The Salmon River Estuary:
Restoring Tidal Inundation and Tracking Ecosystem Response

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Abstract

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The Salmon River estuary offered a unique opportunity to simultaneously evaluate several estuarine restoration projects and make comparisons with a reference, undiked portion of the estuary. Dikes installed in three locations in the estuary during the early 1960s were removed in 1978, 1987 and 1996 creating a "space-for-time substitution" chronosequence. I analyzed marsh community response to assess development state of three recovering marshes and make comparisons with the reference. During spring and summer from 1998-2002, I assessed juvenile salmon habitat development by comparing wild *Oncorhynchus tshawytscha* (Chinook salmon) density, diet composition and modeled growth potential among marshes. I also explored differences in invertebrate abundance and composition among marshes, and determined repeatable and exclusive indicators to marsh age. In my analysis, I applied techniques, such as bioenergetics modeling and indicator analysis, to evaluate ecosystem development. A bioenergetics model was used to compare growth potential with site-specific diet composition and temperature, estuary-specific prey energy, average fish size and calculated consumption rates. Modeled growth potential was positive, ranged from 0.001-0.07 g/g/d, and was unrelated to marsh age. Growth rates approximating the reference were found in the newest restoration site with higher variability. Density and taxonomic richness of benthic macroinvertebrates was related to marsh age, but unrelated to marsh surface insects. Invertebrate communities were also compared among marshes. Benthic communities in the 1996 marsh were different from all other marshes, and insect communities were distinct by site except for several cases when no difference between the reference and the 1978 marsh was found. Using indicator analysis I determined several reliable indicators of marsh age, including Staphylinidae (rove beetles) and Chironomidae larvae (midge flies) in the 1996 marsh. Insect communities were more affected by landscape positioning than benthic communities, but tracking both, and evaluating the key indicators of community response was the most informative for describing and assessing recovery state across marsh conditions. The range of metrics I used to track ecosystem development at the Salmon River estuary provided broad indication as to the state of ecosystem re-development after restoring tidal inundation and revealed differences among the marshes related to landscape position and marsh age.

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Preface

Often referred to as an infant science (Frenkel & Morlan 1990; Palmer et al. 1997) restoration ecology traces its ideological and practical roots to Aldo Leopold and prairie ecosystems in Wisconsin. Leopold (1949) found conservation and restoration incompatible with the idea of land as a commodity, something to be owned. Instead, he believed that land is a community to which we all belong. My views of the land and ecosystems follow Leopold's; a community, a collection of parts which together create a whole that is something more, a system that works through a litany of interactions and relationships, and something not so easily recreated. The science of restoration takes opportunities provided by disturbed and recovering ecosystems to better understand the mechanisms and processes that make systems work in the first place.

Some early and elegant studies in restoration ecology began in response to polluted lakes in the early 1970s. In response to excessive inputs of organic wastes, whole-scale changes of lake ecosystems occurred and amplified algal growth leading to associated problems with over-nutrification. The practical means to restore affected lakes was to divert the sources of organic wastes (Edmondson 1972); but moreover, many ecological questions were raised and answered over the course of these studies.

Mass-balance models were developed to determine algal concentration from nutrient load and determine management criteria. Then model assumptions were tested with whole-lake experimentation (Schindler 1977). Practical and theoretical principles were employed to determine the mechanisms affecting community structure, and only through understanding the eutrophication process could restoration efforts have been as successful as they were in these cases.

Similar concepts may be applied to the restoration of coastal wetlands, which have been heavily impacted by agriculture and development. Policies which dictated impacted wetlands would simply be recreated somewhere else or degraded wetlands would be enhanced, ignoring evident problems from a biodiversity and ecological perspective (Whigham 1999). Formed on a foundation of completely uncertain assumptions regarding our ability to create and remake wetland ecosystems, the policies may have raised public awareness regarding the importance of wetlands and increased support for their protection, but left the impression ecosystems could be made to order. A better understanding of how best to restore, enhance and manage tidal wetlands is needed, especially for their value as critical habitat for key species such as juvenile Chinook salmon.

The Salmon River estuary can serve as an important learning tool in the development of restoration science.

In 1974, the U. S. Congress established Cascade Head Scenic-Research Area (Appendix A: P.L. 93-535) under Forest Service management. Special designation of this area included revitalization and restoration of the Salmon River estuary and associated wetlands, and specific intent to return the estuary to its condition before diking and agricultural use (U. S. Forest Service 1976). In addition to conserving and restoring a beautiful and important marsh, this action presented an ideal situation to test hypotheses surrounding estuarine marsh restoration. Relatively minor agricultural impacts (harvest of marsh vegetation for hay) ceased on the last 27% of undiked marsh, which acts as the reference in this estuarine marsh recovery experiment. Through the restoration activities the U.S. Forest Service created a series of estuarine marshes in varying stages of recovery. Moreover, since the dikes were breached at nine-year intervals and all the dikes were installed within the same short time period (1961-3), diking occurred for varying amounts of time: 17, 26 and 35 years. Duration of diking determines level of subsidence, rate of accretion after restoration and other characteristics of recovered marshes (Frenkel & Morlan 1990). Although recovery of the Salmon River estuarine marshes was purely a restoration program, it aids mitigation and restoration projects alike by filling a void of uncertainty regarding how and if a degraded or diked marsh can be restored to its pre-impact condition, and how success is determined.

Tidally controlled hydrology can be relatively easily reestablished so drained and diked agricultural lands became good prospects for restoration, as one means to recover losses in coastal wetlands (Kusler & Kentula 1989) and rehabilitate depressed salmon populations. However, few studies have looked at the effect of these restoration projects on juvenile salmon populations and none provide the chronosequence of restored habitats available at the Salmon River estuary.

Dr. Robert Frenkel and several graduate students from Oregon State University began research monitoring vegetation, soil and elevation status of the marshes before and after dike breaching began in 1977. Initial restoration changes in plant species, plant composition, marsh elevation, salinity, soil texture, tidal creek geomorphology, and above ground net primary productivity were compared with reference sites and detailed in Mitchell (1981). Frenkel and Morlan (1991) measured marsh surface elevation as 35 cm lower than adjacent controls in 1988. The difference in elevation represents the net effect of a 0.5-1.7 cm/yr accretion rate in low, silty Oregon marshes (Frenkel et al. 1981) and subsidence of diked pasture to compaction and buoyancy loss (Mitchell 1981). Measures of bulk density and soil organic matter in the reference site suggested oxidation, documented in diked wetlands elsewhere (Portnoy 1999; Portnoy & Gilbin 1997), was an unlikely cause of subsidence (Mitchell 1981). Surface elevation largely controls marsh hydrology, vegetation composition and function, inherent elements of estuarine character (Frenkel & Morlan 1990). After restoration, marsh height increased at a rate of 3-7 cm/decade and was dependent upon initial elevation (Frenkel & Morlan 1991). Two years after dike removal, unvegetated soil was the most extensive marsh surface characteristic, but ten years later, primary productivity in the marsh was almost double (2,300 $g/m^2/yr$) the pasture (before dike removal) and an adjacent reference site, suggesting enhanced life support (Frenkel & Morlan 1991). Important refinements in estuarine science were made through these early studies at the Salmon River estuary, including the importance of determination of pre-, post-restoration surface elevations, complete dike removal, reconnection marsh hydrology and long-term monitoring (Frenkel & Morlan 1990, 1991).

It is my attempt with the following research to augment these discoveries and understandings of estuarine marsh recovery. In particular, I am expanding assessment metrics to follow fish and invertebrate communities among marshes and to create more knowledge in regard to the success of restoring estuarine marshes in providing habitat for juvenile Chinook salmon. I hope to better understand the processes involved in marsh recovery, and the mechanisms supporting ecosystem development.

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Dedication

In loving memory

Eddy R. Gory

The spirit of the river

Introduction

The Salmon River estuary, Oregon (45° 01′ N, 123° 58′ W) represents a unique opportunity to enhance and contribute to the science of restoration ecology and apply new understanding to the problem of juvenile *Oncorhynchus tshawyschta* (Chinook salmon) habitat loss.

Problem

Chinook salmon, an immensely important natural resource of the Pacific Northwest, has suffered a devastating fishery decline progressing since the turn of the 20th century. Rapid habitat loss has contributed to this decline, including the loss of estuarine wetlands: an important habitat for juvenile Chinook salmon and among the most severely impacted environments in Oregon.

Juvenile salmon are known to use estuaries during their freshwater-to-the-sea migration to varying degrees as a function of species and life history type (Reimers 1973; Simenstad et al. 1982; Healey 1982a; Levy & Northcote 1982; Iwata & Komatsu 1984; Pearcy et al. 1989; Shreffler et al. 1990; Groot & Margolis 1991; Levings & McDonald 1991; Gray et al. 2002; Miller & Sadro 2003). Juvenile salmon prefer estuarine habitats that are vegetated, heavily channelized, with moderate slope and a range of salinities, low-water refugia, cover (vegetation and large woody debris), and a good prey base (Aitkin 1998). Chinook are thought to use estuarine habitats the most extensively, with chum, coho, sockeye and pink salmon being progressively less marsh-reliant (Simenstad et al. 1982; Healey 1982b). Although, some new evidence reveals wild coho rearing in estuaries (Miller & Sadro 2003), something not previously known; long-term residence is best understood for Chinook.

In Oregon and Washington, 50-80% of salt marshes have been lost, mostly due to diking activities (Oregon Division of State Lands 1972; Washington Department of Ecology 1993). Estuarine marshes in California have suffered a similar fate with some estimates of loss as high as 90% (Denisoff & Movassaghi 1998). Bishop and Morgan (1996) concluded that loss of estuarine marsh was a primary critical habitat issue for juvenile Chinook salmon in 14 of 15 watersheds in Washington.

The Salmon River estuary offers an extraordinary opportunity to assess ecosystem change after landscape-scale disturbances, especially in terms of the mechanisms and processes which support recovery of juvenile Chinook salmon habitat. Marshes in the Salmon River were impacted by dike and tidegate installation and restriction of tidal flow for 17-35 years. Dike and tidegate removal took place at nine-year intervals: in 1978, 1987 and 1996, creating a "space-for-time substitution" which allows for the comparison of marsh conditions in several stages of recovery. In addition, a large portion of the estuarine marsh was never diked, and although used intermittently for pastureland and haying, no agricultural impact has occurred in this marsh since 1974. This marsh is the best estimation of reference conditions.

My five-year study focuses on changing conditions influencing rearing habitat characteristics of juvenile Ocean-type (Healey 1982a) Chinook salmon in these marshes.

Juvenile Chinook salmon are appropriate study subjects because of their extensive use of estuarine wetlands during their seaward migration with mean residence times of 35 days (Bottom et al. 2005), and their integrative nature which combines conditions of habitat quality and reflects effects as growth and survival. The restoration history of the Salmon River enabled tracking of juvenile salmon response to ecosystem recovery essentially over a five to 24-year period. I believe duration of diking and disturbance regimes associated with long-term tidal exclusion to be responsible for the most conspicuous disparities in marsh conditions (Gray et al. 2002). However, I realize differences in marsh position and amount of freshwater input exist and influence system dynamics.

By the early 1960s, 75% of the lower Salmon River marsh was isolated by dikes from tidal inundation and converted to pasture (Frenkel & Morlan 1991; Fig. 1.1). The U. S. Forest Service began restoration activities in 1978 with the partial removal of a dike enclosing 22 ha. The material removed was used to fill a borrow pit created when the dike was built (Frenkel & Morlan 1990). Excavation was used to re-establish major creek connections. In 1987, the entire dike enclosing a 63-ha pasture was removed, and the dike surrounding the marsh restored in 1978 was leveled to historic marsh level (Frenkel & Morlan 1990). In 1996, the final restoration project was completed just west of Highway 101 restoring 135 ha to full tidal inundation; a new cross dike and tidegate were installed to protect adjoining property (Fig. 1.1).

With three marshes in differing stages and conditions of recovery and a reference marsh upon which to make comparisons, the 800-ha Salmon River estuary provides the ideal location to answer questions regarding succession in natural systems, biological and geomorphological marsh response to dike breaching, potential impacts and benefits of restoration activities, and assessment of ecosystem function. As each of these categories represents scores of scientific work and evidence, in this dissertation I will focus on concepts and issues regarding biological response to dike removal and finding appropriate tools for functional assessment.

Study Objective

Estuarine restoration science has significantly progressed in the last seven years with new understandings of the importance of hydrology (Montalto & Steenhuis 2004), elevation (Cornu & Sadro 2002), successional vegetation patterns (Thom et al. 2002), and invasive species (Weis & Weis 2003). The opportunity at the Salmon River is a chance to determine and compare early recovery conditions, characteristics and functions to older marshes and a reference. Lessons garnered from Salmon River research can be applied to a better understanding of the time scale of estuarine marsh recovery and the function of newly restored marshes. The objective is to compare the four marshes of the Salmon River estuary in terms of the following metrics (all essential elements related to juvenile fish habitat): juvenile Chinook diet composition and prey energy content, modeled growth potential, temperature and salinity, and invertebrate abundance and composition. The following manuscripts describe juvenile Chinook habitat development (Chapter 1), modeling juvenile Chinook growth potential (Chapter 2), and exploring patterns in invertebrate abundance and composition (Chapter 3). I followed ecosystem response across a range of recovery stages and applied the science to the problem of juvenile Chinook salmon decline. My objectives were to measure ecosystem response in an effective way and assess estuarine marsh function in terms of invertebrates and fish.

The final discussion puts the work from the Salmon River estuary into the context of habitat conservation and restoration ecology, explores processes involved in estuarine marsh recovery, and provides recommendations for future work.

Experimental Design

To study the effects of tidal restriction and restoring tidal inundation to marshes, I developed a protocol which assessed the state of the recovering and reference wetlands in terms of fish and invertebrate abundance and composition, juvenile Chinook salmon diet composition, prey energy, and modeled growth potential, temperature/salinity regimes and channel sediment organic matter. Data included systematic collections of fish from each of the marshes using a modified fyke net set at high tide and always in two-, or three-day periods each month. All collections were standardized appropriately when necessary; by channel area (fish collections), sampling area or volume (invertebrates), and weight (diet composition and prey energy). Invertebrate communities were compared with count data, as all sample sizes were equal.

The overall design included monthly sampling during the periods of highest juvenile Chinook abundance (March-July) in each of the four marshes. Sampling commenced in April 1998 and was completed in August 2002. Prey energy was determined from readily available invertebrates across the season from 2001-2003.

Statistical analysis of my data involved a range of techniques depending on the properties of the data. In general, Kruskal-Wallis Nonparametric Comparisons of Means was used in cases where our data were not normally distributed. Data transformations were performed in some cases and appropriate statistics were applied, including One-way ANOVA, Multi-Response Permutation Procedure (MRPP) and Nonmetric Multidimensional Scaling (NMS). My comparisons were searching for significant differences in several parameters among reference and restoration marshes. Multivariate statistics were used to illustrate differences in invertebrate communities. NMS was used as the best available ordination technique for assessing differences among ecological communities (McCune & Mefford 1999).

Chapter 1: Contrasting Functional Performance of Juvenile Salmon Habitat in Recovering Wetlands of the Salmon River Estuary, Oregon. *Restoration Ecology* 10(3):514-526.

Introduction

In recent years, ecosystem restoration activities have been undertaken in many coastal watersheds in an effort to remedy biological impoverishment and degradation resulting from such practices as forest clearing, hydrological manipulation, and agricultural and urban-industrial land conversion. Restoration projects typically aim to restore functions (e.g., production, sediment retention, nutrient transformation) lost or diminished when ecosystems are disturbed. The impetus in many cases is conservation of specific habitat types to rehabilitate threatened or endangered species.

In the Pacific Northwest, estuarine marshes are habitats of particular restoration interest. The precipitous decline of many populations of anadromous salmon has lent new urgency to ongoing efforts to restore productive estuarine wetlands lost to decades of diking and filling of intertidal habitats. Several species (and life history types) of juvenile salmon occupy estuarine habitats, and particularly emergent marshes, before completing their seaward migration (Groot & Margolis 1991). Critical questions remain, however, about the mechanisms and rates of marsh restoration that most benefit juvenile salmon. Following patterns in specific parameters indicative of marsh function through time creates functional trajectories, which are assumed to eventually approach reference conditions (Morgan & Short 2002). This paper summarizes results of fish utilization, prey resource, and diet composition of juvenile Chinook salmon from breached-diked wetlands of different ages in the Salmon River estuary, Oregon. Our objective is to determine the functional trajectory of estuarine marsh recovery, and the benefits of early recovery stages to juvenile Chinook salmon and to depressed salmon populations.

Interpreting whether wetland restoration projects enhance ecological conditions and rehabilitate depressed species populations requires assessment of functional state or "performance". This is particularly important to adaptively improving restoration projects (Zedler 1992). The common paradigm that "function follows form" dictates most wetland restoration designs and evaluations: projects usually attempt to reproduce the structural attributes of the mature ecosystem, and success is most often measured by assessing the comparability of the restoring system's structure to that of a reference system. Plant or macrofauna density and composition compared to pre-disturbance or reference levels have been the usual criteria for measuring performance (Sinicrope et al. 1990; Fell et al. 1991; Barrett & Neiring 1993; Merritt et al. 1996; Weinstein et al. 1997; Thom et al. 2002; Roman et al. 2002). This approach assumes the functional responses of fish and wildlife is relatively coincident in space and time with structural characteristics. When explicitly tested, this assumption has often proved invalid. In manipulative experiments in Oregon, Cornu and Sadro (2002) found vegetative recruitment and fish response to oppose each other in relation to marsh surface elevation. Moy and Levin (1991) determined structural attributes (sediment properties, macrofauna densities) to resemble reference levels after only a few years, but the complex interactions (fish abundance and diets) indicative of ecological functioning did not necessarily follow at the same rapid rate. Particularly when restoration is focused on higher trophic level species and communities, explicit measures of functional performance (e.g., trophic linkages and surrogates of production) are needed to provide a more integrated assessment of ecosystem processes and functional equivalency with reference sites. Focusing on the "end-point" structure of a mature system assumes a static linkage between form and function, and disregards the underlying seral processes, their prospective necessity, or the potential benefit to target species of early recovery stages.

The need for a more integrated assessment is coupled with the fact that few studies have evaluated both structural and functional development of recovering systems older than 20 years. Moy and Levin (1991) were among the first to compare functional equivalency between a man-made and a natural marsh by integrating substrate characteristics and marsh utilization of organisms representing two trophic levels. Zedler (1993) linked the structure of *Spartina foliosa* (cordgrass) vegetation to the failure of a mitigation site to support nesting by *Rallus longirostris levipes* (Light-footed Clapper Rail), an endangered species the mitigation was designed to attract. Simenstad and Thom (1996) examined structural and functional changes in the first six years following restoration of a brackish mitigation site in the Puyallup River estuary, Washington. These research projects all suffer the same limitation in that they follow the effects of a single restorative event through time. A "space-for-time" substitution offers a better scenario by limiting the effects of location while maximizing the time through which a pattern of ecosystem development can be followed. The history of the Salmon River estuary, Oregon offers us the unique opportunity to address knowledge gaps in the patterns and dynamics of estuarine restoration and its affects on the marsh community.

Like many estuaries in the Pacific Northwest, 75.4 % of the historic 337 ha of marshes along the lower Salmon River were altered through the installation of dikes in the early 1960s to create pasturelands (Frenkel & Morlan 1991). However, under a special management designation by the U.S. Forest Service, dikes were removed from three sites in 1978, 1987 and 1996. The resulting series of restoring marshes enables us to study marsh community recovery over a 23-year span and compare functional equivalency to an adjacent, undiked reference site. Although there are landscape-scale differences among the marshes, such as position along the estuarine gradient and amount of freshwater input, duration of diking and disturbance regimes associated with long-term tidal exclusion are responsible for the most conspicuous disparities. The unique condition created at the Salmon River allows us to evaluate the following hypotheses:

- Following dike removal, an estuarine wetland follows a trajectory of physical and biotic development toward reference conditions.
- Dike removal in estuarine wetlands restores juvenile salmon rearing habitat

To evaluate our hypotheses we needed to create a method for functional assessment. Simenstad and Cordell (2000) proposed that ecosystem development following restoration can be tracked based on three types of metrics: habitat capacity, habitat opportunity, and realized function. Capacity metrics include productivity measures such as available invertebrate prey and conditions that maintain these prey communities. Opportunity metrics appraise the ability of an organism to access a habitat's capacity. Capacity and opportunity metrics reflect structural attributes of a system and can be regarded as characteristics that allude to system function. Ecological interactions actually demonstrate the system's response. Metrics of realized function include any direct measure of fish response attributable to marsh occupation that enhances fitness and survival, such as fish foraging success. Diet composition of fish illustrates actual trophic linkages and enables functional comparisons between recovering and reference sites. Diet composition and other measures of foraging (i.e., stomach fullness) are arguably sensitive indicators of the ecological state of recovering estuarine wetlands (Shreffler et al. 1992; Miller & Simenstad 1997).

The function of estuarine environments as temporary residence areas for juvenile salmonids has undoubtedly been diminished by extensive wetland alteration and destruction (Shreffler et al. 1990). Estuaries provide the opportunity for gradual osmotic acclimation, and offer productive foraging habitat and refugia from predators for outmigrating juvenile salmon (Healey 1982a; Simenstad et al. 1982; Iwata & Komatsu 1984). Together these factors enhance growth, which is assumed to correlate directly with survival (Reimers 1973). The precipitous decline of anadromous salmon in the Pacific Northwest has intensified inquiry into marsh ecological functions and the consequences to salmon production and life history diversity of a long history of estuarine wetland alteration. Fifty to 80% of salt marshes in Oregon and Washington have been lost, mostly due to diking activities (Oregon Division of State Lands 1972; Washington Department of Ecology 1993).

Restoring marshes to full tidal inundation is assumed to benefit salmon, however there remain many questions about fish response to transitional stages of restoration. Restoring tidal inundation to formerly diked areas imposes dramatic physical and chemical changes. Frenkel and Morlan (1991) documented a rapid turnover in vegetation immediately following dike removal at the Salmon River. Plant die-backs were also reported by DeLaune et al. (1987) after salt-water intrusion changed the chemistry in Louisiana wetlands, and by Tanner et al. (2002) after dike breaching in the Snohomish River estuary. In microcosm experiments, Portnoy (1999) found tidal restoration of highly organic diked and drained marshes mobilized nutrients, causing changes in estuarine water quality that increased primary production and oxygen demand. Increased primary production is typical of developing ecosystems under conditions of disturbance, high nutrient availability, and low metabolic energy requirements (Odum 1969). These physicochemical changes conceivably influence the density of detritivores, especially taxa tolerant of low oxygen conditions (e.g., larval chironomids). Foraging fish may benefit from the increased production of newly restored estuarine marshes, but the effects of decreased ecosystem quality (e.g., channel structure, water temperature) and stability might temper these benefits.

In the present paper, we summarize the conceptual approach and early results from studies of the ecological functions for salmon in the restoring marshes of the Salmon River estuary. We draw on our assessment of capacity, opportunity, and realized function to answer the following questions:

1. What differences in biotic response are seen in marshes of different ages compared to an undiked reference marsh?
2. Are the differences in biota related to recovery time?
3. Does the diet composition of juvenile Chinook salmon differ significantly among marshes in different stages of recovery?
4. What are the potential consequences of differences in trophic structure?

5. What are the implications of estuarine marsh restoration to recovery of Pacific salmon populations?

The development of more direct indicators of the function of restored wetlands as juvenile salmon habitat should be of direct value to researchers concerned with the contribution of wetland restoration to salmon recovery (Shreffler et al. 1992; Simenstad & Cordell 2000).

Methods

Study Site

The Salmon River estuary is located immediately south of Cascade Head, approximately 6 km north of Lincoln City, Oregon. The watershed is 194 km^2 with an 800 ha estuary, half of which is emergent marsh. We established permanent fish and invertebrate sampling sites within each of three formerly diked areas and a reference portion of marsh that has never been diked (Fig. 1.1). For the purposes of this paper, the three marshes are referred to by the year of dike breaching (1978, 1987 and 1996), while the reference site is abbreviated as REF. Estuarine gradient position and freshwater flow regimes determine the salinity range in each of these areas. Average salinities derived from site-specific water column profiles in each marsh at high tide from April and May 1999 (detailed salinity measurement began in March 1999) are reported in Table 1.1.

The REF marsh contains areas of strikingly different elevation, supporting different plant communities typical of salt marshes. Low marsh exists along channel edges and is dominated by *Carex lyngbyei* (Lyngbye's sedge). High marsh areas are characterized by a thick, matted complex of several plant species, including *Potentilla pacifica* (Pacific silverweed), *Juncus balticus* (Baltic rush), and *Deschampsia caespitosa* (tufted hairgrass). The REF tidal channels are deep and steep-sided, branching into a complicated dendritic network throughout the marsh and lacking connection to upland freshwater sources. At low tides of ≤ 1.0 ft MLLW (mean lower low water), the channels completely dewater.

The 1978 site consists of a monospecific stand of Lyngbye's sedge; high marsh vegetation is present only on the remnant dike. Frenkel and Morlan (1991) reported the surface elevation of this site to be about 35 cm lower than adjacent controls in 1988. This lower elevation, which is caused by subsidence from buoyancy loss, compaction, and organic soil oxidation while diked, accounts for the lack of high marsh at this site. A borrow ditch for the former dike runs parallel to the river and perpendicular to the marsh's tidal channels just within the original dike. The channels of the 1978 marsh are steep-sided with some degree of upland freshwater input.

The 1987 marsh contains areas of variable elevation and vegetation comparable to the REF. The distribution of low marsh is patchy, intermittently flanking tidal channels. *Salicornia virginica* (pickleweed; not found in the REF site) is common at this site and the thick, complex vegetative complex of the REF high marsh is comparatively rare. Channel morphology of the 1987 marsh is also similar to the REF, but some input of upland freshwater contributes to the formation of wide channels at the marsh's mouth.

The higher order channels branch into a series of tidal creeks, some with deep holes that fail to dewater at low tide.

The 1996 marsh has undergone dramatic changes since the return of tidal inundation. In 1998, large unvegetated areas were common throughout the site, interspersed with patches of recruiting vegetation, and stands of dead and decaying material, such as *Phalaris arundinacea* (Reed canary grass) and *Rubus discolor* (blackberry). In 1999, the same areas were covered with small patches of several recruiting species including Lyngbye's sedge, Baltic rush, Pacific silverweed, and *Poa spp.* (grasses). The main channel of the 1996 marsh is wider and shallower than the other sites and as yet has little overhanging vegetation. Like the REF marsh, the 1996 marsh lacks upland freshwater input. Formerly the outlet for Salmon Creek, the 1996 channel lost its upland connection when Highway 101 was constructed in the early 1960s (see Fig. 1.1) and Salmon Creek now enters the estuary through a ditch constructed just upstream of the Highway 101 bridge.

Fish Sampling

Fish species composition and density were assessed in discrete tidal channel drainage systems in each marsh twice a month from March to July, 1998 and 1999. A modified nylon mesh (0.6 cm) fyke net was deployed across a tide channel at high, slack tide (Fig. 1.2). As the tide receded, the fish were collected from the cod end of the net, identified, and counted. Pole seining was required to "herd" residual fish into the trap since most marsh channels failed to dewater completely at low tide. This trapping technique has been used successfully by Levy and Northcote (1982) in the Fraser River estuary and by Simenstad et al. (1997) and Miller and Simenstad (1997) in the Chehalis River estuary. Fork length and wet weight were recorded for all salmonids, and subsamples of juvenile Chinook were retained for stomach contents analysis. Abundances from fish samples were standardized to estimated surface area of the tidal channel systems being sampled and reported as average density per m^2.

Prey Resource Assessment

Prey composition and density along the tidal channels were compared among the three treatment sites and the reference site by sampling marsh insects and benthic invertebrates. Marsh surface invertebrates were sampled using Invertebrate Fallout Traps (IFT). The IFT consists of a plastic box (51.7-cm X 35.8-cm X 14-cm) filled with three liters of soapy water, which as a measure of direct input from the marsh to the aquatic system retains invertebrates that fall from the air or the vegetation. This methodology has successfully been used in other studies of emergent marsh vegetation (Cordell et al. 1994). We sampled a total of six IFT sites: REF Lo, REF Hi, 1978 Lo, 1987 Lo, 1987 Hi, and 1996, where 'Hi' and 'Lo' refer to the high and low elevation marsh vegetation strata. No high marsh vegetation was present in the 1978 site, and the 1996 site vegetation could not be characterized as high or low due to its disturbed state. Five replicate IFTs were placed within each vegetative stratum along the channel where fish were collected, and secured using vertical PVC poles. IFTs were deployed monthly between March and July for three consecutive days and invertebrates were collected and preserved in isopropanol. Samples were returned to the laboratory for identification and enumeration. Abundances from IFT samples were standardized to area and reported as average density of invertebrates per m^2.

Benthic invertebrates were sampled monthly using a 5-cm diameter aluminum corer, sampling to a depth of 10 cm for a volume of 196.25 cm^3. Five replicate cores of channel sediments were taken at low tide from haphazardly selected locations along the channel gradient in proximity to the fyke net sites at each of the four marsh areas (REF, 1978, 1987 and 1996). Samples were fixed in the field with 10% buffered formalin. In the laboratory, samples were washed and the macrofauna retained on a 0.5 mm sieve, transferred to isopropanol and stained with Rose Bengal. All organisms were enumerated and identified to the finest taxonomic resolution possible under an illuminated dissecting scope. Abundances from benthic cores were standardized to core volume and are reported as average density of invertebrates per m^3. Strictly epibenthic prey was not sampled by our methods because in oligohaline estuarine environments, epibenthic prey is usually not as important in the diets of juvenile salmon as benthic prey (Levy & Northcote 1982; Simenstad & Cordell 2000). This conclusion was verified by our pilot studies of 1997.

A nonparametric statistical test, Kruskal-Wallis comparison of means, was used to test intergroup differences between invertebrate taxa at each site and nonparametric analogue to the Tukey multiple comparison of means was used *a posteriori* to specify differences. Nonparametric analyses were more robust because our invertebrate data were not normally distributed (Zar 1999). Linear regression models were used to test the relationship between density of specific invertebrate groups in restored sites to recovery time.

Fish Diet Composition

Stomach contents of juvenile Chinook were characterized using a standard processing procedure (Terry 1977). Fork length and damped wet weight were recorded for each fish, stomach fullness was rated from 1 (empty) to 7 (distended), and stage of digestion was ranked from 1 (all unidentifiable) to 6 (no digestion). The prey items were sorted to the finest taxonomic resolution possible under an illuminated dissecting scope. Each prey category was enumerated and weighed to quantify the frequency of occurrence and gravimetric composition of prey items in the diet. We assessed the importance of each prey taxa using the Index of Relative Importance (IRI; Pinkas et al. 1971). The Percent Similarity Index (PSI) (Hurlbert 1978) was used to determine the percent similarity between diets of fish foraging in different areas and to determine overlap between diet composition and available prey items. Standardized Forage Ratios (SFR) (Manly et al. 1972) were calculated as a measure of fish selectivity for particular prey taxa. Our consistent trapping effort enabled us to use stomach fullness indices to assess relative consumption rate between the marshes.

Results

The following summarizes fish densities from March to July 1998 and March to August 1999. Prey resources and juvenile Chinook diet composition are reported from April, May, June 1998 and April, May 1999. We compiled data for these particular months because juvenile Chinook are in the peak of their seasonal outmigration.

Fish

In both 1998 and 1999, *Leptocottus armatus* (Pacific staghorn sculpin) was the most abundant fish species in the REF, 1978, and 1987 marshes, and *Gasterosteus aculeatus* (threespine stickleback) dominated samples collected in the 1996 marsh (Fig. 1.3). Chinook salmon were most abundant in the REF, 1987, and 1996 marshes. Pacific staghorn sculpins were most dense in the REF marsh (Fig. 1.4). In 1998, there were very few staghorn sculpins in the 1996 marsh, but by mid-April 1999 densities had begun to increase. There was little change in staghorn sculpin abundance at the other sites.

In 1998, densities of juvenile Chinook salmon were consistently higher in the REF than the other marshes and peaked in early May (0.04/m^2) (Fig. 1.5). In both years the densities were consistently lowest in the 1978 marsh. In 1999, the overall density of Chinook sampled in the marsh channels was slightly lower than in 1998. Chinook densities were highest in the 1996 marsh in late May (0.035/m^2) and the density peaks in the REF differed from

1998 not only in number but also in time ($0.014/m^2$ in mid-April and $0.01/m^2$ in early June).

Fallout Invertebrates

No consistent pattern was detected in average total density of marsh IFT invertebrates. Average total density was not significantly different between the marshes in 1998 ($p = 0.22$), but low abundance at the 1987 Hi marsh in 1999 contributed to statistical differences in density ($p = 0.02$) (Fig. 1.6). Chironomidae and Ceratopogonidae dipteran families were used to further assess differences between sites because they occurred most frequently and were commonly consumed by fish (Fig. 1.7). In 1998 and 1999, chironomid densities were greater in the 1996 marsh than in the other sites ($p = 0.01$; $p < 0.001$, respectively). Lowest chironomid densities were found in the 1978 marsh in 1998 and in the 1987 Hi and Lo marshes in 1999. No significant difference in ceratopogonid density was detected between the marshes in 1998 ($p = 0.07$); however, ceratopogonid density was significantly greater in the 1996 marsh in 1999 ($p = 0.00$). Regression of average densities of all fallout invertebrates in restored marshes against marsh recovery age indicated no significant correlation in 1998 ($p = 0.06$, $r^2 = 0.18$) or in 1999 ($p = 0.93$, $r^2 = 0.00$). However, chironomids were negatively correlated with marsh age in both 1998 and 1999 ($p < 0.0001$, $r^2 = 0.57$; $p = 0.03$, $r^2 = 0.23$, respectively). Ceratopogonids were unrelated to recovery age in 1998 ($p = 0.84$, $r^2 = 0.00$), and negatively correlated in 1999 ($p = 0.00$, $r^2 = 0.43$).

Benthic Invertebrates

In 1998 and 1999, densities of benthic invertebrates were consistently greater at the REF marsh ($p = 0.02$; $p = 0.00$, respectively) than the other marsh sites (Fig. 1.8). Further differences in the benthic communities were explored specifically through analysis of polychaetes (Family: Nereidae), and the amphipods *Corophium spp.* and *Eogammarus spp.*, which were important Chinook prey (see below). In 1998 and 1999, nereids were most abundant in the 1987 site ($p < 0.001$; $p = 0.03$, respectively) (Fig. 1.9). They were found in small numbers in the 1996 marsh starting in June 1998. During each of the survey years, average *Corophium spp.* densities were significantly lower in the 1996 marsh, compared to the1987 marsh ($p = 0.02$, for both years), but no such differences were detected in *Eogammarus spp.* density ($p = 0.30$; $p = 0.15$) (Fig. 1.10). Neither of these amphipods was commonly found in the 1996 marsh. Regression of average densities of benthic invertebrates in restored marshes against marsh recovery age indicated no correlation for total benthic invertebrates in 1998 ($p = 0.65$, $r^2 = 0.02$) but a positive correlation in 1999 ($p = 0.00$, $r^2 = 0.59$). Nereid worm densities were unrelated to recovery age for both years ($p = 0.60$, $r^2 = 0.02$; $p = 0.11$, $r^2 = 0.18$). *Corophium spp.* densities were also unrelated to recovery age for both years ($p = 0.29$, $r^2 = 0.09$; $p = 0.06$, $r^2 = 0.25$), and significant differences were only detected for *Eogammarus spp.* in 1999 ($p = 0.02$, $r^2 = 0.34$). However, when the data for all species and both sampling years are considered together, a trend of increasing amphipod abundance with increasing recovery age is suggested.

Diet Composition of Juvenile Chinook Salmon

Diet compositions of juvenile Chinook included both marsh insects and benthic invertebrates, dominated by chironomid adults and larvae, ceratopogonid pupae, other dipterans, trichopterans, the amphipods *Corophium spp.* and *Eogammarus spp.*, and nereid polychaetes (Fig. 1.11). Epibenthic (Mysidae) and planktonic prey (fish larvae) also occurred occasionally. In general, diets from fish foraging in the 1978 and 1996 marshes consisted of insects while fish in the REF and 1987 marshes consumed primarily crustaceans and fish larvae. Comparison of stomach fullness indicated no difference between sites.

Juvenile Chinook diet composition was most similar among fish collected in the REF and 1987 marshes (Table 1.4). The lowest diet similarity between these two sites was in June 1998, when fish collected from the 1987 marsh had consumed mostly fish larvae. Among the restoration sites, the most consistent diet similarity was between fish from the 1978 and 1996 marshes. PSI was 70.75% between diets of fish from the 1987 and 1978 marshes in April 1999, but was very low in other months. In some cases, the numerical representation of prey items in salmon stomachs and in invertebrate collections indicate extremely high overlap in the availability and consumption of prey species (Table 1.2). For example, in June 1998 PSI similarity at the 1996 site was 91%. In other cases, the overlap was low. PSI similarity among prey and available invertebrates in the 1987 marsh was only 9% in June 1998, when 80% of the salmon diet consisted of fish larvae. The average PSI between the diet composition and available prey was 39%. Trichoptera were the most highly selected prey items with an SFR range of 0.13 to 1.00 (Table 1.3). Other selected species included *Corophium spp.*, chironomid larvae, and dipterans.

Discussion

We have documented differences in fish use, invertebrate prey resources, and diets of juvenile salmon in one natural and three recovering estuarine marshes of varying ages. While there are potential confounding factors, we have shown that many of these differences are attributable to variable states of marsh redevelopment. These results, although early in our evaluation of the restoration trajectories of these marshes, provide insight into (1) fundamental assumptions about linkages between structure and function of restoring wetlands, (2) how we assess the functional performance of restoration, (3) whether or not early stages of restoring estuarine marshes can contribute to the recovery of juvenile salmon habitat, and (4) the relative importance of marsh landscape position compared to the age of the restoration site.

We assessed differences in biotic structure by measuring fish densities and diet, and invertebrate composition and abundance. These descriptive attributes evaluate the differences in both capacity and opportunity metrics among the marshes (Simenstad & Cordell 2000). Differences in fish use among the marshes may be related to marsh position in the estuarine gradient, food availability, and changing conditions in a restored system. Pacific staghorn sculpins are commonly found in large numbers throughout the estuary, and there was little or no change in their abundance in the REF, 1978 and 1987 marshes. However, they were seldom found in the 1996 marsh until mid-April 1999. The lack and subsequent increasing abundance of staghorn sculpins may reflect their response to rapidly changing channel conditions, reflecting the instability of a newly restored system and an organism's response to conditions resulting from the rapid decay and flushing of in-channel pasture grasses that died soon after salt-water inundation. Rate of change may be a more appropriate evaluation of system attributes because processes governing the system are constantly changing (Parker 1997).

Juvenile Chinook salmon were found most commonly in the REF site, suggesting a potential fish response to ecosystem maturity or quality. However, they were also found in large numbers in the 1996 site, suggesting a response to estuarine gradient position (see Fig. 1), or the prevalence of desirable food items. The 1996 site is the first major backwater area within the tidal reach of the upper estuary that juvenile salmon encounter during their downstream migration. In contrast to this site, Chinook were rarely found in the 1978 site. However, the presence of a borrow ditch along the remnant dike and only limited access from the river restricts the fish's ability to enter that marsh.

We found no difference in total density of IFT invertebrates among sites, but there were some important differences in community composition. The average density of chironomids and ceratopogonids was significantly highest in the 1996 marsh. Regression analysis revealed these fly families to be negatively correlated to recovery time over our sampling period. Chironomids and ceratopogonids are known to be important prey items for juvenile salmonids (Shreffler et al. 1992), so differences among marshes could translate to important differences in marsh function, i.e., diet composition.

The average density of total benthic macroinvertebrates was highest in the REF marsh. Densities in the restoration sites were not significantly different from each other. Examination of specific benthic invertebrate groups (those most often encountered as prey items) revealed other important differences in community structure between the reference and recovery sites. The amphipods, *Corophium spp*. and *Eogammarus spp*., were rarely found at the 1996 marsh, and regression analysis of amphipod density in restoring sites revealed a positive correlation with recovery age. In our continuing research, we will conduct more detailed analyses of the differences in invertebrate communities between the marshes, linkages to ecosystem processes, and implications to rearing juvenile salmonids.

Differences in biotic structure among sites may translate to differences in growth if (1) a fish's diet reflects and tracks the biotic environment as we have measured it, and (2) there are differences in energy content of the prey items. Diet composition (a direct measure of fish response to the system) reflects the structure of the marshes to some degree. In some cases, we found extremely high PSI overlap between our invertebrate collections and the diets (e.g., 91% similarity in June 1998 at 1996 site); in other cases, similarity was very low (e.g., 9% similarity in June 1998 at 1987 site). The average similarity (39%) is a reasonably high overlap considering the amount of inherent variability between samples and the influence of foraging selectivity, which we interpret to be a measure of how the fish uses the resource. High selectivity may indicate disparity between habitat structure and function.

Based on standardized forage ratios, we found trichopterans to be the most highly selected prey in the estuarine marshes. This may reflect fish choice for the high-energy trichopteran prey, or it may reflect sparse collection of trichopterans by our sampling method. Similarity among juvenile Chinook diets from the four marsh areas was highest between the REF and 1987 marshes. These sites are nearest to each other and have similar geomorphology. This does not necessarily imply that these sites have reached functional equivalency; we consider it important to determine whether the 1987 and REF marshes provide similar energetic contributions to foraging fish. This question will be addressed in future work with prey energy content and bioenergetic modeling.

Quantitative assessment of the ecological effects of restoration projects implies long-term monitoring of replicated control and experimental units at temporal and spatial scales appropriate to test a certain hypothesis. Yet, such a design is rarely possible. With respect to ecological structure and function, "long-term" often means decades to centuries, a period of study not feasible for most research (Michener 1997). Even in the case of the Salmon River estuary, where reference and treatment conditions span 23 years, replication of treatment sites is impossible. These conditions present a scientific challenge to the restoration ecologist, but do not preclude the acquisition of valuable information.

Our results indicate disparity between reference and treatment sites based on metrics for capacity, opportunity, and fish performance (realized function) even after more than two decades of recovery. However, foraging juvenile salmonids may still benefit during early stages of marsh recovery. For example, increased production, such as the high density of chironomids, following dike breaching may increase foraging opportunities for juvenile salmon. On the other hand, trade-offs with ecosystem quality, such as poorly formed channels and increased temperature, could temper the benefits derived from increased prey quantity. Further study is needed to determine the significance of interim benefits during early stages of marsh redevelopment.

By contrasting functional and structural differences of the marshes, our inquiry into development patterns and rates of restoring estuarine marshes gives a broad view of how the marsh is functioning, the possible benefits to target species, and the efficacy of dike breaching for restoring estuarine ecosystems. Increased understanding of the processes and mechanisms of functional development will benefit not only salmonid restoration efforts in the Pacific Northwest, but will also further the science of restoration ecology.

Tables

Table 1.1: Watershed Position and Average Salinity (ppt)

WATERSHED POSITION	SITE	APR-99	MAY-99
Highest	1996	0.33	0.70
⬇	REF	8.83	0.37
	1978	2.43	0.13
Lowest	1987	13.3	6.85

Table 1.2: Percent similarity index between diets and available prey

MONTH	SITE	PSI
Apr-98	1996	48%
May-98	REF	27%
	1978	53%
	1996	24%
Jun-98	REF	15%
	1978	33%
	1987	9%
	1996	91%
Apr-99	REF	23%
	1978	31%
	1987	49%
	1996	45%
May-99	REF	38%
	1978	28%
	1987	67%
	1996	31%
Average PSI		38%

Table 1.3: Standardized Forage Ratios for Important Prey Items
(**bold** indicates positively selected prey items)

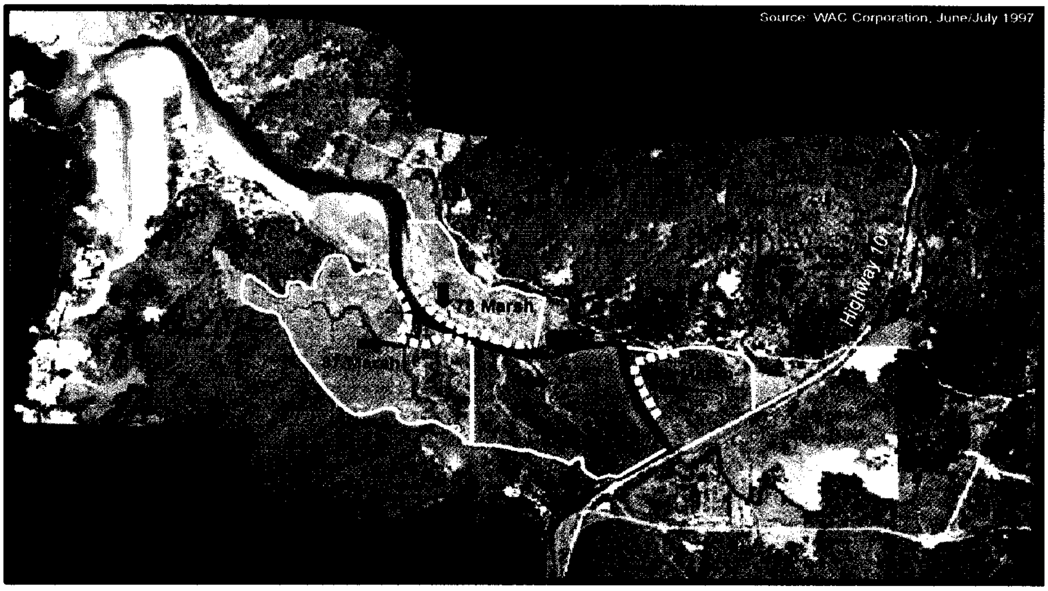
	Apr-98	**May-98**			**Jun-98**				**Apr-99**				**May-99**			
Prey Item	*96*	*CTR*	*78*	*96*	*CTR*	*78*	*87*	*96*	*CTR*	*78*	*87*	*96*	*CTR*	*78*	*87*	*96*
Ceratopogonidae adult	0.11					0.09		0.03		0.01	0.02	0.01			0.01	
Ceratopogonidae pupae												**0.79**				
Chironomidae adult	0.01				0.02	0.04	0.06	0.09	0.01		0.02	0.02		**0.96**	0.02	
Chironomidae larvae						**0.59**		0.05					0.36			
Diptera adult	0.00		0.01			0.09	**0.54**			0.02	0.01			0.04	0.01	
Trichoptera adult	**0.57**	0.44	**0.98**									0.13	0.41		**0.78**	**1.00**
Corophium spp.			0.01	0.33	**0.63**	0.13	0.35	0.10	0.08	0.34	0.08	0.02	0.01		0.03	
Eogammarus spp.		0.16		0.60					**0.84**	0.06	0.2	0.02	0.05		0.12	
Nereidae		0.36							0.07	0.19	**0.67**	0.02	0.12			

Table 1.4: Percent similarity of diet composition* between sites

	Control v. Restored Sites			Comparison of Restored Sites		
Sampling Date	*CTR v. 78*	*CTR v. 87*	*CTR v. 96*	*78 v. 87*	*78 v. 96*	*87 v. 96*
May-98	9.34%	-	19.13%	-	23.74%	-
Jun-98	10.96%	19.25%	7.40%	10.42%	24.49%	5.78%
Apr-99	34.28%	52.90%	26.06%	70.75%	14.52%	29.61%
Early May-99	-	67.17%	1.07%	-	-	7.35%
Late May-99	17.62%	37.30%	21.70%	3.69%	15.60%	9.42%
Average PSI	18.05%	44.16%	15.07%	28.29%	19.59%	13.04%

*Diet composition IRI values were used.

Figures



Source: WAC Corporation, June/July 1997

Figure 1.1: Map of the Salmon River estuary. Dashed lines represent locations of removed dikes. Arrows represent locations of fish and invertebrate collections.

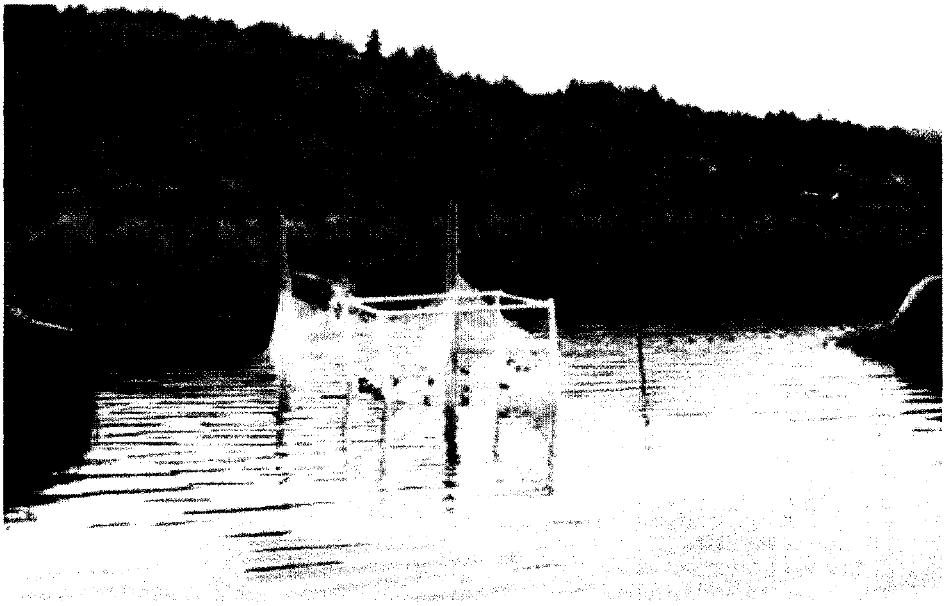


Figure 1.2: Modified fyke net deployed across a tidal channel in the reference marsh.

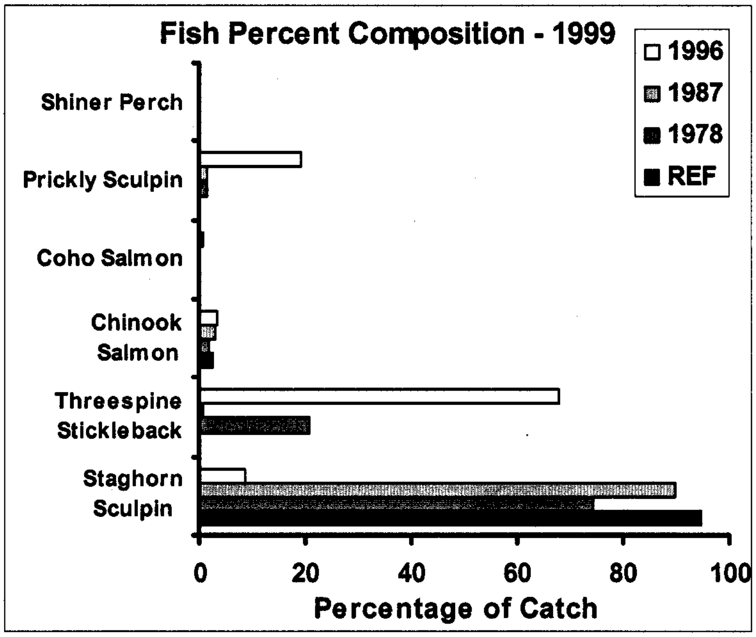


Figure 1.3: Percent composition of fish collected from each of the four marshes. Pacific staghorn sculpins were the most abundant fish in all sites, except for the 1996 marsh where threespine sticklebacks were the most abundant.

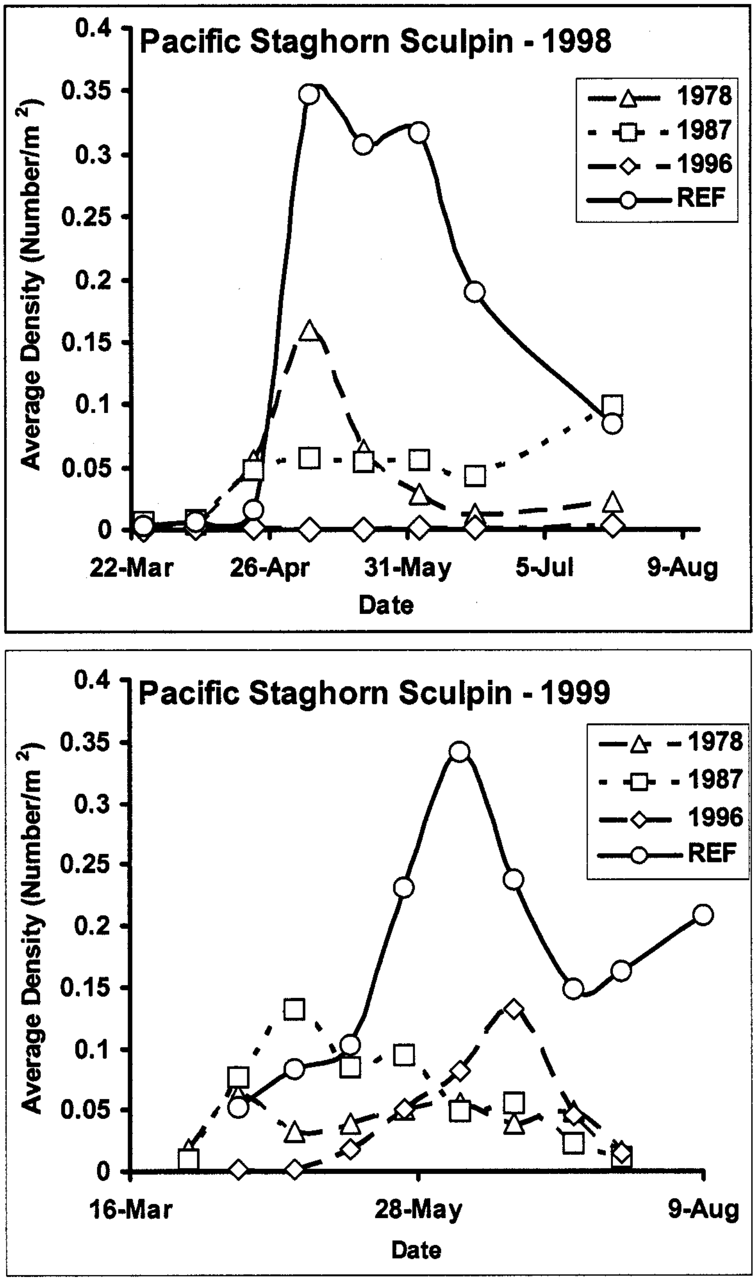


Figure 1.4: Catch per unit area for Pacific staghorn sculpin, a ubiquitous resident of the Salmon River estuary, in 1998 and 1999.

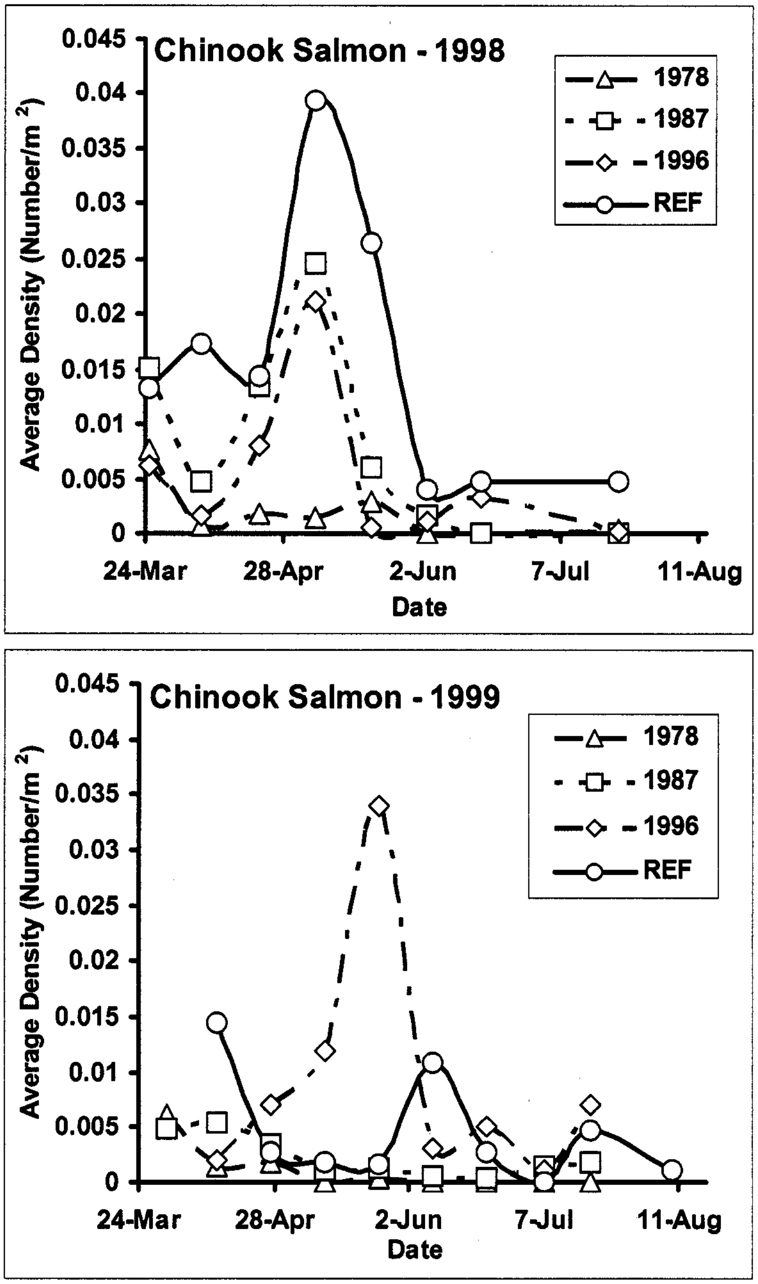


Figure 1.5: Catch per unit area for Chinook salmon in the four marshes in 1998 and 1999.

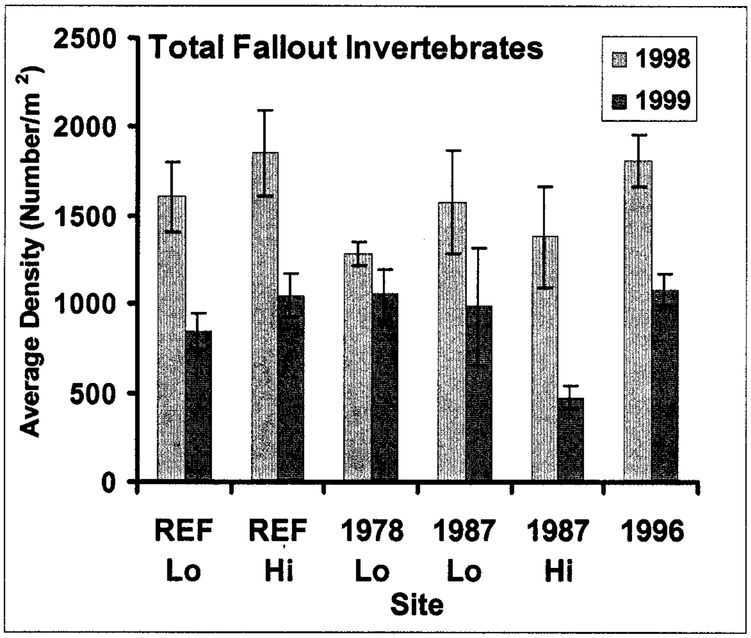


Figure 1.6: Total density of fallout invertebrates from each collection site in the four marshes. No statistical difference was detected in 1998 ($p = 0.22$) and low densities in the 1987 Hi marsh contributed to the statistical difference in 1999 ($p = 0.02$).

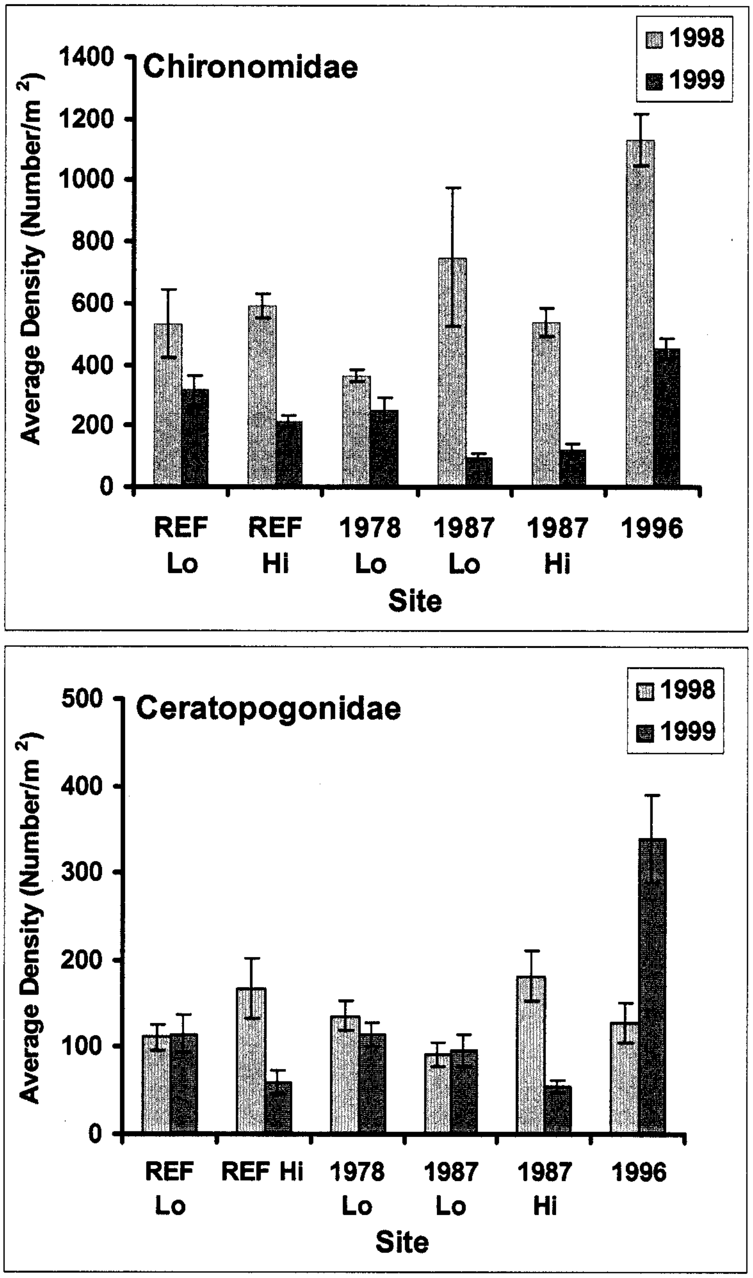


Figure 1.7: Density of the fallout dipterans of family Chironomidae and Ceratopogonidae in four marshes. Only the 1996 marsh is significantly different from all other sites.

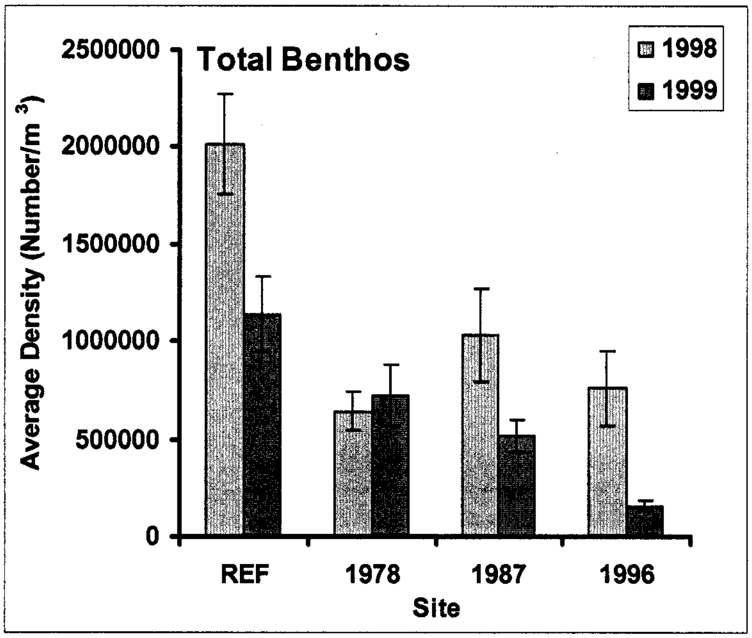


Figure 1.8: Total density of benthic invertebrates in the four marshes. Densities in the REF marsh are significantly different from all other sites. No difference was detected among the restoring marshes.

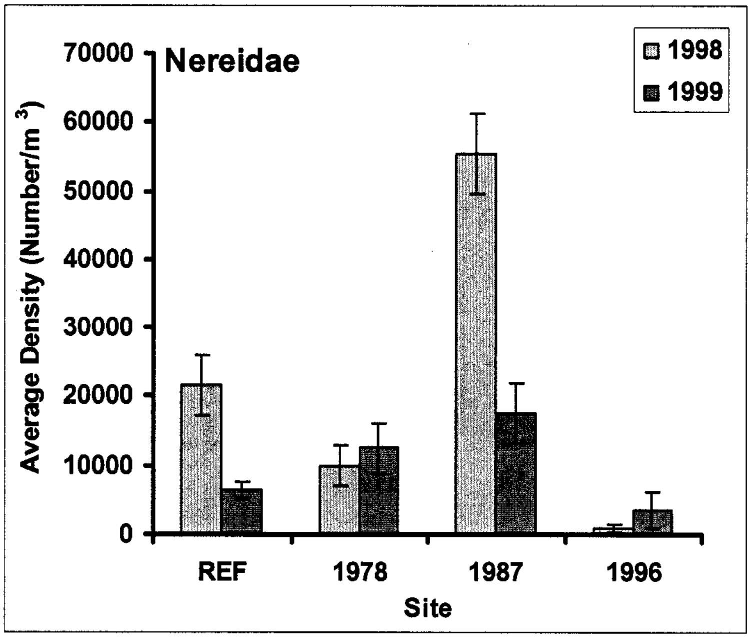


Figure 1.9: Density of nereid polychaetes in the four marshes. Significant differences in nereid densities were detected in the 1987 marsh compared to all other marshes, and the 1996 marsh compared to the REF and 1987.

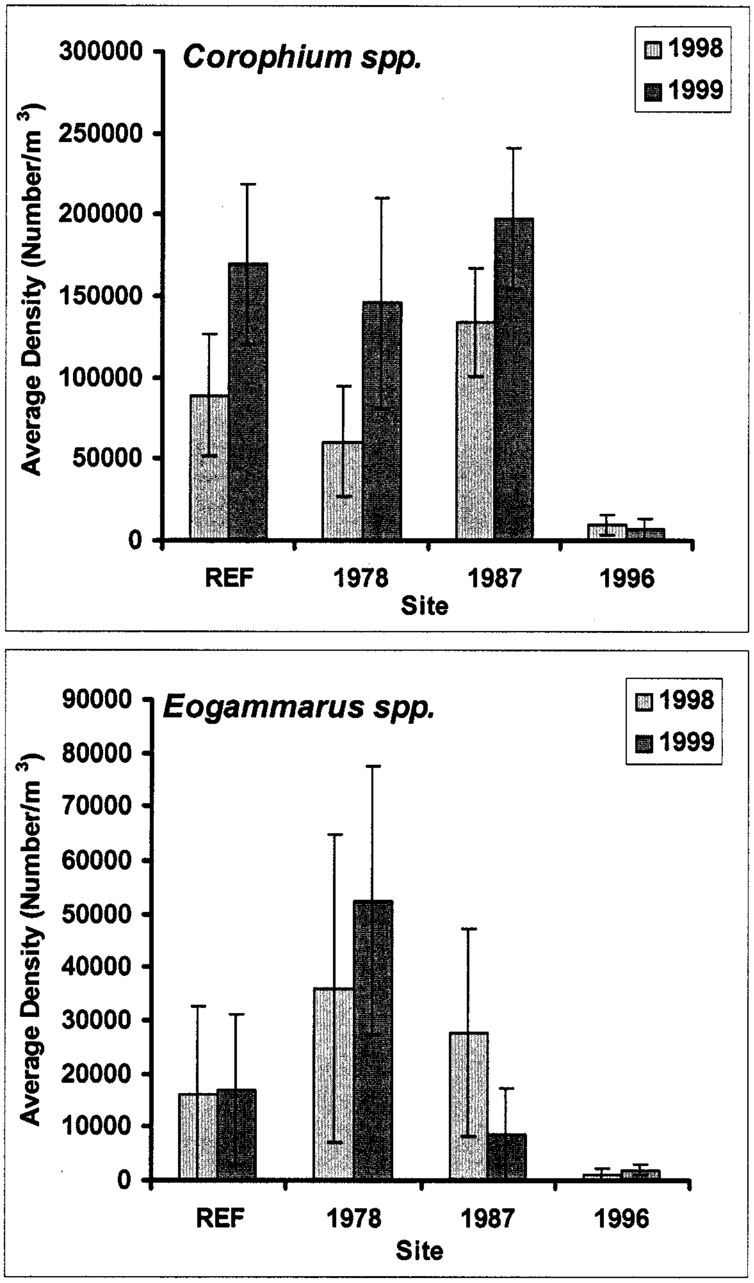


Figure 1.10: Densities of the two amphipods, *Corophium spp.* and *Eogammarus spp.*, in the four marshes. *Corophium spp.* densities were significantly different between the 1996 marsh and all other marshes.

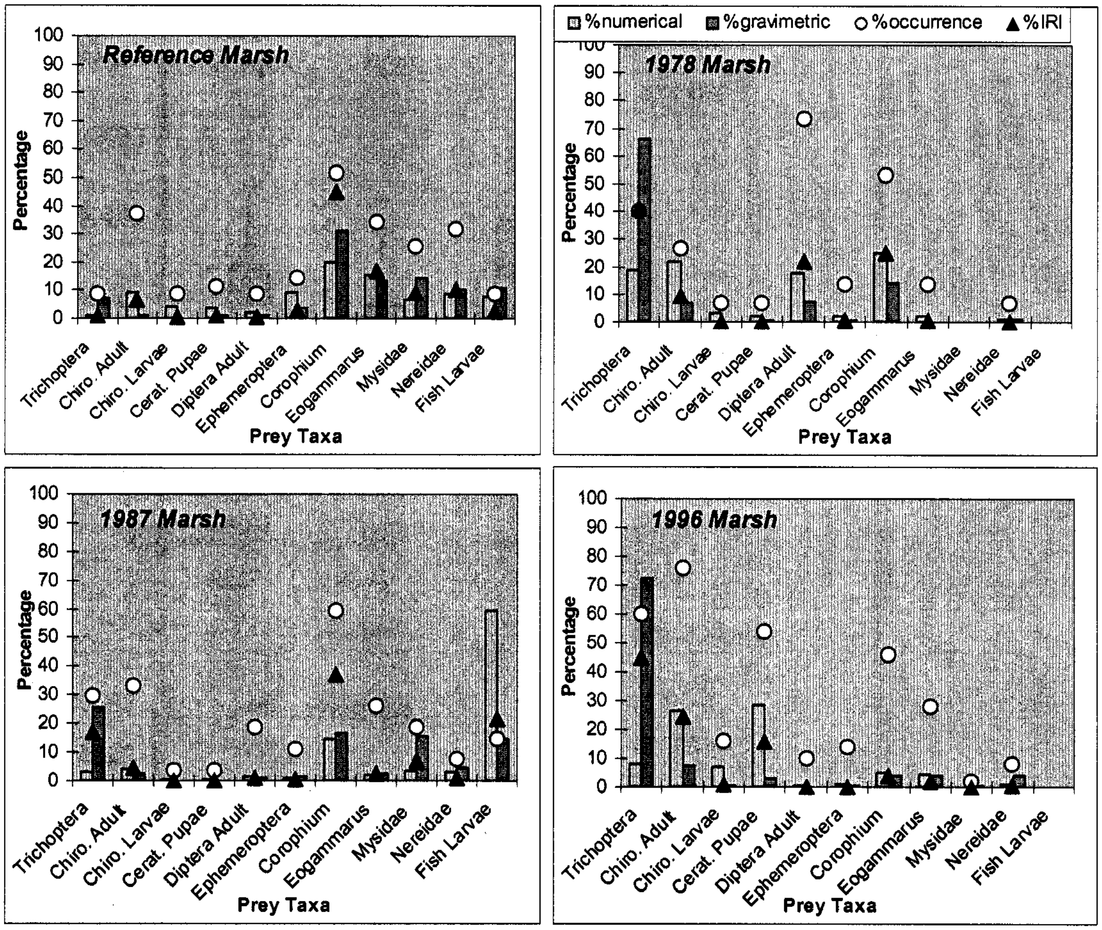


Figure 1.11: Diet composition as percent numeric abundance (light grey bar), percent gravimetric abundance (dark grey bar), percent frequency of occurrence (white dot), and percent total Index of Relative Importance (black triangle) in the four marshes.

Chapter 2: Comparing Performance of Restoring Tidal Wetlands in Terms of Juvenile Chinook Salmon Growth Potential: A Bioenergetic Approach

Introduction

Assessing Restored Ecosystems

Even with the creation of rigorous assessment protocols (Zedler 2001) and numerous scientific studies evaluating restoration projects, debate continues over how to best assess restoration: what is an appropriate reference site; what metrics are most suitable; and what end points constitute "success" (Palmer et al. 1997). Comparability of plant and animal densities, and composition between restored and reference conditions have been the usual criteria. This approach assumes ecosystem function has a dependent relationship with specific measures of ecosystem structure, such as organism density and composition (Gray et al. 2002). Unfortunately, this fundamental assumption is often left unverified. Fell et al. (1998) found prey availability was similar for *Fundulus heteroclitus* (mummichog) in tidal marshes invaded by reed grass and those without, indicating vegetative structure may not be directly linked to trophic processes. Arguably, limiting restoration monitoring to structural attributes fails to answer important questions regarding recovery of critical processes that often support targeted functions of the restoration. To assess functional performance, restoration ecologists need tools and indicators that are more directly indicative of ecosystem complexity.

Comparison of Reference and Restored Systems

Unaltered reference systems provide for reasonable approximation of target conditions in evaluating habitat restoration or creation projects (NRC Restoration of Aquatic Ecosystems 1992). For instance, Levings and Nishimura (1997) compared reference (undisturbed), restoring (revegetated), and disturbed (non-vegetated) sites using a variety of ecological variables to indicate comparative functionality. Higher invertebrate abundance was found in restoring marshes and more Chinook and *Oncorhynchus nerka* (sockeye) smolts were caught in the disturbed marshes, but no difference was found between Chinook and *Oncorhynchus keta* (chum) fry abundances in the three marshes or in species composition among disturbed and reference conditions (Levings & Nishimura 1997). Comparing conditions in a natural and created slough on the Chehalis River estuary, Miller and Simenstad (1997) found no difference in 10-day growth of juvenile Chinook salmon based on otolith microstructure, but found significant differences in invertebrate densities and stomach fullness. Differences in prey densities and stomach fullness may translate into differences in consumer growth (a measure of performance). Subtle differences in diet composition, depending on foraging location, could translate into distinct differences in total energy consumed and weight gained.

Progressive restoration of tidal wetlands in the Salmon River estuary (Frenkel & Morlan 1990; 1991), with three recovering marshes of different ages and a reference, provided an excellent opportunity to assess tidal marshes across a range of ages and landscape positions, and make comparisons with an undiked reference wetland (Gray et al. 2002; Bottom et al. 2005). Gray et al. (2002) showed a productivity boost in the newest restoration site that may or may not be realized in terms of salmon performance. To test the implications of differences in production I used a comparative assessment of juvenile Chinook modeled growth potential among marshes. My objective was to compare the performance of marshes in different stages of recovery in terms of juvenile Chinook growth potential and employ the bioenergetics model as a novel assessment tool. The bioenergetics model condenses a range of impacts and recovering system characteristics into comparable growth potential values which can be used to determine relative functionality as a measure of success.

Bioenergetics Modeling

Bioenergetics modeling has emerged as a new technology in ecosystem science for determining a range of important environmental characteristics: PCB transfer (Jackson & Schindler 1996), nutrient cycling (Schindler & Eby 1997), predation (Stewart et al. 1981; Cartwright et al. 1998), energetic cost of diel migration (Cianelli et al. 1998), and stable isotope ratios (Harvey et al. 2002). An energy-balance model of fish growth rate potential represents ecosystems in the common currency of expected fish growth rate (g/g/d) under specific environmental conditions (Brandt et al. 1992; Tyler & Brandt 2001). Modeling of fish growth potential has recently been proposed as a quantitative tool for linking the physiological conditions of a fish to the prevailing biological and physical conditions of the environment (Brandt et al. 1992; Mason et al. 1995; Tyler & Brandt 2001; Madon et al. 2001). I selected foraging performance of seasonally ubiquitous juvenile Chinook salmon as a metric for comparing ecosystem state and development among the recovering and reference wetlands at the Salmon River.

The method can be used to explore explicit links in fish energetic processes to biotic (prey resource, foraging constraints, body size, and physiological condition) and abiotic variables (water temperature, hydroperiods) (Madon et al. 2001). Fish growth rates are influenced by fish size and consumption rate, prey quality, water temperature, and activity level. Several factors, such as consumption rate, fish size and activity level, were influenced by environmental variables (i.e., water temperature, prey availability), which vary by site. I included site-specific empirical data in the model to evaluate processes affecting fish growth.

Model Application

I used the "Wisconsin" computer model (Hanson et al. 1996) to simultaneously model several parameters of fish growth over time. The energy-balance approach estimates growth as food consumed (C) minus the energetic costs of respiration (R), specific dynamic action (cost of processing a meal) (S), and wastes (egestion (F) and excretion (U)). Wastes are computed as a relatively constant proportion of consumption with a correction for the indigestible percentage of prey (determined from literature and empirical data) (Table 2.1). Consumption and respiration are species-, size-, and temperature-dependent (Hanson et al. 1996; Tyler & Brandt 2001).

In the following chapter, I focused on the following null hypothesis to determine modeled growth potential and compare results among marshes as a reflection of recovery condition.

> H1: *There is no difference in juvenile Chinook salmon performance among the reference and restoring marshes in the Salmon River estuary (as indicated by bioenergetic modeling).*

I also answered the following questions regarding differences in growth potential and consumption in three restoring and reference marshes.

1. Were there differences in modeled growth potential among marshes and were the differences related to marsh age?
2. Were the differences influenced most by temperature, prey quality or prey quantity?
3. Does disparity in fish size among marshes affect modeled growth potential?
4. Does the model provide an effective tool to assess development of juvenile Chinook salmon habitat?

The bioenergetics model represents an approximation of potential growth conditions within the scope of this study, and provides the capability to address research questions regarding differences among marshes, influencing factors, and the utility of a bioenergetics model to compare systems. I gathered as much information as possible about each marsh's unique conditions (temperature, prey resources, prey energy, apparent consumption rate) for foraging juvenile Chinook and translated those data into model inputs to determine relative functional state, or each marsh's capacity to influence growth of foraging fish, as a measure of restoration success. I used data on fish use and residence in marsh habitats, diet composition, prey energy density and water temperature from the peak period of marsh occupation by juvenile Chinook salmon, March to August 1999-2002. Growth is dependent on consumption rate, so exploration of difference in consumption rate among sampling periods and marshes was an important precursor for comparing the relative growth potentials among marshes.

Methods

Site Descriptions

Estuarine marshes in the Salmon River estuary were restored at nine-year intervals: in 1978, 1987 and 1996, creating a "space-for-time substitution" which allows for the comparison of marsh conditions in several stages of recovery (Fig. 2.1). Notably, a large portion of the estuarine marsh (REF) was never diked and, although used intermittently for pastureland and haying, no agricultural impact has occurred in this marsh since 1974. This marsh represents the best estimate of reference conditions available within this estuary.

Refer to Gray et al. (2002) for detailed descriptions of vegetation conditions at all sites from 1998-1999, notable changes since 1999 are mentioned here.

Beginning in 2001 at the 1978 marsh, scattered stands of tufted hairgrass appeared along some channel edges indicative of changes in elevation. The marsh has had lower elevation when compared with references (Frenkel & Morlan 1991), contributing to the proliferation of low marsh vegetation such as Lyngbye's sedge. Accretion rates may be slower here versus the other restoring marshes due to the restriction of laminar tide flow by the remnant dike, which although breached in 1978, was not fully removed until 1987. Tidal channels were steep-sided with significant upland freshwater input into the main sampling channel; however, the former dike and ditch obstruct direct fish access to the marsh from the river. Complicated entry to the 1978 marsh limits the opportunity of a fish to access this marsh and was reflected in low catch numbers compared with the other marshes, especially in the late season when flows drop (Fig. 2.2).

No obvious changes in vegetation characteristics occurred in the 1987 and REF site since we described the sites in Gray et al. (2002), although the 1996 marsh has undergone dramatic changes during our study period. In 1998, large un-vegetated areas interspersed with patches of recruiting vegetation, and stands of dead and decaying material were common throughout the marsh (Gray et al. 2002). Similar plant die-backs are reported in the 1978 and 1987 marshes following restoration (Frenkel & Morlan 1990, 1991). Several ephemeral species, including *Cotula coronopifolia* (mud-disk) and *Eleocharis palustris* (creeping spike-rush), began to form small, monotypic stands in 1999. In 2000, these early recruits disappeared as typical salt marsh species, Lyngbye's sedge, Pacific silverweed, tufted hairgrass, *Distichilis spicata* (salt grass) and *Agrostis spp.* (bentgrass) began to colonize. At present, large stands of Lyngbye's sedge are spreading across the landscape of this marsh (R. Frenkel, pers. comm.). Detailed vegetation changes will be covered in future publications.

The main channel at the 1996 marsh was shallow, wide and undifferentiated in 1998-1999, but by 2000-2002, channels had visibly increased in depth and sinuosity, and decreased in width.

Model Development

Temperature Conditions

I placed TidBit™ loggers manufactured by Onset Corporation at ten locations throughout the estuary (Fig. 2.1) and programmed them to record temperature every 20 minutes. Loggers were in place from March 1999 to July 2002. Average daily temperature was determined over a ten-day neap tide period each month, when the marshes did not dewater fully, so the assumption of continual fish occupation could be made. Some temperature loggers were also exposed at low tides, so using neap tides assured measurements were accurate reflections of water temperature. In most cases it was necessary to combine four, five and/or six-day intervals. Refer to Table 2.2 for temperature input from all 1999 model simulations. The remainder of the temperature input data is available in Appendix B. I also tested the model's sensitivity to different temperature regimes at each site by using average diet conditions, average fish weight through the season and site-specific temperature conditions which varied across the seasons and years.

Fish sampling

Fish sampling occurred on a monthly basis from March-July 1998-1999, March-August 2001 and April-July 2002. A modified fyke net was deployed at high slack tide, and as the tide receded, fish were collected from the cod end of the net, identified, and counted (Gray et al. 2002; Bottom et al. 2005). Trap-net catch was normalized as number of fish per m^2 based on the estimated marsh-channel area above each trap. For all salmonids, weight (g, wet) and length (mm, fork length) was recorded, and a subsample (7-10 individuals) of juvenile Chinook was retained for stomach contents analysis.

From 2000-2002, I conducted a mark and recapture study to determine apparent estuarine growth, timing and residence of juvenile Chinook salmon (Bottom et al. 2005). A 1.5 m rotary screw trap was installed at the head of tide was operated four days/week in 2000 and continuously in 2001-2002 from March-July. Fork length (mm) was recorded from 25 fish per day. Juvenile Chinook salmon were marked using a Panjet™ needleless injector using Liquitex™ artist's quality acrylic paint. Marks were known to last for up to 16 weeks based on laboratory experiments. Full details regarding time of entry, residence and movement can be found in Bottom et al. (2005). Results from the mark and recapture study were used to determine apparent growth rates of juvenile Chinook, were compared with model estimates and used to corroborate model results with actual growth of juvenile Chinook cohorts.

Juvenile Chinook Size

Fish length and weight were recorded at every fyke net collection and a model was used to calculate the relationship between length (L) and weight (W), where $W = a * L^b$. These data were used to determine: 1) growth increments for estimating estuary-wide consumption rate by sampling period (Table 2.3), 2) mean weight of juvenile Chinook sampled from each period across all sites (Table 2.4), and 3) the site-specific adjustments to fish size in the third model simulation (Table 2.4).

Juvenile Chinook Diet Composition

Standard stomach contents processing procedures were used to compare foraging habits of juvenile Chinook in the reference and recovering marshes following Terry (1977), further described in Gray et al. (2002). Average diet proportion was determined using the combined diets of all fish from a sampling period, and were based on damped wet weight (WW) of prey items. Kruskal-Wallis non-parametric comparison of means was used to compare average seasonal abundances of different diet items among sites (Zar 1999). Table 2.51-2.54 summarized diet composition for each site and sampling period. Diet composition model input was based on 33 prey categories and original data are available in Appendix C.

Energy Determination of Prey Items

Energy content of the most common (and available) juvenile Chinook prey was determined by bomb calorimetry. Prey items generally were collected in a single location, providing an estuary-specific approximation of energy content regardless of marsh-recovery status or season. Prey items were collected using various methods (fallout traps, sweep net, vapor light trap), sorted with an illuminated dissecting scope, and heat dried (55°C for 24-48 hours) until a stable weight was obtained.

I used a Parr 1425 Semi-micro Bomb Calorimeter to determine the energy content of important prey items. The dried material was pelletized and combusted following the Parr operating instruction manual procedures for the calorimeter. Final energy values were obtained as dry weight, but all energy values in the bioenergetics model were input as J/g WW (Table 2.1), as diet composition was determined with wet weight analysis. Wet-to-dry weight ratios were necessary to make the conversion and were estimated from lab analysis. My values compared well with literature values (Table 2.1). When wet weight to dry weight ratios were not available (some samples suffered desiccation in transport and storage), I substituted literature values, and similarly substituted literature values when measured energy values were not available.

Modeling Consumption and Growth Potential

I first simulated nominal growth potential where size and consumption rate were constant among marshes, so simulations examined just the effects of differences in prey composition and thermal regime among marshes (Table

2.4). Then, I reran the simulations including differences in prey availability or feeding rate (based on peak gut fullness per marsh). With this approach, I first examined whether growing conditions were better based on temperature and food quality; then, whether food availability/feeding rate compensates or exaggerates differences in growth performance among marshes. Because specific growth rate generally declines with increasing fish weight, I accounted for the potential effects of variable fish size over time and among sites. I assessed the effects of variable fish weight among sites, including situations (only in the REF) when a bimodal size distribution was found. The intent of the second and third model simulations was to insure that specific growth rates were the best approximation of actual growth potential in a site.

Model Simulations

Generalized Monthly Consumption Rates Fitted to Growth for All Marshes Pooled—

Direct measures of consumption rates in these estuarine marshes were extremely difficult to obtain both due to the low numbers of available salmonids for sampling and problems with enclosure nets because of water level, however current studies are evaluating feeding over entire tide cycles (Bieber In prep.). To estimate a generalized average consumption rate for all marshes combined, weight data on fish trapped in the marshes were pooled across all marshes within each sampling period (monthly neap tide series), and the month-to-month weight increment was used with generalized monthly diet and temperature conditions to fit consumption to these average growth increments. Repeated simulations through all weight increments yielded the generalized consumption rates for each period (Table 2.3).

Marsh-specific Diet and Temperature Effects on Growth Potential under a Common Consumption Rate—

My first model simulation used the average weight of fish across all marshes within each period (Table 2.4) and generalized monthly consumption rates (Table 2.3). Marsh-specific diet and temperature inputs were determined by field investigation as detailed above. Empirical values of estuary-specific prey energy were used (except when unavailable, then literature values were substituted) (Table 2.1). Results are reported as specific growth rate by site and year.

Effects of Marsh-specific Diet, Temperature, and Consumption Rate on Growth Potential—

During diet analysis, stomach contents weight was standardized against fish weight and expressed as an instantaneous ration (tidal ration), providing a quantitative surrogate to consumption rate. My second model simulation accounted for marsh-specific differences in consumption rate based on tidal ration (e.g., Fig. 2.3). A comparison of means for these tidal rations was performed with one-way analysis of variance (ANOVA) or Kruskal-Wallis rank test (a non-parametric equivalent) depending on which test was the best fit for those data conditions (Table 2.6). Differences were found in 1999 and 2000 at the 1996 site, which showed higher surrogate consumption in later the sampling season. Those differences were accounted for in the second model simulation by increasing the consumption rate by the percent difference of the higher tidal ration. Refer to Table 2.6 for adjusted consumption rates.

Effects of Marsh-specific Differences in Temperature, Diet, Consumption, and Size on Growth Potential—

The third model simulation attempted to further refine the model by incorporating disparities in average size of fishes foraging among marshes, an important component considering the allometric relationship of growth potential to fish size. Refer to Table 2.4 for mean weights of fish by site.

Empirical and Model Comparisons

I was able to estimate growth of weekly cohorts of juvenile Chinook salmon foraging in the Salmon River estuary using mark and recapture data. Generalized weight data was determined using a linear regression of length records taken from the rotary screw trap. This apparent weight was compared with actual weight upon recapture and used to determine the fish's growth over the residence time. I averaged growth rates from all fish residing in the estuary for > 10 days and compared those values with an average of our modeled growth rates for the same year.

Results

Model Input

Temperature

Average daily river temperatures were similar among the four years and were well within the optimum range (11-19 °C for 3 g fish) for juvenile Chinook (Fig. 2.4). Temperature conditions in the marshes were markedly different than the river, and somewhat related to restoration status. The 1996 and 1978, marshes were generally warmer than the REF, 1987 and River sites in June 1999, for example (Fig. 2.51-2.54).

Juvenile Chinook Density

Fish densities based on catches per unit area of tidal channel surface varied among sites between 1998 and 2002, ranging from 0.003 and 0.02 fish per m^2, but no definitive pattern was evident for the relative rank of each marsh (Fig. 2.2). At the 1978 marsh, fish densities were generally near zero in June-August. Fish densities were generally highest in the REF and 1987 marshes. Fish densities at all marshes were consistently low in 2000.

Juvenile Chinook Size

Fish in 2001-2002 entered the estuary at a larger size than fish in 1998-2000, and continued residence in better condition (Fig. 2.6). A comparison of the regression lines reveals a significant difference between 2001-2002 and 1998-2000 at the 0.05 level. Length and weight frequency graphs from each year illustrate that more small fish by length (40, 55, 60 mm) were caught in 2001 and 2002, but more large fish by weight were also caught in those years (Fig. 2.7). Weight distributions between the 1998-2000 and 2001-2002 groups were compared with a Two-Sample Kolmogorov-Smirnov Goodness of Fit test and a significant difference was detected (p = 0.00).

Diet Composition

Juvenile Chinook salmon foraging in the Salmon River estuary in 1999-2002 ate diverse prey items, including a range of crustaceans and insects, representing 117 taxonomic groups. These data were condensed into 33 groups based on wet weight before percent composition analysis, and further condensed into 9 categories (Crustacean; Diptera adult; Diptera emergent, larvae and pupae; Trichoptera emergent; Homoptera (Leafhoppers); Other marsh insects; Oligo-, Polychaetes; fish larvae; and Other) to simplify graphical and tabular representation (see Table 5). Model input was based upon original 33 taxonomic groups (see Appendix B).

Average monthly percent diet composition across all years for the REF and 1987 sites differed from that of the 1978 and 1996 sites. In general, more crustaceans were found in diets from the REF and 1987 sites, while more dipterans from all life stages were found at the 1978 and 1996 sites (Fig. 2.8). Emergent and larval trichopterans were common in the early season in the 1978 and 1996 sites (Fig.2.9). We found significant seasonal differences (Kruskal-Wallis Comparison of Means) in prey groups (across all years) among sites, including crustaceans and dipterans from April to July and homopterans and cantharid beetles in the late season (Table 2.7). When pooled over all months and years, leafhoppers and polychaetes were found in substantial percentages in the diet composition of fish collected in the REF and 1987 sites (8%, 9% respectively in the REF; 3%, 7% respectively in the 1987 site), but were much less prevalent in the diet (< 1%) from other sites.

In general, late in the juvenile salmon outmigration season (July, August), diets in REF and 1987 sites shifted and were composed of a greater percentage of marsh-derived prey such as dipteran adults, leafhoppers and beetles (Family: Cantharidae). Notable differences in diet composition, e.g. leafhoppers, were even more interesting when prey energy was considered (see Prey Energy). Leafhoppers were also found in small average percent for short periods in the 1978 and 1996 marshes (7% June 2001, 11% July 2000, respectively).

Prey Energy

Overall, semi-aquatic dipterans, trichopterans and other marsh insects were almost twice as energy rich as crustacean prey (Table 2.1). Mean energy densities were for 3.8-8.9 kJ/g WW for dipterans and 5.8-7.8 kJ/g WW for trichopterans, but only 2.5-3.7 kJ/g WW for crustaceans. Terrestrial invertebrates exhibited some of the lowest water content and highest energy content recorded. Energy densities of homopterans and hymenopterans were > 12 kJ/g WW.

Modeled Growth Potential

Question 1: Were there differences in modeled growth potential among marshes and were the differences related to marsh age?

My first model simulation established the relative comparisons in growth rate potential assuming equal fish size and consumption rates among marshes (Fig. 2.10), whereas subsequent simulations accounted for consumption differences and disparities in size among marshes (Fig. 2.11; Fig. 2.12, respectively). Refer to Table 2.6 for adjustments to consumption rate, and Table 2.4 for fish size inputs. Results showed modeled growth rates to be comparable among marsh sites. Both restoring and reference marshes offered foraging and growth opportunities to juvenile Chinook salmon in the simulations. Although clear differences in growth potential among marshes were found, no direct relationship to marsh age was evident. Specific growth rates in the restoring sites ranged from 0.001-0.07 g/g/d, with modeled growth rates generally beginning to decline by June, especially at the 1996 marsh (see 2002). While the REF never achieved the high modeled growth rates found in some periods at the restoring sites (0.07 g/g/d; 1978 site, June 2001), the rates in almost all cases were above 0.015 g/g/d. The REF marsh reflected more stability in modeled growth rates than the restoring sites (except in 2000), which were in general characterized by greater variability (Fig. 2.14). No negative modeled growth rates were observed in the adjusted model.

Question 2: Were the differences influenced most by temperature, prey quality, or prey quantity?

The influence of temperature at each site was evaluated with the model when diet proportion and consumption rate were held constant (Fig. 2.15). Among-site differences in temperature had little effect on modeled specific growth rate, except in June and July in the 1996 and 1978 marshes when modeled growth rates dropped.

The presence of particularly high-energy prey, such as trichopterans, corresponded to extremely high values of modeled specific growth rate. Trichopterans tended to constitute a large proportion of the diet because a single trichopteran larvae or emergent pupae was nearly equal volume to a juvenile Chinook salmon's stomach. Trichopterans contain 5.8 kJ/g WW compared to crustaceans which generally contained about 4 kJ/g. Homopterans were also important high energy (12 kJ/g WW) prey in diets, but only late in the season.

The influence of prey quantity was determined by evaluating tidal ration among sites, and adjusting consumption rates appropriately when tidal rations were significantly different (Fig. 2.11). Adjusting consumption rate drastically affected modeled growth in several cases (Table 2.8) with a several fold increase in some cases.

Question 3: Does disparity in fish size among marshes affect modeled growth potential?

Changes in modeled growth rate were seen among marshes when size disparity was considered (Fig. 2.12). In most cases, the change in modeled growth rate was small. However, notable differences were found in the REF site

(0.02 g/g/d to 0.01 g/g/d) due to accounting for larger than average size fish (which grow slower). An additional simulation was performed with size data from the REF site where a bimodal distribution in fish size was found in July 2000, March 2001 and June 2001, and provided further evidence of faster growth rates for smaller fish (Fig. 2.13).

Question 4: Does the model provide an effective tool to assess development of juvenile Chinook salmon habitat?

The model integrates the potential effects of differences in diet, prey quantities, and temperature regimes among marshes and provides an estimation of relative ecosystem function among recovering and reference marshes. Based on my analysis, I was able to compare function among sites, but also the relative effects of a variety of fish growth controlling factors. Comparisons with empirical data which validate the model will further reveal the utility of bioenergetics modeling in determining estuarine restoration success in terms of juvenile Chinook salmon habitat.

Empirical and Model Comparisons

Empirical and model-derived growth rates were comparable in 2000 and 2002, but the apparent growth rate in 2001 was much higher than the modeled value (Table 2.9). The implication of these numbers was limited due to the fact that they are based on two estimates of size: 1) initial size was based on an average by week, and 2) all weights were based on a linear regression model which varies from year-to-year in the estuary and was unknown for freshwater.

More accurate data were available from 2003 when fish were individually marked and recaptured in the REF marsh where juvenile salmon had comparable growth rates to the model of 0.02 g/g/d (D. Hering, unpublished data). To determine empirical growth mark and recapture of individual fish was required and handling stress may effect growth rate (E. Volk, pers. comm.). Refer to Bottom et al. (2005) for more information regarding growth estimations from mark and recapture studies from the Salmon River.

Discussion

Bioenergetics modeling can be a useful predictive tool to restoration scientists; a tool that can be tested and corroborated, to determine the relative functional state of recovering ecosystems for foraging fish. Using a bioenergetics model I was able to compare performance of three recovering and one reference marsh in terms of modeled juvenile Chinook growth potential. The power of modeling is the ability to distinguish differences among marsh contributions to salmon in comparable ecological terms. After caveats related to disparities in consumption rate and fish size were resolved, differences in modeled growth potential among marshes were evaluated. The most important findings of the work include: 1) positive modeled growth in all marsh environments, 2) high modeled growth rates (0.07 g/g/d) in the newest restoration site, and 3) the relative stability of the REF in terms of modeled growth compared with the more variable restored sites. The model clearly illustrated the utility of restoring estuarine marshes for juvenile salmon habitat as, for the most part, each marsh was occupied by foraging, growing fish. Although restored marshes were variable in performance, average growth conditions equal those found at the reference site. High modeled growth rates in the newest restoration site reflect the abundance of higher-energy insects (e.g., Trichoptera larvae and emergent adults) in the diets of the fish foraging there. Higher energy insect prey may offset effects of increasing water temperature in 1978 and 1996 marshes. Ecological processes associated with a recovering marsh, such as increased sediment organic matter may trigger insect proliferation and aggregation, in turn contributing to increased growth efficiency and relative instability of restoring versus reference sites. These findings aid natural resource managers and advance salmon recovery by contributing to the understanding of early recovery phases to foraging juvenile salmonids, and trade-offs between restored and reference environments.

To evaluate the physiological and ecological trade-offs confronting a juvenile Chinook salmon foraging in the marshes of the Salmon River, an integrative measure was the most informative. Indirect measures of foraging success and density between marshes include fish abundance (upon standardized sampling), diet composition and stomach fullness. Both were useful descriptive measures of fish-habitat interaction, but the integrative quality necessary to assess the difference between physiological characteristics of the fish and physical/biological characteristics of the site was lacking. Although measures of foraging (e.g., stomach fullness) are indicators of the ecological state of recovering estuarine wetlands (Shreffler et al. 1992; Miller & Simenstad 1997), modeling technologies enhance our capability to make analogous evaluation between sites using energy transfer as the common currency.

Physiological and external controls of fish metabolism are well understood, can be measured independently and used to build an effective bioenergetics tool to assess growth or consumption. The growth rate of an individual fish is a highly pliant, species- and size-specific response to environmental conditions and food availability (Demers et al. 2000), so focusing on growth response, we can use the bioenergetics model as an assessment tool for comparing recovering estuarine wetlands with references.

In systems such as the Salmon River estuary, measuring actual growth due to the differing conditions present in the recovering estuarine marshes was difficult because of tidal regimes, which prevent isolation of a fish in a particular environment. A bioenergetics model provided an alternative approach for exploring the linkage between relative fish growth, marsh conditions and restoration time.

Sommer et al. (2001) used a bioenergetics model to assess differences in growth patterns between juvenile Chinook salmon migrating in the Sacramento River mainstem and those using an engineered agricultural floodplain. Juvenile Chinook migrating in the floodplain ate diets consisting of high-energy dipterans compared to diets of fish migrating in the river mainstem that were mostly composed of lower-energy zooplankton (8.9 kJ/g WW; Table 2.1, and 2-3.6 kJ/g WW; Hanson et al. 1997, respectively). Modeled growth rates were higher on the floodplain despite higher temperatures (Sommer et al. 2001). For fish foraging in an estuary, growth depends upon the complex interactions between quality and availability of prey resources at a site, the fish's ability to access that site, water temperature, and other physical factors affecting prey acquisition and fish metabolism (Simenstad & Cordell 2000). Diet and temperature are important external regulators of growth, so optimum temperature conditions and an abundance of desirable prey items increases rearing potential. External temperature determines a fish's metabolic rate, which in turn mandates consumption. Wissmar and Simenstad (1988) suggested that the amount of energy left for growth for migrating *Oncorhynchus keta* (chum salmon) was directly dependent on levels of prey standing stock. Sites with diets dominated by high-energy prey (e.g., trichoptera, homoptera) achieved greater growth potentials than those with relatively higher consumption rates. However, prey quality may be equally or more important than consumption rate when looking at growth potentials (Beauchamp et al. 1989; Boisclair & Leggett 1989). Other factors affecting rearing potential include salinity, levels of dissolved oxygen, photoperiod and flow (Jobling 1994). In restoring estuarine marshes, any number of these factors can be altered from conditions present in reference sites.

Model simulations were designed with the highest conformity to nature as possible. The model simulations revealed important aspects affecting the function of recovering estuarine wetlands. High-energy prey was available in the recovering marshes, so modeled growth rates were very high at times in those areas. The stability of the REF marsh follows common ecological ideas regarding relative stability in natural systems, compared with high variability in disturbed ones (Tilman 1996). However, temperature had little affect on specific growth potential until late in the season (June and July) when growth potential dropped at the 1978 and 1996 marshes, but homopterans (important prey June-August in the 1987 and REF marshes) could be a key component responsible for offsetting the negative growth effects of rising water temperature in the late season.

Whether fish arrive in the estuary early in the migration period or late, timing is also an essential component to gaining maximum possible benefits from estuary rearing. Bioenergetics models allow the consideration of a range of scenarios across seasons and could be used to test where, when, and under what conditions the fish benefit most, in addition to assessing recovery status. This model fails to incorporate natural processes of fish movement with the tide. As the marsh channels dewater, fish are forced into the river mainstem where temperature, diet conditions and predation pressure change. Including this behavior into the model will improve its accuracy and give us a more complete picture of estuarine growth potential and restored marsh function. Further refinements to the model will help to fully understand the spatial and temporal implications of foraging choices, the importance of a mosaic of habitat types arrayed in an estuarine landscape, and the implications of variable life history type on marsh performance.

A variety of growing conditions were prevalent at the Salmon River estuary, both within the various marshes and across the sampling years. Data revealed a significantly higher condition factor for juvenile Chinook salmon caught in the marsh fyke nets in 2001 and 2002, compared with 1998-2000 (see Fig. 2.6). I have made the assumption of constant immigration and emigration from the system across sampling years, and from these data fish apparently arrived in better condition beginning in 2001 and continued to follow an elevated growth trajectory when compared with 1998-2000 fish. These data suggest conditions in the watershed may "set" the growth trajectory, regardless of estuarine conditions. To fully understand the differences in condition between years it would be necessary to know condition of fish upon entry to the estuary. Those data were not available for the Salmon River or any other Oregon coast river (Steve Johnson, pers. comm.).

This model does not duplicate nature, nor does it provide actual measures of growth, but instead reveals potential differences in patterns and processes among sites. Manipulations of model inputs enabled us to detect key elements affecting juvenile salmon in various habitats. Conditions in recovering wetlands may deviate from those observed in undiked marshes due to variability in elevation, structural differences, and invertebrate composition, which differ among marshes (Frenkel & Morlan 1990, Gray et al. 2002). How these factors influence fish growth and their relationship to restorative procedures is of interest to those attempting to rehabilitate depressed Chinook populations with dike removal and predict fish response.

Assessment of restoration at the community level attempts to overlay normal community variability on a restoring ecosystem signal and then make testable predictions. Heatwole et al. (2004) illustrated how a predictive, quantitative model could be used to deduce patterns in invertebrate abundance and composition in restoring pocket estuaries on Whidbey Island, WA. Combining predictive models such as this with a greater understanding of prey energy, energy transfer and potential fish growth represents an example of how bioenergetics models can contribute to understanding the implications of restoration for foraging juvenile salmonids.

Modeling experimentation revealed what benefits and costs for juvenile Chinook were associated with physical, biological and chemical differences found among restoring sites of different ages. Developmental shifts among marshes include the proliferation of high-energy insects in relation to plant die-backs, tidal channel deepening and narrowing, and increased overhanging vegetation which may affect water temperature. The influence of those factors may be generalized in terms of bioenergetic trade-offs. I linked those factors to marsh recovery status and modeled growth potential and increased knowledge regarding the importance of restored estuarine wetlands even in early recovery phases. Examples from the Salmon River provided a new template to apply bioenergetics modeling and resulted in a better understanding of how recovering estuarine marshes contribute to foraging of juvenile Chinook salmon.

Tables

Table 2.1: Prey Water Content, Energy Content and Fraction Indigestible (measured and literature values)*

	Dry Weight to Wet Weight Ratios		Prey Energy by Bomb Calorimetry		Percent Indigest-ible
Prey Item	*Measured*	*Literature*	*kJ/g dw*	*kJ/g ww*	
Corophium spp.	0.23	0.21 (Amphipoda)[a]	13.42	3.09	26[a]
Eogammarus spp.	0.27	0.21 (Amphipoda)[a]	11.49	3.1	
Isopoda	0.27		9.11	2.46	50[g]
Mysidae	0.17	0.11 (Mysids)[c]	20.9	3.55	15[c]
Talitridae	0.24		12.68	3.04	
Chironomidae adult			25.56	3.83	
Dolichopodidae adult	0.27	0.17 (Diptera)[a]	22.98	6.2	
Other Diptera adult	0.40	0.4 (Empididae)[b]	22.31	8.92	12[a]
Diptera larv	0.13		19.85	2.58	5[e]
Trichoptera emerg, larv	0.27	0.21 (Trichoptera)[a]	21.53	5.81	5[e]
Trichoptera adult	0.34	0.59 (Trichoptera)[b]	22.81	7.76	10[e]
Cantharidae adult	0.35	0.37 (Coleoptera)[b]	22.68	7.94	30[d]
Coleoptera adult	0.35	0.37 (Coleoptera)[b]	22.77	7.97	35[d]
Homoptera	0.50	0.48 (Heteroptera)[b]	24.54	12.27	20[d]
Hemiptera adult and immature	0.46	0.48 (Heteroptera)[b]	23.75	10.93	15[e]
Lepidoptera adult, larv	0.36	0.50 (adult); 0.25 (larvae)[b]	23.62	8.5	15[e]
Hymenoptera	0.58		21.85	12.67	30[e]
Arachnids	0.26	0.31 (Araneae)[b]	20.46	5.32	15[e]
Oligo-, Polychaetes	0.12	0.18 (Oligochaetes)[a]	16.51	1.98	2[e]
Fish	0.20	0.21 (Pisces)[a]	17.84	3.57	0[e]
Algae and Other Plant	0.22		17.52	3.85	25[e]

*Sources: [a]Groot et al. 1995; [b]Bradman & Reyer 1999; [c]Woods & Valentino 2003; [d]Empirical data from bomb calorimetry; [e]Best Guess

Table 2.2: Model Temperature Inputs for 2002

Year	Date	*1978*	*1987*	*1996*	*REF*	*River*
Apr-02	4/4	12.19	11.72	13.93	11.83	10.02
	4/5	11.14	11.05	12.02	11.09	10.00
	4/6	11.66	11.15	12.00	11.51	9.63
	4/7	12.39	11.44	12.70	11.61	9.54
	4/8	11.88	10.86	11.99	10.35	9.10
	4/9	11.53	10.19	11.75	10.77	9.88
	4/10	11.40	10.31	11.47	10.75	8.95
	4/11	10.90	9.92	10.59	10.43	8.72
	4/12	11.79	10.70	11.70	11.21	9.23
	4/13	11.58	10.77	11.21	11.16	9.79
May-02	5/4	12.01	13.36	13.68	10.58	10.10
	5/5	17.16	12.53	12.27	11.19	10.01
	5/6	12.57	11.46	12.16	10.41	9.17
	5/7	12.66	11.49	13.26	10.01	9.10
	5/8	14.86	11.30	12.85	9.85	8.94
	5/9	12.65	11.13	13.47	10.78	9.63
	5/10	14.61	11.60	15.00	12.27	10.46
	5/21	15.30	12.62	13.80	12.96	11.56
	5/22	15.86	13.19	14.29	13.17	12.02
	5/23	16.43	13.06	14.37	13.12	11.69
Jun-02	6/2	17.48	15.73	17.24	15.03	13.62
	6/3	16.52	15.24	16.04	14.80	13.66
	6/4	16.55	14.97	16.64	15.89	13.78
	6/5	16.21	14.75	16.10	15.68	13.92
	6/6	17.11	15.58	15.61	14.64	13.63
	6/7	16.28	14.29	14.34	13.31	12.88
	6/18	15.65	14.70	14.96	14.86	13.71
	6/19	17.23	15.89	15.34	14.75	13.94
	6/20	18.54	15.07	16.94	15.14	14.04
	6/30	17.60	16.42	18.08	16.93	14.39
Jul-02	7/1	18.56	17.40	18.20	16.14	14.43
	7/2	18.24	17.21	17.78	15.61	14.26
	7/3	18.63	16.58	17.81	15.89	14.60
	7/4	19.13	16.52	17.48	15.68	14.74
	7/5	19.03	16.14	17.54	15.90	14.68
	7/6	20.34	15.73	18.25	16.54	15.28
	7/16	19.30	14.24	17.26	16.55	13.92
	7/17	19.31	14.90	17.47	17.40	14.77
	7/18	19.65	15.62	18.06	17.98	15.19
	7/19	19.39	15.82	18.26	18.22	15.58

Table 2.3: Proportion of Maximum Consumption (P)*

Month Interval	Year	Start Weight	End Weight	P-value
March-April	1999	0.96	1.47	**0.470**
April-May	1999	1.47	3.01	**0.601**
May-June	1999	3.01	5.17	**0.486**
June-July	1999	5.17	7.05	**0.588**
March-April	2000	0.74	1.42	**0.472**
April-May	2000	1.42	2.7	**0.493**
May-June	2000	2.7	3.98	**0.484**
June-July	2000	3.98	6.96	**0.581**
March-April	2001	0.64	0.96	**0.314**
April-May	2001	0.96	2.83	**0.494**
May-June	2001	2.83	4.11	**0.409**
June-July	2001	4.11	7.96	**0.511**
July-August	2001	7.96	12.02	**0.542**
April-May	2002	1.65	3.36	**0.508**
May-June	2002	3.36	6.74	**0.552**
June-July	2002	6.74	9.66	**0.439**

*P-values are based on average growth increment of trap net fish, and average diet and temperature conditions.

Table 2.4: Fish Weights as Used in Various Model Simulations

	Model Simulation 1 & 2	***Model Simulation 3***				***Model Simulation 4***
	Mean Weight of Trapnet Fish	Mean Weight of Trapnet Fish by Sampling Site				Mean Weights of Bimodal Dis-tribution
Sampling Period	All Sites	1978	1987	1996	REF	
Mar-99	*0.96*		*1.08*	*0.68*	*0.9*	
Apr-99	1.47	1.24	**2.04**	1.34	0.81	
May-99	3.01	2.4	3.27	3.06	2.71	
Jun-99	5.17		3.8	3.3	**5.59**	
Jul-99	7.05		6	5.7	**8.17**	
Mar-00	0.74	0.6	0.6	0.74	**0.8**	
Apr-00	1.42	1.6	1.43	1.45	1.37	
May-00	2.7	2.6	**3.75**	2.49	2.71	
Jun-00	3.98	4.7	4.42	4.2	3.65	
Jul-00	6.96		7.09	6.62	7.17	6.26; 10.54
Mar-01	0.64			0.6	2.68	0.65; 10.8
Apr-01	0.96		1.15	0.82	1.11	
May-01	2.83	1.63	**3.41**	2.45	**3.68**	
Jun-01	4.11	3.3	3.88	3.8	**4.91**	4.47; 13
Jul-01	7.96		7.62	5.99	8.72	
Aug-01	12.02		12.34		11.68	
Apr-02	1.65	1.31	**1.8**	**1.57**	**1.89**	
May-02	3.36	3.05	3.2	3.03	**3.9**	
Jun-02	6.74		6.82	6.28	6.86	
Jul-02	9.66	9.57	9.93	8.52	**11.44**	

****bold** indicates site responsible for a significant difference; *italics* indicates values taken from 1998, since no 1999 values were available; underline indicates bimodal distribution in fish size

Table 2.51: Average Percent Composition of Major Prey Items from Each Sampling Period by Wet Weight among Marshes

REF *n* =	*11*	*8*	*9*	*8*	*9*	*10*	*8*	*10*	*10*	*10*	*10*
Prey Items	**Mar-99**	**Apr-99**	**eMay-99**	**lMay-99**	**Jun-99**	**Jul-99**	**Mar-00**	**Apr-00**	**May-00**	**Jun-00**	**Jul-00**
Crustaceans	0.28	**0.5B**	**0.84**	**0.60**	**0.46**	**0.67**	**0.40**	**0.52**	**0.87**	0.30	0.07
Diptera adult	0.00	0.05	0.00	0.11		0.02	0.08	0.03	0.01	0.24	0.02
Diptera larv,pup,emer				0.06		0.01		0.01	0.05	0.05	0.00
Trichoptera	0.17		0.10	0.06			0.28	0.10			
Homoptera (Leafhoppers)						0.06				0.02	**0.49**
Coleoptera adult				0.03	0.33	0.10				0.02	0.14
Other marsh insects	0.02	0.04	0.01		0.00	0.06		0.02		0.11	0.03
Oligo-, Polychaetes	0.31	0.09	0.05				0.15	0.01			
Fish/Fish larvae	0.00	0.23					0.08	0.22		0.08	
Other	0.21	0.02	0.00	0.14	0.21	0.07	0.00	0.10	0.06	0.17	0.26
1996 *n* =	*10*	*10*	*10*	*10*	*10*	*7*	*10*	*10*	*10*	*10*	*10*
Crustaceans	0.08	0.15	0.01	0.02	**0.98**	**0.91**	0.27	0.12	0.16	0.28	0.28
Diptera adult	0.15	0.14	0.10	0.02	0.01	0.01	0.11	0.15	0.18	0.09	0.15
Diptera larv,pup,emer	0.23	**0.47**	0.21	0.00	0.01	0.07	0.28	**0.45**	0.32	**0.53**	0.19
Trichoptera	0.37	0.06	**0.61**	**0.93**			0.20	0.12	0.30		
Homoptera (Leafhoppers)											0.11
Coleoptera adult									0.03	0.02	0.01
Other marsh insects	0.04	0.02		0.02		0.01	0.03	0.12	0.00	0.07	0.08
Oligo-, Polychaetes	0.02	0.15									
Fish/Fish larvae							0.09				
Other	0.12		0.07	0.00	0.01	0.00	0.02	0.03	0.01	0.01	0.18

Table 2.52: Average Percent Composition of Major Prey Items from Each Sampling Period by Wet Weight among Marshes, Continued

1987 *n =*	*8*	*10*	*4*	*7*	*0*	*6*	*2*	*3*	*6*	*5*	*10*
Prey Items	**Mar-99**	**Apr-99**	**eMay-99**	**lMay-99**	**Jun-99**	**Jul-99**	**Mar-00**	**Apr-00**	**May-00**	**Jun-00**	**Jul-00**
Crustaceans	0.24	0.37	**0.64**	0.30		0.24	0.91		**0.78**	0.19	**0.44**
Diptera adult	0.00	0.06	0.24	0.00		0.21		0.05	0.20	0.19	0.11
Diptera larv,pup,emer	0.22	0.00	0.01	0.00					0.01	0.04	0.01
Trichoptera	0.33	**0.43**		0.15				0.27		0.09	
Homoptera (Leafhoppers)						0.04					0.04
Coleoptera adult			0.01			0.16					0.04
Other marsh insects	0.04					0.17		0.02	0.01	0.12	0.02
Oligo-, Polychaetes	0.17	0.12									0.02
Fish/Fish larvae		0.00				0.02	0.09	**0.66**			
Other (mostly algae)		0.03	0.10	**0.55**		0.16			0.00	0.37	0.32
1978 *n =*	*10*	*5*	*1*	*0*	*0*	*0*	*3*	*1*	*2*	*0*	*0*
Crustaceans	0.22	**0.44**					**0.64**		0.06		
Diptera adult	0.06	0.12	**0.99**				0.33	0.05	0.17		
Diptera larv,pup,emer	0.19	0.04					0.02		0.02		
Trichoptera	0.39	0.38									
Homoptera (Leafhoppers)											
Coleoptera adult									0.15		
Other marsh insects	0.08	0.00							**0.58**		
Oligo-, Polychaetes	0.01	0.02							0.03		
Fish/Fish larvae								**0.95**			
Other	0.05		0.01								

Table 2.53: Average Percent Composition of Major Prey Items from Each Sampling Period by Wet Weight among Marshes, Continued

REF *n* =	*4*	*7*	*7*	*7*	*7*	*7*	*8*	*7*	*7*	*7*
Prey Items	**Mar-01**	**Apr-01**	**May-01**	**Jun-01**	**Jul-01**	**Aug-01**	**Apr-02**	**May-02**	**Jun-02**	**Jul-02**
Crustaceans	0.25	0.33	**0.43**	0.16	0.11	0.09	**0.46**	0.17	0.18	0.08
Diptera adult	0.25	0.12	0.01	**0.46**	0.07	0.02	0.05	0.11	0.05	0.05
Diptera larv,pup,emer		0.02	0.02	0.03	0.01		0.01	0.30	0.02	0.01
Trichoptera	0.17	0.14								
Homoptera (Leafhoppers)			0.00	0.03	**0.46**	**0.53**		0.01	0.39	0.10
Coleoptera adult			0.07	0.06	0.09	0.17	0.01	0.07	0.14	**0.44**
Other marsh insects		0.06		0.17	0.07	0.02	0.08	0.19	0.08	0.08
Oligo-, Polychaetes	0.08	0.23	0.33	0.01			0.39	0.05		
Fish/Fish larvae		0.10								
Other		0.01	0.06		0.17	0.16		0.10	0.14	0.24
1996 *n* =	*1*	*7*	*8*	*7*	*7*	*0*	*6*	*6*	*6*	*7*
Crustaceans	**0.80**	0.01	0.09	0.09	0.23		0.09	0.07	0.13	0.19
Diptera adult		**0.60**	0.10	0.06	0.08		0.36	0.21	0.01	0.27
Diptera larv,pup,emer	0.20	0.14	**0.71**	**0.77**	**0.40**		0.25	0.13	**0.48**	0.35
Trichoptera		0.24	0.02	0.01	0.06			**0.51**		
Homoptera (Leafhoppers)		0.00								
Coleoptera adult			0.01	0.00	0.03			0.07	0.00	0.06
Other marsh insects		0.00		0.06	0.06		0.25	0.02	0.01	0.09
Oligo-, Polychaetes		0.00			0.00					
Fish/Fish larvae										
Other			0.06		0.13		0.06		0.37	0.04

Table 2.54: Average Percent Composition of Major Prey Items from Each Sampling Period by Wet Weight among Marshes, Continued

1987 *n* =	*0*	*5*	*7*	*7*	*7*	*7*	*7*	*7*	*7*	*7*
Prey Items	**Mar-01**	**Apr-01**	**May-01**	**Jun-01**	**Jul-01**	**Aug-01**	**Apr-02**	**May-02**	**Jun-02**	**Jul-02**
Crustaceans		0.15	0.29	**0.48**	**0.80**	0.21	**0.75**	0.37	**0.75**	0.18
Diptera adult		0.22	0.13	0.03	0.04	0.06	0.07	0.06	0.05	0.26
Diptera larv,pup,emer		0.05		0.04		0.01	0.01	0.01	0.00	0.00
Trichoptera										
Homoptera (Leafhoppers)				0.00	0.01	**0.43**			0.04	0.03
Coleoptera adult			0.01			0.12		0.15	0.01	0.18
Other marsh insects		0.14	0.19	0.21	0.16	0.05	0.00	0.01	0.11	0.16
Oligo-, Polychaetes		**0.40**	0.30	0.10			0.08	0.12		
Fish/Fish larvae				0.09			0.02	0.13		
Other (mostly algae)		0.03	0.09	0.05		0.11	0.07	0.15	0.05	0.20
1978 *n* =	*0*	*0*	*6*	*7*	*0*	*0*	*7*	*6*	*0*	*2*
Crustaceans			0.19	0.03			0.33	0.24		0.11
Diptera adult			0.32	0.12			0.23	**0.43**		0.20
Diptera larv,pup,emer			0.08	0.17			0.05	0.01		0.02
Trichoptera							0.05			
Homoptera (Leafhoppers)			0.01	0.07						
Coleoptera adult			0.15	0.00			0.09	0.07		0.08
Other marsh insects			0.18	0.60			0.23	0.04		0.05
Oligo-, Polychaetes								0.06		
Fish/Fish larvae										0.34
Other			0.07				0.02	0.15		0.20

Table 2.6: Comparison of Means* for Tidal Ration and Consumption Rate Adjustments

Sampling Period	**Sites with Samples**	**Tidal Ration Significance Level**	**Site with Higher Value**	**Nominal Consumption Rate (from Table 4)**	**Adjusted Consumption Rate**
Mar-99	all	0.36		0.470	
Apr-99	all	0.70		0.601	
May-99	all	**0.04**	1996	0.486	0.689
Jun-99	REF, 1996	**0.02**	1996	0.588	0.847
Jul-99	REF, 1987, 1996	**0.01**	1996	0.609	0.918
Mar-00	all	0.16		0.472	
Apr-00	all	0.33		0.493	
May-00	all	0.92		0.484	
Jun-00	REF, 1987, 1996	**0.00**	1987 & 1996	0.581	0.763
Jul-00	REF, 1987, 1996	0.52		0.617	
Mar-01	REF, 1996	1.00		0.314	
Apr-01	REF, 1987, 1996	0.24		0.494	
May-01	all	**0.01**	REF	0.409	0.575
Jun-01	all	**0.00**	1978	0.511	0.712
Jul-01	REF, 1987, 1996	0.31		0.542	
Aug-01	REF, 1987	0.29		0.500	
Apr-02	all	0.29		0.508	
May-02	all	0.93		0.552	
Jun-02	REF, 1987, 1996	0.06		0.439	
Jul-02	all	0.34		0.432	

*One-way Analysis of Variance (ANOVA) or Kruskal-Wallis (KW) Comparison of Means

****bold** indicates significance at the 0.05-level

Table 2.7: Kruskal-Wallis Comparison of Mean Weight of Prey Items Among Marshes (1999-2002)

Prey Item	March	April	May	June	July	August
Crustacean	0.64	**0.00**	**0.00**	**0.00**	**0.03**	0.41
Diptera adult	0.15	**0.00**	**0.00**	**0.02**	**0.01**	0.34
Diptera immature	**0.01**	**0.00**	**0.00**	**0.00**	**0.00**	0.73
Trichoptera	0.93	0.16	**0.00**	0.60	0.55	
Homoptera		0.55	0.23	**0.00**	**0.00**	0.57
Cantharidae		0.07	0.88	**0.00**	**0.00**	**0.02**
Insects	0.73	0.22	**0.00**	**0.00**	**0.00**	0.18
Polychaetes	0.07	0.19	**0.01**	**0.05**	0.75	
Fish larvae	0.71	**0.01**	0.36	0.32	**0.00**	
Other	0.49	0.81	**0.00**	0.17	0.36	0.52

Table 2.8: Change in Growth Rate by Increasing Consumption Rate

Site	Sampling Period	Nominal Consumption Rate	Adjusted Consumption Rate	Nominal Modeled Growth Rate	Adjusted Modeled Growth Rate
1996	May-99	0.486	0.689	0.027	0.043
1996	Jun-99	0.588	0.847	0.005	0.014
1996	Jul-99	0.609	0.918	-0.002	0.006
1987	Jun-00	0.581	0.763	0.018	0.027
REF	Jun-00	0.581	0.763	0.010	0.019
REF	May-01	0.409	0.575	0.011	0.020
1978	Jun-01	0.511	0.712	0.047	0.070

Table 2.9: Comparison of Apparent and Modeled Growth Rates

Year	Number of Fish Recaptured	Average Residence Time (days)*	Average Apparent Growth Rate (g/g/d)	Average Modeled Growth Rate (g/g/d)
2000	25	20.24	0.0295	0.0218
2001	59	18.63	0.0812	0.0231
2002	109	31.35	0.0193	0.0178

*Residence time averages include all recaptured fish

Fig. 2.1: Composite aerial photograph of the Salmon River estuary from 2000 showing sampling sites (C) and locations of temperature loggers (T).

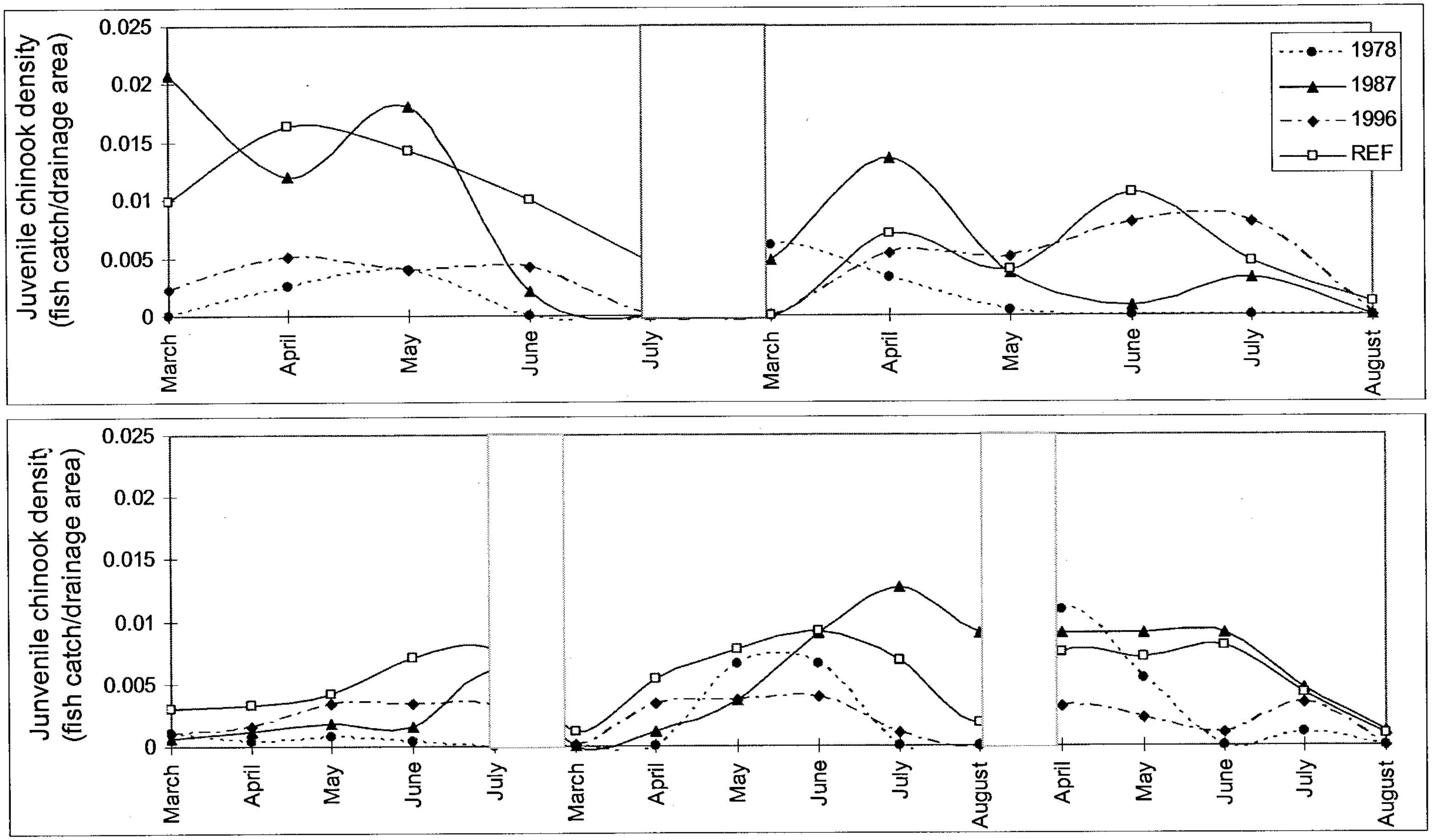


Figure 2.2: Density (normalized to tidal drainage area) of juvenile Chinook salmon caught in each of the marsh sites, 1998-2002.

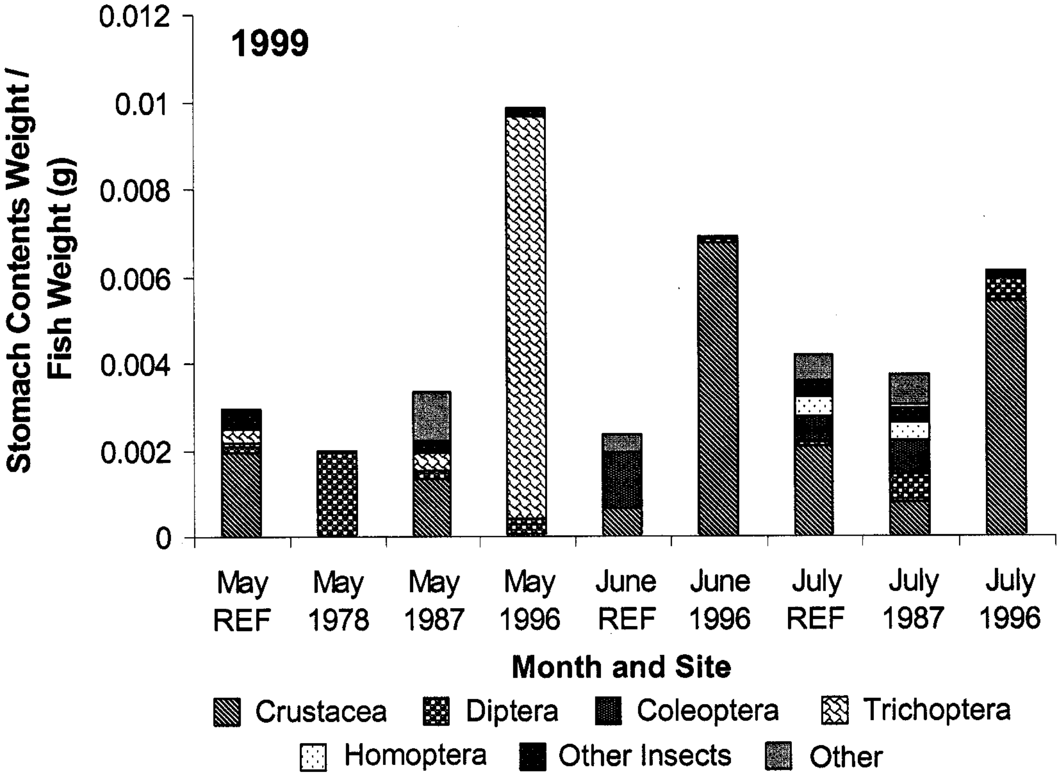


Figure 2.3: Stomach contents weight (g) standardized to fish weight (g) (tidal ration) for May-July 1999. Tidal ration was significantly greater at the 1996 marsh in each month.

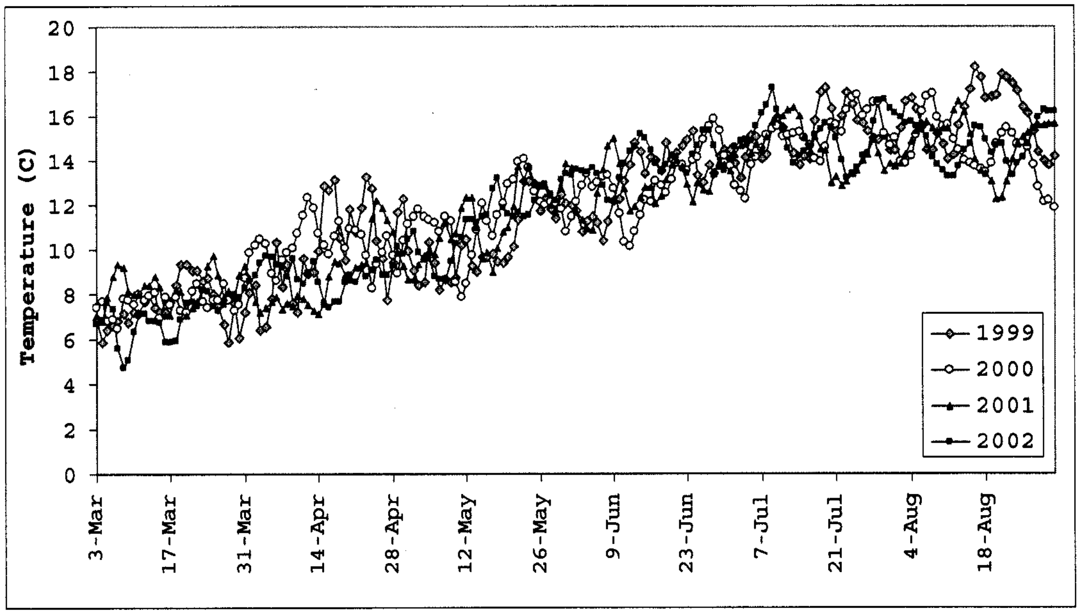


Figure 2.4: A comparison of daily averages of river temperature during the sampling season (March-August 1999-2002).

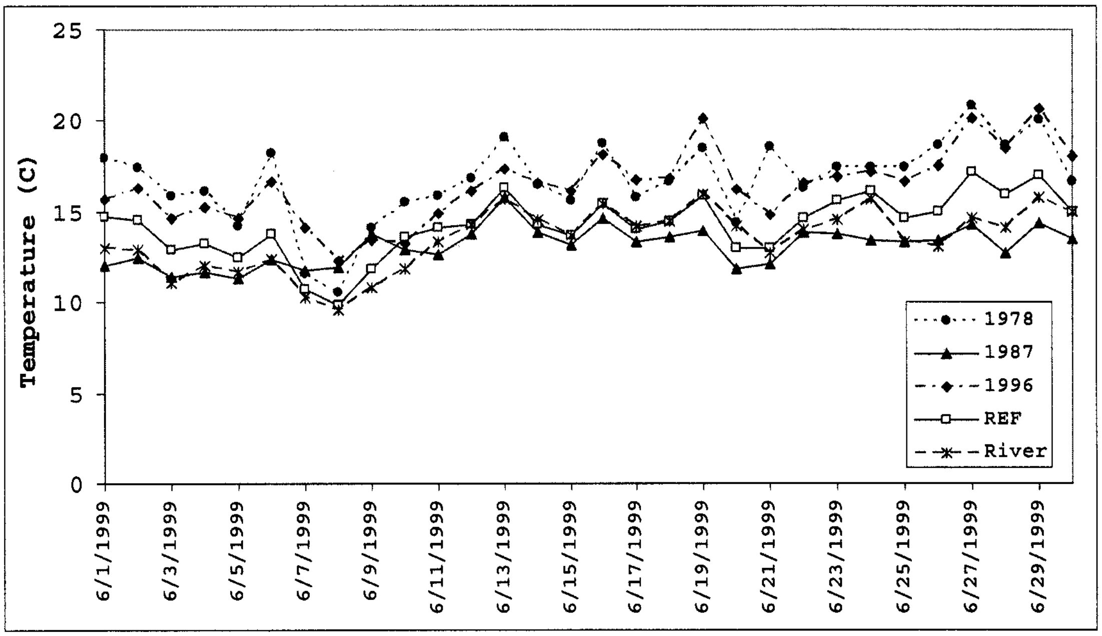


Figure 2.5: Average daily high tide temperature from each marsh site and the river for June 1999.

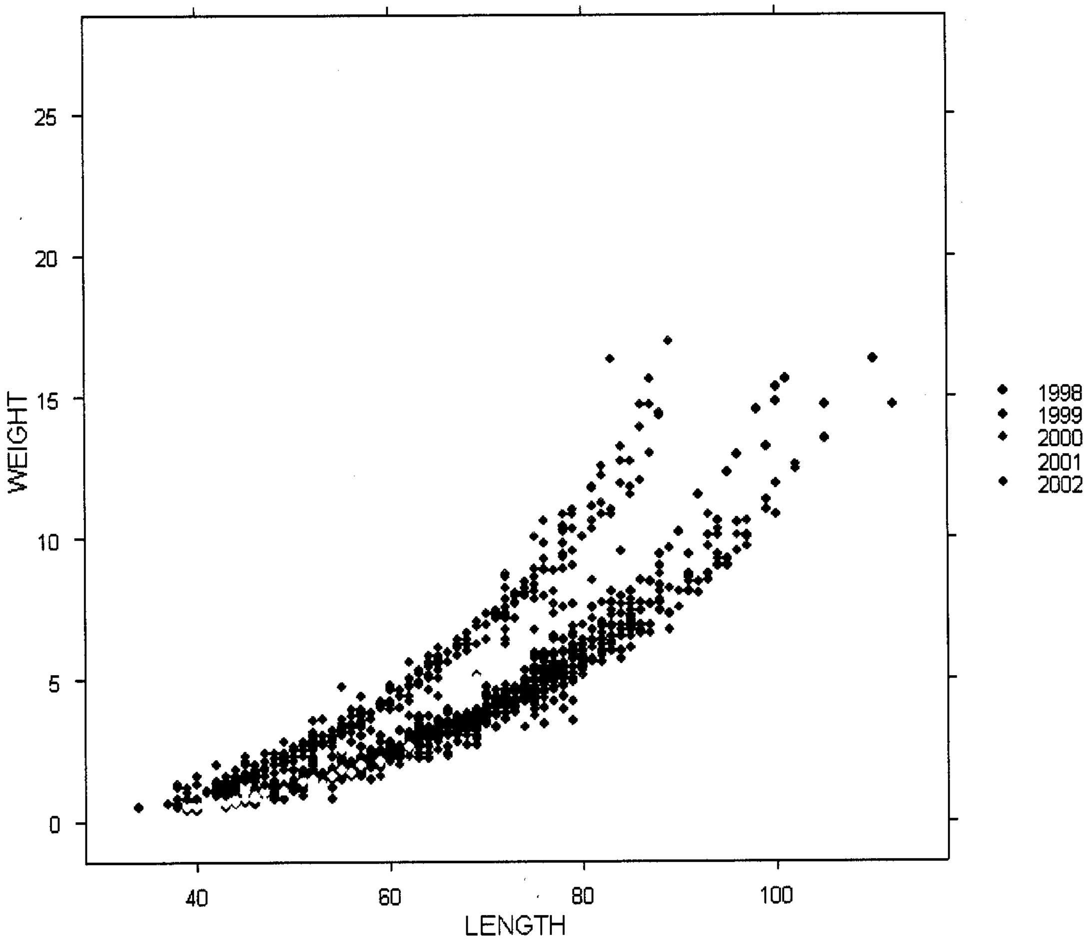


Figure 2.6: Length and weight data for all juvenile Chinook salmon captured at trap net sites during the sampling period (1998-2002).

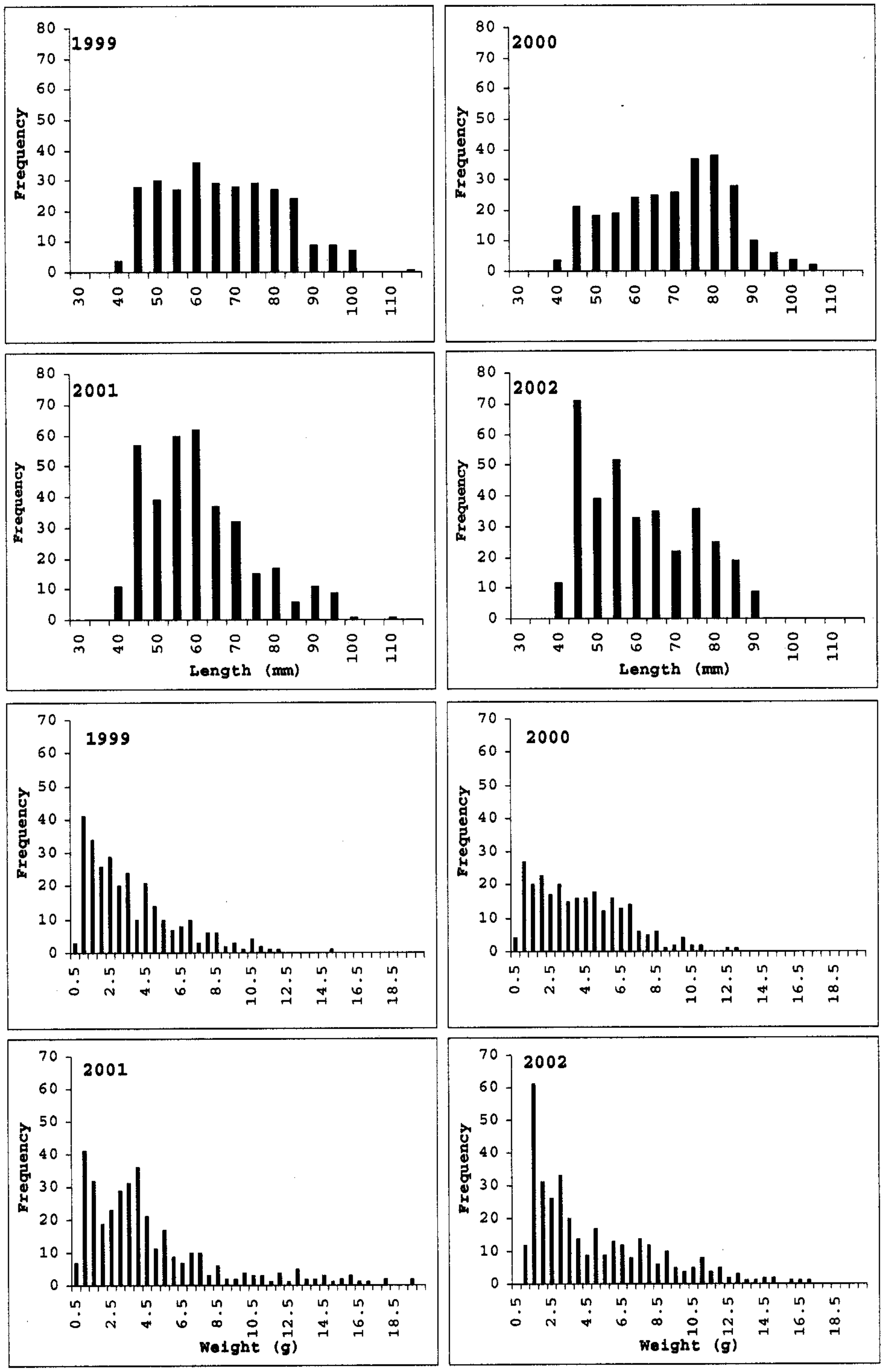


Figure 2.7: Length and weight frequency distributions for all juvenile Chinook salmon collected in the marsh trap nets (1999-2002).

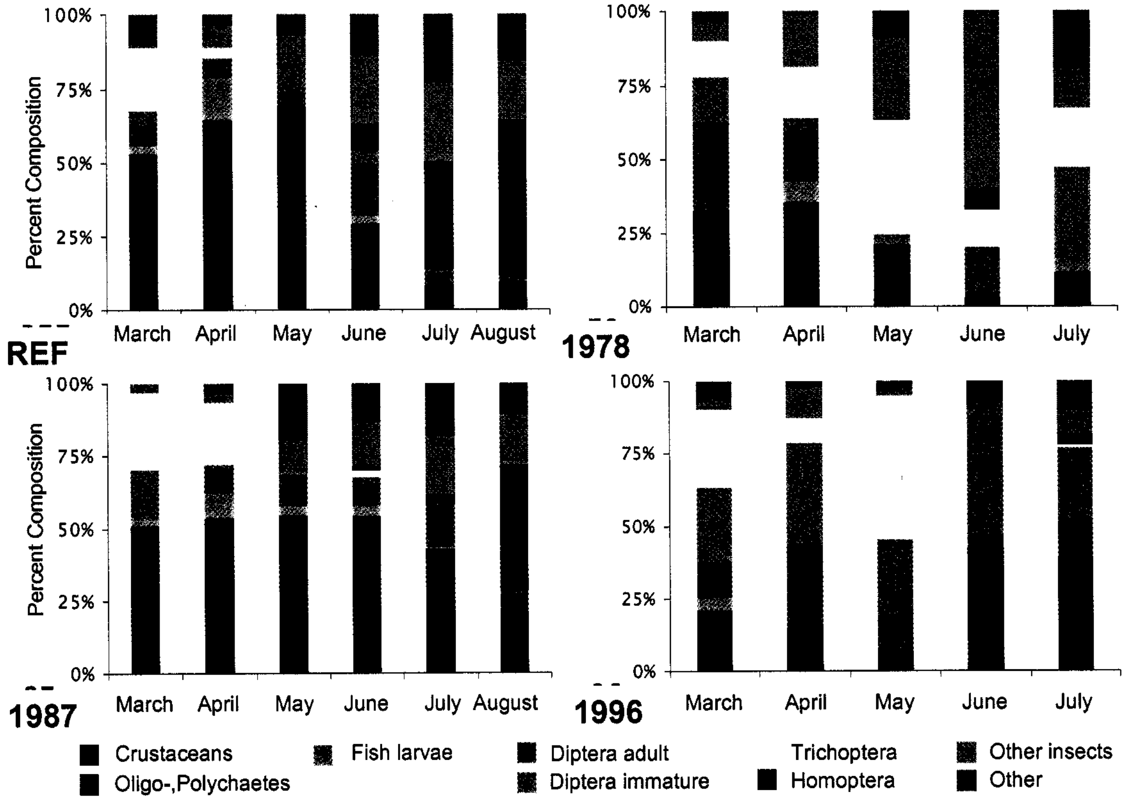


Figure 2.8: Average diet composition by month across all years for each marsh.

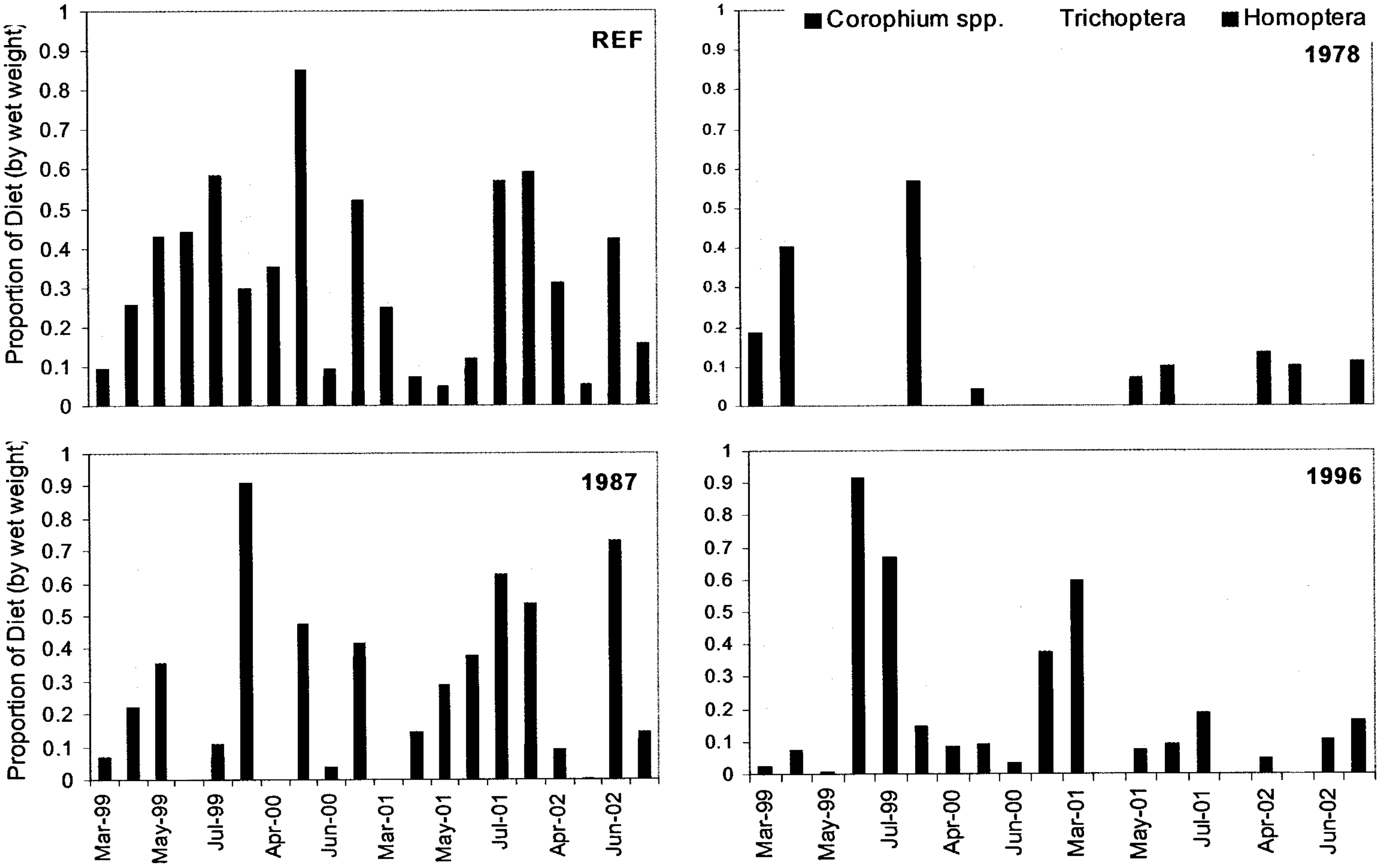


Figure 2.9: Average diet proportion by sampling period and site for *Corophium spp.* (fully aquatic), Trichoptera (semi-aquatic) and Homoptera (terrestrial).

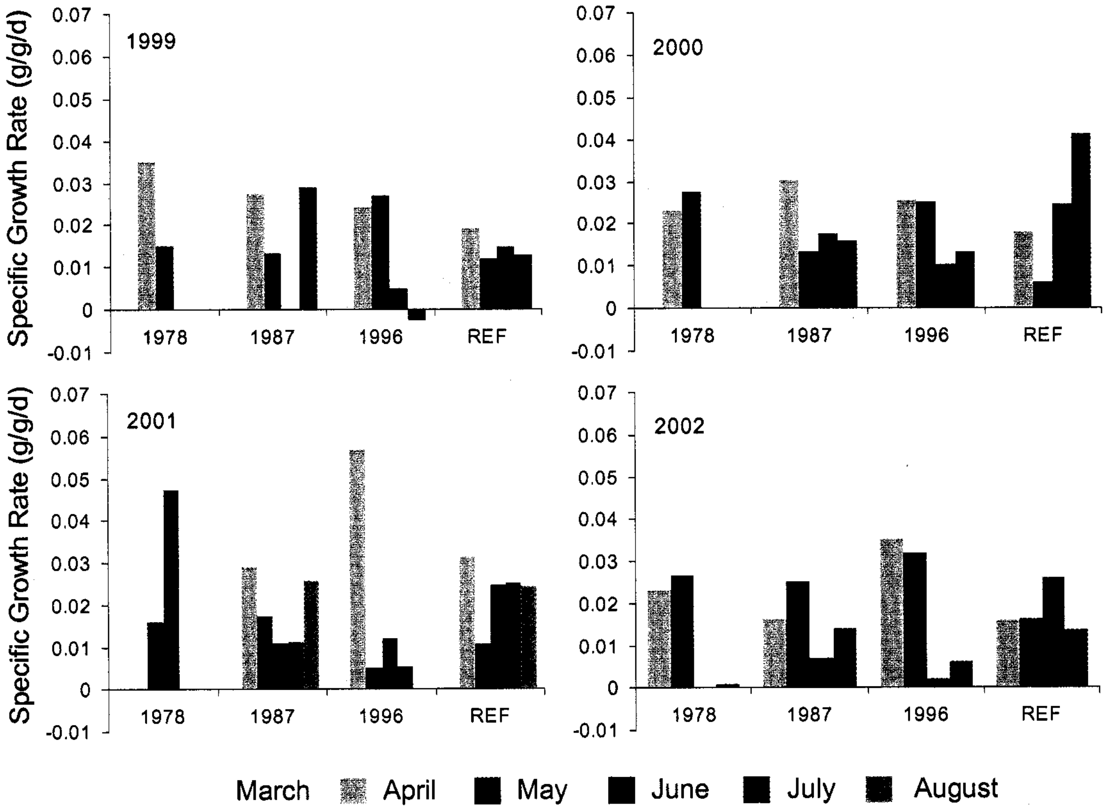


Figure 2.10: Model Simulation 1: Marsh-specific diet, marsh-specific temperature, equal (variable mean) start weight, and equal, generalized consumption rate.

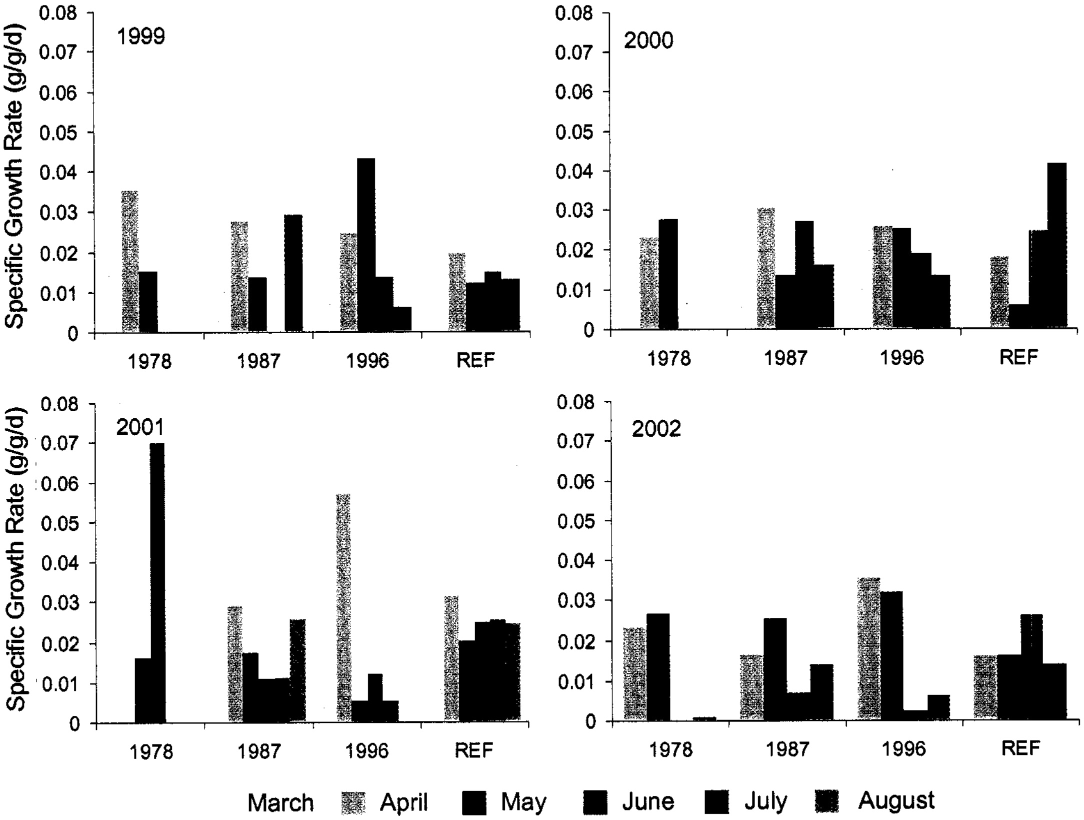


Figure 2.11: Model Simulation 2: Consumption rates were adjusted (indicated with an *) to reflect observed differences in tidal ration among sites.

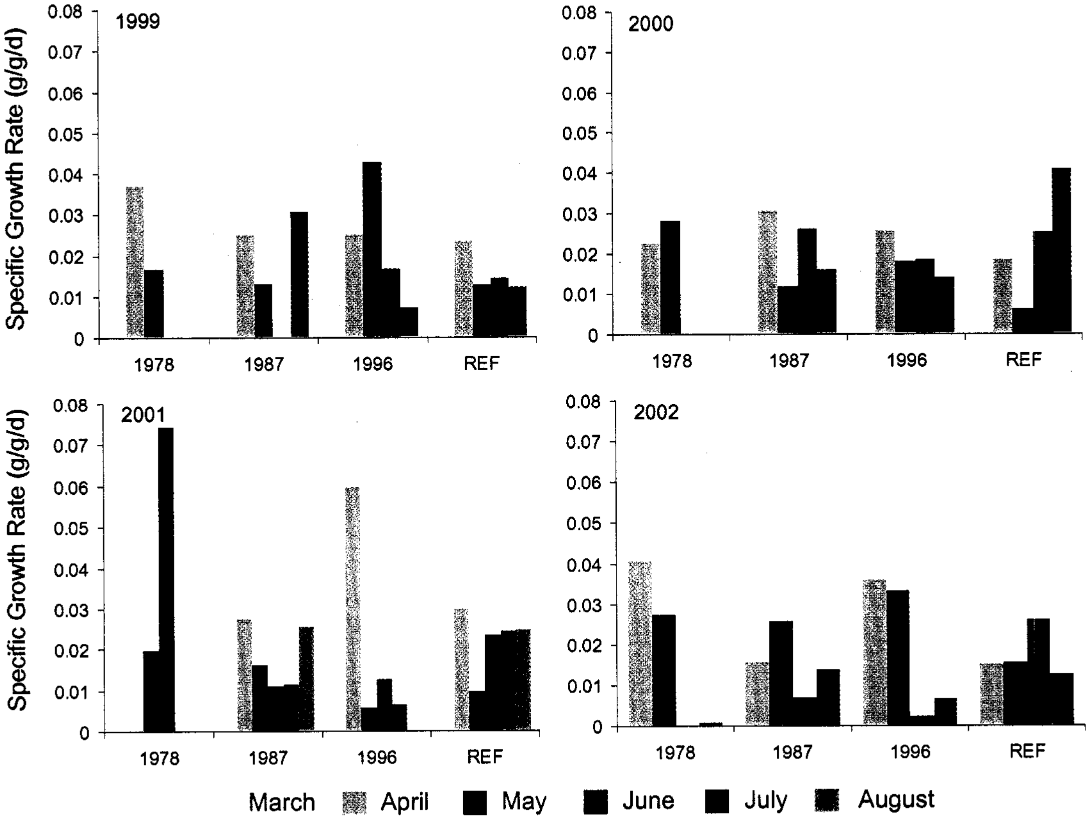


Figure 2.12: Model Simulation 3: Model simulation 2 was adjusted to include observed site-specific size of fish.

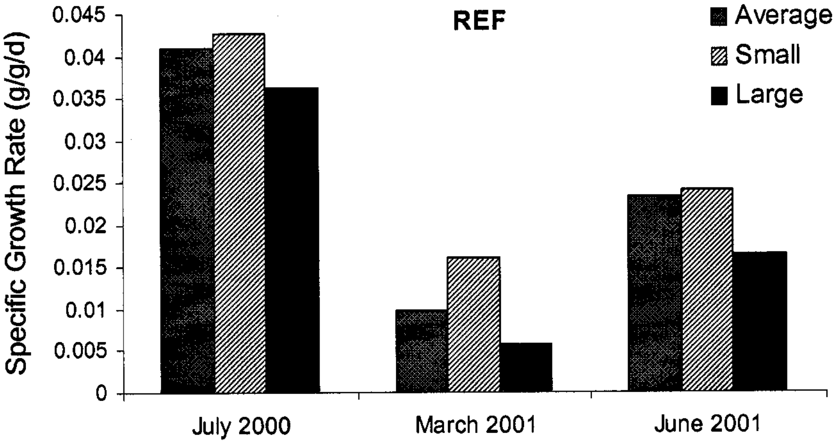


Figure 2.13: Model Simulation 4: Growth performance of size classes was compared when a bimodal distribution of fish was found.

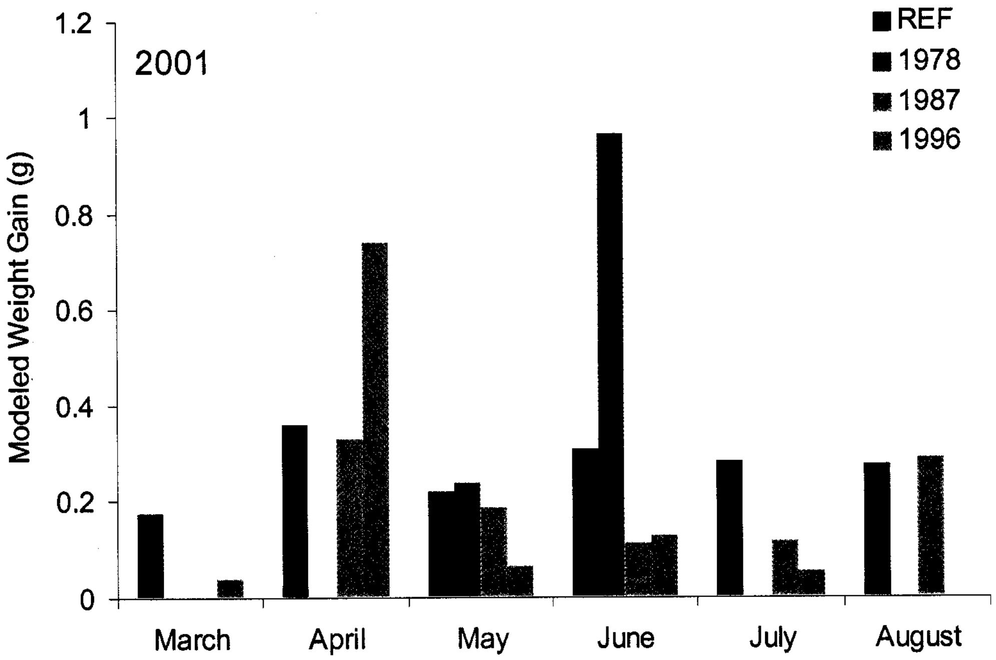


Figure 2.14: Modeled weight gain by month and site.

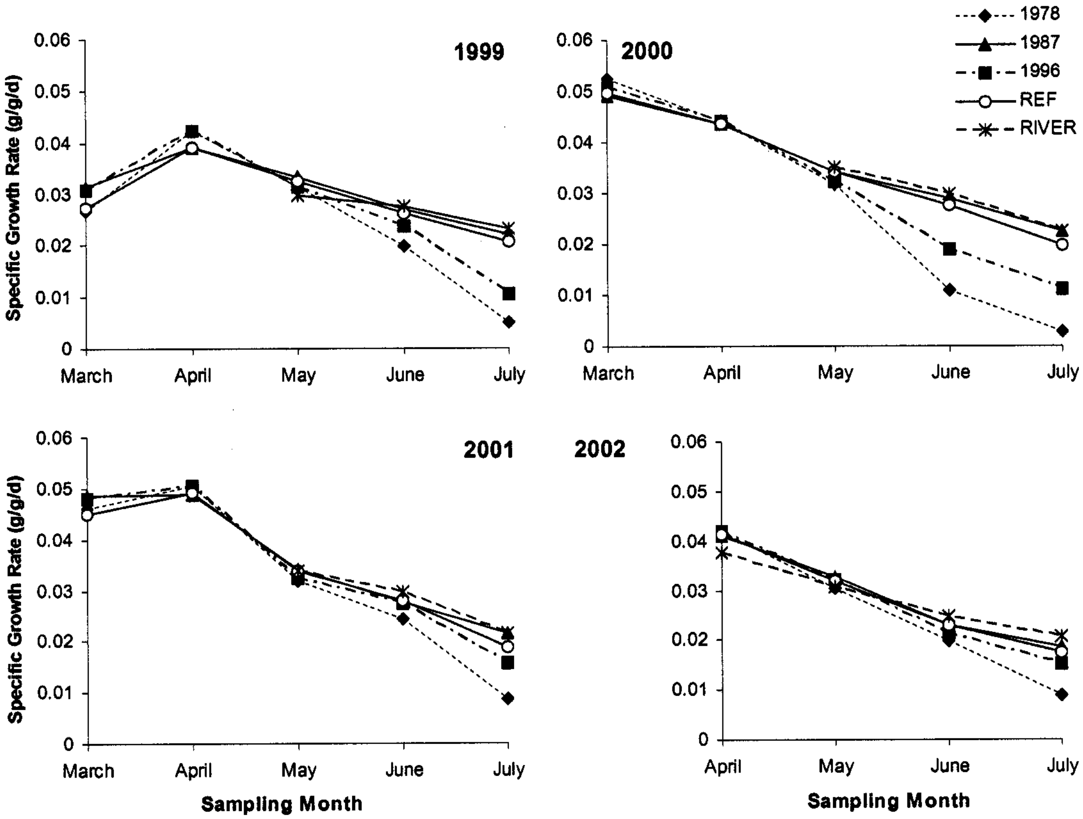


Figure 2.15: Specific growth rate of juvenile Chinook salmon (3 g) with a constant diet composition and consumption rate, and variable temperature regimes from each of the four marsh sites.

Chapter 3: Can Marsh Recovery Status be Determined with Tidal Marsh Invertebrate Assemblages from the Salmon River Estuary, Oregon?

Introduction

Determining Appropriate Indicators

Tracking the biotic response to a fundamental change in ecosystem conditions (such as dike breaching) is important in understanding how rapidly changing conditions influence structural and functional development in biotic communities. "Assembly rules" determine how species fit with each other, how a community resists invasion and/or how the community responds to local conditions of the physical environment (Weiher & Keddy 2004). The physiology of certain organisms will govern their response to disturbance conditions, and designations such as "residuals", "ephemeral" and "permanent" colonizers have been used to describe establishing plant taxa in recovering estuarine marshes (Frenkel & Morlan 1991).

Functional feeding group is an attribute of invertebrates that may be indicative of environmental variations, causing proliferation/extinction of local species and change in assemblage patterns. Appropriate indicators are taxonomic groups which respond reliably to changes in environmental conditions. The important differences in ecoregions or geographic variation must also be considered to determine appropriate indicator taxa (Mykra et al. 2004). A consistent indicator that captures variability across temporal and geographic gradients, while also being sensitive to local habitat conditions, would be useful to natural resource managers interested in assessing recovery status in restored estuarine marshes. Criteria for choosing a biotic indicator have generally relied on *a priori* situations and convenience; more extensive and quantitative guidelines are necessary (McGeoch 1998). A protocol which selects indicators based on statistical significance, equally applied over many sampling events across temporal, spatial and recovery status gradients would be the most informative. In this paper, I used a multiple-year data set on insects and benthic macroinvertebrates in an age-series of restoring marshes and a reference marsh to identify indicators of recovery status. Comparing invertebrate communities among recovering marshes of different ages and a reference may lend insight into determining ecological similarity when assessing a restoration project's success, the time necessary to achieve equivalency in terms of invertebrate community structure, and the processes which most effectively lead to successful restoration.

Opportunity at the Salmon River Estuary

The Salmon River estuary provided an ideal opportunity to relate estuarine marsh recovery age to invertebrate assemblage structure, and determine specific taxonomic indicators related to marsh age. A 'space-for-time' substitution enabled us to explore invertebrate assemblage patterns in wetlands of differing ages (coincident in the landscape), and make comparisons with a reference. Environmental conditions act like a sieve, filtering out species lacking certain requisite traits required to exist in a given location (Weiher & Keddy 2004). The effects of this process, as it relates to estuarine marsh recovery age, may reflect in the assemblages of tidal marsh invertebrates collected. The course of our study was sufficiently long to provide a measure of variability at each site. Gray et al. (2002) contains further details regarding restoration activities and vegetation descriptions.

Invertebrates as Indicators

Biotic indicators represent popular "litmus" tests for determining ecosystem status, and have been widely applied with a variety of taxa in many different ecosystems (see review, Carignan & Villard 2002). In aquatic environments, biotic metrics have been applied to freshwater biomonitoring to assess the condition of stream environments (Karr & Chu 1999). Groups of indicator species, or assemblages, have been used to determine biotic integrity (Karr 1981). Ecological health is inferred by the appropriate taxonomic and functional composition. Since invertebrates have a variety of physiological needs, their presence/absence reflects the condition of the surrounding environment, and for this reason they have been thought of as integrators of ecosystem variability and possible descriptors of ecosystem function, responding to a variety of ecosystem structuring factors (Garono & Kooser 2001). Invertebrates may be useful indicators of marsh recovery age as they are strongly influenced by environmental variation and react to disturbances on fine spatial scales (Carignan & Villard 2002). My aim was to determine if invertebrate assemblages differed among marsh sites, and find appropriate and sensitive indicators to marsh age. I determined an indicator was appropriate if it appeared consistently and exclusively in a particular location. I evaluated indicators based on functional feeding groups (e.g., predaceous, herbivore, detritivore, piercer, shredder, etc.); and, in addition to age, site vegetation, channel sediment organic matter, and estuarine gradient location were considered. Schoener (1986) suggested that physical environment was more influential in shaping species assemblages among invertebrates compared with biological relationships, such as competition or predation. Single species and assemblages of a variety of insects have been used to determine environmental change in habitats ranging from forests to grasslands to urban areas and mine sites (McGeoch 1998). Wetland characteristics have been elucidated using invertebrate indicators (Dufrene & Legendre 1997), and invertebrate communities are often compared among disturbed, restored and reference wetland sites (Craft et al. 2003; Zajac & Whitlatch 2001; Lerberg et al. 2000; Greenwood et al. 1991). However, few studies have used invertebrates to characterize marsh recovery age in Pacific Northwest wetlands.

Studies of Invertebrates in Estuarine Marshes

When benthic invertebrates were sampled in reference and restored marshes in the Snohomish River estuary, Washington, two site-specific taxa were found: Collembola (reference) and Ephydridae (restoring) (Cordell et al.

1998; Tanner et al. 2002). Chironomids were found in both reference and restoring sites, but were especially abundant ($5,000/m^2$ in April 1998) near the wooded edge of the restoring marsh (Cordell et al. 1998). Heatwole (2004) assessed insects in barrier salt marshes on Whidbey Island, Washington, and found variation in assemblage structure related to plant composition, porewater depth and tidal flooding duration. As these characteristics are impacted when marshes are disturbed, determination of insect indicators could be useful in assessing tidal marshes that are recovering from diking, muted tidal exchange or other disturbance. Distinctions among vegetation types within salt marshes were determined by invertebrate indicators, as well as variability in the same vegetation type among marshes (Heatwole 2004).

Hypotheses and Questions

My objective was to determine whether there were significant differences in invertebrate assemblages among the recovering and reference marshes, and determine site-specific invertebrate indicators that recurred across season and year. I tested the significance of relationships between recovery status and indicator taxa. I sought to evaluate the following hypotheses:

H1: There are differences in invertebrate assemblage structure between restoring and reference estuarine wetlands at the Salmon River estuary.

H2: Specific ("indicator") taxonomic groups drive the differences in the invertebrate assemblages among sites.

H3: Life histories distinctions in functional feeding group of indicator taxonomic groups relate to physical conditions (e.g., sediment organic matter) determined by recovery status among marsh sites.

And, the following questions:

1. Are there estuary-wide differences in mean density, taxonomic richness and invertebrate assemblages among years?
2. Are there among-site differences in mean density, taxonomic richness, and percent composition of invertebrate assemblages by month (across years) and by sampling period?
3. Which taxonomic groups drive the differences detected in invertebrate assemblages per sampling period, and which are the most exclusive indicators for each marsh?
4. What is the indicator assemblage for each site, based on repeated and exclusive indicators?
5. How do the life histories of those taxa reflect the condition of the indicated site?

Methods

Site Description

Estuarine marshes in the lower Salmon River, Oregon were diked to create pasturelands during the 1960s and restored by breaching and/or removing dikes in 1978, 1987 and 1996 (Frenkel & Morlan 1990, 1991; Gray et al. 2002). The resulting marshes represent a series of restoration experiments, and coupled with an undiked reference site, provided an ideal template to evaluate the differences in invertebrate assemblage patterns across a recovery age range and determine indicators of recovery status (Fig. 3.1). The marshes all occur along the central region of the estuarine gradient in the 800-ha estuary. The 1996 marsh is the most landward along the estuarine gradient, with the 1978 marsh and the reference (REF) occurring in nearly the same locations on the north and south side of the river, respectively. The 1987 marsh is the closest to the mouth and the most heavily influenced by the sea. Salinity range for each site at high and low tide is reported in Table 3.1.

In general, lower elevations at the restored marshes have resulted in a dominant Lyngbye's sedge assemblage. Pickleweed and salt grass form another distinctive community in the 1987 and 1978 marshes, but are absent in the fresher water 1996 marsh where *Agrostis alba* is prominent (Robert Frenkel, pers. comm.). More detailed descriptions of vegetation composition and change can also be found in Frenkel and Morlan (1990, 1991) and Gray et al. (2002).

A dendritic network of deep, step-sided tidal channels characterize the REF, 1987 and 1978 marsh sites, with abundant overhanging vegetation (Fig. 3.2). In contrast, tide channels at the 1996 site were wide, shallow, and undifferentiated shortly after dike breaching (Gray et al.

2002) (Fig. 3.3). The 1996 marsh tide channels remained shallower than the other sites, but have deepened and increased in sinuosity since 1998. Anthropogenic impact on tidal marsh channels in the 1996 marsh was also evident by aerial photo (see Fig. 3.1).

Physical Characteristics

Salinity

Salinity was recorded periodically at several stations (see Fig. 3.1) within the estuary at high and low tide throughout the sampling period. Measurements were recorded with a YSI-30 salinity/conductivity/temperature sensor from the surface of the water to the bottom at 0.5 m intervals.

Organic Matter

Five 5-cm diameter, 10 cm depth cores were taken in channel sediments at each site April-June 2000, and March-June 2001. Sediments were ground in a blender and a subsample was dried at 60°C until a stable weight was obtained. Dry material was then combusted in a muffle furnace at 600°C to burn all organic matter. The difference in dry weight/final weight determined percentage of organic matter. Samples were averaged by sampling period and compared using a One-way ANOVA.

Invertebrate Collections

Insect Fallout Traps (IFT)

Invertebrate density and composition was compared among sites and using Insect Fallout Traps (IFT) (Fig. 3.4). Each IFT consisted of a (51.7-cm X 35.8-cm X 14-cm) plastic box set on a PVC platform, filled with 3 L of water and several drops of unscented soap to break the surface tension of the water allowing capture of insects that landed on the surface. Abundances were calculated as density (number/m^2) for reporting purposes. This methodology has successfully been used in other studies of emergent marsh insects (Cordell et al. 1994; Gray et al. 2002; Heatwole 2004; Lott 2004).

Five traps were set in each vegetation type that occurred at each marsh site; this resulted in ten traps total deployed in the 1987 and REF marshes, five traps at the 1978 marsh, and five traps in the changing vegetation at the 1996 marsh. Collection sites were referred to as follows ('Hi' and 'Lo' refer to marsh elevations): 87 Lo, 87 Hi, REF Lo, REF Hi, 78 Lo, and 96 MT (marsh transition). All 30 traps were set on the same day during each sampling period and deployed for 72 hours. Traps were set, and collected within a two-hour window. Insects were collected by pouring the contents of the IFT through a 106-μm sieve and preserving the retained insects in a labeled sample jar with 70% isopropanol. In the laboratory, insects were generally sorted to family with some taxonomic groups (e.g., Hymenoptera and other less abundant taxonomic groups) sorted to order. Total density per sample, insect indicator values and ordination analysis were determined excluding the group Acarina (mites and ticks) due to extreme densities (8177.8 in one sample) and variability (SD 2533). However, Acarina was included in determining taxonomic richness.

Fallout insect sampling began in April 1998 and was continued monthly during the spring and summer (March-June). Insects were collected in 18 periods from 1998-2002.

Benthic Cores

Benthic invertebrates were sampled monthly using a 5-cm diameter aluminum core. Cores were taken to a depth of 10 cm yielding a volume of 196.25 cm^3. Five replicate cores of channel sediments were taken at low tide at each of the four marsh areas (REF, 1978, 1987 and 1996). Samples were fixed in the field in a 10% solution of buffered formalin. In the laboratory, following safety procedures for formalin use, sample contents were washed through a 0.5 mm sieve, macrofauna was retained, transferred to isopropanol and stained with Rose Bengal. All organisms were counted and identified to species in all cases when possible under an illuminated dissecting scope.

Benthic macroinvertebrate samples were analyzed from 13 sampling periods from 1998-2001.

Statistical Analysis

A One-Way ANOVA was used to compare total invertebrate (fallout insects and benthos) density and taxonomic richness among sites. Total invertebrate density was log transformed and taxonomic richness was square root transformed to meet the assumptions regarding normality and equality of variances necessary for parametric statistical tests (Zar 1999). Linear regression was used to evaluate relationships between total invertebrate density or taxonomic richness and site age. To evaluate trends among sites, years, and months, data were combined and analyzed in several ways. Estuary-wide trends in total invertebrate density and taxonomic richness among years were evaluated when all data were combined and grouped by month. Month-to-month trends among sites were evaluated when data were combined by month across all years and grouped by site. Finally, total invertebrate density and taxonomic richness were evaluated across all sampling periods independently.

In addition to traditional univariate statistics, invertebrate communities were analyzed with multivariate procedures using PC-ORD 4.25 (McCune & Mefford 1999). Multivariate analysis allowed for a visual representation of data in *n*-dimensional space (Kruskal & Wish 1978), where the greater distance between points reflects fewer similarities. Significant differences among invertebrate assemblages were tested using multi-response permutation procedure (MRPP) with Sorensen distance measures (McCune & Grace 2002). Due to the complexity of our sampling design, the data were segregated several different ways: all combined, by year, by month and by sampling period. Piecewise analysis with MRPP allows us to answer different questions, but this approach sacrifices the ability to analyze interaction terms (McCune & Grace 2002).

Definition of groups varied depending on segregated data sets and data source (fallout insects or benthos). When all data were combined, definition of groups included site, location (high marsh or low marsh) (IFT only), month and year. For yearly data sets, groups were defined based on site, location and month. For monthly data sets, groups were defined based on site, location and year. Site and location were evaluated when the data were segregated by sampling period. The magnitude of effect is described by the chance-corrected within-group agreement (A) and represents a scaled measure of similarity among data matrices. An $A > 0.2$ was considered ecologically significant (high dissimilarity). The p-value reflects how likely an observed difference is due to chance.

The same data sets were used in the MRPP and the ordination analysis. NMS depends principally on variation within the dataset being tested and is therefore the most effective ordination method for ecological data (McCune & Grace 2002). All data matrices were first transformed with the following formula to increase normality and enhance performance of the NMS: $b_{ij} = \log (x_{ij} + 1)$. Rare taxonomic groups (those representing less than 5% of the total density) were excluded from analysis.

Following Heatwole (2004), I tested for indicator invertebrates by sampling period using the Dufrene and Legendre (1997) method of calculating taxonomic indicator values (INDVAL). The method determines which taxonomic groups were most abundant and occurred most frequently in a site. A perfect indicator (value 100) would occur faithful and exclusive at one site, while never appearing at other sites. Statistical significance of INDVAL was determined using a Monte Carlo randomization procedure with 1000 permutations ($p \leq 0.01$) (McCune & Grace 2002). INDVAL results were evaluated over all sampling periods (18 for fallout insects; 13 for benthic macroinvertebrates) to determine which taxonomic groups most appropriately indicated the state of the recovering or reference marshes over time, and would strongly (value > 70) and consistently indicate marsh age. The indicators were assessed according to the number of times the group was determined to be significant by INDVAL, their exclusivity to a particular site or to a group of similar sites (i.e., low marsh or restoring site indicators) and their absence from a site. Special attention was also paid to how indicators changed over the study period.

Results

Salinity

Distribution of salinity was strongly influenced by season and freshwater flow, and the estuary was often flooded after sustained winter-spring storm events. High tide surface salinities demonstrate the typical differences among sites (Table 3.1). At high tide a salt-wedge was typically present, typical of drowned-river mouth estuaries. However, most other estuaries along the Oregon and Washington coasts do not have classic salt wedges, but have extensive salt-fresh mixing zones (e.g., Chehalis, Willapa, Naselle, Tillamook, Alsea, Siuslaw, Yaquina, Coos, Umpqua and Coquille Rivers) (Jeff Cordell, pers. comm.). Circulation differences may affect the comparability of the Salmon River estuary to these other coastal estuaries.

Channel Sediment Organic Matter

The 1996 marsh consistently contained significantly higher amounts of channel sediment organic matter than the other three marshes (Fig. 3.5). Based on One-way ANOVA, all p-values were equal to zero except May 2001 (p = 0.23).

Fallout Insects

Invertebrates from IFT samples were represented by three classes (Insecta, Arachnida and Crustacea) with 81 families of insects across 14 orders, three orders of arachnids and two orders of crustaceans.

In estuary-wide comparisons in mean density among years by collecting month, significantly more invertebrates occurred in March 2000 (p = 0.03) compared with the same month in 1999 and 2001, but no other significant differences in invertebrate density were found (April, p = 0.16; May, p = 0.20; or June, p = 0.84) (Fig. 3.6). Significant differences in taxonomic richness were found for each month among years although no discernable pattern was detected (Fig. 3.6). Taxonomic richness was significantly greater in March 2000, compared with March 1999 and 2001 (p = 0.00). In April 1999 and June 1998, taxonomic richness was significantly less than all other years (p = 0.00, for both). Taxonomic richness in May 1998 was significantly lower than May 2000 and 2002 (p = 0.02).

Evaluating among-site mean density and taxonomic richness by months (across all years) demonstrated significant differences in all cases (Fig. 3.7; Table 3.2). The composition of the REF, 1987 and 1978 marshes progressed from high numbers of dipterans in March to increasing homopterans in June (Fig. 3.8). The 1996 site differs from this pattern in March and May when 20-40% of density was due to staphylinid beetles. Mean density and taxonomic richness were also evaluated independently by sampling period (Figs. 3.9, 3.10). Mean densities were significantly different in April 1998 ($p = 0.02$), May 1999 ($p = 0.05$), April and June 2000 ($p = 0.00$, $p = 0.00$, respectively), and April 2001 ($p = 0.00$) (Table 3.3). Mean taxonomic richness was also significantly different among sites in several sampling months (Table 3.3). Linear regression between mean densities or taxonomic richness and restoration age revealed no relationship ($r^2 = 0.00$, $p = 0.94$; $r^2 = 0.01$, $p = 0.21$) (Fig. 3.11).

Using MRPP, fallout insect communities were evaluated at several different levels of data segregation to determine which factors (site, location, month and/or year) influenced the patterns of assemblage assembly the most. The strongest distinctions in insect assemblages among sites and locations were found when all sampling sites were evaluated independently, completely removing the effects of yearly and monthly variation (Table 3.4). When all data were combined and evaluated, insect communities varied widely among sites ($A = 0.08$, $p = 0.00$), locations ($A = 0.03$, $p=0.00$), years ($A = 0.05$, $p = 0.00$), and months ($A = 0.15$, $p=0.00$). When yearly variation was removed, interannual differences in the monthly effect size were ecologically significant in most cases, although the site differences were only detected in 1998 (Table 3.5). The month effect was significantly different in May, with the highest effect value for year found in March.

The assemblage patterns described by the MRPP statistics were supported by NMS. As the effect size was generally greatest in May, NMS of all May data and each May period were used to illustrate differences. When all May data were combined, separation among sites was evident between 78 Lo and 96 MT along Axis 1, and between the 87 sites and 78 Lo and 96 MT along Axis 2. Separation along Axis 3 was most distinct between REF Hi and 87 Lo (Fig. 3.12). Sites were distinct in May of each sampling year. In May 1998, both REF sites and 78 Lo were separate from the other sites along Axis 2 and in May 1999, the REF sites were again separate from the restoring sites along Axis 2 (Fig. 3.13). In May 2000 and 2001, the 1996 marsh was distinct from the other sites along Axis 1 (Fig. 3.14). In May 2002, the 1996 site and the 1978 marsh were separate along Axis 2 (Fig. 3.15). Rarely were high and low marshes distinguishable in ordination space from each other and the 1996 marsh was responsible for significant differences among sites.

Significant taxonomic groups determined by INDVAL occurred as consistent indicators (determined in at least five of the 18 sampling periods) arrayed along a gradient of exclusivity (Appendix D; Table 3.6). The rove beetles (Coleoptera: Staphylinidae) were the most consistent and exclusive indicators, occurring in 11 of the 18 sampling periods exclusively in the 1996 marsh. Dance flies (Diptera: Empididae) were also exclusive indicators for the 1996 marsh in six of the 18 sampling periods, and shore flies (Diptera: Ephydridae) were exclusive indicators for the 1996 marsh in eight of 18 sampling periods. Click beetles (Coleoptera: Elateridae) and leaf beetles (Coleoptera: Chrysomelidae) were exclusive indicators for the REF Hi site, found in eight and five, of the 18 sampling periods, respectively. Sciarid flies (Diptera: Sciaridae) and collembolans were mostly exclusive indicators for the REF Hi site. Small fruit flies (Diptera: Drosophilidae) were exclusive indicators in the 87 Lo site, but there were no exclusive indicators for the 87 Hi site. The muscid family (Diptera: Muscidae) was an exclusive indicator for the 78 Lo site in five sampling periods. Homopterans (Homoptera: Cicadellidae) were also mostly exclusive indicators for the 78 Lo site, occurring seven out of 18 times and only once as an indicator elsewhere.

The most common indicator taxa, Chironomidae and Cecidomyiidae, were found to be indicators in 12 of the 18 sampling periods, but were not exclusive to one site. Many insects were never found as indicators in the reference site. These taxa included rove beetles, long-legged flies (Diptera: Dolichopodidae), dance flies, small fruit flies and muscid flies. Cantharid beetles (Coleoptera: Cantharidae), on the other hand, were found as an indicator in five sampling periods in the REF site and once in the 1978 site.

Benthic Invertebrates

Benthic invertebrates encompassed seven classes (Turbellaria, Oligochaeta, Polychaeta, Insecta, Arachnida, Crustacea and Bivalvia) and two phyla (Nematoda and Nemertea). Five families of polychaetes were identified and two groups were identified to species. Insects were identified from three orders; Diptera (three families), Trichoptera and Collembola. Two orders of arachnids were found, four orders of crustaceans (Ostracoda, Amphipoda, Isopoda and Cumacea) and one family of bivalve (Tellinidae).

Estuary-wide differences in mean density and taxonomic richness were evaluated among years with benthic invertebrate data grouped by collecting month (Fig. 3.16). Variability was quite high, and the only significant difference in mean density occurred in March ($p = 0.01$). There were no other significant differences in mean density or taxonomic richness among years, by month.

Mean density and taxonomic richness of benthic invertebrates analyzed by month across all years showed that density and taxonomic richness increased with marsh age (Fig. 3.17). Differences between the REF and the 1996 marsh, and in some cases between the 1978 and 1996 marshes, were significant (Table 3.2). Differences in mean density or taxonomic richness were not significant between the REF and 1978 marsh. When composition was evaluated by month, there were few differences among the REF, 1978 and 1987 marshes (Fig. 3.18); however the 1996 marsh differs in the number of insects and lack of polychaete worms.

Mean density and taxonomic richness of benthic invertebrates were also evaluated based on sampling period (Figs. 3.19, 3.20). Mean density and taxonomic richness were significantly different in several sampling periods (Table 3.3). Mean density and taxonomic richness were inversely related to marsh age ($r^2 = 0.44$, $p = 0.00$; $r^2 = 0.65$, $p = 0.00$, respectively), with the highest densities consistently found in the REF and 1978 marshes and lowest densities found in the 1996 marsh (Fig. 3.11).

In contrast to the insect fallout data that were strongly affected by month and year, MRPP analysis showed little relationship between benthic invertebrate assemblages and month and year (A = 0.00, p = 0.00; A = 0.05, p = 0.10, respectively). Instead, site differences were responsible for a larger effect size for this level of data segregation (A = 0.16, p=0.00).

When analyzed separately by year, month and site, site-related differences were more obvious, and differences due to year and month were still not significant (Table 3.5). The strongest distinctions among sites were evident when all sampling sites were evaluated independently (Table 3.6).

Assemblage patterns in benthic invertebrates described by the MRPP statistics were supported with NMS. Results from May and June 2001, when the effect was generally the greatest, illustrate site differences (Fig. 3.21).

Appendix E indicates the significant taxonomic groups of benthic invertebrates as determined by INDVAL. Benthic invertebrates that were consistent and repeatable indicators (determined in at least three of the 13 sampling periods) were evaluated along a gradient of exclusivity (Table 3.7). The introduced Asian cumacean, *Nippoleucon hinumensis*, and a polychaete worm (Polychaeta: Capitellidae) were repeatable and exclusive indicators at the REF marsh, occurring in 11 and eight of 13 sampling periods, respectively. Chironomid larvae (Diptera: Chironomidae) were repeatable and exclusive indicators of the 1996 marsh in 11 of 13 sampling periods. Another polychaete worm (Polychaeta: Spionidae) was exclusive to the 1987 marsh, but occurred less frequently (five of 13 sampling periods). No exclusive indicators were found in the 1978 marsh; indicators found there, the polychaetes *Manayunkia aestuarina* and *Hobsonia florida* and ostracod crustaceans, were also indicators at the REF marsh (Table 3.7).

Indicator Assemblage for Marsh Age

When taxonomic groups were found to be repeatable indicators over the study period and were mostly exclusive to a particular site, they were added to a group designated the indicator assemblage (Table 3.8). Indicator assemblages were determined for each marsh site.

Discussion

For the fallout insects that we sampled at the Salmon River restored and references marshes, among-site differences in mean density and taxonomic groups by sampling periods revealed few strong patterns, other than a spring-summer increase in abundance and low taxonomic richness in the 1987 marsh. However, for benthic invertebrates, mean density and taxonomic richness were both inversely proportional to recovery age especially at the newest restoration site, and thus appeared to be more responsive to marsh recovery stage. There were more among-site differences in insect fallout assemblages than for benthic invertebrates. Although benthic invertebrate composition in the 1996 marsh was consistently different from the other sites, the other sites were usually similar to each other. Among the reasons for these differences between fallout and benthic assemblages might be: (1) the benthic assemblage stabilizes earlier in the development of restored marshes; (2) fallout insects are inherently more variable due to their mobility, dispersal by wind, etc.; (3) fallout traps probably integrate individuals from multiple habitats, while benthic sampling is more habitat specific; and, (4) landscape position affect fallout insect composition while benthic invertebrates are relatively unaffected. When IFT assemblages were assessed by sampling periods, distinctions among sites were evident. I found the 1996 marsh to be distinct from the other marshes in most cases, and the greatest separation among sites was evident both by NMS and MRPP analysis in the May sampling periods. Overall patterns in assemblage composition may be more instructive in determining site-specific environmental differences than mere density measures (Heino 2003). Ecosystem change may allow migrant species access which leads to greater species density and richness; understanding endemic taxonomic assemblages is essential to avoiding misdiagnosing ecosystem health with ambiguous indicators.

Macroinvertebrate response to restoration is often evaluated in estuarine marsh environments as part of ecological assessment protocols (Simenstad et al. 1991; Zedler 2001). In some cases macroinvertebrates are similar at restored and reference sites, even though there are differences in other organisms such as fishes (Moy & Levin 1991). In other cases, some species of macroinvertebrates recover in a relatively short time frame (< 5 years), but others (e.g., the snail, *Melampus*) may require decades to reach reference densities (Warren et al. 2002). Warren et al. (2002) determined that invertebrate populations recover at different rates depending on the individual marsh, and that recovery rates are not necessarily related to changes in vegetation. Evidence of macrofaunal succession among created Salicornia marshes (aged 16-months, five-years, six-years, and 10-years) was seen in California as younger insect-dominated sites progressed to older oligochaete-dominated sites (Levin & Talley 1999). My benthic data also revealed a higher percent composition of insects across all sampling periods in the 1996 marsh, but the site was also often dominated by oligochaetes. Polychaetes, however, were never more than a small percent of benthic macrofauna at the 1996 marsh, while ranging from 20-30% in all other sites.

Indicators have been useful in detecting environmental conditions in wetland environments (distribution of carabid beetles (Coleoptera: Carabidae) (Dufrene & Legendre 1997)), and pinpointing mechanisms of assemblage change, but have been used less often to determine restoration status of recovering estuarine marshes. I grouped repeated and mostly exclusive indicators and determined an indicator assemblage for each marsh (see Table 3.8). Due to the duration of my study, I was able to determine invertebrates that were consistent indicators of marsh age. Groups of organisms were identified which repeatedly indicated restoring and reference conditions and, although the level of taxonomic resolution may limit interpretation based on life history characteristics, some potentially useful site-specific information emerged from my analysis.

The indicator assemblage in the newest restoration site consisted of staphylinid beetles, dance and shore flies, and chironomid larvae. Rove beetles are the largest beetle family and occupy a variety of habitats. They are predaceous and commonly found in intertidal environments (Merritt & Cummins 1996). The biology of this family is poorly understood, but it is known they occur in a variety of decaying habitats, e.g. dung, shoreline wrack, fungi, leaf litter (Borror & DeLong 1971; Merritt & Cummins 1996). Rove beetles have been used as bioindicators of anthropogenic effects (Bogach 1993), and in the assessment of different floodplain habitats (Greenwood et al. 1991). Dance flies are predaceous, found in moist environments with larvae occur in decaying vegetation (Merritt & Cummins 1996). Shore flies are adapted to living in brackish environments and contain larvae who feed on decaying, fine organic matter (Merritt & Cummins 1996). Shore flies have also been found to be associated with restored Pacific Northwest tidal wetlands in the Snohomish River estuary, Washington (Cordell et al. 1998). Due to the increase in decaying vegetation at the 1996 site (Gray et al. 2002), and the higher amount of organic matter in the sediments (Fig. 3.5), the insect assemblage at this site may be an early response to the recent die-back of pasture vegetation there. Chironomids may break down organic matter, changing channel sediments and shifting faunal density and diversity. Similar processes have been recorded in other ecosystems. Levin and Talley (1999) and Moseman et al. (2004) also found high densities of insects in created *Salicornia*-marshes in southern California. The results at the Salmon River suggest this breakdown of residual freshwater detritus and other organic material happens rapidly (<15-24 years).

Although amphipods and long-legged flies were repeatable indicators in the 1987 marsh, they were not exclusive and therefore not included in the indicator assemblage for the site. Small fruit flies (Diptera: Drosophilidae) and spionid worms (Polychaeta: Spionidae) comprise the indicator assemblage for the 1987 marsh. Small fruit flies are generally known to occur around decaying vegetation or fruit, and a few species are predaceous on small Homoptera (Borror & DeLong 1971), which were also found as repeated indicators in the 1987 marsh. The spionid worms, are more likely to be found in saline conditions and may reflect the estuarine gradient position of the 1987 marsh.

The reference marsh had the largest indicator assemblage, including click beetles (Coleoptera: Elateridae), leaf beetles (Coleoptera: Chrysomelidae), sciarids (Diptera: Sciaridae), collembolans, cumaceans (Crustacea: Cumacea), capitellid worms (Polychaeta: Capitellidae) and nematodes. Click and leaf beetles are phytophagus on flowers and foliage, and are vascular plant shredders, preferring a specific host plant, or a few host plants that occupy the same habitat (Borror & DeLong 1971; Merritt & Cummins 1996). The determination of these families as indicators in the high elevation reference site may be an indication of greater plant diversity compared with restoring sites. However, the family is not well-enough understood, especially semi-aquatic forms, to determine the host plant species (Merritt & Cummins 1996). Click beetles are known to occur on bark and rotting logs (Borror & DeLong 1971), so the presence of large woody debris near the high reference site may partially account for their indicator status there. Cordell et al. (1998) associated collembolans with reference Pacific Northwest marshes conditions, and Collembola were indicated seven out of 18 times in the high reference and twice in the 1987 higher elevation site. A range of factors may be responsible for the proliferation of cumaceans in the REF marsh.

The 1978 marsh indicator insect assemblage consisted of muscid flies (Diptera: Muscidae), leafhoppers (Homoptera: Cicadellidae) and wasps (Hymenoptera). Repeated indicators were found in the benthic cores consisted of the polychaete worms *Manayunkia aestuarina* and *Hobsonia florida* and Ostracoda in the 1978 marsh, but also at the reference marsh. Muscid flies contain few aquatic species, but members of the family are widespread and occur in a variety of habitats (Merritt & Cummins 1996), with larvae breeding in excrement and various types of decaying material (Borror & DeLong 1971). They are generally predators feeding on a variety of organisms, including oligochaetes and mosquito larvae. Leafhoppers pierce the stem of vegetation with a long proboscis and feed on plant sugars (Borror & DeLong 1971). Wasps are generally predators or parasites. Most indicators in the restored marshes had known links to decaying vegetation, while the indicators in the reference marsh (and some in the 1978 marsh) were associated with living plant material. This could be the result of either higher standing stock of marsh vegetation at the older and reference sites, or larger amounts of plant detritus at the younger sites, or a combination of these factors. We demonstrated higher sediment organic matter at the 1996 marsh. I did not measure vegetation communities, and so cannot relate them to the invertebrate indicator assemblages here; however, future work can incorporate the long history of plant data.

Resource managers, funding agencies, policy-makers and scientists often view restoration ecology as more of an "art" than a "science", relying on intuition rather than well-documented knowledge (Michener 1997). Applied scientific knowledge can develop more with each restoration action, if restoration projects are treated as experiments with definable questions, hypotheses, and measurements. The time series of restoring marshes at the Salmon River allowed us to explore differences in invertebrate communities across a 24-year recovery gradient with an intact reference site, determine the relationship between invertebrate density and taxonomic composition and marsh age, and establish reliable and appropriate indicators of restoration status. Although scientists are often reluctant to make management recommendations based on data from a specific case-studies that have not undergone replication (Michener 1997), they often comprise the only available information about cause and effect relationships between restoration techniques and rate and magnitude of recovery. Conducting ecology-based management within the limitations of classical experimental science would require long-term monitoring, replicated control and treatments, and experimental units at temporal and spatial scales appropriate to the hypotheses. Unfortunately, such conditions are rarely if ever achievable in the natural environment. The Salmon River represents one of the best examples of progressive restoration where a range of hypotheses may be tested. With the acquisition of methodically collected data and the continuing development of new approaches and techniques here and in other restored estuarine environments, the skill and practice of restoration science will continue to discover useful ecological patterns and links among abiotic and biotic environmental factors.

Tables

Table 3.1: Surface Salinity (%o) in Marsh Sites

	Estuarine Gradient →				
Date	Mouth	1987	1978	REF	1996
29-Mar-99	30.7	4.4	0.8	0.4	0.0
27-Apr-99	32.5	3.6	2.4	1.7	0.3
25-May-99	32.0	1.7	0.1	0.3	0.7
24-Jun-99	31.5	10.5	5.1	6.3	3.0
18-Apr-00	27.6	6.3	3.3	2.3	1.7
20-May-00	23.5	3.5	5.4	1.5	0.2
7-Jun-00	14.6	7.2	0.0	3.9	2.2
22-Mar-01	33.0	0.1	0.0	0.0	0.0
21-Apr-01	28.0	2.5	0.0	0.5	0.0
6-May-01	27.2	3.0	2.3	1.8	0.0
10-Jun-01	27.0	6.5	3.0	2.5	0.0
15-Apr-02	14.4	2.8	1.7	1.1	0.2
26-May-02	21.4	13.1	14.5	11.7	10.4
30-Jun-02	15.9	4.0	2.8	2.3	0.4

Table 3.2: Significance Level (p-value) from ANOVA on Density and Taxonomic Richness (1998-2002)

	Fallout Insects		**Benthic Invertebrates**	
Month	Density	Taxonomic Richness	Density	Taxonomic Richness
March	**0.00**	**0.01**	**0.01**	**0.00**
April	**0.00**	**0.00**	**0.00**	**0.00**
May	**0.00**	**0.00**	**0.00**	**0.00**
June	**0.00**	**0.00**	**0.00**	**0.00**

Table 3.3: Comparison of Mean Total Abundance and Taxa Richness among Sites by Sampling Period

	Fallout Insects		Benthic Invertebrates	
	Density	Taxonomic Richness	Density	Taxonomic Richness
Sampling Period	*p-value*	*p-value*	*p-value*	*p-value*
April 1998	**0.02**	**0.00**	**0.01**	**0.00**
May 1998	0.80	**0.00**	0.15	**0.00**
June 1998	0.08	**0.02**	**0.00**	**0.00**
March 1999	0.81	0.53		
April 1999	0.57	**0.01**	**0.04**	**0.02**
May 1999	**0.05**	0.09	**0.00**	**0.01**
June 1999	0.13	0.57		
March 2000	0.63	**0.00**	0.11	**0.00**
April 2000	**0.00**	**0.00**	**0.02**	**0.04**
May 2000	0.83	0.75	**0.01**	**0.01**
June 2000	**0.00**	**0.04**	**0.01**	**0.03**
March 2001	0.16	**0.02**	**0.02**	0.16
April 2001	**0.00**	0.61	**0.03**	0.38
May 2001	0.14	0.29	**0.00**	**0.04**
June 2001	0.58	**0.05**	**0.02**	0.26
April 2002	0.16	0.11		
May 2002	0.60	0.38		
June 2002	0.16	0.26		

Table 3.4: MRPP for Marsh Surface and Benthic Invertebrates by Sampling Period

	Effect Size - A>0.2 is Ecologically Significant		
	Fallout Invertebrates		**Benthos**
Sampling Period	*By Site*	*By Location*	*By Site*
April 1998	**0.42**	0.13	**0.35**
May 1998	**0.45**	**0.23**	**0.25**
June 1998	**0.36**	0.16	**0.45**
March 1999	0.17	0.08	
April 1999	**0.26**	0.13	**0.28**
May 1999	**0.41**	0.14	**0.29**
June 1999	**0.34**	0.12	
March 2000	**0.29**	0.13	**0.23**
April 2000	**0.39**	0.18	**0.30**
May 2000	**0.43**	**0.25**	0.19
June 2000	**0.39**	0.15	**0.31**
March 2001	0.19	0.08	**0.27**
April 2001	**0.36**	0.17	**0.23**
May 2001	**0.32**	0.15	**0.32**
June 2001	**0.32**	0.13	**0.40**
April 2002	**0.38**	0.13	
May 2002	**0.39**	0.15	
June 2002	**0.37**	0.15	

Table 3.5: Multi-Response Permutation Procedure (MRPP) for Marsh Surface and Benthic Invertebrates [Effect Size (A)]

	Effect Size - A>0.2 is Ecologically Significant				
	Fallout Insects			*Benthic Invertebrates*	
Year	*By Month*	*By Site*	*By Location*	*By Month*	*By Site*
1998	0.10	**0.22**	0.11	0.00	**0.27**
1999	**0.33**	0.06	0.03	-0.01	**0.27**
2000	**0.21**	0.13	0.06	0.00	**0.20**
2001	**0.21**	0.09	0.05	0.00	**0.24**
2002	0.16	0.16	0.07		
Month	*By Year*	*By Site*	*By Location*	*By Year*	*By Site*
March	0.16	0.08	0.04	0.04	0.19
April	0.14	0.14	0.07	0.03	0.16
May	0.10	**0.21**	0.09	0.06	0.16
June	0.10	0.15	0.06	0.05	**0.25**

Table 3.6: Occurrences of Invertebrate Indicators by Site from all Sampling Periods

Taxonomic Group	REF Lo	REF Hi	78 Lo	87 Lo	87 Hi	96 MT	TOTAL	Indicator Level
Coleoptera: Staphylinidae	0	0	0	0	0	**11**	11	Exclusive Indicators
Coleoptera: Elateridae	0	**8**	0	0	0	0	8	
Diptera: Empididae	0	0	0	0	0	**6**	6	
Diptera: Drosophilidae	0	0	0	**6**	0	0	6	
Coleoptera: Chrysomelidae	0	**5**	0	0	0	0	5	
Diptera: Muscidae	0	0	**5**	0	0	0	5	
Diptera: Ephydridae	0	1	0	0	0	**8**	9	Mostly Exclusive Indicators
Diptera: Sciaridae	1	**7**	0	0	0	0	8	
Homoptera: Cicadellidae	0	0	**7**	0	0	1	8	
Hymenoptera	0	1	**8**	0	1	0	10	
Collembola	0	**7**	0	0	2	0	9	
Diptera: Psychodidae	2	0	**5**	0	0	0	7	
Amphipoda	0	1	0	**3**	1	0	5	
Diptera: Chironomidae	**2**	**1**	**1**	0	0	**8**	12	Not Exclusive Indicators
Diptera: Cecidomyiidae	0	**4**	**8**	0	0	0	12	
Diptera: Sphaeroceridae	0	**1**	**3**	0	0	**5**	9	
Thysanoptera	0	**3**	0	0	0	**2**	5	
Diptera: Ceratopogonidae	**3**	0	**1**	0	**1**	**4**	9	Broad Indication
Coleoptera: Cantharidae	**3**	**2**	**1**	0	0	0	6	Indicator for REF
Diptera: Dolichopodidae	0	0	**2**	**4**	0	**2**	8	Indicator of Restoration

Table 3.7: Number of Occurrences of Benthic Invertebrate Indicators by Site for all Sampling Periods

Taxonomic Group	REF	1978	1987	1996	TOTAL	Indicator Level
Cumacea: *Nippoluecon humenensis*	**11**	0	0	0	11	Exclusive Indicators
Polychaeta: Capitellidae	**8**	0	0	0	8	
Diptera: Chironomidae (larvae)	0	0	0	**8**	8	
Polychaeta: Spionidae	0	0	**5**	0	5	
Nematoda	**4**	0	0	0	4	
Polychaeta: *Manayunkia estuarina*	**4**	**4**	0	0	8	Indicator in Two Sites
Polychaeta: *Hobsonia florida*	**3**	**5**	0	0	8	
Crustacea: Ostracoda	**2**	**2**	0	0	4	
Gastropoda: *Macoma spp.*	3	0	3	0	6	
Amphipoda: *Corophium spp.*	2	3	1	**0**	6	No Indicator
Polychaeta: Nereidae	0	0	3	0	3	

Table 3.8: Taxonomic Groups Composing Indicator Assemblage for Each Site

REF	**1978**	**1987**	**1996**
Coleoptera: Elateridae Coleoptera: Chrysomelidae Diptera: Sciaridae Collembola Cumacea Polychaeta: Capitellidae Nematoda	Diptera: Muscidae Homoptera: Cicadellidae Hymenoptera	Diptera: Drosophilidae Polychaeta: Spionidae	Coleoptera: Staphylinidae Diptera: Empididae Diptera: Ephydridae Diptera: Chironomidae (larvae)

Figures

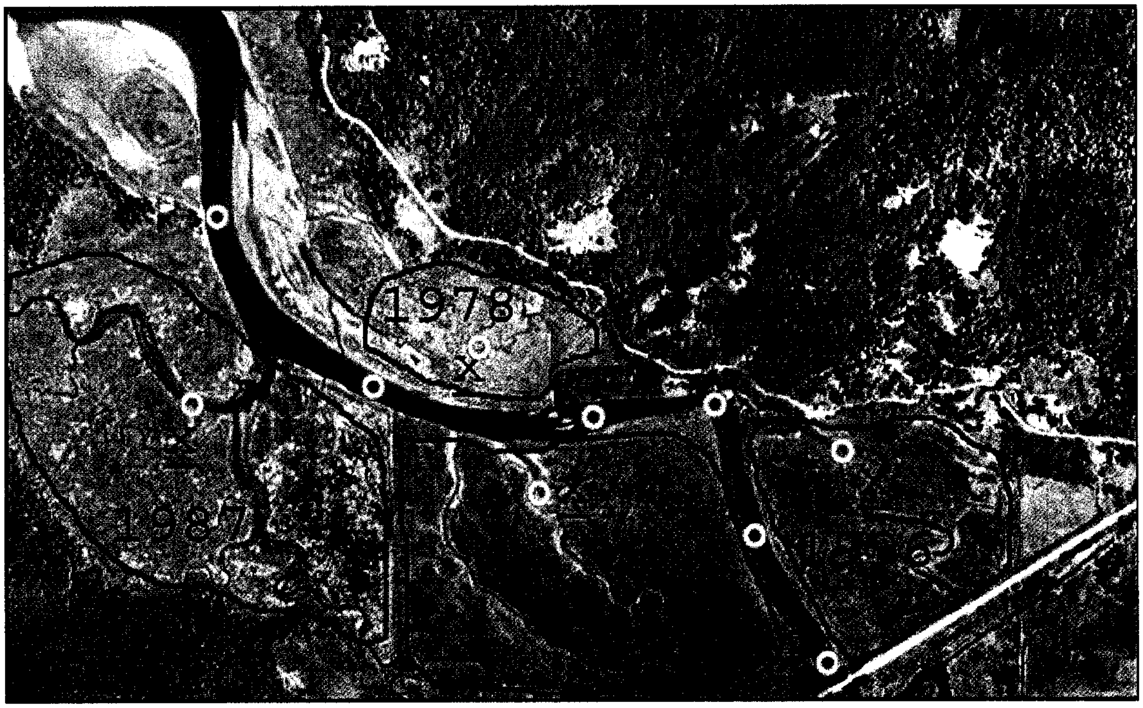


Figure 3.1: Invertebrate sampling sites (x) and salinity sampling sites (o) in the Salmon River estuary.

Figure 3.2: Steep-sided tidal channel with overhanging vegetation.

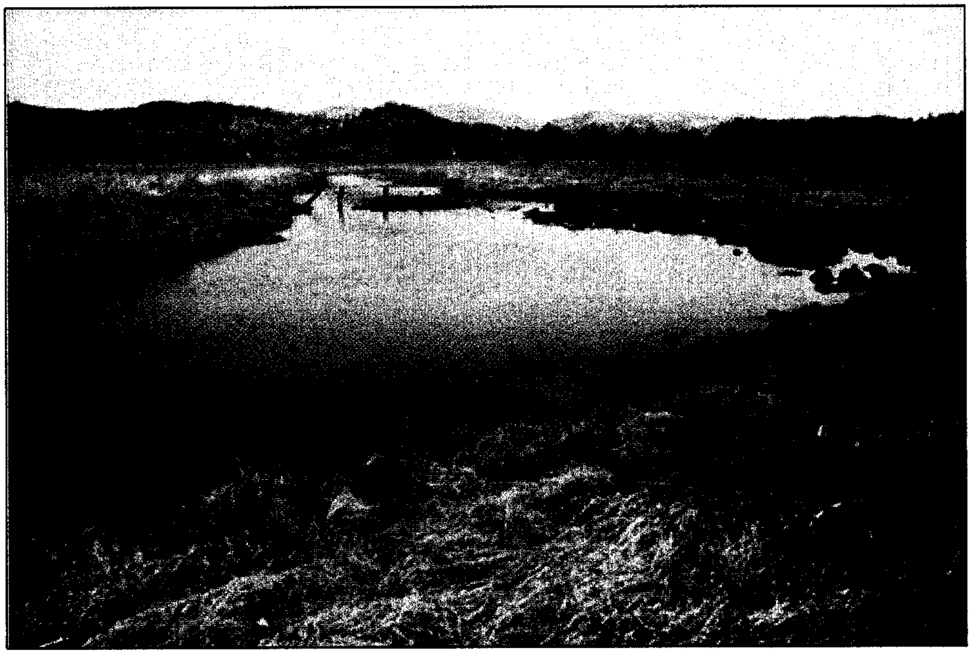


Figure 3.3: Wide, shallow tide channels of 1996 marsh in 1999.

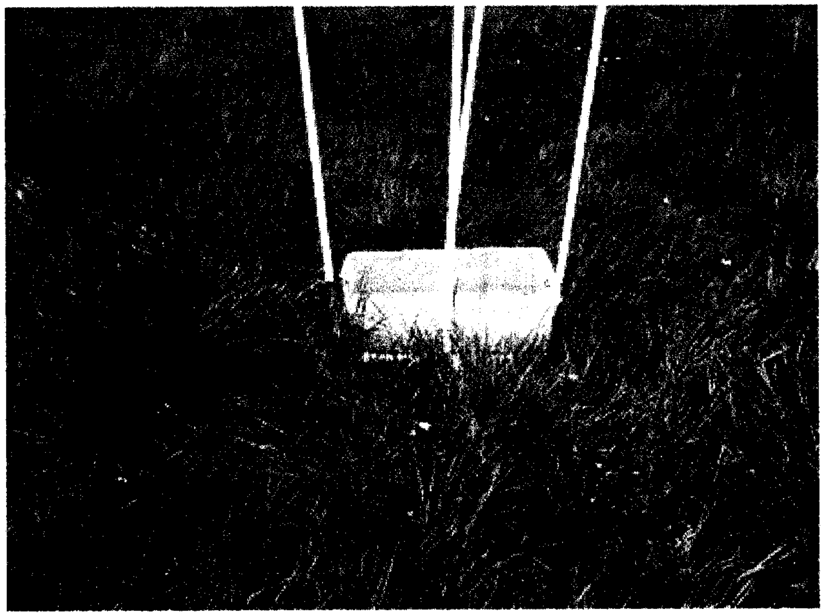


Figure 3.4: The Invertebrate Fallout Trap (IFT).

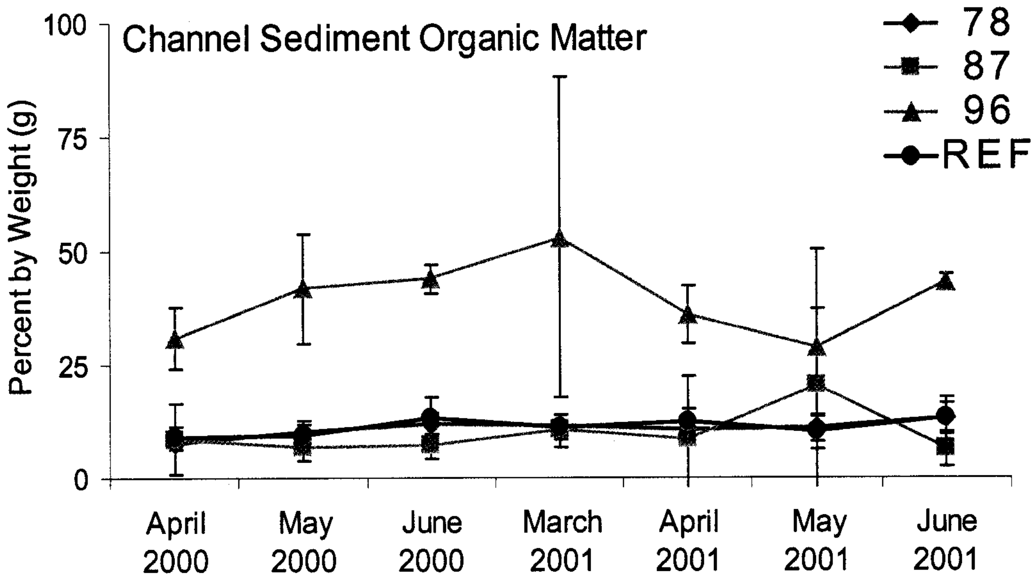


Figure 3.5: Channel sediment organic matter average percent composition by weight (g).

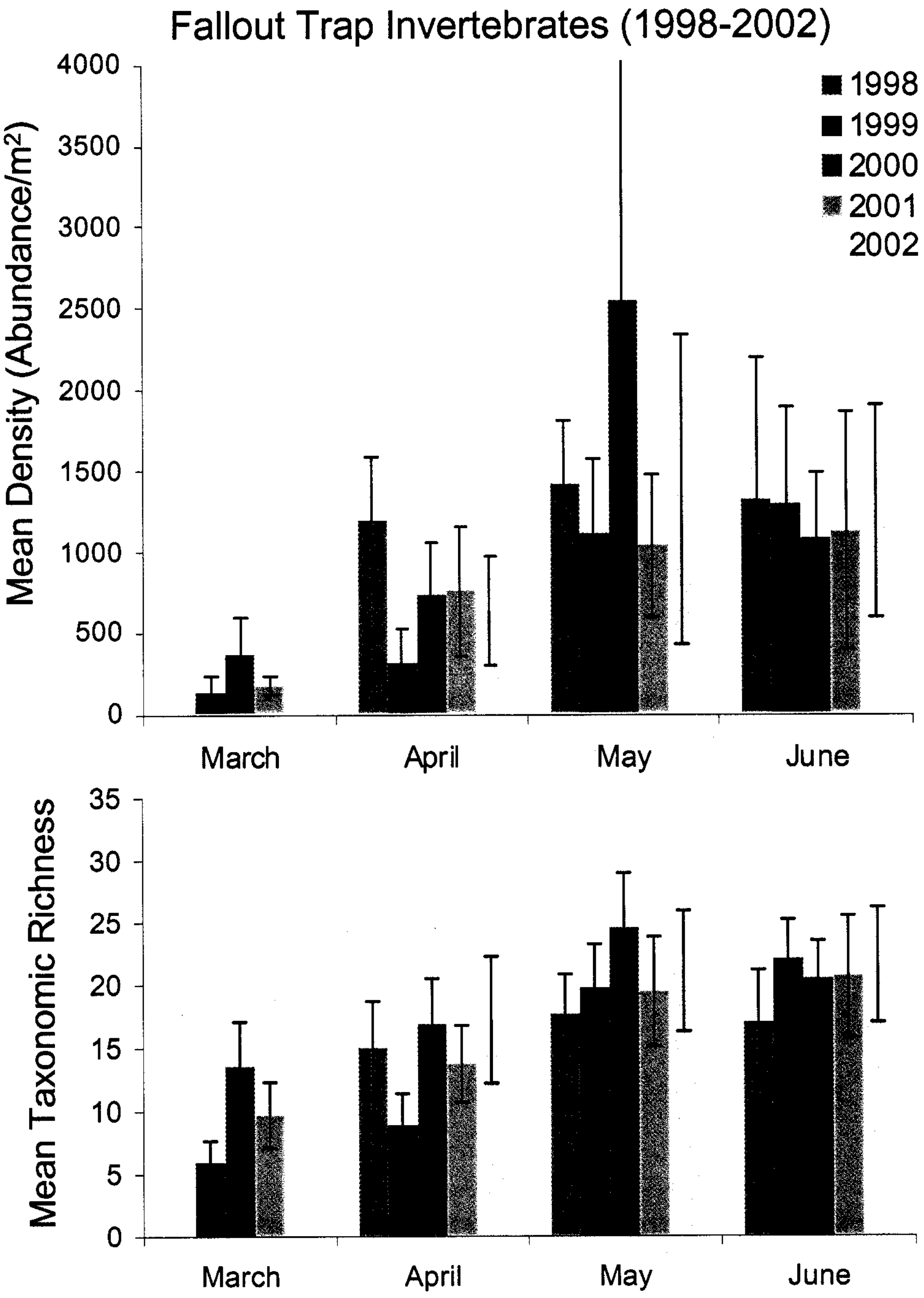


Figure 3.6: Estuary-wide differences in mean density and taxonomic richness among years of fallout trap invertebrates.

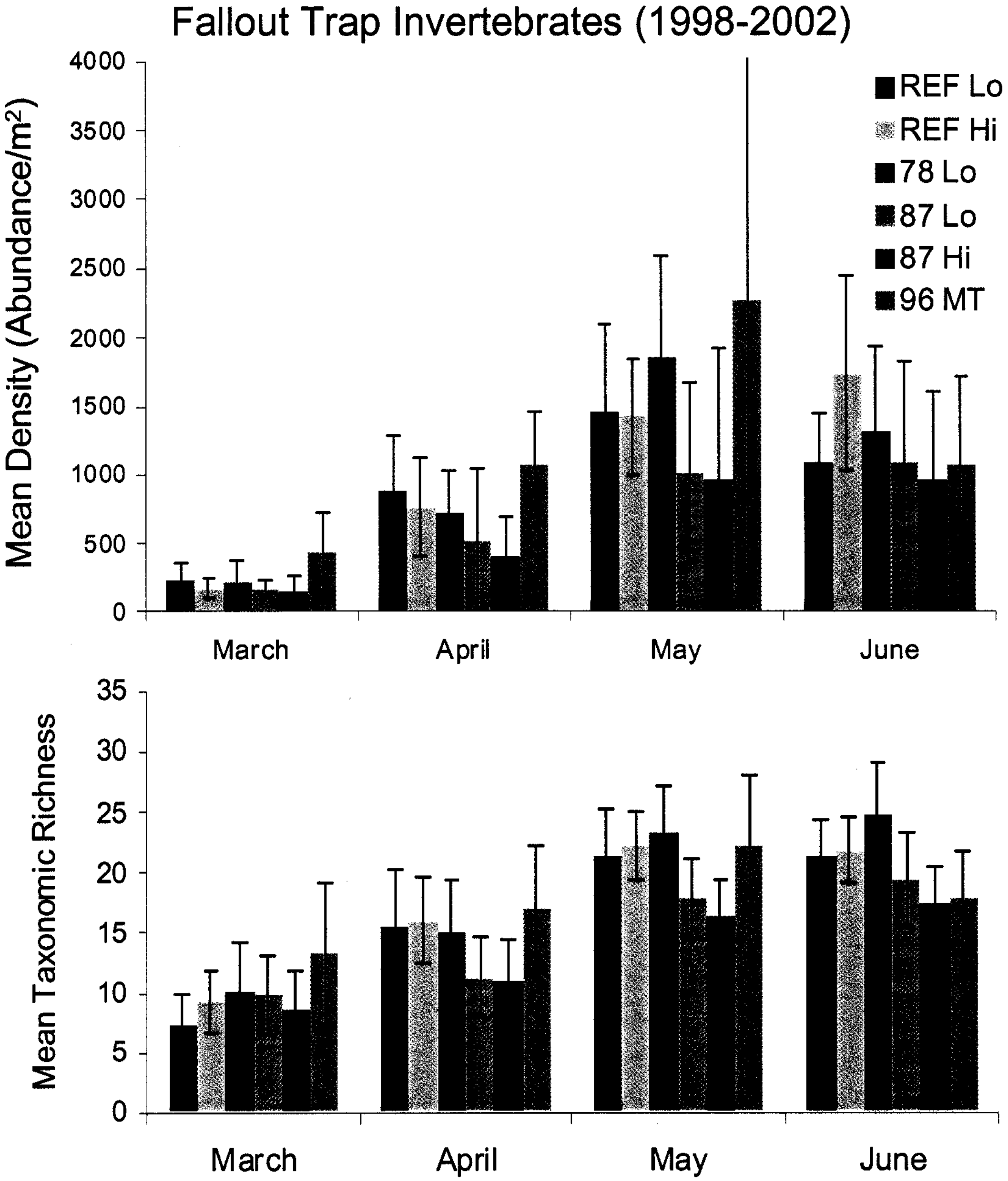


Figure 3.7: Among-site differences in mean density and taxonomic richness of fallout invertebrates at each sampling site.

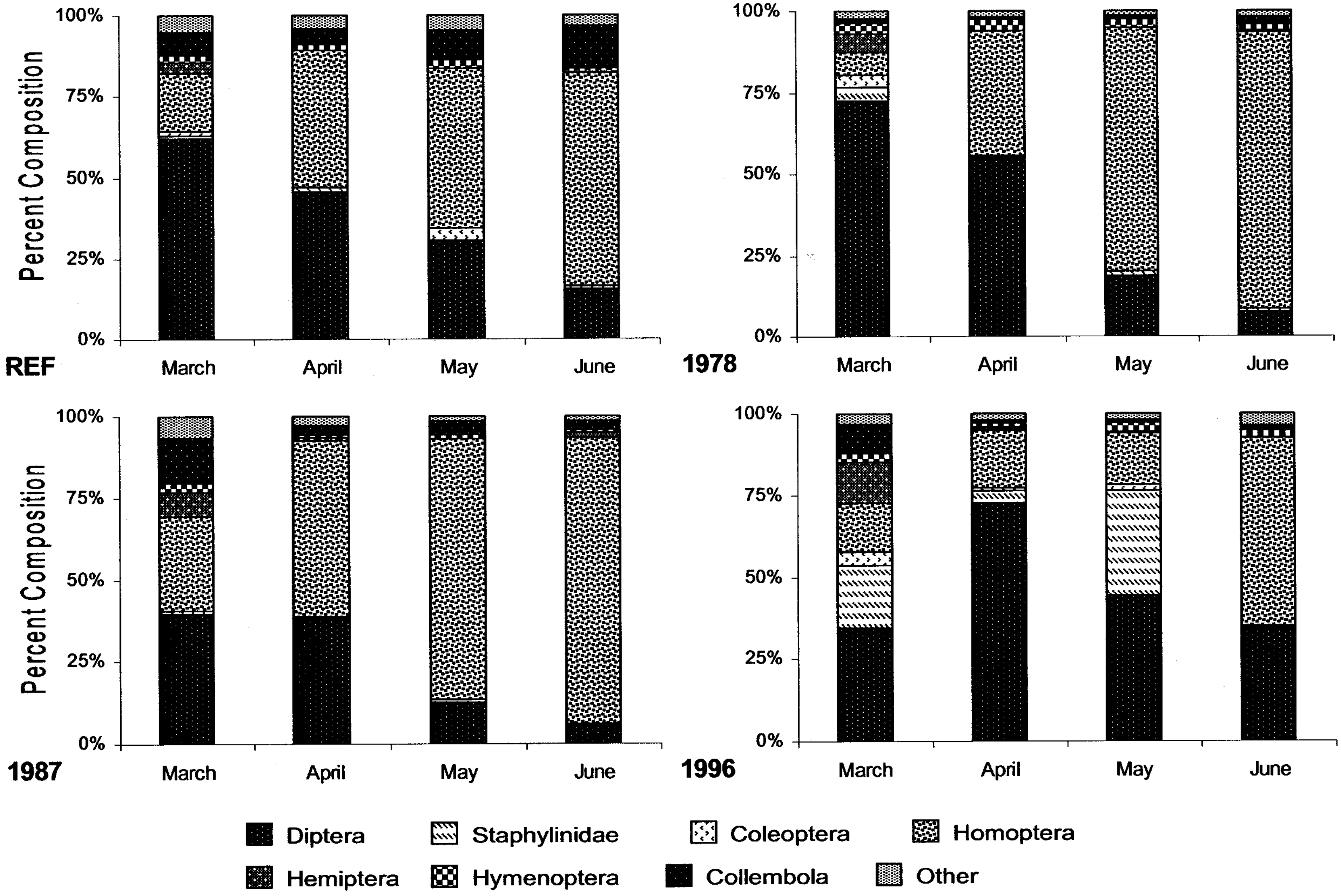


Figure 3.8: Average percent composition of fallout insects by month (1998-2002).

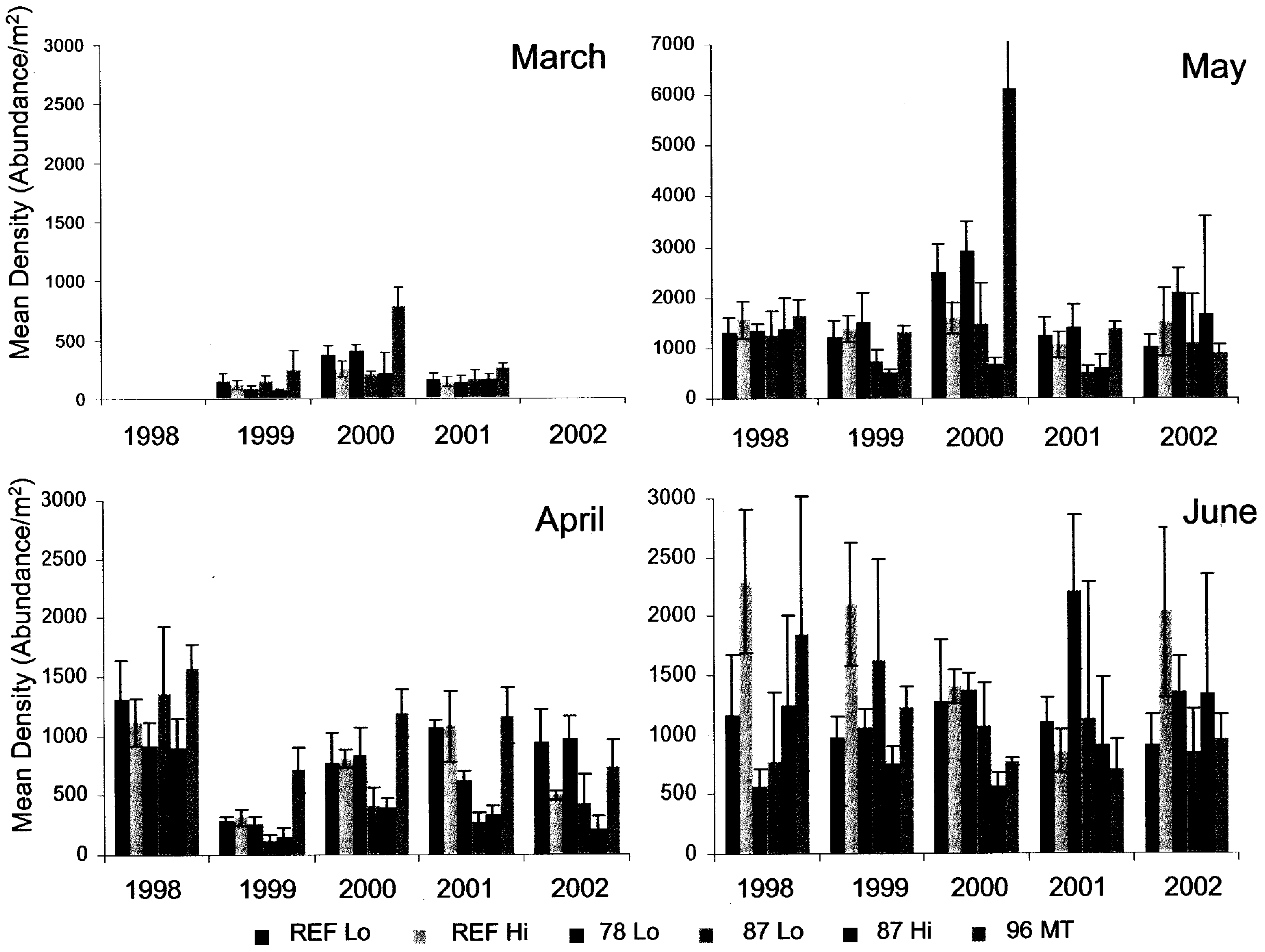


Figure 3.9: Total mean density of IFT invertebrates by site and sampling period (note: different scale on y-axis for May).

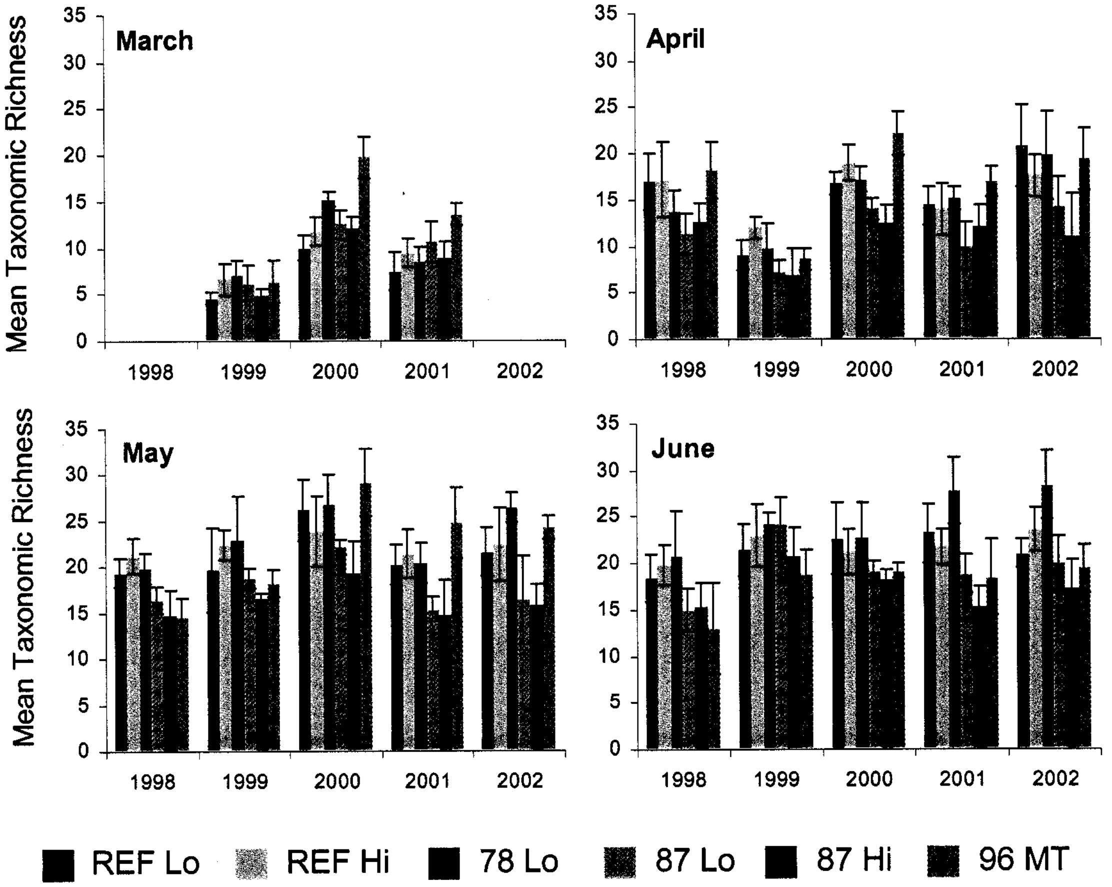


Figure 3.10: Mean taxonomic richness of fallout insects by site and sampling period.

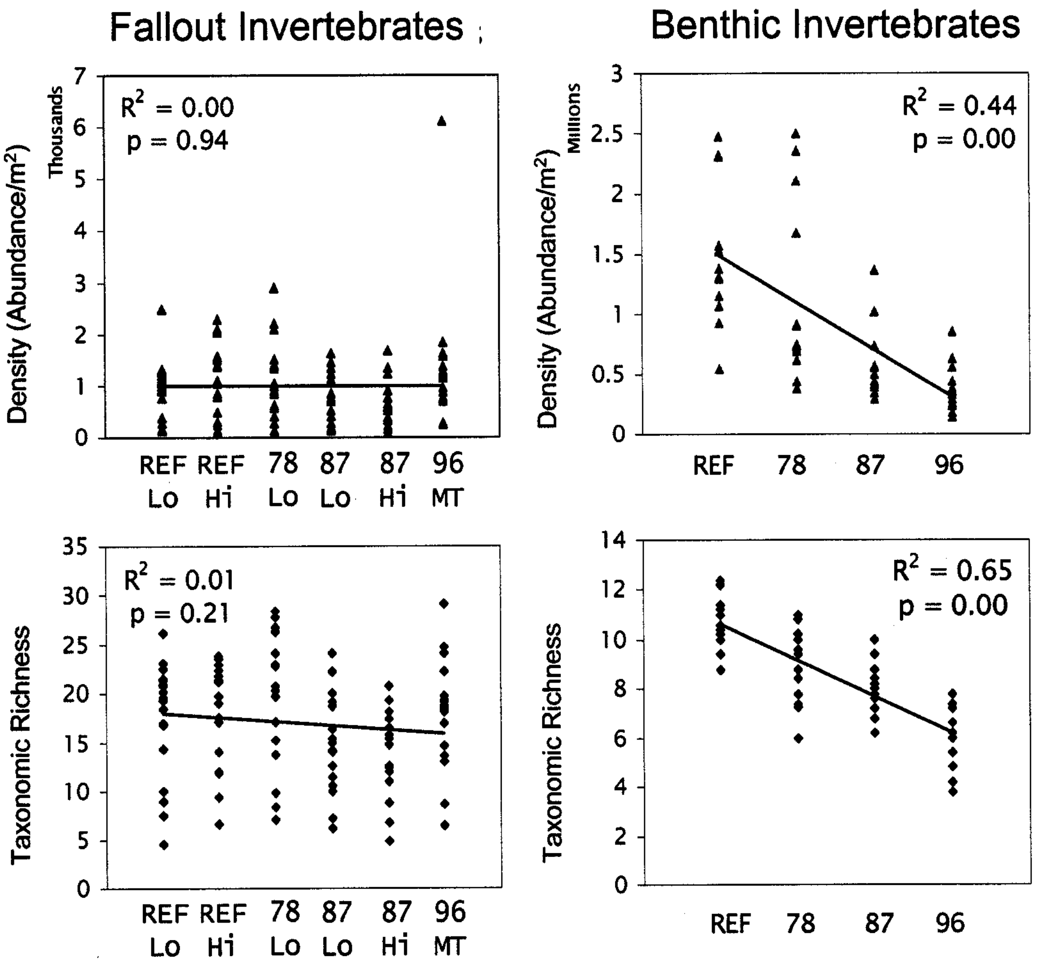


Figure 3.11: Linear regression for density and taxonomic richness of invertebrates among sites.

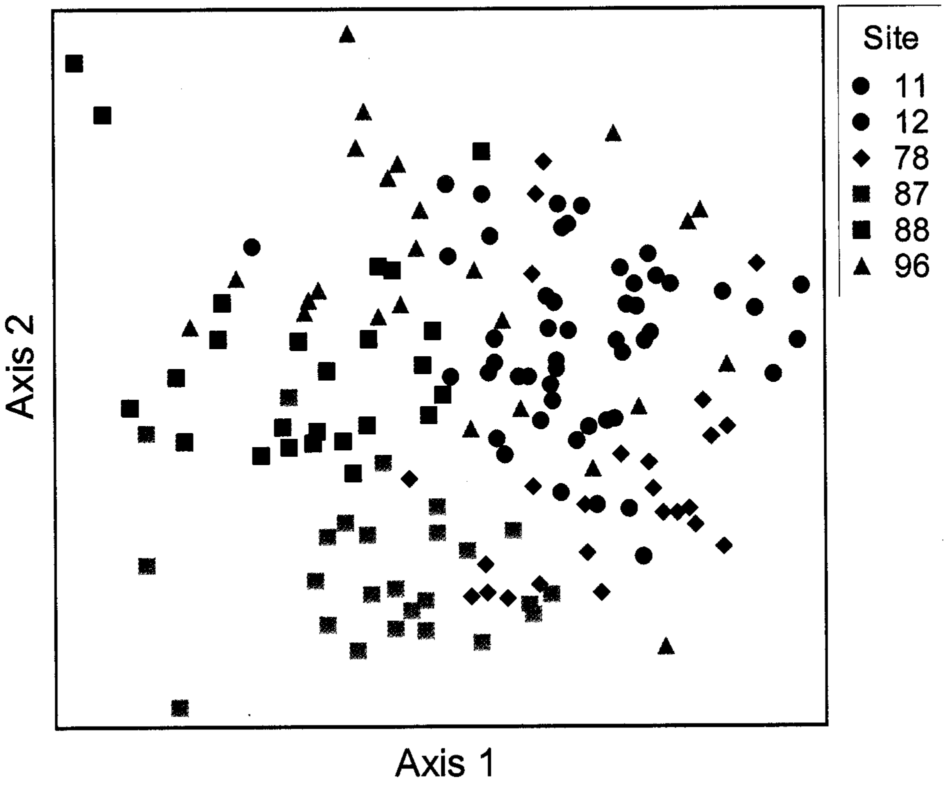
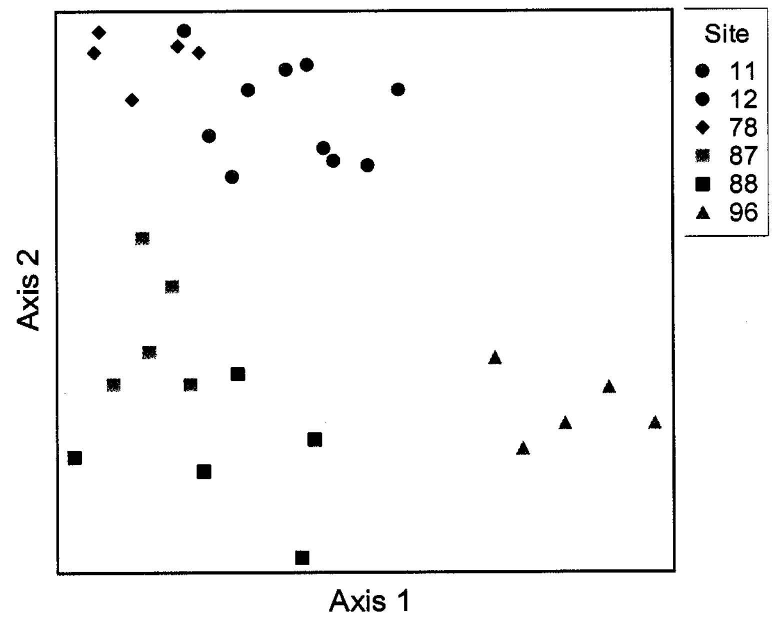
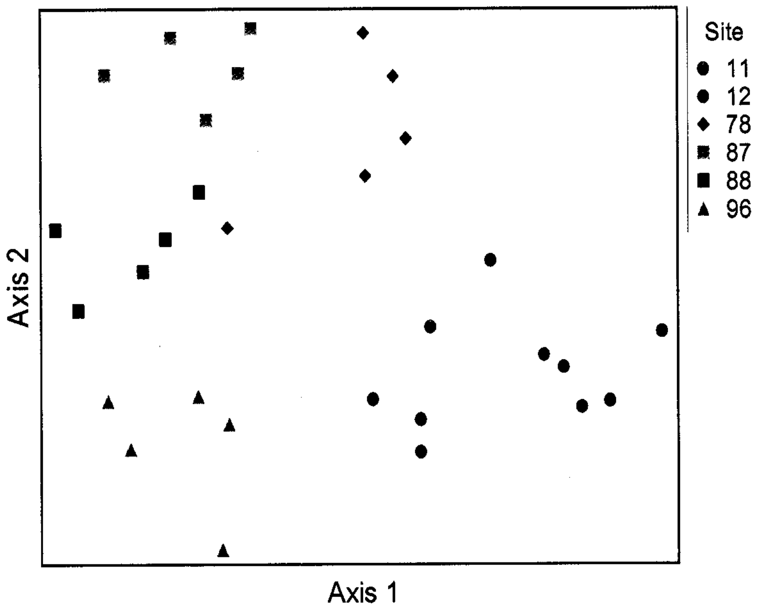


Figure 3.12: Insect community ordination (Nonmetric Multidimensional Scaling - NMS) for all May samples in taxonomic group space by site. Sites are coded as follows: CTR Lo (11), CTR Hi (12), 78 Lo (78), 87 Lo (87), 87 Hi (88), and 96 MT (96).

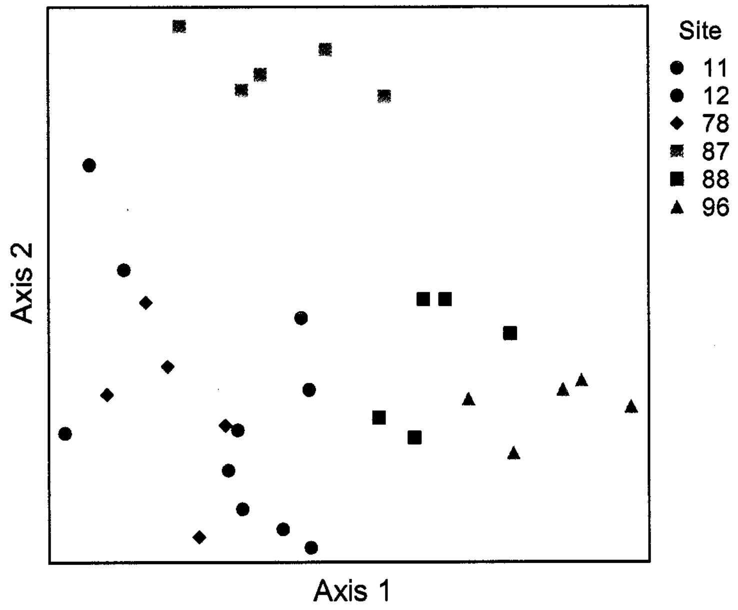


May 1998

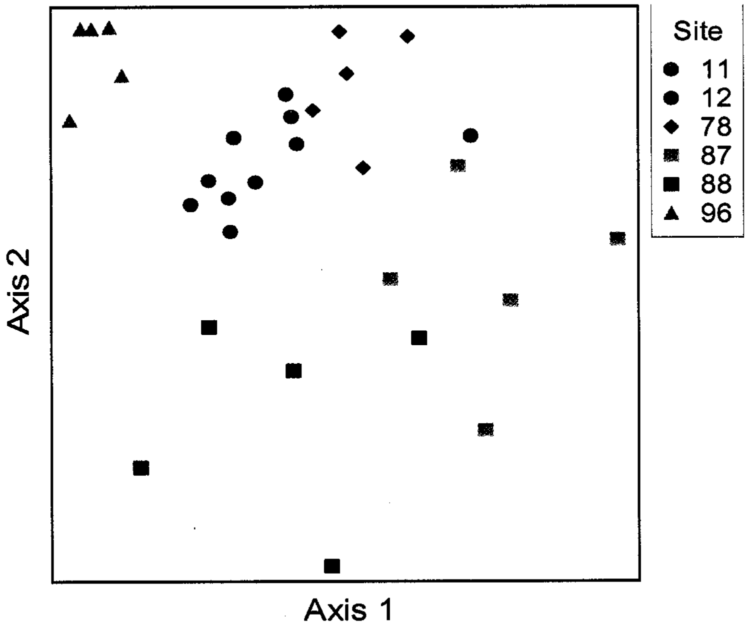


May 1999

Figure 3.13: Insect community ordination (NMS) from May 1998-1999 in taxonomic space by site: REF Lo (11), REF Hi (12), 78 Lo (78), 87 Lo (87), 87 Hi (88), and 96 MT (96).

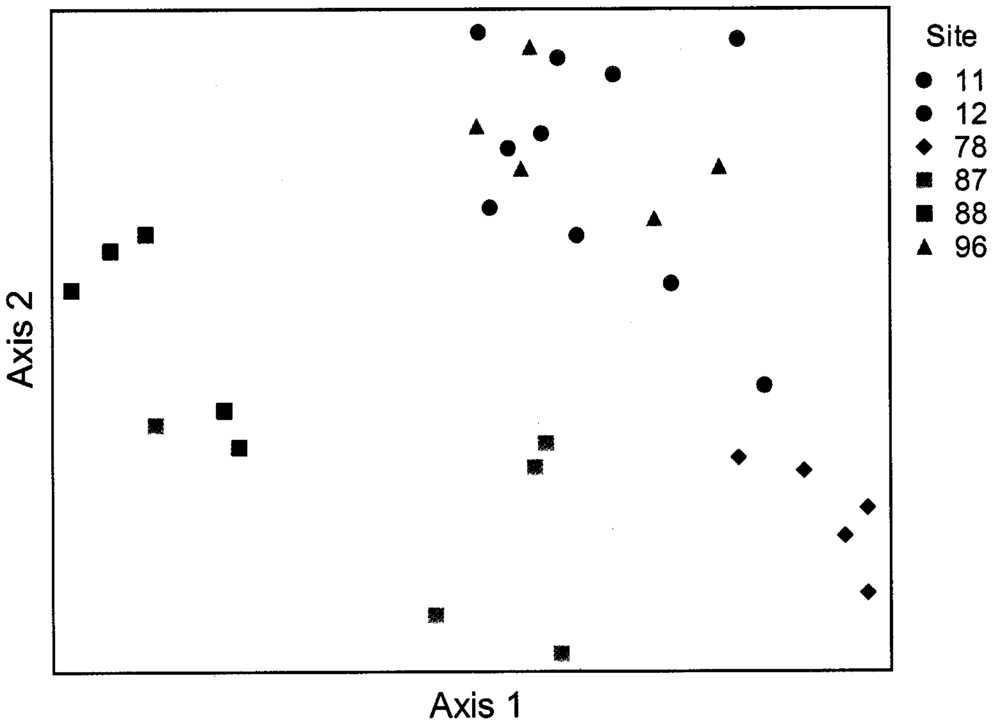


May 2000



May 2001

Figure 3.14: Insect community ordination (NMS) from May 2000-2001 in taxonomic space by site: REF Lo (11), REF Hi (12), 78 Lo (78), 87 Lo (87), 87 Hi (88), and 96 MT (96).



May 2002

Figure 3.15: Insect community ordination (NMS) from May 2002 in taxonomic space by site: REF Lo (11), REF Hi (12), 78 Lo (78), 87 Lo (87), 87 Hi (88), and 96 MT (96).

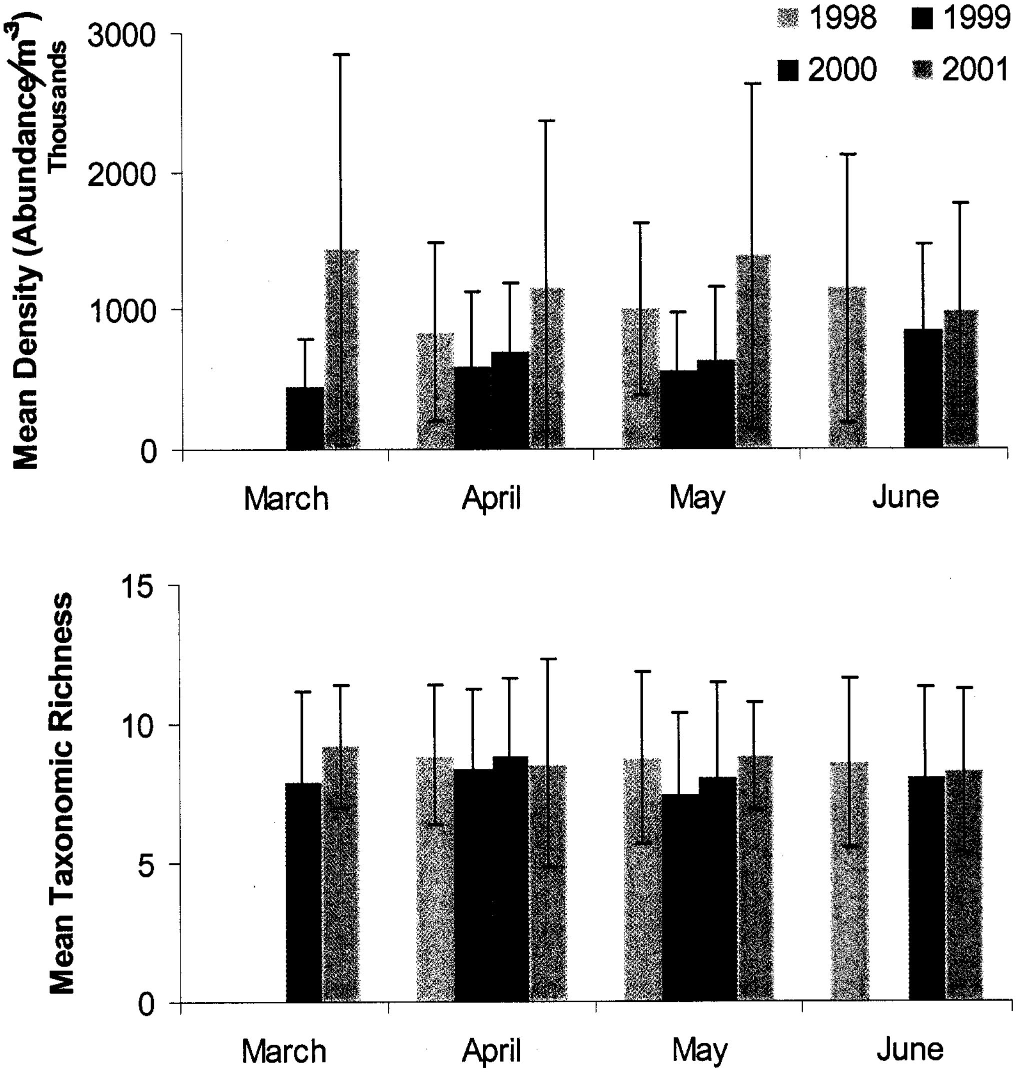


Figure 3.16: Estuary-wide differences in mean density and taxonomic richness for benthic invertebrates.

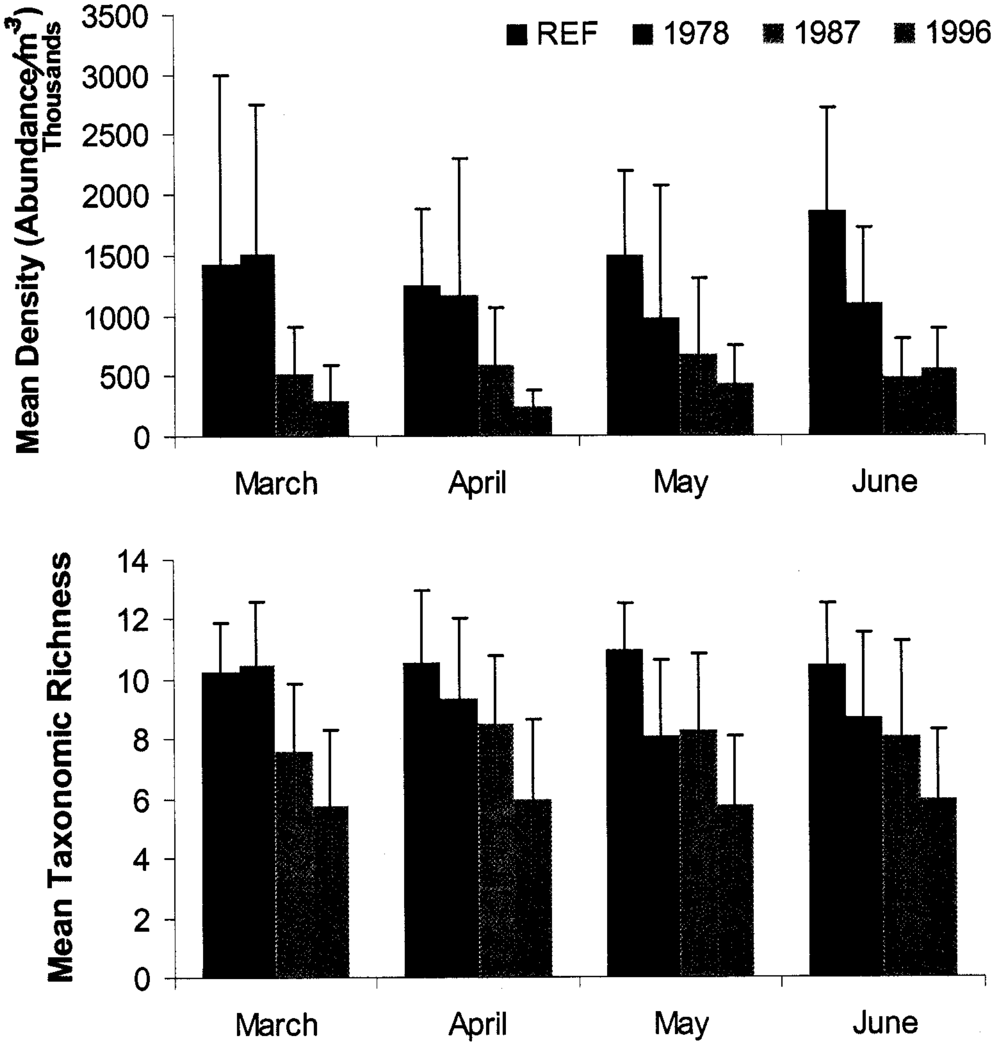


Figure 3.17: Mean density and taxonomic richness of benthic invertebrates by month (1998-2001).

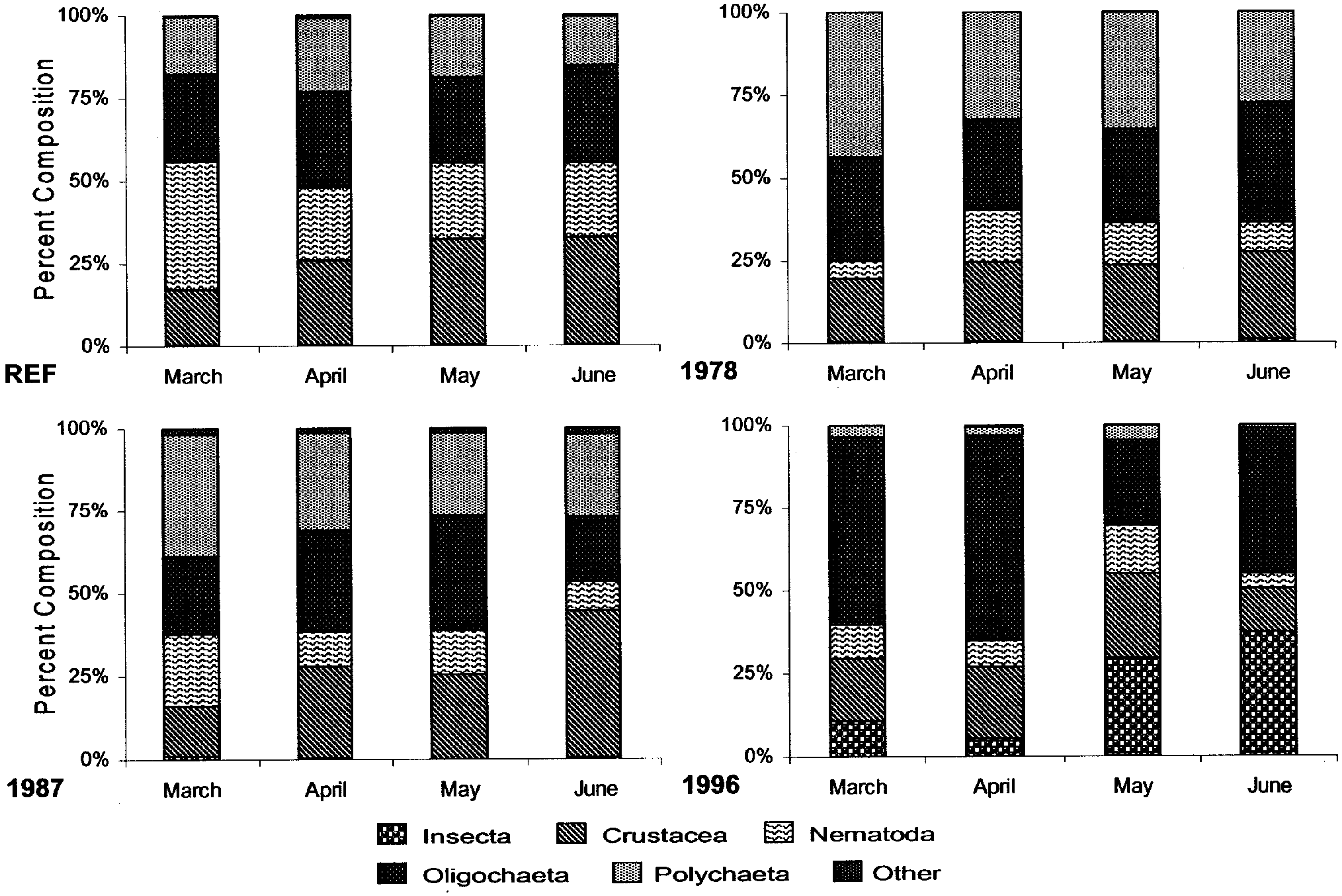


Figure 3.18: Average percent composition of benthic invertebrates by month (1998-2001).

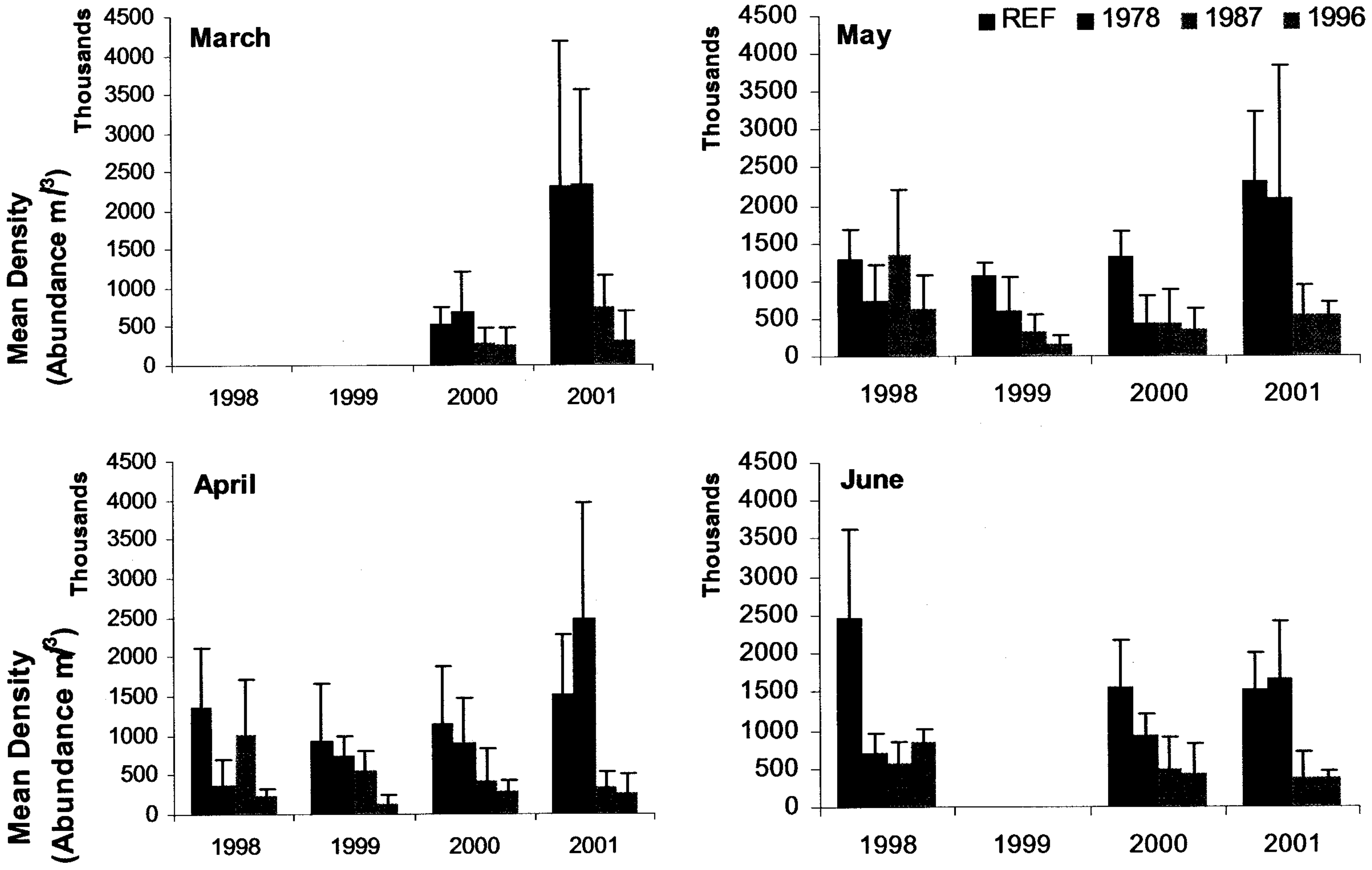


Figure 3.19: Mean density of benthic invertebrates by site and sampling period.

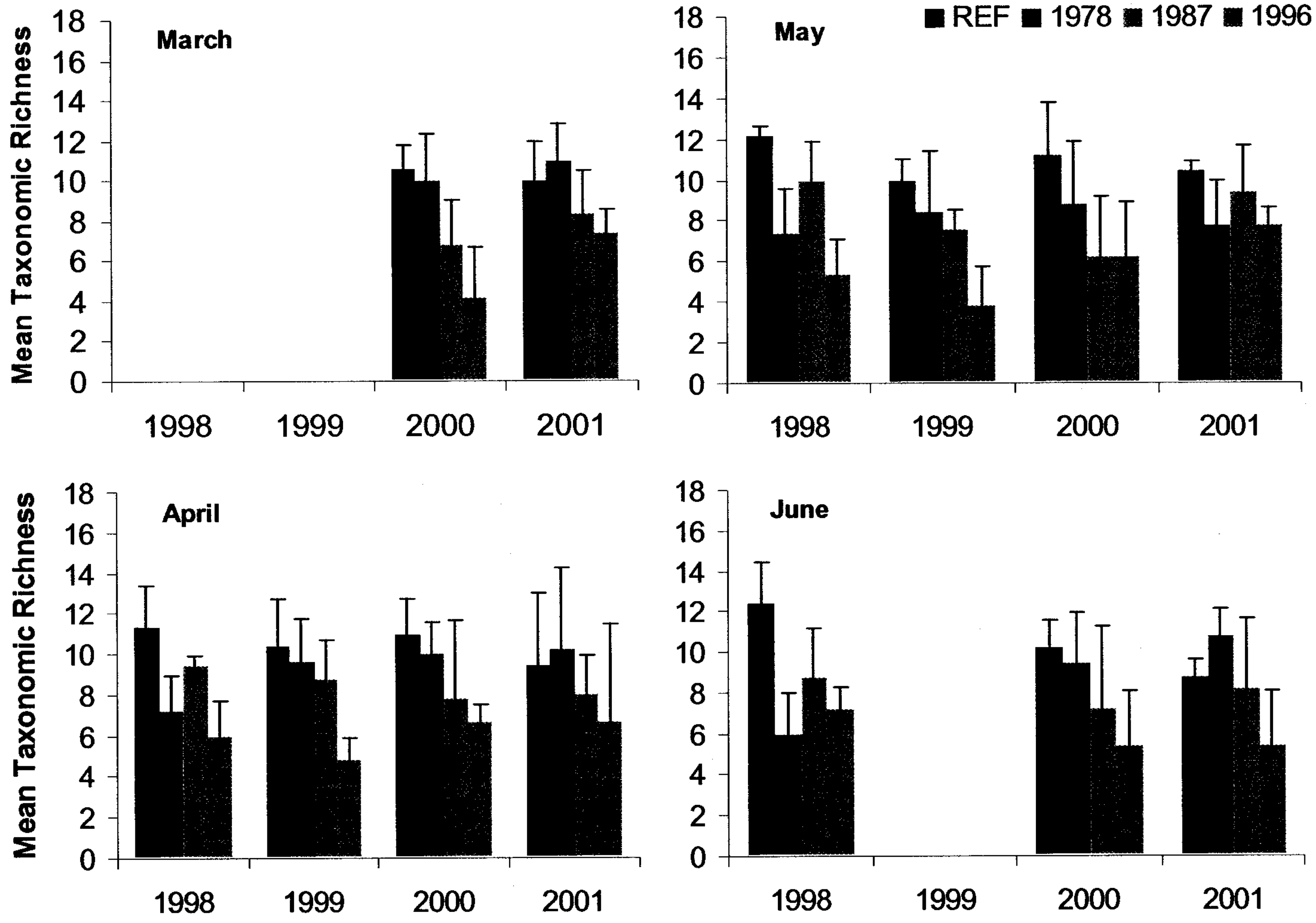


Figure 3.20: Mean taxonomic richness of benthic invertebrates by site and sampling period.

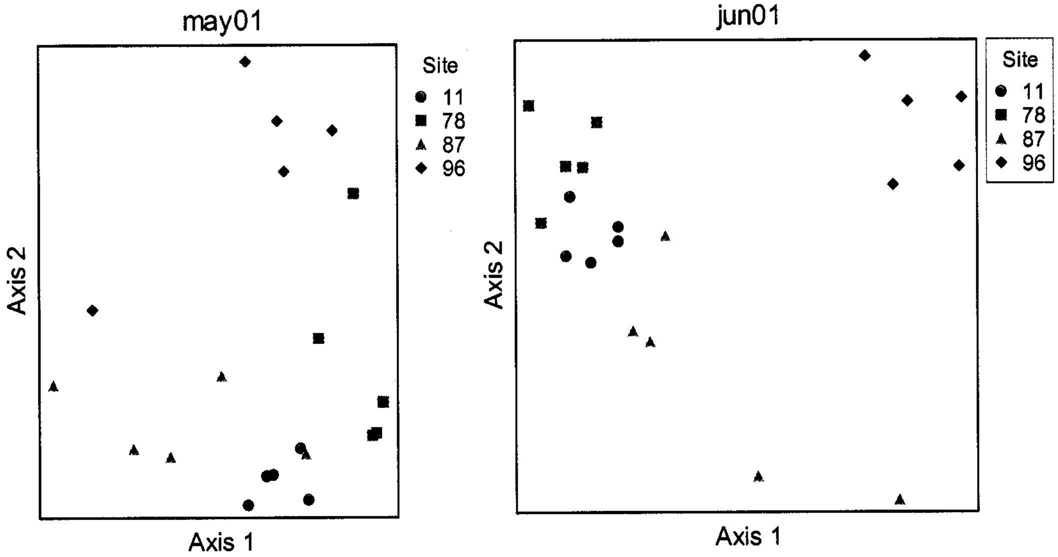


Figure 3.21: Benthic community ordination (NMS) for May and June 2001: REF (11), 1978 marsh (78), 1987 marsh (87), and 1996 marsh (96).

Final Discussion

> "The old prairie lived by the diversity of its plants and animals, all of which were useful because of the sum of their total co-operations and competitions achieved continuity."
>
> -- Aldo Leopold 1949

Living systems are structural phenomena built and operated by the interrelationships created among constituent parts. When ecosystems are disturbed, the incredibly complex web of relationship breaks down and new associations are created. To return an ecosystem to its former state is difficult, as is determining the characteristics and acceptable thresholds of recovered or restored habitat. However, as anything is only understood when disassembled and returned to working condition. If well applied, the study of disturbed, restored and reference ecosystems can serve to further the science of ecology in general. By tracking ecosystem response in a variety terms ecological relationships can be linked to recovering processes.

My aim at the Salmon River estuary was to evaluate ecological processes related to estuarine marsh recovery following dike removal. Habitat conservation and recovery are topics receiving increased attention as disturbance continues to impact species of special concern, such as juvenile Chinook salmon. Through my examination of prey resource and diet composition, I found differences in habitat conditions between reference and recovering marshes, most notably in the abundance of chironomids of all life stages. The diets of fish foraging in the newest and oldest restoring sites were dominated by insect prey, while fish foraging in the reference and 1987 marsh had crustacean-dominated diets. Differences in prey resource were also detected with my invertebrate collections. I also used a bioenergetics model, in an effort to understand the potential implication of diet differences, and to project juvenile chinook growth patterns in each of the recovering and reference marshes. The reference site, while never reflecting the highest growth potential, did reflect consistent growth conditions. The recovering sites were all variable in terms of modeled growth potential, in direct relation to invertebrate prey (e.g., trichoptera larvae) and/or sampling month. Finally, I examined invertebrate community differences to find a relationship with recovering marsh age. In terms of marsh surface invertebrates, site-to-site differences were obscured by yearly and monthly variation, however, by assessing sampling period independently, I detected differences among sites, and repeatable and exclusive invertebrate indicators to marsh age. The benthic invertebrate communities were more strongly influenced by site-to-site differences than by yearly or seasonal variation and, in general, the benthic invertebrate communities were similar among the REF, 1987 and 1978 sites, but distinct differences in the community occurred in the 1996 site. The most repeatable and exclusive indicator in the 1996 site were chironomid larvae. In the "shredder" (CPOM ≥ 1 mm diameter) functional feeding group (Merritt & Cummins 1996). The proliferation of chironomids in the newest restoration site may be due to the higher channel sediment organic matter content (Fig. 3.5), present due to the die-back of pasture grasses extant pre-restoration.

By assessing the marshes in this series of ways, I have determined that recovering sites are more variable in terms of modeled growth potential, have marsh surface invertebrates which covary with season, year and site, and benthic invertebrate communities reach equivalency with reference conditions after an initial period of channel sediment organic matter breakdown. Understanding ecological processes involved in restoration of estuarine ecosystems lends itself to juvenile salmon habitat recovery. As estuarine marshes represent important components of early life history for juvenile Chinook and coho salmon, effective recovery of diked pasturelands may help rehabilitate depressed Oregon coast populations.

This body of work, all part of the cooperative project to better understand restorative processes within the frame work of the Salmon River estuary, attempts to approach the practice of restoration assessment from a flexible viewpoint. Ecological assembly rules work to establish the nature and characteristics of a living system (Weiher & Keddy 2004). Determining what assembles an ecosystem's processes requires an investigation into several aspects of what ecosystems do: the interactions among constituent parts, the performance of specific species, and variation among communities. Simenstad and Cordell (2000) suggested estuarine assessment for juvenile salmon should proceed with evaluation of capacity, opportunity and realized performance. Bottom et al. (2005) overlayed the importance of life history structure in determining the importance of estuarine marshes in salmon migration. These concepts form the foundation of my thoughts in this dissertation. I have gone on to extend my ideas into the realm of ecosystem mechanics, in the absence of species rehabilitation. From learning about the interactions among juvenile Chinook salmon and invertebrate communities, and the marshes of varying ages, I could begin to see compositional differences exist between the reference and restored sites. How important those differences were to the task of salmon rehabilitation required a detailed understanding of how realized performance, or production, would be affected. A computer model helped evaluate the effects of temperature, prey resource and energy, fish size and consumption. The disparity in interactions had led to a sustained difference in function between restored and reference sites. In some cases the restored sites performed at a much higher level than the reference, and other times at much lower levels. Relative stability in environmental conditions may be expected from ecosystems less subject to disturbance.

Cumulatively, the value of stability versus variability is difficult to determine. In the case of rehabilitation for juvenile salmon, introducing a variable source of habitat is certainly preferred to the lack of habitat, and is likely (Hendry et al. 2000) able to enhance diversity in a population of fish narrowed into hatchery-dominated life history structures (Bottom et al. 2005). Finally, I used the invertebrate community to investigate variation among marsh age and reference conditions.

Two phytophagus beetles, (Chrysomelidae and Elateridae), were exclusive and consistent indicator for the REF site. Vegetation structure may be adjusted and biogeochemical fluxes changed to adapt to prevailing environmental conditions by native phytophagus insects (Schowalter 2000). The presence of phytophagus beetles, as exclusive indicator to reference conditions, supports the idea of an unphased (Abugov 1982) environmental disturbance (constant herbivory) leading to increased biodiversity. This relationship also supports the link between a consistent system characteristic, or constituent part, and an ecological process. The predaceous beetle (Staphylinidae) indicated conditions in the newest recovering site. Without the intricate complex of vegetation found in the reference conditions, the invertebrate community shifted dominating functional feeding groups. Feeding groups related to decaying vegetation (e.g., chironomid larvae) and predaceous beetles (e.g., rove beetles) were found to be exclusive indicators. While the presence of chironomid larvae likely relates to increased channel sediment organic matter; the rove beetles may be responding to the trophic cascade initiated by dying pasture grasses.

Hypotheses, theories and laws upon which experimentation and testing are determined develop under a paradigmatic framework. This frame of understanding and thinking is necessary for the scientific method to operate, but if the structure is too rigid fissures may develop jeopardizing the truth and applicability of the science. When appraising the dynamic nature of an ecosystem, understanding must arise from constantly changing ideas not static theories and laws. Paradigmatic thinking has led some restoration ecologists to develop their understanding of recovering systems based on the theories of succession (Zedler 2001), expecting an orderly development of characteristics (especially floristic characteristics) over time. Ehrenfeld (2000) suggests ecologists should stop expecting to find simple rules or Newtonian laws to apply in the quantum world of nature, by instead develop probabilistic understanding. Scientists need to acknowledge how applied and adaptive solutions will have to be practiced and that no one paradigm or context for setting restoration goals exists. The freedom to use all available tools, methods, perspectives and designs allows ecologists to shape their knowledge in a constantly evolving form and develop a dynamic acuity more suitable to understanding the processes, mechanisms, patterns and phenomenas of the natural world.

Recommendations

It is only through the most arduous things that we learn of what we are truly capable. The long and detailed work necessary to understand ecosystem dynamics is enlightening and worthwhile in developing a better response to our world. The foundation of ecological science is built on the detailed work of many years. Reimers (1973) represents the seminal investigation into Chinook salmon life history and opened many doors into new understanding.

Those understandings contributed significantly to the thoughts and work of many others and helped develop better management schemes for Pacific salmon (Lichatowich 1999). Furthermore, as more restoration monies are spent in concert with the progressing environmental alteration, the need to conceptualize the living system and ourselves within it increases in importance. Studies of juvenile salmon on the Oregon coast (and beyond where applicable) should be conducted with high resolution in locations instructive to natural resource management and scientists. The Salmon River estuary offered the perfect conditions, a formerly great run of wild Chinook and coho, estuarine marsh protections and restorations, and a hatchery. The conditions at the Salmon River provided us an opportunity to answer a range of inter-tangled questions. Simpler approaches may have been employed had the hatchery not been there, but questions regarding how hatcheries can change life history characteristics would not have been possible. The knowledge acquired may be useful to a range of rivers with some of the same conditions. Situations such as this may provide better instruction in restoration science than the more unique, less common conditions in other watersheds.

Methods of assessment should continue to include spatial and temporal variation, and measures of parameters in interrelationship among components (i.e., diet composition), ecosystem performance and community structure. Additions could include consistent length and weight measures of captured juvenile salmon, as our best estimation regarding upper watershed production differences among years arose from the variation in juvenile salmon condition factor. Natural resource managers should also focus on landscape positioning when choosing and designing restoration projects. Habitats occur as a mosaic across the landscape, and the qualities and values of each contributes to overall system mechanics. Enhancing values and qualities in ecosystems may fit management goals, but the overarching tenet of restoration should include the detailed understanding of the mechanics and intervention efforts to remove and recover processes lost under disturbance regimes. Taking advantage of a restoration project to further understand the relationship of elevation to marsh recovery was performed in South Slough National Estuarine Research Reserve, and included long term monitoring (9 years monitoring since project completion) of vegetation and invertebrate communities (Cornu & Sadro 2002). A dual goal was met with the project: dikes were removed to recovery estuarine marsh; and by design, an experiment looking at processes controlled by marsh elevation was performed. The joint effort of scientists and managers should be to include consideration of research questions in every project design. If projects are designed as experiments, the underlying processes and mechanisms of natural organization may be more clearly seen.

Epilogue

"There are, however, parts of every drowned tree that refuse to become part of this cycle. There is, in every log, a series of cross-grained, pitch-hardened masses where long-lost branches once joined the trees trunk. 'Knots' they are called, in a piece of lumber. But in the bed of a river, after the parent log has broken down and vanished, these stubborn masses take on a very different appearance, and so perhaps deserve a different name. 'River teeth' is what we called them as kids, because that's what they look like. Like enormous fangs, often with a connected, cross-grained root.

I don't know how long these teeth last, but even on the rainy coast I'd guess centuries: you sense antiquity when you heft one. Because their pitch content is so high, and hardened pitch outlasts the grainy wood fiber, the oldest teeth lost much of their resemblance to wood. Some look like Neolithic hand tools, others like mammals-miniature seals, otters, manatees. Still other resemble art objects-something intelligently worked, not just worn. And to an extent this is what they become. There is life in rivers and strength; there are countless grinders and sanders: in a relic the waters have shaped so long, why wouldn't we begin to glimpse the river's mind and blind artistry?"

David James Duncan
1995

"One fish, two fish, red fish, blue fish."

Dr. Suess

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Public Law 93-535
93rd Congress, H. R. 8352
December 22, 1974

An Act

88 STAT. 1732

To establish the Cascade Head Scenic-Research Area in the State of Oregon, and for other purposes.

Cascade Head Scenic-Research Area, Oreg. Establishment. 16 USC 541.

Be it enacted by the Senate and House of Representatives of the United States of America in Congress assembled, That in order to provide present and future generations with the use and enjoyment of certain ocean headlands, rivers, streams, estuaries, and forested areas, to insure the protection and encourage the study of significant areas for research and scientific purposes, and to promote a more sensitive relationship between man and his adjacent environment, there is hereby established, subject to valid existing rights, the Cascade Head Scenic-Research Area (hereinafter referred to as "the Area") in the Siuslaw National Forest in the State of Oregon.

Administration. 16 USC 541a.

Sec. 2. The administration, protection, development, and regulation of use of the Area shall be by the Secretary of Agriculture (hereinafter referred to as the "Secretary") in accordance with the laws, rules, and regulations applicable to national forests, in such manner as in his judgment will best contribute to attainment of the purposes of this Act.

Boundaries. 16 USC 541b.

Sec. 3. (a) The boundaries of the Area, and the boundaries of the subareas included therein, shall be those shown on the map entitled "Proposed Cascade Head Scenic-Research Area", dated June 1974, which is on file and available for public inspection in the office of the Chief, Forest Service, United States Department of Agriculture: *Provided*, That, from time to time, the Secretary may, after public hearing or other appropriate means for public participation, make adjustments in the boundaries of subareas to reflect changing natural conditions or to provide for more effective management of the Area and each of the subareas in accordance with the purposes and provisions of this Act.

Comprehensive management plan.

(b) As soon as practicable after the enactment of this Act, the Secretary shall, with provisions for appropriate public participation in the planning process, develop a comprehensive management plan for the Area. Said plan shall prescribe specific management objectives and management controls necessary for the protection, management, and development of the Area and each of the subareas established pursuant to subsection (c) hereof.

Subareas, establishment.

(c) Within the Area, the following subareas shall be established and shall be managed in accord with the following primary management objectives which shall be supplemental to the general management objectives applicable to the entire Area:

(1) Estuary and Associated Wetlands Subarea: An area managed to protect and perpetuate the fish and wildlife, scenic, and research-education values, while allowing dispersed recreation use, such as sport fishing, nonmotorized pleasure boating, waterfowl hunting, and other uses which the Secretary determines are compatible with the protection and perpetuation of the unique natural values of the subarea. After appropriate study, breaching of existing dikes may be permitted within the subarea.

(2) Lower Slope-Dispersed Residential Subarea: An area managed to maintain the scenic, soil and watershed, and fish and wildlife values, while allowing dispersed residential occupancy, selective recreation use, and agricultural use.

(3) Upper Timbered Slope and Headlands Subareas: Areas managed to protect the scenic, soil and watershed, and fish and wildlife values while allowing selective recreation and extensive

Appendix A: Law Creating Cascade Head Scenic-Research Area

Public Law 93-535
93rd Congress, H. R. 8352
December 22, 1974

An Act

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Cascade Head Scenic-Research Area, Oreg. Establishment. 16 USC 541.

Be it enacted by the Senate and House of Representatives of the United States of America in Congress assembled, That in order to provide present and future generations with the use and enjoyment of certain ocean headlands, rivers, streams, estuaries, and forested areas, to insure the protection and encourage the study of significant areas for research and scientific purposes, and to promote a more sensitive relationship between man and his adjacent environment, there is hereby established, subject to valid existing rights, the Cascade Head Scenic-Research Area (hereinafter referred to as "the Area") in the Siuslaw National Forest in the State of Oregon.

Administration. 16 USC 541a.

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Boundaries. 16 USC 541b.

Sec. 3. (a) The boundaries of the Area, and the boundaries of the subareas included therein, shall be those shown on the map entitled "Proposed Cascade Head Scenic-Research Area", dated June 1974, which is on file and available for public inspection in the office of the Chief, Forest Service, United States Department of Agriculture: *Provided*, That, from time to time, the Secretary may, after public hearing or other appropriate means for public participation, make adjustments in the boundaries of subareas to reflect changing natural conditions or to provide for more effective management of the Area and each of the subareas in accordance with the purposes and provisions of this Act.

Comprehensive management plan.

(b) As soon as practicable after the enactment of this Act, the Secretary shall, with provisions for appropriate public participation in the planning process, develop a comprehensive management plan for the Area. Said plan shall prescribe specific management objectives and management controls necessary for the protection, management, and development of the Area and each of the subareas established pursuant to subsection (c) hereof.

Subareas, establishment.

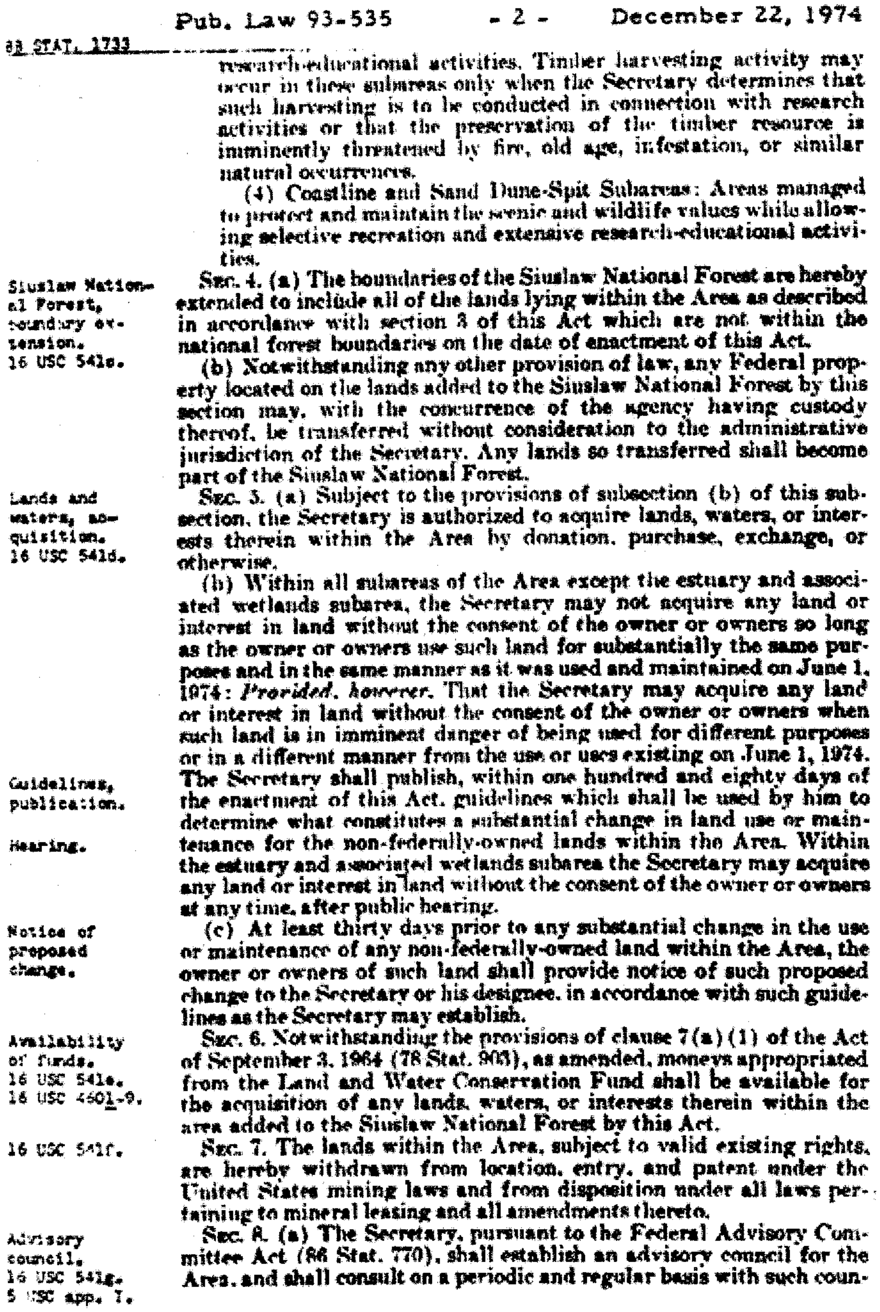
(c) Within the Area, the following subareas shall be established and shall be managed in accord with the following primary management objectives which shall be supplemental to the general management objectives applicable to the entire Area:

(1) Estuary and Associated Wetlands Subarea: An area managed to protect and perpetuate the fish and wildlife, scenic, and research-education values, while allowing dispersed recreation use, such as sport fishing, nonmotorized pleasure boating, waterfowl hunting, and other uses which the Secretary determines are compatible with the protection and perpetuation of the unique natural values of the subarea. After appropriate study, breaching of existing dikes may be permitted within the subarea.

(2) Lower Slope-Dispersed Residential Subarea: An area managed to maintain the scenic, soil and watershed, and fish and wildlife values, while allowing dispersed residential occupancy, selective recreation use, and agricultural use.

(3) Upper Timbered Slope and Headlands Subareas: Areas managed to protect the scenic, soil and watershed, and fish and wildlife values while allowing selective recreation and extensive

Appendix A, Cont.: Law Creating Cascade Head Scenic-Research Area



Pub. Law 93-535 - 2 - December 22, 1974

88 STAT. 1733

research-educational activities. Timber harvesting activity may occur in these subareas only when the Secretary determines that such harvesting is to be conducted in connection with research activities or that the preservation of the timber resource is imminently threatened by fire, old age, infestation, or similar natural occurrences.

(4) Coastline and Sand Dune-Spit Subareas: Areas managed to protect and maintain the scenic and wildlife values while allowing selective recreation and extensive research-educational activities.

Siuslaw National Forest, boundary extension.
16 USC 541c.

Sec. 4. (a) The boundaries of the Siuslaw National Forest are hereby extended to include all of the lands lying within the Area as described in accordance with section 3 of this Act which are not within the national forest boundaries on the date of enactment of this Act.

(b) Notwithstanding any other provision of law, any Federal property located on the lands added to the Siuslaw National Forest by this section may, with the concurrence of the agency having custody thereof, be transferred without consideration to the administrative jurisdiction of the Secretary. Any lands so transferred shall become part of the Siuslaw National Forest.

Lands and waters, acquisition.
16 USC 541d.

Sec. 5. (a) Subject to the provisions of subsection (b) of this subsection, the Secretary is authorized to acquire lands, waters, or interests therein within the Area by donation, purchase, exchange, or otherwise.

(b) Within all subareas of the Area except the estuary and associated wetlands subarea, the Secretary may not acquire any land or interest in land without the consent of the owner or owners so long as the owner or owners use such land for substantially the same purposes and in the same manner as it was used and maintained on June 1, 1974: *Provided, however*, That the Secretary may acquire any land or interest in land without the consent of the owner or owners when such land is in imminent danger of being used for different purposes or in a different manner from the use or uses existing on June 1, 1974.

Guidelines, publication.

The Secretary shall publish, within one hundred and eighty days of the enactment of this Act, guidelines which shall be used by him to determine what constitutes a substantial change in land use or maintenance for the non-federally-owned lands within the Area.

Hearing.

Within the estuary and associated wetlands subarea the Secretary may acquire any land or interest in land without the consent of the owner or owners at any time, after public hearing.

Notice of proposed change.

(c) At least thirty days prior to any substantial change in the use or maintenance of any non-federally-owned land within the Area, the owner or owners of such land shall provide notice of such proposed change to the Secretary or his designee, in accordance with such guidelines as the Secretary may establish.

Availability of funds.
16 USC 541e.
16 USC 460l-9.

Sec. 6. Notwithstanding the provisions of clause 7(a)(1) of the Act of September 3, 1964 (78 Stat. 903), as amended, moneys appropriated from the Land and Water Conservation Fund shall be available for the acquisition of any lands, waters, or interests therein within the area added to the Siuslaw National Forest by this Act.

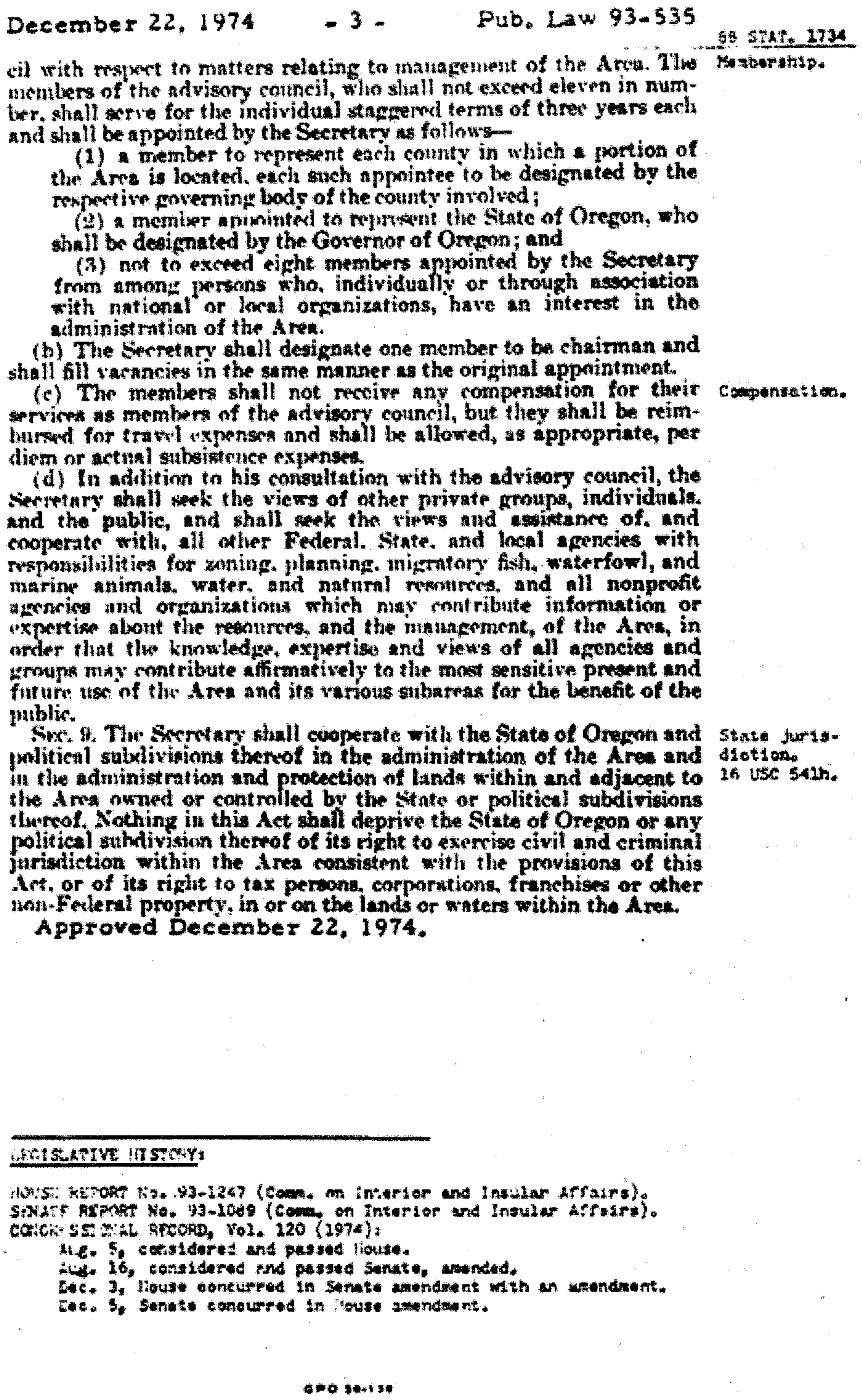
16 USC 541f.

Sec. 7. The lands within the Area, subject to valid existing rights, are hereby withdrawn from location, entry, and patent under the United States mining laws and from disposition under all laws pertaining to mineral leasing and all amendments thereto.

Advisory council.
16 USC 541g.
5 USC app. I.

Sec. 8. (a) The Secretary, pursuant to the Federal Advisory Committee Act (86 Stat. 770), shall establish an advisory council for the Area, and shall consult on a periodic and regular basis with such coun-

Appendix A, Cont.: Law Creating Cascade Head Scenic-Research Area



cil with respect to matters relating to management of the Area. The members of the advisory council, who shall not exceed eleven in number, shall serve for the individual staggered terms of three years each and shall be appointed by the Secretary as follows—

Membership.

(1) a member to represent each county in which a portion of the Area is located, each such appointee to be designated by the respective governing body of the county involved;

(2) a member appointed to represent the State of Oregon, who shall be designated by the Governor of Oregon; and

(3) not to exceed eight members appointed by the Secretary from among persons who, individually or through association with national or local organizations, have an interest in the administration of the Area.

(b) The Secretary shall designate one member to be chairman and shall fill vacancies in the same manner as the original appointment.

Compensation.

(c) The members shall not receive any compensation for their services as members of the advisory council, but they shall be reimbursed for travel expenses and shall be allowed, as appropriate, per diem or actual subsistence expenses.

(d) In addition to his consultation with the advisory council, the Secretary shall seek the views of other private groups, individuals, and the public, and shall seek the views and assistance of, and cooperate with, all other Federal, State, and local agencies with responsibilities for zoning, planning, migratory fish, waterfowl, and marine animals, water, and natural resources, and all nonprofit agencies and organizations which may contribute information or expertise about the resources, and the management, of the Area, in order that the knowledge, expertise and views of all agencies and groups may contribute affirmatively to the most sensitive present and future use of the Area and its various subareas for the benefit of the public.

State jurisdiction.
16 USC 541h.

SEC. 9. The Secretary shall cooperate with the State of Oregon and political subdivisions thereof in the administration of the Area and in the administration and protection of lands within and adjacent to the Area owned or controlled by the State or political subdivisions thereof. Nothing in this Act shall deprive the State of Oregon or any political subdivision thereof of its right to exercise civil and criminal jurisdiction within the Area consistent with the provisions of this Act, or of its right to tax persons, corporations, franchises or other non-Federal property, in or on the lands or waters within the Area.

Approved December 22, 1974.

LEGISLATIVE HISTORY:

HOUSE REPORT No. 93-1247 (Comm. on Interior and Insular Affairs).
SENATE REPORT No. 93-1089 (Comm. on Interior and Insular Affairs).
CONGRESSIONAL RECORD, Vol. 120 (1974):
    Aug. 5, considered and passed House.
    Aug. 16, considered and passed Senate, amended.
    Dec. 3, House concurred in Senate amendment with an amendment.
    Dec. 5, Senate concurred in House amendment.

Appendix B: Model Temperature Inputs

Temperature Inputs for 2000

Year	Date	*1978*	*1987*	*1996*	*REF*	*River*
Mar-00	3/22	8.90	9.34	8.85	8.85	
	3/23	9.32	9.17	8.65	9.07	
	3/24	10.01	9.30	9.17	9.57	
	3/25	10.37	9.83	9.46	9.60	
	3/26	11.03	9.81	10.12	10.33	
	3/27	10.91	10.02	10.41	10.28	
	3/28	10.18	9.56	10.00	9.61	
	3/29	10.24	8.68	10.37	9.10	
	3/30	10.88	8.63	10.88	9.19	
	3/31	11.84	10.01	12.35	9.88	
Apr-00	4/1	12.55	10.66	13.26	11.18	
	4/2	12.52	10.67	12.95	10.17	
	4/3	12.69	10.54	12.94	10.91	
	4/4	11.84	10.17	11.68	10.46	
	4/25	11.62	11.35	11.61	10.72	10.04
	4/26	14.02	12.60	14.23	12.77	10.67
	4/27	13.10	12.73	13.48	12.32	11.28
	4/28	12.64	12.52	12.52	11.77	10.85
	4/29	12.38	12.00	12.42	11.05	9.26
	4/30	13.77	12.79	13.76	12.00	10.59
May-00	5/1	13.56	12.51	13.38	12.23	11.54
	5/2	13.17	12.49	13.20	12.34	11.74
	5/13	13.52	12.14	12.92	12.48	9.82
	5/14	14.09	12.85	14.45	13.76	11.17
	5/24	18.44	16.85	18.74	16.10	14.08
	5/25	16.85	15.53	16.28	14.62	12.79
	5/26	15.77	13.59	15.23	14.01	12.07
	5/27	15.11	12.95	14.51	13.75	12.00
	5/28	16.09	13.60	15.24	14.65	12.27
	5/29	16.60	14.27	16.16	14.74	12.18
Jun-00	6/9	14.45	13.33	14.22	13.39	12.71
	6/10	12.39	12.88	13.12	12.55	12.13
	6/11	11.80	11.85	12.24	11.78	10.87
	6/12	12.38	12.19	12.35	12.54	10.22
	6/23	19.81	15.58	18.57	15.48	13.50
	6/24	20.60	15.67	18.77	16.52	13.79
	6/25	20.37	15.87	19.52	16.28	13.75
	6/26	20.64	15.36	20.64	16.72	14.47
	6/27	21.04	15.13	19.64	17.04	14.98
	6/28	20.19	15.68	19.31	17.54	14.98
Jul-00	7/8	19.89	15.02	18.18	17.11	14.35
	7/9	20.76	15.68	18.95	17.61	14.83
	7/10	20.17	15.23	18.82	17.28	15.11
	7/11	19.44	14.88	17.98	16.08	14.42
	7/12	19.13	14.56	17.97	15.93	14.26
	7/22	18.97	14.31	18.55	16.77	14.58
	7/23	20.01	16.03	19.48	16.97	15.94
	7/24	21.25	16.50	20.64	17.21	16.58
	7/25	20.30	15.54	20.23	17.44	16.49
	7/26	19.75	15.22	18.39	16.78	15.68

Appendix B, Cont.: Model Temperature Inputs

Temperature Inputs for 2001

Year	Date	*1978*	*1987*	*1996*	*REF*	*River*
Mar-01	3/1	6.71	7.93	7.28	7.04	7.71
	3/2	8.24	8.33	8.15	7.81	7.84
	3/3	8.24	8.55	8.65	8.31	7.89
	3/13	9.06	9.19	8.95	8.70	8.92
	3/14	10.83	10.15	10.14	10.44	9.45
	3/15	8.90	9.24	9.44	8.63	8.76
	3/16	6.92	8.47	7.88	6.93	7.38
	3/17	8.20	8.43	8.53	8.16	7.59
	3/18	9.44	9.06	9.52	8.98	8.74
	3/19	9.78	9.50	9.88	9.18	8.58
Apr-01	4/14	10.69	9.47	11.06	10.49	7.60
	4/15	10.52	9.43	11.29	10.21	8.07
	4/16	11.78	10.52	11.99	11.09	9.43
	4/17	12.57	11.73	12.51	11.79	10.20
	4/18	13.32	12.31	13.00	12.00	10.13
	4/19	12.65	11.83	12.25	10.76	9.66
	4/20	11.50	11.02	11.35	9.74	9.45
	4/21	12.61	11.20	11.53	10.70	9.92
	4/22	11.51	10.49	11.25	10.26	9.99
	4/23	12.59	10.28	10.81	10.45	10.06
May-01	5/4	12.24	10.86	12.66	10.83	10.26
	5/13	17.21	14.74	16.03	15.32	12.95
	5/14	12.43	12.96	13.83	12.14	11.56
	5/15	12.51	12.62	13.03	12.29	10.08
	5/16	14.97	13.87	14.48	13.93	10.42
	5/17	12.87	12.89	13.70	11.98	9.53
	5/18	15.23	13.72	14.58	13.10	10.67
	5/19	15.87	13.70	15.97	13.00	11.40
	5/20	16.05	13.63	16.03	13.27	11.52
	5/31	17.92	15.34	17.37	16.39	13.78
Jun-01	6/1	16.20	12.90	16.15	14.60	12.71
	6/2	13.55	11.80	14.42	12.76	12.13
	6/12	16.24	15.97	15.14	14.95	10.87
	6/13	17.49	16.85	15.66	15.34	10.22
	6/14	17.20	16.32	15.12	14.77	13.50
	6/15	18.61	16.22	15.99	16.04	13.79
	6/16	17.82	15.31	15.29	14.60	13.75
	6/17	17.22	13.91	14.13	13.92	14.47
	6/28	16.13	15.05	15.85	16.06	14.98
	6/29	17.70	15.74	16.70	16.76	14.98
Jul-01	7/11	19.52	16.25	19.09	17.70	15.58
	7/12	20.31	16.29	19.57	17.87	16.03
	7/13	19.58	16.52	18.39	16.81	16.09
	7/14	19.05	16.31	17.90	16.64	15.70
	7/15	17.98	14.56	16.94	15.89	14.71
	7/16	17.95	14.38	17.33	16.91	14.33
	7/26	19.23	14.31	17.80	16.89	14.50
	7/27	20.30	14.86	18.07	17.47	14.93
	7/28	20.21	14.59	18.07	17.86	15.68
	7/29	17.04	13.52	15.91	14.85	14.10
Aug-01	8/9		15.35		17.94	15.70
	8/10		15.30		17.32	15.96
	8/11		15.52		17.95	16.01
	8/12		15.98		18.65	16.79
	8/13		15.90		18.46	16.88
	8/24		15.97		17.05	15.19
	8/25		15.86		16.85	15.31
	8/26		16.12		17.09	15.74
	8/27		16.55		16.92	15.74
	8/28		16.78		18.26	16.26

Appendix B, Cont.: Model Temperature Inputs

Model Temperature Inputs for 2002

Year	Date	*1978*	*1987*	*1996*	*REF*	*River*
Apr-02	4/4	12.19	11.72	13.93	11.83	10.02
	4/5	11.14	11.05	12.02	11.09	10.00
	4/6	11.66	11.15	12.00	11.51	9.63
	4/7	12.39	11.44	12.70	11.61	9.54
	4/8	11.88	10.86	11.99	10.35	9.10
	4/9	11.53	10.19	11.75	10.77	9.88
	4/10	11.40	10.31	11.47	10.75	8.95
	4/11	10.90	9.92	10.59	10.43	8.72
	4/12	11.79	10.70	11.70	11.21	9.23
	4/13	11.58	10.77	11.21	11.16	9.79
May-02	5/4	12.01	13.36	13.68	10.58	10.10
	5/5	17.16	12.53	12.27	11.19	10.01
	5/6	12.57	11.46	12.16	10.41	9.17
	5/7	12.66	11.49	13.26	10.01	9.10
	5/8	14.86	11.30	12.85	9.85	8.94
	5/9	12.65	11.13	13.47	10.78	9.63
	5/10	14.61	11.60	15.00	12.27	10.46
	5/21	15.30	12.62	13.80	12.96	11.56
	5/22	15.86	13.19	14.29	13.17	12.02
	5/23	16.43	13.06	14.37	13.12	11.69
Jun-02	6/2	17.48	15.73	17.24	15.03	13.62
	6/3	16.52	15.24	16.04	14.80	13.66
	6/4	16.55	14.97	16.64	15.89	13.78
	6/5	16.21	14.75	16.10	15.68	13.92
	6/6	17.11	15.58	15.61	14.64	13.63
	6/7	16.28	14.29	14.34	13.31	12.88
	6/18	15.65	14.70	14.96	14.86	13.71
	6/19	17.23	15.89	15.34	14.75	13.94
	6/20	18.54	15.07	16.94	15.14	14.04
	6/30	17.60	16.42	18.08	16.93	14.39
Jul-02	7/1	18.56	17.40	18.20	16.14	14.43
	7/2	18.24	17.21	17.78	15.61	14.26
	7/3	18.63	16.58	17.81	15.89	14.60
	7/4	19.13	16.52	17.48	15.68	14.74
	7/5	19.03	16.14	17.54	15.90	14.68
	7/6	20.34	15.73	18.25	16.54	15.28
	7/16	19.30	14.24	17.26	16.55	13.92
	7/17	19.31	14.90	17.47	17.40	14.77
	7/18	19.65	15.62	18.06	17.98	15.19
	7/19	19.39	15.82	18.26	18.22	15.58

Appendix C: Model Diet Inputs

n=	11	8	17	9	10	8	10	10	10	10	4	7	7	7	7	7	8	7	7	7
REF marsh	Sampling Period																			
Prey Items	**Mar-99**	**Apr-99**	**May-99**	**Jun-99**	**Jul-99**	**Mar-00**	**Apr-00**	**May-00**	**Jun-00**	**Jul-00**	**Mar-01**	**Apr-01**	**May-01**	**Jun-01**	**Jul-01**	**Aug-01**	**Apr-02**	**May-02**	**Jun-02**	**Jul-02**
Corophium spp.	0.095	0.257	0.432	0.444	0.525	0.300	0.355	0.854	0.072	0.036	0.250	0.072	0.050	0.095	0.108	0.066	0.314	0.049	0.032	0.060
Eogammarus spp.	0.014	0.261	0.140	0.018	0.085	0.102	0.018		0.062			0.002				0.028	0.033	0.117		0.008
Isopoda					0.007									0.001			0.009		0.024	
Mysidae	0.162	0.019	0.124		0.001	0.001		0.021	0.126			0.056	0.054						0.064	
Talitridae			0.029				0.145			0.030									0.050	0.012
Cumacea									0.005			0.194	0.324	0.029						
Copepoda	0.004	0.038							0.031			0.003	0.007	0.032			0.004	0.001	0.002	
Other Crustacean					0.049									0.004			0.095		0.011	
Chironomidae	0.004	0.047	0.008			0.018	0.001	0.014	0.108			0.044	0.014	0.225	0.008		0.018	0.027	0.003	
Ceratopogonidae							0.029		0.064	0.001		0.035		0.029			0.012	0.048		0.001
Dolichopodidae			0.001		0.015					0.008				0.063	0.005	0.012			0.028	0.049
Empididae																0.001		0.006		
Other Diptera adult			0.043			0.060	0.003		0.067	0.008	0.250	0.045		0.139	0.056	0.004	0.024	0.025	0.017	0.003
Dipt emergent		0.009	0.001		0.012									0.045		0.002				
Dipt larvae	0.007		0.004			0.002	0.001	0.001	0.015	0.000	0.250		0.079	0.037	0.019	0.001		0.010		0.001
Dipt pupae			0.028		0.013		0.005	0.048	0.045	0.001		0.016	0.023	0.026	0.010	0.002	0.006	0.301	0.015	0.010
Trichop immature	0.174					0.281					0.174	0.136								
Trichoptera adult			0.083				0.097													
Cantharidae			0.013	0.163	0.081				0.019	0.132			0.066	0.061	0.093	0.152		0.075	0.085	0.369
Coleoptera adult				0.166	0.016				0.003	0.008				0.001		0.019	0.014		0.056	0.052
Coleo larvae				0.002	0.004		0.020		0.001			0.002		0.057	0.007			0.185	0.010	0.001
Aphididae					0.001							0.029		0.018	0.002	0.005		0.004	0.003	0.002
Homoptera					0.063				0.023	0.487			0.002	0.026	0.462	0.530		0.005	0.392	0.100
Hemiptera										0.003						0.014			0.023	0.029
Lepidoptera									0.006	0.003		0.010		0.022	0.033		0.082			0.006
Hymenoptera	0.014		0.001		0.005				0.079	0.022		0.009		0.031	0.001	0.002			0.040	0.019
Collembola	0.004				0.008				0.021					0.004				0.005		
Oligo-, Polychaet	0.309	0.092	0.027			0.154	0.006				0.076	0.230	0.326	0.014			0.388	0.046		
Arachnids		0.036	0.004		0.046				0.005			0.006		0.041	0.029	0.003			0.007	0.018
Fish/Fish larvae		0.225				0.080	0.222		0.078			0.101								
Algae and Other Plant		0.002		0.159	0.070		0.052		0.170	0.261					0.045	0.131		0.077	0.110	0.235
Non-nutritious				0.048	0.000										0.121	0.027		0.006	0.004	0.003
Other	0.213	0.013	0.064		0.003	0.002	0.047	0.063				0.008	0.056					0.012	0.024	

Appendix C, Cont.: Model Diet Inputs

n=	10	5	1	3	1	2	6	7	7	6	2
1978 Marsh	Sampling Period										
Prey Items	**Mar-99**	**Apr-99**	**May-99**	**Mar-00**	**Apr-00**	**May-00**	**May-01**	**Jun-01**	**Apr-02**	**May-02**	**Jul-02**
Corophium spp.	0.188	0.404		0.571		0.045	0.067	0.029	0.138	0.102	0.114
Eogammarus spp.		0.030		0.071			0.009		0.029	0.141	
Isopoda		0.007					0.095		0.030		
Mysidae	0.037										
Talitridae											
Cumacea							0.017				
Copepoda						0.015			0.008		
Other Crustacean									0.125		
Chironomidae	0.006		0.957	0.333	0.049	0.061	0.250	0.002	0.029	0.052	
Ceratopogonidae		0.010					0.002	0.014	0.042	0.001	
Dolichopodidae						0.106		0.047		0.156	0.166
Empididae											
Other Diptera adu	0.054	0.114	0.032				0.069	0.061	0.155	0.217	0.036
Dipt emergent								0.135			
Dipt larvae	0.184	0.041		0.024			0.046	0.011	0.042	0.001	
Dipt pupae	0.003					0.015	0.033	0.026	0.010	0.006	0.020
Trichop immature	0.392	0.378							0.045		
Trichoptera adult											
Cantharidae									0.095	0.069	0.031
Coleoptera adult						0.152	0.147				0.046
Coleo larvae							0.016	0.087	0.042	0.014	
Aphididae						0.076		0.477		0.001	
Homoptera							0.008	0.073			
Hemiptera		0.001					0.100			0.019	
Lepidoptera										0.004	0.030
Hymenoptera							0.006	0.025	0.188		0.012
Collembola							0.013	0.008		0.001	
Oligo-, Polychaet	0.006	0.016				0.030				0.061	
Arachnids	0.083					0.500	0.048	0.001		0.001	0.012
Fish/Fish larvae					0.951						0.337
Algae and Other Plant			0.011					0.001		0.029	0.195
Non-nutritious	0.048							0.002		0.007	
Other							0.073		0.073	0.118	

Appendix C, Cont.: Model Diet Inputs

n=	8	10	11	6	2	3	6	5	10	5	7	7	7	7	7	7	7	7
1987 Marsh	Sampling Period																	
Prey Items	**Mar-99**	**Apr-99**	**May-99**	**Jul-99**	**Mar-00**	**Apr-00**	**May-00**	**Jun-00**	**Jul-00**	**Apr-01**	**May-01**	**Jun-01**	**Jul-01**	**Aug-01**	**Apr-02**	**May-02**	**Jun-02**	**Jul-02**
Corophium spp.	0.074	0.223	0.323	0.073	0.909		0.478	0.042	0.377	0.148	0.290	0.381	0.621	0.108	0.095	0.007	0.695	0.116
Eogammarus spp.	0.095	0.088	0.004	0.006				0.149	0.006		0.003	0.051	0.031	0.021	0.399	0.049	0.046	0.062
Isopoda																		
Mysidae	0.067	0.057	0.092				0.157	0.002	0.000			0.002			0.163		0.008	
Talitridae							0.146		0.053				0.143	0.084				
Cumacea												0.011						
Copepoda										0.007		0.023		0.000	0.090	0.319		
Other Crustacean				0.159								0.012						
Chironomidae	0.002	0.057	0.086	0.007		0.011	0.038	0.038		0.075	0.012	0.023	0.009		0.036	0.016	0.002	
Ceratopogonidae		0.001	0.001			0.014	0.002		0.003	0.027		0.003		0.000		0.002		0.002
Dolichopodidae				0.164			0.065	0.048	0.036	0.114	0.002	0.006		0.028				0.108
Empididae																		0.001
Other Diptera adult		0.001	0.003	0.041		0.024	0.091	0.102	0.075	0.004	0.113		0.033	0.031	0.036	0.040	0.044	0.149
Dipt emergent	0.001									0.046				0.001			0.001	
Dipt larvae	0.159	0.001						0.075	0.000	0.007		0.009		0.009	0.003	0.002	0.001	
Dipt pupae	0.057		0.002				0.010	0.016	0.006			0.027		0.002	0.005	0.003		0.001
Trichop immature	0.333	0.426				0.270		0.086										
Trichoptera adult			0.094															
Cantharidae																0.119		
Coleoptera adult			0.004	0.156					0.038		0.007			0.124		0.033	0.009	0.179
Coleo larvae				0.017				0.012	0.014	0.008	0.005	0.137	0.034	0.004		0.009	0.034	0.033
Aphididae				0.073		0.006				0.045		0.066	0.071	0.002		0.002	0.021	0.001
Homoptera				0.038					0.044			0.001	0.006	0.434			0.037	0.028
Hemiptera				0.078				0.045	0.006		0.102	0.001	0.052	0.006			0.042	0.103
Lepidoptera																		0.001
Hymenoptera				0.005			0.007	0.042	0.002	0.081	0.077			0.040			0.012	0.004
Collembola	0.042						0.006			0.008			0.001		0.001			0.001
Oligo-, Polychaet	0.172	0.117							0.020	0.396	0.296	0.104			0.078	0.121		
Arachnids						0.017		0.020	0.002		0.005	0.003		0.000				0.011
Fish/Fish larvae		0.001		0.020	0.091	0.659						0.091			0.020	0.130		
Algae and Other Plant		0.004	0.351	0.163				0.256	0.313	0.026	0.002	0.048		0.106	0.075	0.146	0.050	0.187
Non-nutritious		0.022						0.118	0.000			0.001		0.000		0.003		
Other			0.037				0.002			0.007	0.087			0.000				0.009

Appendix C, Cont.: Model Diet Inputs

n=	8	10	11	6	7	3	4	5	10	5	7	7	7	7	7	7	7	7	7
1996 Marsh	Sampling Period																		
Prey Items	**Mar-99**	**Apr-99**	**May-99**	**Jun-99**	**Jul-99**	**Mar-00**	**Apr-00**	**May-00**	**Jun-00**	**Jul-00**	**Mar-01**	**Apr-01**	**May-01**	**Jun-01**	**Jul-01**	**Apr-02**	**May-02**	**Jun-02**	**Jul-02**
Corophium spp.	0.356	0.776	0.953	0.929	0.671	0.150	0.847	0.950	0.350	0.262	0.600	0.232	0.776	0.969	0.188	0.577		0.197	0.168
Eogammarus spp.		0.529	0.747	0.293	0.229	0.924	0.663	0.659	0.332	0.124	0.200	0.687				0.339	0.659	0.197	
Isopoda		0.156			0.123				0.425	0.625					0.364				0.262
Mysidae		0.219		0.297					0.224										
Talitridae															0.752				
Cumacea																			
Copepoda	0.560					0.278	0.377						0.139			0.269			
Other Crustacean				0.333															0.162
Chironomidae	0.499	0.865	0.519	0.418		0.848	0.134	0.136	0.772	0.135		0.628	0.146	0.843	0.149	0.227	0.647	0.773	0.278
Ceratopogonidae	0.283	0.570	0.688				0.863	0.100	0.614	0.288		0.360		0.935		0.197			0.736
Dolichopodidae			0.512		0.647		0.325		0.522	0.132		0.135		0.916	0.624		0.146	0.121	0.187
Empididae												0.433				0.193			
Other Diptera adu	0.718		0.128	0.750	0.223	0.238	0.263	0.295	0.289	0.159		0.567		0.295	0.197	0.422	0.457		0.769
Dipt emergent					0.559				0.366			0.482		0.479				0.194	
Dipt larvae	0.220	0.432		0.252	0.263	0.264	0.424	0.347	0.792	0.420	0.200		0.353	0.224	0.750	0.652	0.341	0.438	0.269
Dipt pupae	0.314	0.467	0.574	0.334	0.672	0.283	0.411	0.289	0.159	0.147		0.137	0.358	0.269	0.326	0.152	0.932	0.242	0.344
Trichop immature	0.365		0.777			0.196	0.116	0.335				0.236	0.231		0.567		0.569		
Trichoptera adult		0.627												0.122					
Cantharidae																	0.672		0.278
Coleoptera adult								0.295	0.223	0.127			0.128	0.134	0.349		0.800	0.573	0.304
Coleo larvae										0.192					0.260				0.130
Aphididae						0.333			0.187	0.155		0.158		0.376			0.342	0.288	0.597
Homoptera										0.114		0.272							
Hemiptera			0.724		0.800				0.448	0.395				0.141				0.115	0.369
Lepidoptera							0.944			0.726									
Hymenoptera		0.133			0.264				0.168	0.117				0.242	0.446	0.458	0.578	0.578	0.191
Collembola		0.376						0.125				0.674		0.733					
Oligo-, Polychaet	0.157	0.154										0.281			0.130				
Arachnids	0.429	0.333	0.296				0.299		0.935	0.343		0.212		0.282	0.527	0.272	0.798	0.267	0.253
Fish/Fish larvae						0.926													
Algae and Other Plant				0.287	0.190				0.116	0.169					0.123	0.440		0.259	0.477
Non-nutritious	0.174				0.114			0.588		0.222					0.287				
Other	0.105		0.329	0.229		0.167	0.322			0.657			0.573		0.747	0.116		0.387	0.143

Appendix D: Energy Content from All Bomb Trials

Bomb Subject	*Average Energy content (kJ/g ww)*	*dw:ww*
Amphipoda: Corophium spp.	24.71	0.2
Amphipoda: Eogammarus spp.	23.83	0.2
Amphipoda: Talitridae	30.66	0.25
Amphipoda: Talitridae	40.82	
Araneae	57.76	0.25
Araneae (< 3 mm abdomen width)	47.31	0.23
Araneae (< 5 mm abdomen width)	50.68	0.24
Araneae (> 5 mm abdomen width)	49.22	0.24
Coleoptera	42.06	0.18
Coleoptera (> 5 mm)	69.25	0.32
Coleoptera larvae	34.60	0.17
Coleoptera: Cantharidae	79.37	0.35
Coleoptera: Coccinellidae	120.47	0.47
Coleoptera: Coccinellidae larvae	89.38	0.35
Coleoptera: Elateridae	60.16	0.27
Coleoptera: Staphylinidae larvae	18.66	0.08
FISH ORDER: Cottidae (< 20 mm)	34.43	0.2
FISH ORDER: Cottidae (> 20 mm)	35.99	0.2
Deschampsia capesatosa	110.30	0.61
Diptera: Mixed	177.67	0.74
Diptera: Mixed	148.66	0.67
Diptera: Nematocera	48.59	0.21
Diptera: TIGER FLY	171.34	0.77
Diptera: Chironomidae	44.30	0.14
Diptera: Chironomidae larvae (fresh)	24.75	0.13
Diptera: Chironomidae, Ceratopogonidae, Cecidomyiidae	21.89	0.1
Diptera: Dolichopodidae	65.46	0.27-0.30
Diptera: Chironomidae larvae (frozen)	28.72	0.14
Diptera: Chironomidae larvae (frozen)	28.01	0.14
Diptera: Muscidae	43.39	0.2
Diptera: Scatopsidae	150.08	0.66
Diptera: Tabanidae	129.20	0.59
Diptera: Tipulidae	73.53	0.34-0.4
Hemiptera: GREEN HEMI	91.63	0.6
Hemiptera: Lygaeidae	84.23	0.54
Hemiptera: Lygaeidae nymph	93.80	0.47
Hemiptera: Pentatomidae	68.32	0.31
Hemiptera: Pentatomiidae nymph	92.58	0.37
Homoptera: Mixed	76.45	0.31
Homoptera: Cercropidae	62.31	0.53
Homoptera: Cicadellidae	74.12	0.65
Hymenoptera: Mixed	54.71	0.25
Hymenoptera (< 5 mm)	174.78	0.8
Hymenoptera (> 5 mm)	148.46	0.68
Hymenoptera: Formicidae	111.25	0.52
Isopoda: Saduria entomon	33.35	0.28
Lepidoptera (< 5 mm)	63.07	0.26
Lepidoptera larvae	74.05	0.3
Lepidoptera: Arctiidae	87.58	0.38
Neomysis	35.52	0.17
Opionidae	68.46	0.35
Orthoptera	68.28	0.23-0.36
Polychaeta: Nereidae	19.81	0.12
Psocoptera	173.33	0.74
Trichoptera (estuarine)	43.21	0.22
Trichoptera larvae	68.71	0.31
Trichoptera: Hydropsyche	88.12	0.39
Trichoptera: Phryg/Limnephilidae	65.06	0.29
Ulva spp.	37.29	0.23

Appendix E: Prey Energy Model Input

Prey Categories	*Prey Energy (J/g ww)*	*% Indigestible*
Corophium spp.	3086.83	26
Eogammarus spp.	3103.13	26
Gnorimosphaeroma spp.	2458.61	50
Mysidae	3552.17	15
Talitridae	3042.44	26
Cumacean adult*	4981.54	8
Copepoda*	4981.54	8
Other Crustacean**	3048.64	20
Chironomidae adult	3833.66	12
Ceratopogonidae adult	3833.66	12
Dolichopodidae adult	6203.61	12
Empididae adult	9684.77	12
Other Diptera adult	8922.63	12
Diptera emergent*	3140.06	10
Diptera larvae	2580.68	5
Diptera pupae*	3140.06	10
Trichop emergent, larvae	5813.84	5
Trichoptera adult	7756.09	9
Cantharidae adult	7937.48	30
Coleoptera adult	7968.37	35
Coleoptera larvae	2405.86	10
Aphididae adult and immature	11915.44	2
Homoptera (Leafhoppers)	12268.21	20
Hemiptera adult and immature	10926.82	15
Lep adult and larvae	8501.89	15
Hymenoptera	12672.37	30
Collembola	5621.65	0
Oligo-, Polychaetes	1980.84	2
Arachnids	5320.89	15
Fish/Fish larvae	3567.69	0
Algae and Other Plant	3853.37	25
Non-nutritious	0.00	100
Other**	2512.04	0

Appendix F: INDVAL Fallout Insect Indicators

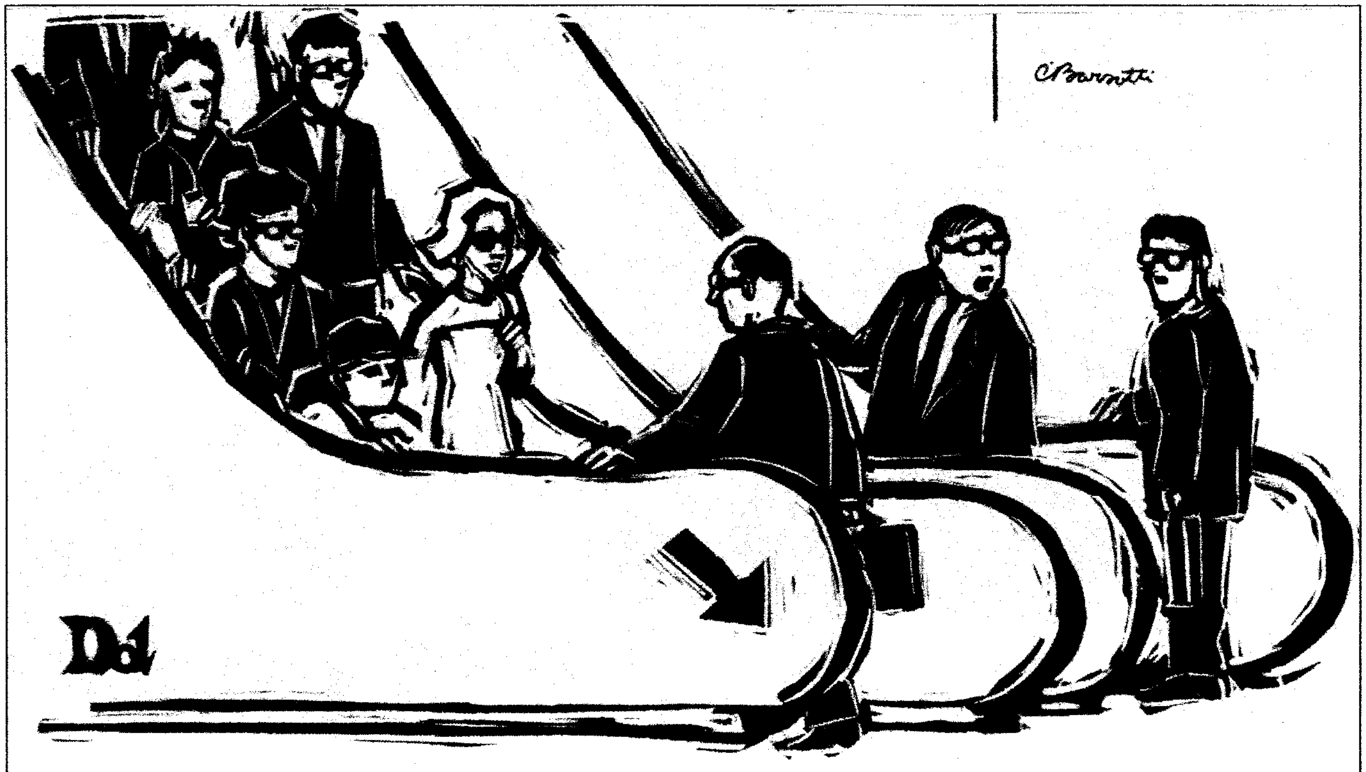
Sampling Period	**REF Lo**	**REF Hi**	**78 Lo**	**87 Lo**	**87 Hi**	**96 MT**
April 1998		Cecidomy (48) Sciarida (53) Sphaeroc (46) Lep larv (45)		Dolichop (71)	Limnephi (52)	Ceratopo (34) Staphyli (63) Ephydrid (93) Empidida (87) Carabida (80) Scaphidi (60)
May 1998	Cantharid (47)	Collembo (67) Thysanop (75) Sciarida (59) Chrysome (59) Eucnemid (60)	Ceratopo (35) Psychodi (61)	Drosophi (59) Amphipod (64)		Chironom (25) Ephydrid (99)
June 1998	Cantharid (52) Pentatom (80)	Collembo (67) Cecidomy (63) Thysanop (42) Sciarida (57) Lep larv (59)		Drosophi (67)	Ceratopo (39) Homo imm (91) Saldidae (85) Hemipter (59)	Chironom (47) Ephydrid (74)
March 1999						Staphylin (80)
April 1999		Chironom (24) Collembo (34) Hymenopt (52) Araneae (63) Sciarida (43)				Ceratopo (67) Empidida (46)
May 1999	Canthari (61)	Thysanop (75) Elaterid (64)	Dolichop (63) Psychodi (44)	Chloropi (49)		Staphyli (96) Ephydrid (83)
June 1999		Collembo (79) Cecidomy (39) Elaterid (88) Chrysome (53) Cercopid (60)	Hymenopt (33) Cicadell (48) Sphaeroc (65) Delphacid (64) Muscidae (42)	Homo imm (75) Drosophi (37) Chloropi (67) Aphidida (86)	Scatopsi (77)	Chironom (60) Staphyli (94) Ephydrid (86)
March 2000	Chironom (32)		Psychodi (63)		Collembo (91)	Staphyli (87) Sphaeroc (48) Saldidae (53) Coleopte (73) Carabida (86) Lygaeida (60)
April 2000		Amphipod (46) Cantharid (66) Sciarida (49) Elaterid (67) Chrysome (69) Thrip imm (76)	Cicadell (80) Cecidomy (59)	Hemi imm (69)		Ceratopo (48) Dolichop (70) Staphyli (96) Ephydrid (98) Sphaeroc (51) Saldidae (80) Phoridae (60) Chamaemy (60)
May 2000	Ceratopo (40)	Elaterid (57)	Cicadell (86) Cecidomy (52) Canthari (51) Phoridae (57)	Drosophi (60) Amphipod (70) Hemi imm (92) Heteroce (87)		Staphyli (99) Ephydrid (63) Sphaeroc (63)
June 2000		Collembo (51) Cecidomy (53) Sciarida (56) Elaterid (63) Cercopid (75)	Dolichop (36) Cicadell (78) Muscidae (74)	Delphaci (86)	Hymenopt (33) Hemi imm (69) Coccinel (60)	Chironom (28) Thysanop (53) Empidida (58)

Appendix F, Cont.: INDVAL Fallout Insect Indicators

Sampling Period	REF Lo	REF Hi	78 Lo	87 Lo	87 Hi	96 MT
March 2001	Psychodi (43)	Ephydrid (48) Sciarida (41)		Dolichop (59)	Collembo (88) Amphipod (62)	Chironom (28) Staphyli (67)
April 2001	Ptiliida (50)	Collembo (56)	Hymenopt (36) Cecidomy (46) Psychodi (61) Lepidopt (56)			Dolichop (52) Staphyli (60) Araneae (41) Sphaeroc (58) Empidida (73)
May 2001	Chironom (25) Ceratopo (32)	Elaterid (73)	Hymenopt [illegible] Cicadell (83) Cecidomy (74)	Dolichop (63) Amphipod (49)		Staphyli [illegible] Sphaeroc (68) Empidida (45)
June 2001	Psychodi (54) Pentatom (69)	Elaterid (54) Chrysome (76)	Hymenopt (40) Cicadell (70) Sphaeroc (42) Muscidae (66) Diptera (46) Lygaeida (87)	Delphacid (66)		Phoridae (47)
April 2002	Sciarida (52) Ptiliida (58)		Cecidomy (71) Psychodi (59)	Dolichop (77)		Staphyli (87) Empidida (45)
May 2002		Elaterid (77) Orthopte (60)	Chironom (37) Hymenopt (40) Cicadell (61) Araneae (40) Cecidomy (55) Psychodi (69)	Homo imm (80) Drosophi (59)	Hemi imm (60) Micropez (78)	Thysanop (40) Ephydrid (43) Phoridae (39)
June 2002		Dolichop (45) Canthari (53) Sciarida (51) Elaterid (69) Cercopid (79)	Hymenopt (35) Cecidomy (45) Sphaeroc (56) Muscidae (49) Endomych (88)	Tipulida (36) Drosophi (60)	Scatopsi (42) Delphaci (49) Miridae (88)	Chironom (42) Ceratopo (38) Cicadell (58)

Appendix G: INDVAL Benthic Invertebrate Indicators

Sampling Period	REF	1978	1987	1996
April 1998	Cumacea (54) Nematoda (75) Capitellidae (66) Macoma (50)			
May 1998	Cumacea (92) Capitellidae (54) Manayunkia (58) Hobsonia (60)		Oligochaete (50) Nereidae (64) Spionidae (99)	Chiro larvae (93)
June 1998	Cerat larvae (60) Ostracod (51) Nematoda (91) Oligochaete (51) Capitellidae (84) Manayunkia (87) Hobsonia (87) Macoma (53)		Corophium (65) Nereidae (62) Spionidae (80)	Chiro larvae (97) Chiro pupae (60)
April 1999	Capitellidae (82) Manayunkia (64)	Ostracod (73) Hobsonia (74)	Nereidae (44) Spionidae (92)	
May 1999	Cumacea (93) Nematoda (71) Oligochaete (48) Capitellidae (67)			Chiro larvae (70)
March 2000	Corophium (45) Cumacea (100) Capitellidae (49)	Eogammarus (80) Hobsonia (71)		
April 2000	Cumacea (100) Capitellidae (56) Manayunkia (61) Hobsonia (51)			Gnorimosphaeroma (69)
May 2000	Eogammarus (57) Cumacea (96) Nematoda (67)	Hobsonia (75)		Chiro larvae (90)
June 2000	Corophium (56) Eogammarus (73) Cumacea (88) Capitellidae (74)	Gnorimosphaeroma (71) Ostracod (52) Manayunkia (84) Hobsonia (95)		Chiro larvae (81)
March 2001	Cumacea (93) Macoma (59)	Corophium (91) Eogammarus (82) Manayunkia (75)		Cerat larvae (75)
April 2001	Cumacea (93)	Corophium (67) Manayunkia (92)	Macoma (53)	Chiro larvae (67)
May 2001	Cumacea (86) Ostracod (70)	Corophium (62)	Spionidae (79) Macoma (58)	Chiro larvae (94) Eogammarus (64) Gnorimosphaeroma (59)
June 2001	Cumacea (65) Nematod (59)	Manayunkia (96) Hobsonia (56)	Spionidae (62) Macoma (68)	Chiro larvae (87)



"Bob was raised in the wilderness by salmon."

Appendix H: New Yorker Magazine, November 2004

Vita

Ayesha Gray was born in Arlington, Texas in 1971. She received a Bachelor of Science in Zoology from the University of Texas, Austin in 1994. Living in Costa Rica she rediscovered her love of science and came to Seattle to pursue a doctorate in 1998. For six years she lived between Seattle and the gorgeous Salmon River estuary, Oregon. In the spring 2005 she earned a Doctor of Philosophy from the University of Washington's School of Aquatic and Fishery Sciences.