknitz, K. Neely, S. T. Lindley, and R. S. Waples. 1998. Status review of chinook salmon from Washington, Idaho, Oregon, and California. Tech. Memo. NMFS-NWFSC-35, U.S. Dept. Commer., NOAA, Seattle, WA.

Myers, K. W., and H. F. Horton. 1982. Temporal Use Of An Oregon Estuary By Hatchery And Wild Juvenile Salmon. Pages 377-392 in V. S. Kennedy, editor. Estuarine Comparisons. Academic Press, New York.

Naiman, R. J., C. A. Johnston, and J. C. Kelly. 1988. Alteration of North American streams by beaver. Bioscience 38:753-761.

Naiman, R. J., K. L. Fetherson, S. J. McKay, and J. Chen. 1998. Riparian Forests. Pages 289-323 in R. J. Naiman and R. E. Bilby, editors. River Ecology and Mangement: Lessons From the Pacific Coastal Ecoregion. 1st edition. Springer-Verlag, New York.

Nehlsen, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. Fisheries 16:4-21.

Nelson, T. S., G. Ruggerone, H. Kim, R. Schaefer, and M. Boles. 2004. Juvenile Chinook migration, growth and habitat use in the lower Green River, Duwamish River and nearshore of Elliott Bay 2001-2003. Draft report, King County Department of Natural Resources and Parks, Seattle, Washington.

Newton, J. A., T. Mumford, J. Hohrmann, J. West, R. Llanso, H. Berry, and S. Redman. 2000. A conceptual model for environmental monitoring of a marine system. Puget Sound Ambient Monitoring Program, Olympia, Washington.

Newton, J. A. 2002. Low dissolved oxygen in Hood Canal—a science primer. http://www.hoodcanal.washington.edu/aboutHC/scienceprimer.jsp

NOAA. 1978. MESA Marine Ecosystems Analysis Program. The Puget Sound Project. FY77 - Anual Report. National Oceanic and Atmospheric Administration, Boulder, CO.

NRC (National Research Council). 1990. Managing Troubled Waters, the Role of Marine Environmental Monitoring. National Academy Press, Washington, D.C.

NRC (National Research Council). 1996. Upstream: Salmon and Society in the Pacific Northwest. National Academy Press, Washington DC.

NRC (National Research Council). 2000. Ecological Indicators for the Nation. National Academy of Sciences, Washington, DC.

Nyswander, D. R., J. R. Evenson, B. L. Murphie, and T. A. Cyra. 2002. Report of Marine Bird and Mammal Component, Puget Sound Ambient Monitoring Program, for July 1992 to December 1999 Period. Agency Report. Washington Department of Fish and Wildlife, Olympia, Washington.

O'Connell, T. J., L. E. Jackson, and R. P. Brooks. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecological Applications 10:1706-1721.

Palsson, W. A., J. C. Hoeman, G. G. Bargmann, and D. E. Day. 1997. 1995 Status of Puget Sound Bottomfish Stocks (revised). Washington Department of Fish and Wildlife, Olympia, WA.

Parker, R. R. 1971. Size selective predation among juvenile salmonid fishes in a British Columbia inlet. Journal of the Fisheries Research Board of Canada 28:1503-1510.

Parsons, T. R., and C. M. Lalli. 2002. Jellyfish population explosions: Revisiting a hypothesis of possible causes. La mer 40:111-121.

Pearcy, W. G. 1992. Ocean ecology of North Pacific salmonids. University of Washington Press, Seattle.

Pearcy, W. G., K. Y. Aydin, and R. D. Brodeur. 1999. What is the carrying capacity of the North Pacific Ocean for salmonids? Pices Press 7:17-23.

Penttila, D. E. 1973. Observations on some Puget Sound spawning beaches of the surf smelt (*Hypomesus pretiosus*). Supplemental Progress Report. Washington Department of Fisheries, Olympia, WA.

Penttila, D. E. 1978. Studies of the surf smelt (*Hypomesus pretiosus*) in Puget Sound. Technical Report No. 42, Washington Department of Fisheries, Olympia, Washington.

Penttila, D. E. 1995. The WDFW's Puget Sound intertidal baitfish spawning beach survey project. Puget Sound Research '95. Conference proceedings:235-241.

Penttila. D. E. 1997. Investigations of intertidal spawning habitats of surf smelt and Pacific sand lance in Puget Sound, Washington. Pages 395-408 *in* Alaska Sea Grant, editor. 1997. Forage Fishes in Marine Ecosystems: Proceedings of the International Symposium on the Role of Forage Fishes in Marine Ecosystems. Alaska Sea Grant College Program, University of Alaska, Fairbanks, Anchorage, AK

Penttila, D. E. 2001. Effects of shading upland vegetation on egg survival for summer-spawning surf smelt on upper intertidal beaches in Puget Sound. Puget Sound Research 2001 Conference Proceedings. http://www.psat.wa.gov/Publications/01_proceedings/sessions/oral/2a_pentt.pdf (accessed July 17, 2005).

Pess, G. R., D. R. Montgomery, T. J. Beechie, and L. Holsinger. 2003. Anthropogenic alterations to the biogeography of Puget Sound salmon. Pages 129-154 *in* D. R. Montgomery, S. Bolton, D. B. Booth, and L. Wall, editors. Restoration of Puget Sound rivers. University of Washington Press, Seatte, WA.

Peterman, R. M. 1984. Density-dependent growth in early ocean life of sockeye salmon (*Oncorhynchus nerka*). Canadian Journal of Fisheries and Aquatic Sciences 41:1825-1829.

Peterson, B., R. Emmett, R. Goericke, E. Venrick, A. Mantyla, S. J. Bograd, F. B. Schwing, S. Ralston, K. A. Forney, R. Hewitt, N. Lo, W. Watson, J. Barlow, M. Lowry, B. E. Lavaniegos, F. Chavez, W. J. Sydeman, D. Hyrenbach, R. W. Bradley, P. Warzybok, K. Hunter, S. Benson, M. Weise, and J. Harvey. 2006. The state of the California current, 2005-2006: Warm in the North, cool in the South. California Cooperative Oceanic Fisheries Investigations Reports 47:30-74.

Peterson, C. H., and J. A. Estes. 2001. Conservation and Management of marine communities. Pages 469-508 *in* M. D. Bertness, S. D. Gaines, and M. E. Hay, editors. Marine Community Ecology. Sinauer Associates, Inc., Sunderland, MA.

Peterson, I., and J. S. Wroblewski. 1984. Mortality rate of fishes in the pelagic ecosystem. Canadian Journal of Fisheries and Aquatic Sciences 41:1117-1120.

Phillips, R. C. 1984. The Ecology of Eelgrass Meadows in the Pacific Northwest: A Community Profile. FWS/OBS-84/24, U.S. Fish and Wildlife Service, Washington, D.C.

Pikitch, E. K., C. Santora, E. A. Babcock, A. Bakun, R. Bonfil, D. O. Conover, P. Dayton, P. Doukakis, D. Fluharty, B. Heneman, E. D. Houde, J. Link, P. A. Livingston, M. Mangel, M. K. McAllister, J. Pope, and K. J. Sainsbury. 2004. Ecosystem-based fishery management. Science 305:346-347.

Pilkey, O. H., and L. Pilkey-Jarvis. 2007. Uselss Arithmetic: Why Environmental Scientists Can't Predict the Future. Columbia University Press, New York, NY.

Pinnix, W. D. 1999. Climate and Coho: A Puget Sound Perspective. M.S. thesis. University of Washington, Seattle, WA.

POC (Pew Oceans Commission). 2003. America's living oceans: charting a course for sea change. Summary report., Pew Oceans Commission, Arlington, VA.

Polis, G. A., and S. D. Hurd. 1996. Linking marine and terrestrial food webs: Allochthonous input from the ocean supports high secondary productivity on small islands and coastal land communities. American Naturalist 147:396-423.

Pollock, M. M., G. R. Pess, T. J. Beechie, and D. R. Montgomery. 2004. The importance of beaver ponds to coho salmon production in the Stillaguamish River basin. North American Journal of Fisheries Management 24:749-760.

Porter, S. C., and T. W. Swanson. 1998. Radiocarbon age constraints on rates of advance and retreat of the Puget Lobe of the Cordilleran ice sheet during the last glaciation. Quaternary Research 50:205-213.

Preikshot, D., and A. Beattie. 2001. Fishing for answers: analysis of ecosystem dynamics, trophic shifts and salmonid population changes in South Puget Sound, WA, 1970-1999. The Fisheries Center, University of British Columbia, Vancouver, BC.

PSAMP Management Committee. 2005. 2005 review of the Puget Sound Ambient Monitoring Program. Final Report. Puget Sound Action Team, Olympia, Washington.

PSAT (Puget Sound Action Team). 2007a. 2007-2009 Puget Sound conservation & recovery plan. Puget Sound Action Team, Olympia, Washington. http://www.psp.wa.gov/our_work/pscrp.htm

PSAT (Puget Sound Action Team). 2007b. Puget Sound Update: Ninth Report of the Puget Sound Ambient Monitoring Program. Puget Sound Action Team, Olympia, Washington. http://www.psp.wa.gov/puget_sound/update.htm

PSAT (Puget Sound Action Team). 2007c. State of the Sound 2007 executive summary and report card. Puget Sound Action Team, Olympia, Washington. http://www.psp.wa.gov/publications/puget_sound/sos/07sos/sections/2007_st ateofthesound_executive.pdf

PSNERP (Puget Sound Nearshore Ecosystem Restoration Project). 2004. PSNERP work plan: A strategy for restoration of Puget Sound. Working draft. Puget Sound Nearshore Ecosystem Restoration Project, Seattle.

PSP. 2006. Sound Health, Sound Future. Protecting and Restoring Puget Sound. Puget Sound Partnership.

PSWQAT (Puget Sound Water Quality Action Team). 2002. 2002 Puget Sound Update: Eighth Report of the Puget Sound Ambient Monitoring Program. Puget Sound Water Quality Action Team. Olympia, WA.

PSWQA (Puget Sound Water Quality Authority). 1987. 1987 Puget Sound Water Quality Management Plan. Puget Sound Water Quality Authority, Seattle, Washington.

PSWQA (Puget Sound Water Quality Authority). 1988. Committee on research in Puget Sound. Final Report. Puget Sound Water Quality Authority, Seattle, Washington.

Pugh, P. J. A., and H. E. Macalister. 1994. Acari of the Supralittoral Zone on Sub-Antarctic South Georgia. Pedobiologia 38:552-565.

Purcell, J. E. in review. Anthropogenic causes of jellyfish blooms and their direct consequences for humans: a review.

Purcell, J. E., and M. N. Arai. 2001. Interactions of pelagic cnidarians and ctenophores with fish: a review. Hydrobiologia 451:27-44.

Quinn, T. P. 2005. The Behavior and Ecology of Pacific Salmon and Trout. American Fisheries Society, Bethesda, Maryland.

Quinn, T. P., B. R. Dickerson, and L. A. Vollestad. 2005. Marine survival and distribution patterns of two Puget Sound hatchery populations of coho (*Oncorhynchus kisutch*) and chinook (*Oncorhynchus tshawytscha*) salmon. Fisheries Research 76:209-220.

Rafaelli, D., and S. Hawkins. 1996. Intertidal Ecology. 1st edition. Chapman and Hall, London.

Reimers, P. E. 1973. The length of residence of juvenile fall chinook in the Sixes River. Oregon Fish Game Commisions Research 4:1-43.

Rhodes, L. D., C. Durkin, S. L. Nance, and C. A. Rice. 2006. Prevalence and analysis of Renibacterium salmoninarum infection among juvenile Chinook salmon Oncorhynchus tshawytscha in North Puget Sound. Diseases of Aquatic Organisms 71:179-190.

Rice, C. A. 2006. Effects of shoreline modification on a northern Puget Sound beach: microclimate and embryo mortality in surf smelt (*Hypomesus pretiosus*). Estuaries and Coasts 29:63-71.

Rice, C. A., W. G. Hood, L. M. Tear, C. A. Simenstad, L. L. Johnson, G. D. Williams, P. Roni, and B. E. Feist. 2005. Monitoring Rehabilitation in Temperate North American Estuaries. Pages 167-207 *in* P. Roni, editor. Monitoring Stream and Watershed Restoration. American Fisheries Society, Bethesda, MD.

Rice, J. 2003. Environmental health indicators. Ocean & Coastal Management 46:235-259.

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

Rich, W. H. 1920. Early history and seaward migration of chinook salmon in the Columbia and Sacramento Rivers. U.S. Burea of Fisheries Bulletin 37:2-73.

Ricketts, E. F., J. Calvin, J. W. Hedgpeth, and D. W. Phillips. 1985. Between Pacific Tides. 5th edition. Stanford University Press, Standford, CA.

Ricklefs, R. E., and G. Miller. 2000. Ecology, 4th edition. W. H. Freeman, New York.

Romanuk, T. N., and C. D. Levings. 2003. Associations between arthropods and the supralittoral ecotone: Dependence of aquatic and terrestrial taxa on riparian vegetation. Environmental Entomology 32:1343-1353.

Romanuk, T. N., and C. D. Levings. 2005. Stable isotope analysis of trophic position and terrestrial vs. marine carbon sources for juvenile Pacific salmonids in nearshore marine habitats. Fisheries Management and Ecology 12:113-121.

Roni, P., editor. 2005. Monitoring Stream and Watershed Restoration. American Fisheries Society, Bethesda, MD.

Ruckelshaus, M. H., K. P. Currens, W. H. Graeber, R. R. Fuerstenberg, K. Rawson, N. J. Sands, and J. B. Scott. 2006. Independent populations of Chinook salmon in Puget Sound. NOAA Technical Memorandum NMFS-NWFSC-78, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Washington, D.C., USA.

Ruckelshaus, M. H., and M. M. McClure, editors. 2007. Sound Science: Synthesizing ecological and socioeconomic information about the Puget Sound ecosystem. U.S. Department of Commerce, National Oceanic and Atmospheric Administration (NMFS), Northwest Fisheries Science Center, Seattle, Washington.

Ruggerone, G., and F. Goetz. 2004. Survival of Puget Sound chinook salmon (Oncorhynchus tshawytscha) in response to climate-induced competition with pink salmon (*O. gorbuscha*). Canadian Journal of Fisheries and Aquatic Sciences 61:1756-1770.

Ruggerone, G., T. S. Nelson, J. Hall, and E. Jeanes. 2006. Habitat utilization, migration, timing, growth, and diet of juvenile Chinook salmon in the Duwamish River Estuary. Natural Resource Consultants, Inc., Seattle, Washington.

Ruggerone, G., and J. L. Nielsen. 2004. Evidence for competive dominance of pink salmon (*Oncorhynchus gorbuscha*) over other salmonids in the North Pacific Ocean. Reviews in Fish Biology and Fisheries 14:371-390.

Ruiz, G. M., P. W. Fofonoff, J. T. Carlton, M. J. Wonham, and A. H. Hines. 2000. Invasion of coastal marine communities in North America: Apparent patterns, processes, and biases. Annual Review of Ecology and Systematics 31:481-531.

Scharf, F. S., F. Juanes, and M. Sutherland. 1998. Inferring ecological relationships from the edges of scatter diagrams: Comparison of regression techniques. Ecology 79:448-460.

Schmitt, C., J. Schweigert, and T. P. Quinn. 1994. Anthropogenic influences on fish populations of the Georgia Basin. Pages 218-255 *in* R. C. H. Wilson, R. J. Beamish, F. Aitkens, and J. Bell, editors. Review of the marine environment and biota of Strait of Georgia, Puget Sound and Juan de Fuca Strait: Proceedings of the BC/Washington symposium on the marine environment, January 13 & 14, 1994. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1948.

Scholz, N. L., N. K. Truelove, B. L. French, B. A. Berejikian, T. P. Quinn, E. Casillas, and T. K. Collier. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 57:1911 - 1918.

Shaefer, M. B. 1936. Contributions to the life history of the surf smelt (*Hypomesus pretiosus*) in Puget Sound. Washington Department of Fisheries, Biological Report 35-B. Olympia, Washington.

Shen, G. T. 1995. Panel findings and recommendations. Based on the first comprehensive review of the Puget Sound Ambient Monitoring Program. Puget Sound Water Quality Authority, Olympia, Washington.

Sibert, J. R. 1979. Detritus and juvenile salmon production in the Nanaimo estuary: II. Meiofauna available as food to juvenile chum salmon (*Oncorhynchus keta*). Journal of the Fisheries Research Board of Canada 36:497-503.

Simenstad, C. A. 1983. The ecology of estuarine channels of the Pacific Northwest: A community profile. FWS/OBS-83/05, US Fish and Wildlife Service Biological Services Program.

Simenstad, C. A., K. L. Fresh, and E. O. Salo. 1982. The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. Pages 343-364 *in* V. S. Kennedy, editor. Esuarine Comparisons. Academic Publishers, New York.

Simenstad, C. A., R. Garono, T. Labbe, A. Mortimer, R. Robertson, C. Weller, S. Todd, J. Toft, J. Burke, D. Finlayson, J. Coyle, M. Logsdon, and C. Russell. 2007. Assessment of Intertidal Eelgrass Habitat Landscapes for Threatened Salmon in the Hood Canal and Eastern Strait of Juan de Fuca, Washington State. Report for the Point No Point Treaty Council. Kingston, Washington.

Simenstad, C. A., D. A. Jay, and C. R. Sherwood. 1992. Impacts of watershed management on land-margin ecosystems: the Columbia River Estuary as a case study. Pages 266-306 *in* R. Naiman, editor. Watershed management : balancing sustainability and environmental change. Springer-Verlag, New York.

Simenstad, C. A., M. Logsdon, K. Fresh, H. Shipman, M. Dethier, and J. Newton. 2006. Conceptual model for assessing restoration of Puget Sound nearshore ecosystems. Technical Report 2006-03. Puget Sound Nearshorp Partnership, Seattle.

Simenstad, C. A., B. S. Miller, C. F. Nyblade, K. Thornburgh, and L. J. Bledsoe. 1979. Food web relationships of Northern Puget Sound and the Straight of Juan de Fuca. EPA 600/7-79-259, Office of Engineering and Technology, Office of Research and Development, U.S. Environmental Protection Agency, Washington, D.C.

Simenstad, C. A., and E. O. Salo. 1982. Foraging success as a determinant of estuarine and nearshore carrying capacity of juvenile chum salmon (*Oncorhynchus keta*) in Hood Canal, Washington. Pages 21-37 *in* B. R. Melteff and R. A. Neve, editors. Proceedings of the North Pacific Aquaculture Symposium, 18-27 August 1980. Anchorage, Alaska and Newport, Oregon. Alaska Sea Grant Rep. 82 2. University of Alaska, Fairbanks, AK.

Simenstad, C. A., C. D. Tanner, R. M. Thom, and L. L. Conquest. 1991. Estuarine Habitat Assessment Protocol. Report to the U.S. Environmental Protection Agency, Region 10. EPA 910/9-91-037, Fisheries Research Institute, University of Washington, Seattle,WA.

Simon, T. P., editor. 1999. Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities. CRC, Boca Raton, FL.

Skalski, J. R. 2005. Monitoring the Skagit River estuary by estimating cumulative fish-days of usage. Draft report prepared for the Watershed Program of the Northwest Fisheries Science Center, NOAA. University of Washington, Seattle, Washingtn.

Slobodkin, L. B. 1994. The connection between single species and ecosystems. Pages 75-87 *in* D. W. Sutcliffe, editor. Water Quality and Stress Indicators: Linking Levels of Organization. Freshwater Biological Association, Ambleside, UK.

Smith, E. V., and M. G. Anderson. 1922. A preliminary biological survey of the Skagit and Stillaguamish rivers. University of Washington School of Fisheries, Seattle, Washington.

Smyder, E. A., and K. L. M. Martin. 2002. Temperature effects on egg survival and hatching during the extended incubation period of California grunion, *Leuresthes tenuis*. Copeia 2:313-320.

Snover, A. K., P. W. Mote, L. Whitely Binder, A. F. Hamlet, and N. J. Mantua. 2005. Uncertain Future: Climate Change and its Effects on Puget Sound. A Report for the Puget Sound Action Team by the Climate Impacts Group (Center for Science in the Earth System, Joint Institute for the Study of the Atmosphere and Oceans. University of Washington, Seattle, WA.

Sobocinski, K. L. 2003. The Impact of Shoreline Armoring on Supratidal Beach Fauna of Central Puget Sound. MS Thesis. University of Washington, Seattle.

Sogard, S. M. 1997. Size-selective mortality in the juvenile stages of teleost fishes: a review. Bulletin of Marine Science 60:1129-1157.

SSPS (Shared Strategy for Puget Sound). 2005. Puget Sound Salmon Recovery Plan.

Steneck, R. S., and J. T. Carlton. 2001. Human alterations of marine communities: Students beware! Pages 445-468 *in* M. D. Bertness, S. D. Gaines, and M. E. Hay, editors. Marine Community Ecology. Sinauer Associates, Inc., Sunderland, MA.

Stick, K. C. 2005. 2004 Washington State Herring Stock Status Report. Washington Department of Fish and Wildlife, Olympia, Washington.

Stickney, R. R. 1989. Flagship. A History of Fisheries at the University of Washington. Kendall/Hunt Publishing Company, Dubuque, Iowa.

Stober, Q. J., and E. O. Salo. 1973. Ecological studies of the proposed Kiket Island nuclear power site. FRI-UW-7304, University of Washington, Seattle, WA.

Stouder, D. J., P. A. Bisson, and R. J. Naiman, editors. 1997. Pacific Salmon and Their Ecosystems - Status and Future Options. Chapman and Hall, New York.

Strickland, R. M. 1983. The Fertile Fjord: Plankton in Puget Sound. University of Washington Press, Seattle, WA. Available online at: http://www.wsg.washington.edu:16080/publications/online/fjord/fjord.html.

Sturtevant, W. C., editor. 1996. Handbook of North American Indians. Vol. 7, Nothwest Coast. Smithsonian Institution, Washington, D.C.

Sweeting, R., R. J. Beamish, and C. M. Neville. 2003a. Juvenile Salmon in Puget Sound and the Strait of Georgia. Puget Sound Research 2003, Proceedings.

Sweeting, R., R. J. Beamish, D. J. Noakes, and C. M. Neville. 2003b. Replacement of wild coho salmon by hatchery-reared coho salmon in the Strait of Georgia over the past three decades. North American Journal of Fisheries Management 23:492-502.

Sweeting, R., R. J. Beamish, and C. A. Cooper. 2007. Comparison of juvenile salmon diets in the Strait of Georgia and Puget Sound 1997-2006. Proceedings of the 2007 Georgia Basin Puget Sound Research Conference.

Syvitski, J. P. M., D. C. Burrell, and J. M. Skei, editors. 1987. Fjords: Processes and Products. Springer-Verlag, New York.

Taylor, J. E. 1999. Making Salmon: An Environmental History of the Northwest Fisheries Crisis. University of Washington Press, Seattle, Washington.

TBI (The Bay Institute). 2003. The Bay Institute Ecological Scorecard: San Francisco Bay Index 2003. The Bay Institute, San Francisco, CA.

Thom, R. M., and D. K. Shreffler. 1994. Shoreline Armoring Effects on Coastal Ecology and Biological Resources in Puget Sound, Washington. Report 94-80. Washington Department of Ecology, Olympia, WA.

Thompson, W. F., and associates. 1936. The spawning of the silver smelt, Hypomesus pretiosus. Ecology 17:158-168.

Thomson, J. D., G. Weiblen, B. A. Thomson, S. Alfaro, and P. Legendre. 1996. Untangling multiple factors in spatial distributions: Lilies, gophers, and rocks. Ecology 77:1698-1715.

Thorpe, J. E. 1994. Salmonid fishes and the estuarine environment. Estuaries 17:76-93.

Toft, J. D., C. A. Simenstad, J. R. Cordell, C. Young, and V. A. Stamatiou. 2004. Fish distribution, abundance, and behavior along city shoreline types in Puget Sound. North American Journal of Fisheries Management 27:465-480.

Toft, J. D., C. A. Simenstad, J. R. Cordell, C. Young, and V. A. Stamatiou. 2007. Fish distribution, abundance, and behavior along city shoreline types in Puget Sound. North American Journal of Fisheries Management 27:465-480.

USCB (United States Census Bureau). 1995. Historical population counts, 1900 to 1990, for all counties in Washington. http://www.census.gov/population/cencounts/wa190090.txt. U.S. Census Bureau.

USEPA (United States Environmental Protection Agency). 2002. Summary of biological assessment programs and biocriteria development for states, tribes, territories, and interstate commissions: streams and wadeable rivers. EPA-822-R-02-048. U.S. Environmental Protection Agency.

USFWS (United States Fish and Wildlife Service). 1999. Endangered and threatened wildlife and plants: determination of threatened status for bull trout in the coterminous United States. Federal Register volume 64, number 210, pages 58909-58933. Washington, D.C.

USGAO (United States Government Accounting Office). 2005. Environmental information: Status of programs that support ecological indicators. GAO-05-376. United States Government Accounting Office, Washington, DC.

Valiela, I. 1995. Marine Ecological Processes. Second edition, 2nd edition. Springer-Verlag, New York.

Valiela, I., and P. Martinetto. 2007. Changes in bird abundance in eastern North America: Urban sprawl and global footprint? Bioscience 57:360-370.

Verity, P. G., V. Smetacek, and T. J. Smayda. 2002. Status, trends and the future of the marine pelagic ecosystem. Environmental Conservation 29:207-237.

Wahl, T. R., S. M. Speich, D. A. Manuwal, K. V. Hirsch, and C. Miller. 1981. Marine bird populations of the Strait of Juan de Fuca, Strait of Georgia, and adjacent waters in 1978 and 1979. U.S. Environmental Protection Agency, DOC/EPA Interagency Energy/Environment R&D Program Report. EPA/600/f-81-156.

Walters, C. 1997. Information requirements for salmon management. Pages 61-68 *in* D. J. Stouder, P. A. Bisson, and R. J. Naiman, editors. Pacific Salmon and Their Ecosystems: Status and Future Options. Chapman and Hall, New York.

WDFW (Washington Department of Fish and Wildlife). 2006. 2007 – 2009 biennium strategic plan budget submittal. Washington Department of Fish and Wildlife, Olympia, Washington.

WDFW, and PSTT (Washington Department of Fish and Wildlife and Puget Sound Treaty Tribes). 2004. Puget Sound Chinook Salmon Hatcheries Resource Management Plan. Washington Department of Fish and Wildlife, Olympia, Washington.

WDNR (Washington Department of Natural Resources). 2001. Washington State ShoreZone Inventory. Washington Department of Natural Resources, Nearshore Habitat Program, Olympia, Washington.

Webb, W., S. Szarek, W. Lauenroth, R. Kinerson, and M. Smith. 1978. Primary productivity and water use in native forest, grassland, and desert ecosystems. Ecology 59:1239-1247.

Weber, E. D., and K. D. Fausch. 2003. Interactions between hatchery and wild salmonids in streams: differences in biology and evidence for competition. Canadian Journal of Fisheries and Aquatic Sciences 60:1018-1036.

Weins, J. A., J. F. Addicott, T. J. Case, and J. Diamond. 1986. The importance of spatial and temporal scale in ecological investigations. Pages 145-153 *in* J. Diamond and T. J. Case, editors. Community Ecology. Harper & Row, New York.

Weir, G. E. 2001. An Ocean in Common: American Naval Officers, Scientists, and the Ocean Environment. Texas A & M Press, College Station.

Weitkamp, L. A., T. C. Wainwright, G. C. Bryant, G. B. Milner, D. J. Teel, R. G. Kope, and R. S. Waples. 1995. Status Review of Coho Salmon from Washington, Oregon, and California. Tech. Memo. NMFS-NWFSC-24, U.S. Dept. Commer., NOAA, Seattle, WA.

Welch, D. W. 1997. Anatomical specialization in the gut of Pacific salmon (*Oncorhynchus*): Evidence for oceanic limits to salmon production? Canadian Journal of Zoology-Revue Canadienne De Zoologie 75:936-942.

Welch, D. W., and T. R. Parsons. 1993. Delta super(13)C- delta super(15)N values as indicators of trophic position and competetive overlap for Pacific salmon. Fisheries Oceanography 2:11-23.

Wertheimer, A. C., W. W. Smoker, T. L. Joyce, and W. R. Heard. 2001. Comment: A review of the hatchery programs for pink salmon in Prince William Sound and Kodiak Island, Alaska. Transactions of the American Fisheries Society 130:712-720.

West, J. E. 1997. Protection and Restoration of Marine Life in the Inland Waters of Washington State. Puget Sound/Georgia Basin Environmental Report Series No.6.

White, R. 1992. Land Use, Environment, and Social Change: The Shaping of Island County, Washington. University of Washington Press, Seattle, Washington.

White, R. J., J. R. Karr, and W. Nehlsen. 1995. Better roles for fish stocking in aquatic resource management. American Fisheries Society Symposium 15:527-547.

Williams, G. D., and R. M. Thom. 2001. Marine and estuarine shoreline modification issues: white paper submitted to the Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Wilson, R. C. H., R. J. Beamish, F. Aitkens, and J. Bell. 1994. Review of the Marine Environment and Biota of Strait of Georgia, Puget Sound, and Juan de Fuca Strait. Proceedings of the BC/Washington Symposium on the Marine Environment, January 13 & 14, 1994. Canadian Technical Report of Fisheries and Aquatic Sciences 1948:398.

Wingert, R. C., and B. S. Miller. 1979. Distributional Analysis of Nearshore and Demersal Fish Species Groups and Nearshore Fish Habitat Associations in Puget Sound. Univ. Washington, Fish. Res. Inst. FRI-UW-7901. University of Washington, Seattle, WA.

Wissmar, R., and C. A. Simenstad. 1988. Energetic constraints of juvenile chum salmon (*Oncorhynchus keta*) migrating in estuaries. Canadian Journal of Fisheries and Aquatic Sciences 45:1555-1560.

Wissmar, R., and C. A. Simenstad. 1998. Variability of estuarine and riverine ecosystem productivity for supporting Pacific salmon. Pages 253-301 *in* G. R. McMurry and R. J. Bailey, editors. Change in Pacific Northwest coastal ecosystems. NOAA Coastal Ocean Program, Decision Analysis Series No. 11, NOAA Coastal Ocean Office, Silver Spring, Maryland.

WSOFM. 2005. Washington State Data Book. Washington State Office of Financial Management Forecasting Division, Olympia Washington.

Yap-Chiongco, J. V. 1941. Hypomesus pretiosus, its development and early life history. Ph.D. dissertation. University of Washington, Seattle, Washington.

Zabel, R. W., and S. Achord. 2004. Relating size of juveniles to survival within and among populations of Chinook salmon. Ecology 85:795-806.

Zabel, R. W., and J. G. Williams. 2002. Selective mortality in chinook salmon: What is the role of human disturbance? Ecological Applications 12:173-183.

Zar, J. H. 1996. Biostatistical analysis. 3rd edition. Prentice-Hall, Englewood Cliffs, NJ.

Table A1. Taxonomic codes used in the aerial survey (SPPCODE; DEFINITION) and in Chapter 5. "Group" is the taxonomic designation used in the analysis. Diet codes: o = omnivore; h = herbivore; c = carnivore.

WDFW Species Code	Definition	Family	Group	Diet	Primary Food	Feeding Behaviour	Feeding Mode
AMCO	American Coot	RALLIDAE	Rails	o	vegetation	dabble	dabbling herbivore
AMWI	American Wigeon	ANATIDAE	Dabbling Ducks	o	vegetation	dabble	dabbling herbivore
ANMU	Ancient Murrelet	ALCIDAE	Auks	c	invertebrates	surface dive	diving invertivore
BAEA	Bald Eagle	ACCIPITRIDAE	Raptors	c	fish	capture/scavenge	opportunistic carnivore
BAGO	Barrows Goldeneye	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
BBPL	Black-Bellied Plover	CHARADRIIDAE	Shorebirds	o	invertebrates	peck	pecking invertivore
BEKI	Belted Kingfisher	ALCEDINIDAE	Kingfishers	c	fish	plunge dive	diving piscivore
BLBR	Black Brant	ANATIDAE	Geese	h	vegetation	grazing	grazing herbivore
BLOY	Black Oystercatcher	HAEMATOPODIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
BLSC	Black Scoter	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
BLTU	Black Turnstone	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
BOGU	Bonapartes Gull	LARIDAE	Small Gulls	c	fish	plunge dive/surface-sieze	diving piscivore
BRCO	Brandts Cormorant	PHALACROCORACIDAE	Cormorants	c	fish	surface dive	diving piscivore
BRPE	Brown Pelican	PELECANIDAE	Pelicans	c	fish	plunge dive	diving piscivore
BTPI	Band-Tailed Pigeon	COLUMBIDAE	Pigeons	h	vegetation	peck	pecking herbivore
BUFF	Bufflehead	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore

BWTE	Blue-Winged Teal	ANATIDAE	Dabbling Ducks	o	vegetation	dabble	dabbling herbivore
CAGO	Canada Goose	ANATIDAE	Geese	h	vegetation	grazing	grazing herbivore
CAGU	California Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic carnivore
CANV	Canvasback	ANATIDAE	Diving Ducks	o	vegetation	surface dive	diving omnivore
CATE	Caspian Tern	LARIDAE	Terns	c	fish	plunge dive	diving piscivore
COGO	Common Goldeneye	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
COLO	Common Loon	GAVIIDAE	Loons	c	fish	surface dive	diving piscivore
COME	Common Merganser	ANATIDAE	Diving Ducks	c	fish	surface dive	diving piscivore
COMU	Common Murre	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
CORA	Common Raven	CORVIDAE	Crows	o	carrion	capture/scavenge	opportunistic omnivore
COTE	Common Tern	LARIDAE	Terns	c	fish	plunge dive	diving piscivore
DCCO	Double-Crested Cormorant	PHALACROCORACIDAE	Cormorants	c	fish	surface dive	diving piscivore
DUNL	Dunlin	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
GADW	Gadwall	ANATIDAE	Dabbling Ducks	o	vegetation	dabble	dabbling herbivore
GBHE	Great Blue Heron	CICONIIFORMES	Herons	c	fish	wade	pecking piscivore
GRSC	Greater Scaup	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
GRYE	Greater Yellowlegs	SCOLOPACIDAE	Shorebirds	c	invertebrates	wade	pecking invertivore
GWGU	Glaucous-Winged Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
GWTE	Green-Winged Teal	ANATIDAE	Dabbling Ducks	o	vegetation	dabble	dabbling omnivore
HADU	Harlequin Duck	ANATIDAE	Diving Ducks	c	invertebrates	surface dive	diving invertivore

HEGU	Herring Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic carnivore
HOGR	Horned Grebe	PODICIPEDIDAE	Grebes	c	invertebrates	surface dive	diving carnivore
HOME	Hooded Merganser	ANATIDAE	Diving Ducks	c	fish	surface dive	diving piscivore
HRGU	Heermann's Gull	LARIDAE	Large Gulls	o	fish	capture/scavenge	opportunistic carnivore
KILL	Killdeer	CHARADRIIDAE	Shorebirds	o	invertebrates	peck	pecking invertivore
LESP	Leachs Storm-Petrel	HYDROBATIDAE	Seabirds	c	fish	plunge dive	diving piscivore
MAGO	Godwit	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
MALL	Mallard	ANATIDAE	Dabbling Ducks	o	vegetation	dabble	dabbling omnivore
MAMU	Marbled Murrelet	ALCIDAE	Auks	c	fish	dive	diving piscivore
MEGU	Mew Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
NOFL	Northern Flicker	PICIDAE	Woodpeckers	o	invertebrates	peck	pecking invertivore
NOFU	Northern Fulmar	PROCELLARIIDAE	Seabirds	c	fish	surface dive	diving piscivore
NOHA	Northern Harrier	ACCIPITRIDAE	Raptors	c	mammals	pounce	pouncing carnivore
NOPI	Northern Pintail	ANATIDAE	Dabbling Ducks	o	invertebrates	dabble	diving invertivore
NWCR	Northwestern Crow	CORVIDAE	Crows	o	invertebrates	capture/scavenge	opportunistic omnivore
OLDS	Oldsquaw	ANATIDAE	Diving Ducks	o	invertebrates	dive	diving invertivore
OSPR	Osprey	ACCIPITRIDAE	Raptors	c	fish	plunge dive	diving piscivore
PALO	Pacific Loon	GAVIIDAE	Loons	c	fish	surface dive	diving piscivore
PBGR	Pied-Billed Grebe	PODICIPEDIDAE	Grebes	c	invertebrates	surface dive	diving invertivore
PECO	Pelagic Cormorant	PHALACROCORACIDAE	Cormorants	c	fish	surface dive	diving piscivore

PEFA	Peregrine Falcon	FALCONIDAE	Raptors	c	birds	pounce	pouncing carnivore
PIGU	Pigeon Guillemot	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
POJA	Pomarine Jaeger	LARIDAE	Seabirds	c	fish	capture/scavenge	opportunistic carnivore
RBGU	Ring-Billed Gull	LARIDAE	Large Gulls	o	fish	capture/scavenge	opportunistic omnivore
RBME	Red-Breasted Merganser	ANATIDAE	Diving Ducks	c	fish	surface dive	diving piscivore
REPH	Red Phalarope	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
RHAU	Rhinoceros Auklet	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
RLHA	Rough-Legged Hawk	ACCIPITRIDAE	Raptors	c	mammals	pounce	pouncing carnivore
RNGR	Red-Necked Grebe	PODICIPEDIDAE	Grebes	c	fish	surface dive	diving carnivore
RNPH	Red-Necked Phalarope	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
RODO	Rock Dove	COLUMBIDAE	Pigeons	o	vegetation	peck	pecking herbivore
RTHA	Red-Tailed Hawk	ACCIPITRIDAE	Raptors	c	mammals	pounce	pouncing carnivore
RTLO	Red-Throated Loon	GAVIIDAE	Loons	c	fish	surface dive	diving piscivore
RUDU	Ruddy Duck	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
RUTU	Ruddy Turnstone	SCOLOPACIDAE	Shorebirds	o	invertebrates	peck	pecking invertivore
SAND	Sanderling	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
SCAU	Unidentified Scaup	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
SEPL	Semi-Palmated Plover	CHARADRIIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore

SNGO	Snow Goose	ANATIDAE	Geese	h	vegetation	grubbing	grazing herbivore
SOSH	Sooty Shearwater	PROCELLARIIDAE	Seabirds	c	fish	dive/surface-sieze	dipping carnivore
SPSA	Spotted Sandpiper	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
SURF	Surfbird	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
SUSC	Surf Scoter	ANATIDAE	Diving Ducks	c	invertebrates	surface dive	diving invertivore
TEAL	Unidentified Teal	ANATIDAE	Dabbling Ducks	o	vegetation	dabble	dabbling herbivore
THGU	Thayers Gull	LARIDAE	Large Gulls	o	fish	dip/scavenge	opportunistic omnivore
TRSW	Trumpeter Swan	ANATIDAE	Swans	o	vegetation	graze/dabble	grazing herbivore
TUPU	Tufted Puffin	ALCIDAE	Puffins	c	invertebrates	surface dive	diving piscivore
TURN	Unidentified Turnstone	SCOLOPACIDAE	Shorebirds	o	invertebrates	peck	pecking invertivore
TUSW	Tundra Swan	ANATIDAE	Swans	o	vegetation	graze/dabble	grazing herbivore
TUVU	Turkey Vulture	CATHARTIDAE	Vultures	c	mammals	scavenge	opportunistic carnivore
UBWG	Unidentified Black-Wing Tip Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
UCTE	Unidentified Commic Tern	LARIDAE	Terns	c	fish	plunge dive	diving piscivore
ULAC	Unidentified Large Alcid	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
ULGR	Unidentified Large Grebe	PODICIPEDIDAE	Grebes	c	invertebrates	surface dive	diving carnivore
ULGU	Unidentified Large Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
ULSD	Unidentified Large Shorebirds	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore

UMAC	Unidentified Medium Alcid	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
UMSD	Unidentified Medium Shorebirds	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
UNAC	Unidentified Alcid	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
UNCO	Unidentified Cormorant	PHALACROCORACIDAE	Cormorants	c	fish	surface dive	diving piscivore
UNDD	Unidentified Diving Duck	ANATIDAE	Diving Ducks	o	invertebrates	surface dive	diving invertivore
UNDO	Unidentified Dowitcher	SCOLOPACIDAE	Shorebirds	o	invertebrates	peck	pecking invertivore
UNDU	Unidentified Duck	ANATIDAE	Ducks	o	invertebrates	dive/dabble	dabbling/diving omnivore
UNGO	Unidentified Goldeneye	ANATIDAE	Diving Ducks	o	invertebrates	dive	diving invertivore
UNGR	Unidentified Grebe	PODICIPEDIDAE	Grebes	c	invertebrates	dive	diving invertivore
UNGU	Unidentified Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
UNJA	Unidentified Jaeger	LARIDAE	Seabirds	c	fish	capture/scavenge	opportunistic carnivore
UNLO	Unidentified Loon	GAVIIDAE	Loons	c	fish	surface dive	diving piscivore
UNME	Unidentified Merganser	ANATIDAE	Diving Ducks	c	fish	surface dive	diving piscivore
UNML	Unidentified Murrelet	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
UNMU	Unidentified Murre	ALCIDAE	Auks	c	fish	surface dive	diving piscivore
UNPD	Unidentified Dabbling/Pond Duck	ANATIDAE	Dabbling Ducks	o	invertebrates	dabble	dabbling omnivore
UNPH	Unidentified Phalarope	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
UNSB	Unidentified Sea Bird		Seabirds	c	fish	dive/surface-sieze	diving piscivore

253

UNSC	Unidentified Scoter	ANATIDAE	Diving Ducks	o	invertebrates	dive	diving invertivore
UNSD	Unidentified Sandpiper	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
UNSH	Unidentified Shearwater	PROCELLARIIDAE	Seabirds	c	fish	dive/surface-sieze	pecking carnivore
UNSP	Unidentified Storm-Petrel	HYDROBATIDAE	Seabirds	c	fish	surface-sieze	pecking carnivore
UNSW	Unidentified Swan	ANATIDAE	Swans	o	vegetation	graze/dabble	grazing herbivore
UNTE	Unidentified Tern	LARIDAE	Terns	c	fish	dive	diving piscivore
UNYE	Unidentified Yellowlegs	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
USAC	Unidentified Small Alcid	ALCIDAE	Auks	c	fish	dive	diving piscivore
USGR	Unidentified Small Grebe	PODICIPEDIDAE	Grebes	c	invertebrates	dive	dabbling omnivore
USGU	Unidentified Small Gull	LARIDAE	Small Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
USSD	Unidentified Small Shorebirds	CHARADRIIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
WATA	Wandering Tattler	SCOLOPACIDAE	Shorebirds	c	invertebrates	peck	pecking invertivore
WEGR	Western Grebe	PODICIPEDIDAE	Grebes	c	fish	dive	diving piscivore
WEGU	Western Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
WGGU	Western X Glaucous Winged Gull	LARIDAE	Large Gulls	o	invertebrates	capture/scavenge	opportunistic omnivore
WHIM	Whimbrel	SCOLOPACIDAE	Shorebirds	o	invertebrates	peck	pecking invertivore
WWSC	White-Winged Scoter	ANATIDAE	Diving Ducks	c	invertebrates	dive	diving invertivore

254

Appendix B

SIMPER results by season and basin
(tables presented in order from north to south)

Table B1. Percent taxon contribution (based on relative frequency of occurrence) to
within group assemblage similarity during summer and winter in the
Rosario sub-basin (1993-1996).

Rosario Sub-basin					
Summer Average similarity across years: 90			**Winter** Average similarity across years: 92		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	29	29	Diving Ducks	25	25
Auks	14	42	Large Gulls	17	42
Herons	13	55	Dabbling Ducks	9	51
Cormorants	6	62	Cormorants	8	59
Diving Ducks	6	68	Loons	8	66
Small Gulls	5	73	Grebes	7	74
Terns	4	77	Auks	7	81
Crows	4	81	Geese	6	87
Loons	3	85	Raptors	3	90
Geese	3	88			
Raptors	3	91			

Table B2. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the San Juan sub-basin (1993-1996).

San Juan Sub-basin					
Summer			**Winter**		
Average similarity across years: 90			Average similarity across years: 92		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	28	28	Diving Ducks	24	24
Auks	23	51	Large Gulls	21	45
Cormorants	8	60	Auks	14	60
Herons	8	67	Cormorants	10	69
Small Gulls	5	72	Loons	7	77
Diving Ducks	5	77	Grebes	7	83
Crows	5	82	Raptors	3	86
Raptors	4	86	Herons	3	89
Shorebirds	4	90	Crows	2	91
Seabirds	4	94	Diving Ducks	24	24
Large Gulls	28	28			

Table B3. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Straits sub-basin (1993-1996).

Straits Sub-basin					
Summer Average similarity across years: 89			**Winter** Average similarity across years: 89		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	29	29	Diving Ducks	27	27
Auks	22	51	Large Gulls	18	45
Cormorants	10	61	Auks	14	59
Diving Ducks	8	70	Grebes	8	67
Herons	5	75	Cormorants	7	74
Shorebirds	5	80	Loons	6	81
Seabirds	4	83	Dabbling Ducks	5	85
Crows	3	86	Herons	3	88
Raptors	3	89	Shorebirds	3	91
Small Gulls	2	91			

Table B4. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Whidbey sub-basin (1993-1996).

Whidbey Sub-basin					
Summer Average similarity across years: 87			Winter Average similarity across years: 89		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	26	26	Diving Ducks	23	23
Diving Ducks	10	37	Large Gulls	19	42
Terns	10	47	Grebes	10	52
Herons	10	57	Dabbling Ducks	9	61
Auks	7	64	Loons	7	68
Crows	5	69	Cormorants	7	75
Cormorants	5	74	Auks	5	80
Small Gulls	5	79	Herons	4	84
Shorebirds	3	83	Shorebirds	3	87
Kingfishers	3	86	Small Gulls	3	89
Geese	3	89	Crows	3	92
Raptors	3	92			

Table B5. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the North Main sub-basin (1993-1996).

North Main Sub-basin					
Summer Average similarity across years: 87			**Winter** Average similarity across years: 89		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	30	30	Diving Ducks	26	26
Auks	22	52	Large Gulls	17	43
Herons	9	61	Auks	9	53
Small Gulls	8	69	Cormorants	8	61
Crows	5	74	Grebes	8	68
Terns	5	79	Dabbling Ducks	6	74
Cormorants	5	84	Loons	6	80
Diving Ducks	4	88	Herons	4	84
Shorebirds	3	91	Crows	3	87
			Shorebirds	3	90

Table B6. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the West Main sub-basin (1993-1996).

West Main Sub-basin					
Summer Average similarity across years: 86			Winter Average similarity across years: 87		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	31	31	Diving Ducks	27	27
Herons	13	44	Large Gulls	19	47
Crows	11	54	Cormorants	10	57
Auks	10	64	Grebes	10	67
Cormorants	7	71	Dabbling Ducks	9	76
Geese	5	76	Auks	5	81
Raptors	4	80	Loons	5	86
Terns	3	83	Small Gulls	3	89
Diving Ducks	3	87	Crows	3	91
Kingfishers	3	90			
Small Gulls	3	92			

Table B7. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the East Main sub-basin (1993-1996).

East Main Sub-basin					
Summer			Winter		
Average similarity across years: 80			Average similarity across years: 88		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	47	47	Diving Ducks	28	28
Terns	9	56	Large Gulls	25	53
Herons	9	65	Cormorants	11	63
Crows	7	72	Grebes	11	74
Auks	7	79	Auks	6	80
Geese	4	83	Dabbling Ducks	4	84
Shorebirds	4	87	Loons	3	87
Small Gulls	4	90	Geese	3	90

Table B8. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Hood Canal sub-basin (1993-1996).

Hood Canal Sub-basin					
Summer			**Winter**		
Average similarity across years: 82			Average similarity across years: 88		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	28	28	Diving Ducks	27	27
Herons	13	42	Large Gulls	17	44
Auks	12	54	Grebes	11	55
Crows	9	62	Dabbling Ducks	8	63
Diving Ducks	5	68	Cormorants	6	69
Kingfishers	5	73	Loons	6	75
Cormorants	4	77	Herons	5	80
Raptors	4	82	Auks	4	84
Small Gulls	3	85	Geese	3	87
Shorebirds	3	88	Small Gulls	2	90
Geese	3	91	Crows	2	92

Table B9. Percent taxon contribution (based on relative frequency of occurrence)
to within group assemblage similarity during summer and winter in the
South Sound sub-basin (1993-1996).

South Sound Sub-basin					
Summer			Winter		
Average similarity across years: 88			Average similarity across years: 92		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	28	28	Diving Ducks	29	29
Herons	14	42	Large Gulls	18	47
Auks	13	55	Grebes	8	55
Crows	10	66	Cormorants	8	63
Diving Ducks	5	71	Dabbling Ducks	7	70
Terns	4	75	Loons	6	77
Shorebirds	4	79	Auks	5	82
Kingfishers	4	83	Herons	4	87
Seabirds	4	87	Small Gulls	4	90
Cormorants	3	90			
Raptors	3	92			

Appendix C

SIMPER results by season and estuary
(Tables presented in order from north to south)

Table C1. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Skagit River estuary (1993-1996).

Skagit River Estuary					
Summer			**Winter**		
Average similarity across years: 79			Average similarity across years: 83		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	41	41	Diving Ducks	23	23
Herons	17	58	Dabbling Ducks	21	44
Terns	12	70	Large Gulls	19	63
Dabbling Ducks	7	78	Herons	7	69
Diving Ducks	6	83	Shorebirds	6	76
Raptors	5	88	Cormorants	6	82
Shorebirds	4	92	Loons	5	87
			Raptors	5	92

Table C2. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Stillaguamish (1993-1996).

Stillaguamish River Estuary					
Summer			**Winter**		
Average similarity across years: 64			Average similarity across years: 81		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	47	47	Diving Ducks	26	26
Terns	15	62	Large Gulls	18	43
Herons	10	72	Dabbling Ducks	17	60
Dabbling Ducks	7	79	Loons	7	68
Small Gulls	6	85	Herons	5	73
Diving Ducks	5	89	Raptors	5	79
Grebes	3	93	Cormorants	5	84
			Grebes	5	89
			Shorebirds	4	93

Table C3. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Snohomish River estuary (1993-1996).

Snohomish River Estuary					
Summer Average similarity across years: 83			**Winter** Average similarity across years: 81		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	35	35	Large Gulls	28	28
Terns	20	55	Diving Ducks	20	48
Herons	11	65	Grebes	15	63
Geese	9	74	Cormorants	14	77
Cormorants	8	82	Dabbling Ducks	8	86
Raptors	6	87	Loons	5	90
Crows	6	93			

Table C4. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Duwamish River estuary (1993-1996).

Duwamish River Estuary					
Summer			**Winter**		
Average similarity across years: 78			Average similarity across years: 75		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	96	96	Large Gulls	34	34
			Diving Ducks	23	57
			Grebes	23	81
			Cormorants	17	98

Table C5. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Puyallup River estuary (1993-1996).

Puyallup River Estuary					
Summer			**Winter**		
Average similarity across years: 78			Average similarity across years: 80		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	61	61	Large Gulls	31	31
Terns	17	77	Diving Ducks	23	54
Geese	15	93	Grebes	19	73
			Cormorants	14	87
			Herons	6	92

Table C6. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Skokomish River estuary (1993-1996).

Skokomish River Estuary					
Summer Average similarity across years: 64			**Winter** Average similarity across years: 79		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	50	50	Diving Ducks	30	30
Herons	19	68	Large Gulls	17	47
Crows	15	83	Grebes	14	61
Diving Ducks	8	92	Dabbling Ducks	12	73
			Loons	7	80
			Geese	6	86
			Herons	4	90

Table C7. Percent taxon contribution (based on relative frequency of occurrence) to within group assemblage similarity during summer and winter in the Nisqually River estuary (1993-1996).

Nisqually River Estuary					
Summer			Winter		
Average similarity across years: 74			Average similarity across years: 83		
Taxon	Contrib. %	Cumul. %	Taxon	Contrib. %	Cumul. %
Large Gulls	30	30	Diving Ducks	33	33
Auks	16	46	Large Gulls	28	61
Terns	12	58	Dabbling Ducks	11	71
Herons	11	69	Cormorants	8	79
Diving Ducks	8	78	Grebes	7	86
Crows	7	85	Loons	3	90
Shorebirds	6	91	Small Gulls	2	92

VITA

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