Contrasting Functional Performance of Juvenile Salmon Habitat in Recovering Wetlands of the Salmon River Estuary, Oregon, U.S.A.

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Abstract

For an estuarine restoration project to be successful it must reverse anthropogenic effects and restore lost ecosystem functions. Restoration projects that aim to rehabilitate endangered species populations make project success even more important, because if misjudged damage to already weakened populations may result. Determining project success depends on our ability to assess the functional state or "performance" and the trajectory of ecosystem development. Mature system structure is often the desired "end point" of restoration and is assumed to provide maximum benefit for target species; however, few studies have measured linkages between structure and function and possible benefits available from early recovery stages. The Salmon River estuary, Oregon, U.S.A., offers a unique opportunity to simultaneously evaluate several estuarine restoration projects and the response of the marsh community while making comparisons with a concurring 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. Analysis of the marsh community responses enables us to use the development state of the three recovering marshes to determine a trajectory of estuarine recovery over 23 years and to make comparisons with a reference marsh. We assessed the rate and pattern of juvenile salmon habitat development in terms of fish density, available prey resources, and diet composition of wild juvenile *Oncorhynchus tshawytscha* (chinook salmon). Results from the outmigration of 1998 and 1999 show differences in fish densities, prey resources, and diet composition among the four sites. Peaks in chinook salmon densities were greatest in the reference site in 1998 and in the youngest (1996) site in 1999. The 1996 marsh had higher densities of chironomids (insects; average $864/m^2$) and lower densities of amphipods (crustaceans; average $8/m^3$) when compared with the other sites. Fauna differences were reflected in the diets of juvenile chinook with those occupying the 1978 and 1996 marshes based on insects (especially chironomids), whereas those from the 1987 and reference marshes were based on crustaceans (especially amphipods). Tracking the development of recovering emergent marsh ecosystems in the Salmon River estuary reveals significant fish and invertebrate response in the first 2 to 3 years after marsh restoration. This pulse of productivity in newly restored systems is part of the trajectory of development and indicates some level of early functionality and the efficacy of restoring estuarine marshes for juvenile salmon habitat. However, to truly know the benefits consumers experience in recovering systems requires further analysis that we will present in forthcoming publications.

Key words: dike removal, estuary, juvenile salmon, restoration, salt marsh, wetland function.

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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 issue). We summarize results of fish utilization, prey resource, and diet composition of juvenile *Oncorhynchus tshawytscha* (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 with pre-disturbance or reference levels have been the usual criteria for measuring performance (Sinicrope et al. 1990; Fell et al. 1991; Barrett & Niering 1993; Merritt et al. 1996; Weinstein et al. 1997; Roman et al. 2002, this issue; Thom et al. 2002, this issue). 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 & Sadro (2002, this issue) 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 an artificial 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 6 years after 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 enabled 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 allowed us to evaluate the following hypotheses: (1) after dike removal an estuarine wetland follows a trajectory of physical and biotic development toward reference conditions and (2) 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 after 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 1982; 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 after 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, this issue) 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.

Here 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:

- What differences in biotic response are seen in marshes of different ages compared with an undiked reference marsh?
- Are the differences in biota related to recovery time?
- Does the diet composition of juvenile chinook salmon differ significantly among marshes in different stages of recovery?
- What are the potential consequences of differences in trophic structure?
- 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 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 (45° 01′ N, 123° 58′ W). 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). For the purposes of this study the three marshes are referred to by the year of dike breaching (1978, 1987, and 1996), and 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
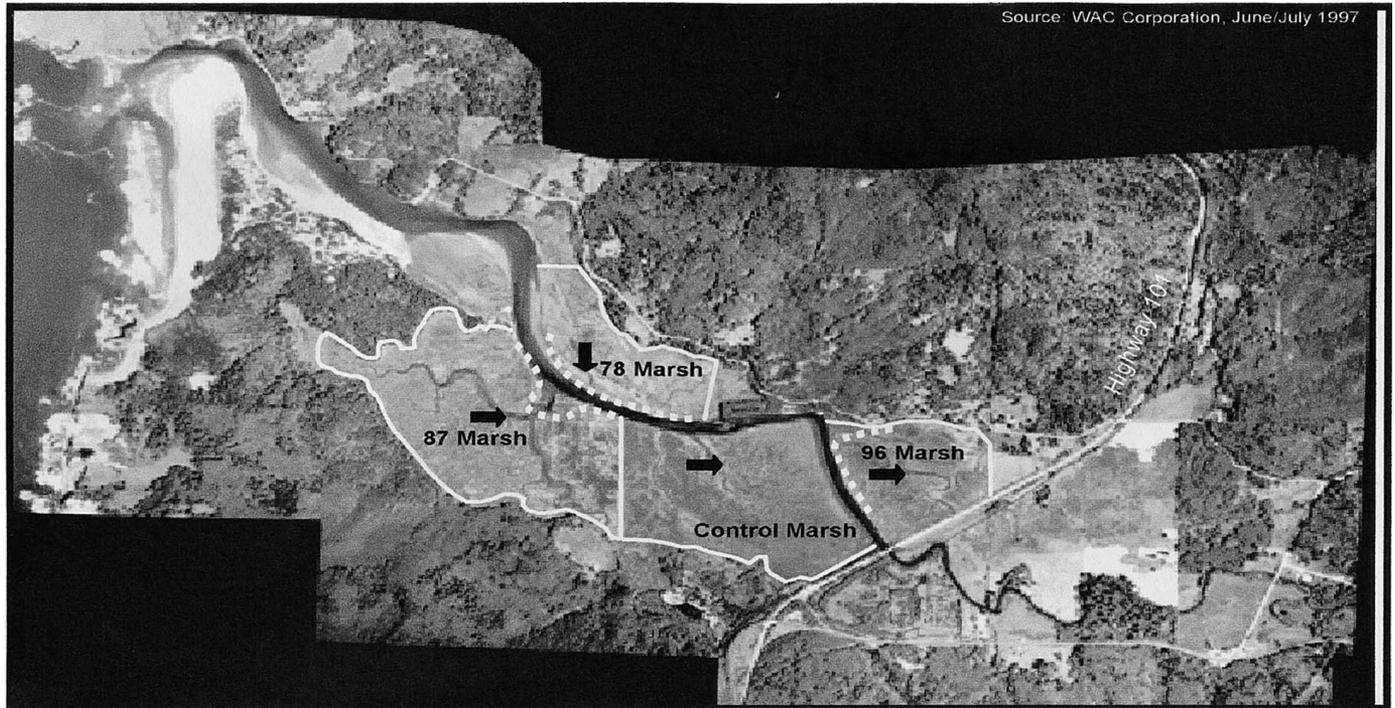
Figure 1. Map of the Salmon River estuary. Dashed lines represent locations of removed dikes. Arrows represent locations of fish and invertebrate collections.

from April and May 1999 (detailed salinity measurement began in March 1999) are reported in Table 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 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 with 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 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,

Table 1. Watershed position and average salinity (ppt).

Watershed Position	Site	April 1999	May 1999
Highest	1996	0.3	0.7
↓	REF	8.8	0.4
	1978	2.4	0.1
Lowest	1987	13.3	6.9

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 (Fig. 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. 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 because 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 m².

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 (IFTs). The IFT consists of a plastic box (51.7 × 35.8 × 14 cm) filled with 3 L 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 been used successfully 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 polyvinyl chloride poles. IFTs were deployed monthly between March and July for 3 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 m².

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 cm³. 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

Figure 2. Modified fyke net deployed across a tidal channel in the reference marsh.

benthic cores were standardized to core volume and are reported as average density of invertebrates per m^3. Strictly epibenthic prey were not sampled by our methods because in oligohaline estuarine environments, epibenthic prey are 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 1984). 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 (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, and June of 1998 and April and May of 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. 3). Chinook salmon were most abundant in the REF, 1987, and 1996 marshes.

Pacific staghorn sculpins were most dense in the REF marsh (Fig. 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. 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 ($p = 0.221$), but low abundance at the 1987 hi marsh in 1999 contributed to statistical differences in density ($p = 0.022$) (Fig. 6). Chironomidae and Ceratopogonidae dipteran families were used to further assess
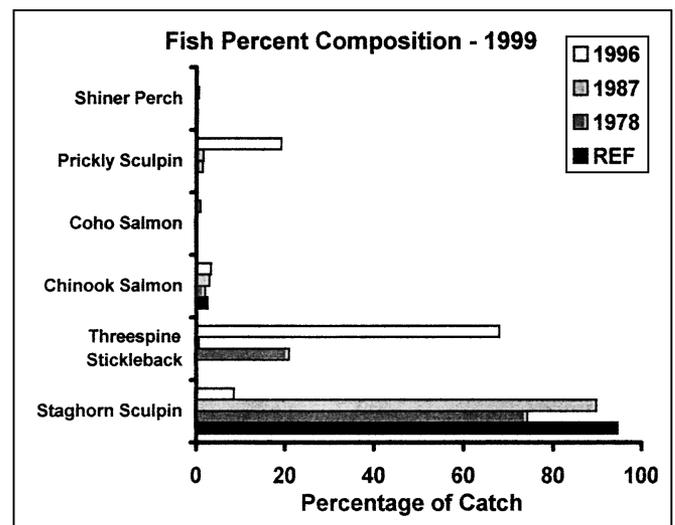

Figure 3. Percent composition of fish collected from each of the four marshes. Pacific staghorn sculpins were the most abundant fish in all sites, except for the 1996 marsh where threespine sticklebacks were the most abundant.
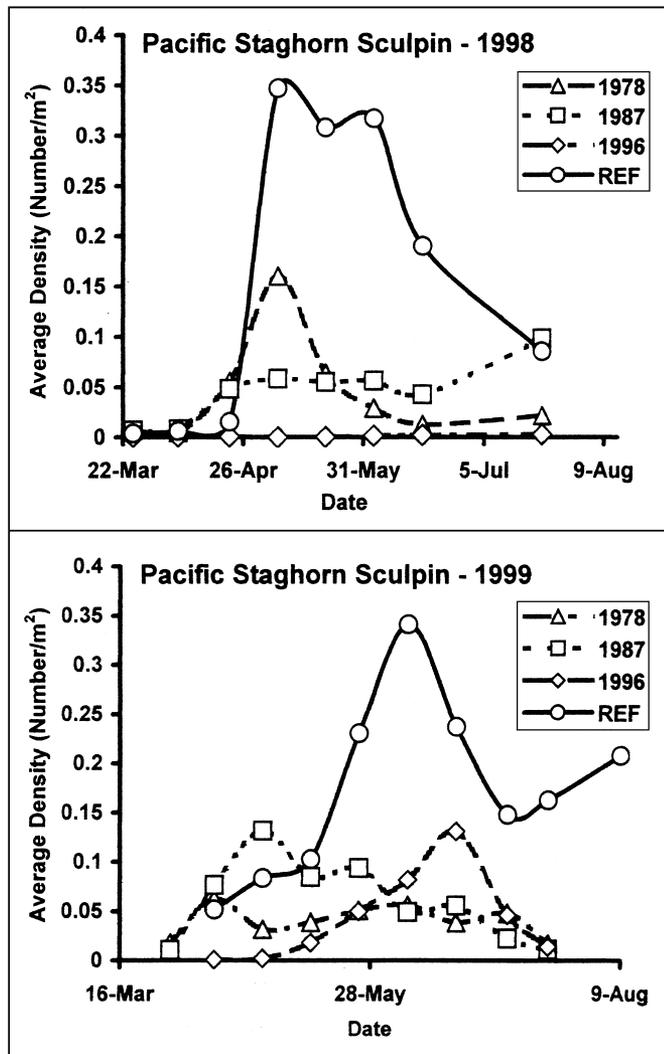
Figure 4. Catch per unit area for Pacific staghorn sculpin, a ubiquitous resident of the Salmon River estuary, in 1998 and 1999.

Figure 5. Catch per unit area for chinook salmon in the four marshes in 1998 and 1999.

differences between sites because they occurred most frequently and were commonly consumed by fish (Fig. 7). In 1998 and 1999, chironomid densities were greater in the 1996 marsh than in the other sites ($p = 0.008$ and $p < 0.001$, 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 ($p = 0.071$); however, ceratopogonid density was significantly greater in the 1996 marsh in 1999 ($p = 0.001$). Regression of average densities of all fallout invertebrates in restored marshes against marsh recovery age indicated no significant correlation in 1998 ($p = 0.063$, $r^2 = 0.179$) or in 1999 ($p = 0.927$, $r^2 = 0.001$). However, chironomids were negatively correlated with marsh age in both 1998 and 1999 ($p < 0.0001$, $r^2 = 0.566$; $p = 0.033$, $r^2 = 0.229$, respectively). Ceratopogonids were unre-
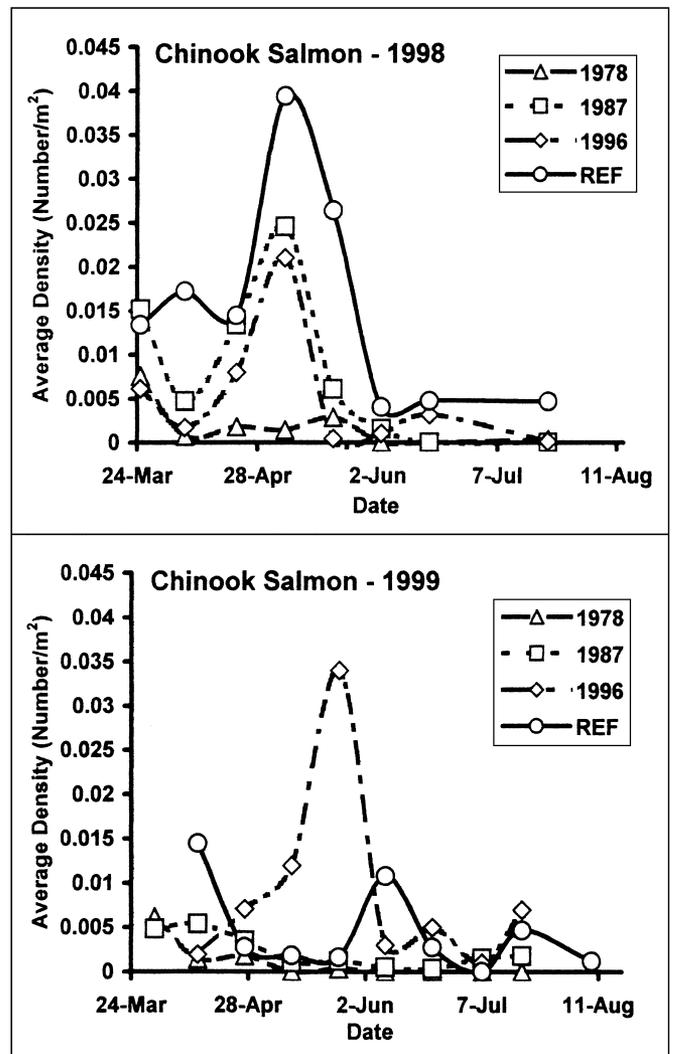
lated to recovery age in 1998 ($p = 0.838$, $r^2 = 0.002$) and negatively correlated in 1999 ($p = 0.002$, $r^2 = 0.428$).

Benthic Invertebrates

In 1998 and 1999, densities of benthic invertebrates were consistently greater at the REF marsh ($p = 0.022$ and $p = 0.002$, respectively) than the other marsh sites (Fig. 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 ($p < 0.001$ and $p = 0.027$, respectively) (Fig. 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
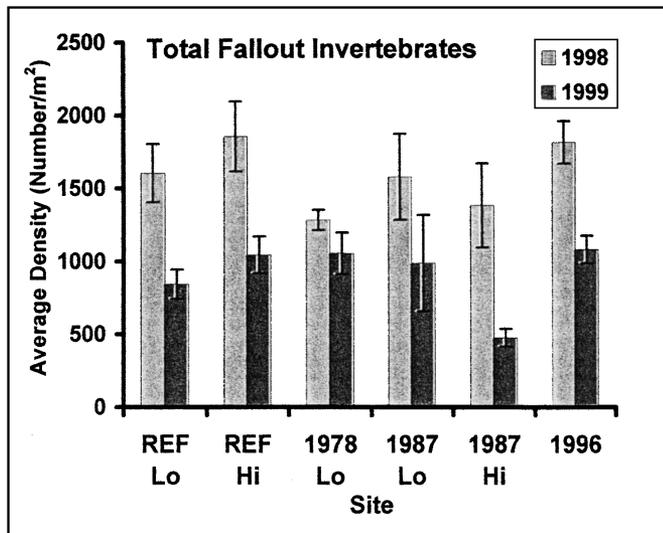
Figure 6. Total density of fallout insects in the four marshes. "Hi" and "Lo" (REF and 1987) refer to areas of differing elevation that support different plant communities. No statistical difference was detected in 1998 ($p = 0.221$), and low densities in the 1987 Hi marsh contributed to the statistical difference in 1999 ($p = 0.022$).

significantly lower in the 1996 marsh, compared with the 1987 marsh ($p = 0.020$ for both years), but no such differences were detected in *Eogammarus* spp. density ($p = 0.304$ and $p = 0.146$, respectively) (Fig. 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 ($p = 0.653$, $r^2 = 0.016$) but a positive correlation in 1999 ($p = 0.001$, $r^2 = 0.592$). Nereid worm densities were unrelated to recovery age for both years ($p = 0.598$, $r^2 = 0.022$; $p = 0.113$, $r^2 = 0.182$). *Corophium* spp. densities were also unrelated to recovery age for both years ($p = 0.292$, $r^2 = 0.085$; $p = 0.059$, $r^2 = 0.247$), and significant differences were only detected for *Eogammarus* spp. in 1999 ($p = 0.023$, $r^2 = 0.339$). However, when the data for both amphipods and both sampling years are considered together, a trend of increasing 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. 11). Epibenthic (mysids) and planktonic prey (fish larvae) also occurred occasionally. In general, diets from fish foraging in the 1978 and 1996 marshes con-
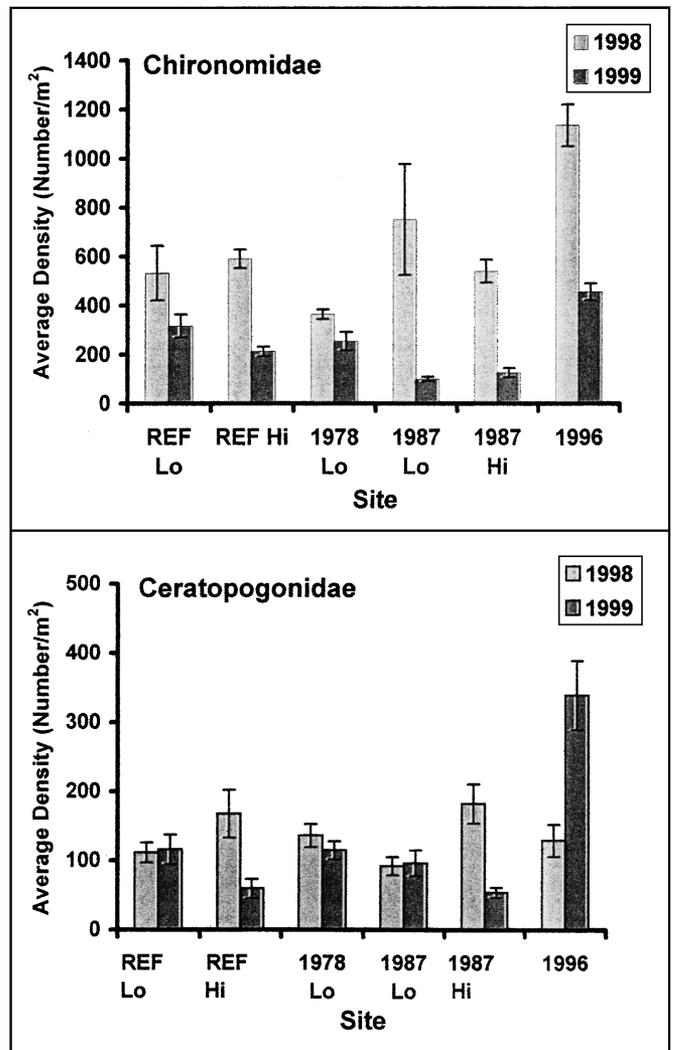

Figure 7. Density of the fallout dipterans of family Chironomidae and Ceratopogonidae in four marshes. Only the 1996 marsh is significantly different from all other sites based on a pair-wise comparison of means.

sisted of insects, whereas 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 2). 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 col-
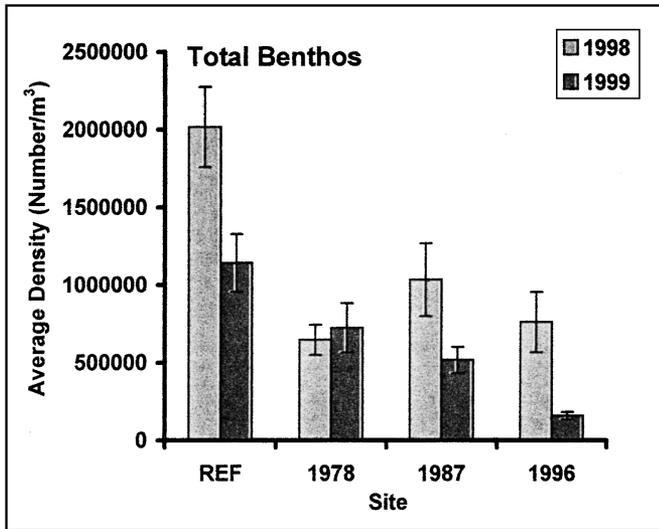
Figure 8. Total density of benthic invertebrates in the four marshes. Densities in the REF marsh are significantly different from all other sites based on a pair-wise comparison of means. No difference was detected between the restoring marshes.

lections indicate extremely high overlap in the availability and consumption of prey species (Table 3). 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
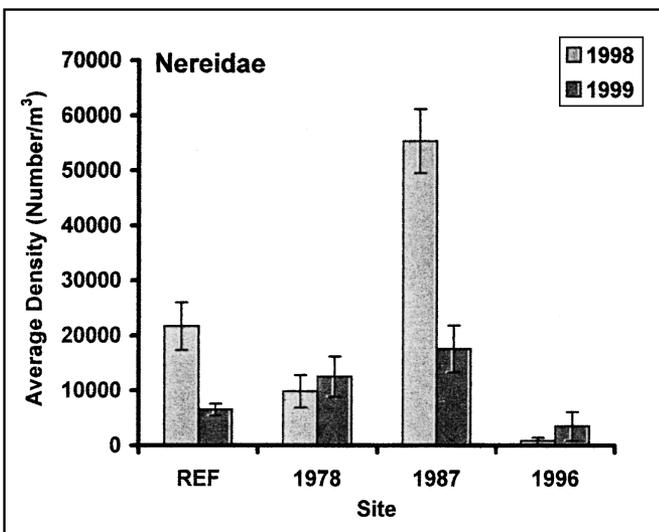

Figure 9. Density of nereid polychaetes in the four marshes. Pair-wise comparison of means detected significant differences in nereid densities in the 1987 marsh compared with all other marshes, and in the 1996 marsh compared with the REF and 1987.
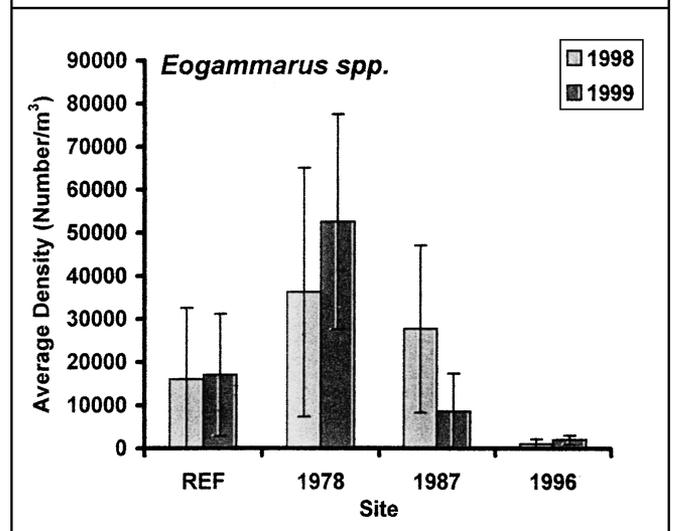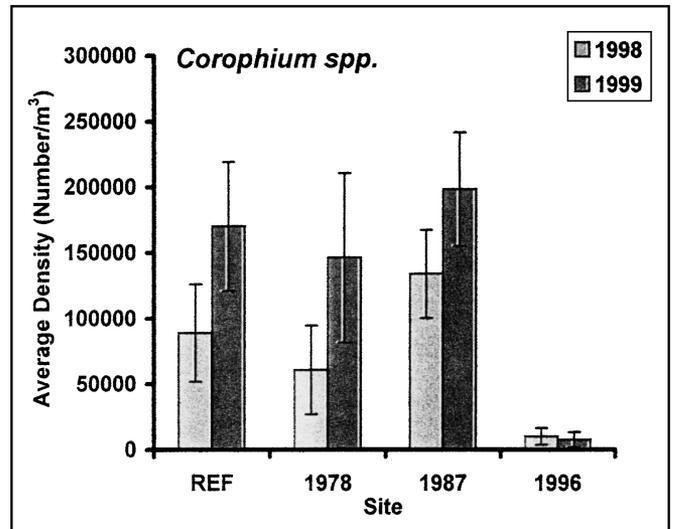

Figure 10. Densities of the two amphipods, *Corophium* spp. and *Eogammarus* spp., in the four marshes. Pair-wise comparison of *Corophium* spp. densities detected differences between the 1996 marsh compared with all other marshes and the 1978 and 1987 marshes.

38%. Trichoptera were the most highly selected prey items with an standardized forage ratios range of 0.13 to 1.00 (Table 4). Other selected species included *Corophium* spp., chironomid larvae, and dipterans.

Discussion

We documented differences in fish use, invertebrate prey resources, and diets of juvenile salmon in one natural and three recovering estuarine marshes of varying ages. Although there are potential confounding factors, we show that many of these differences are attributable to variable states of marsh redevelopment. These results, although early in our evaluation of the restoration
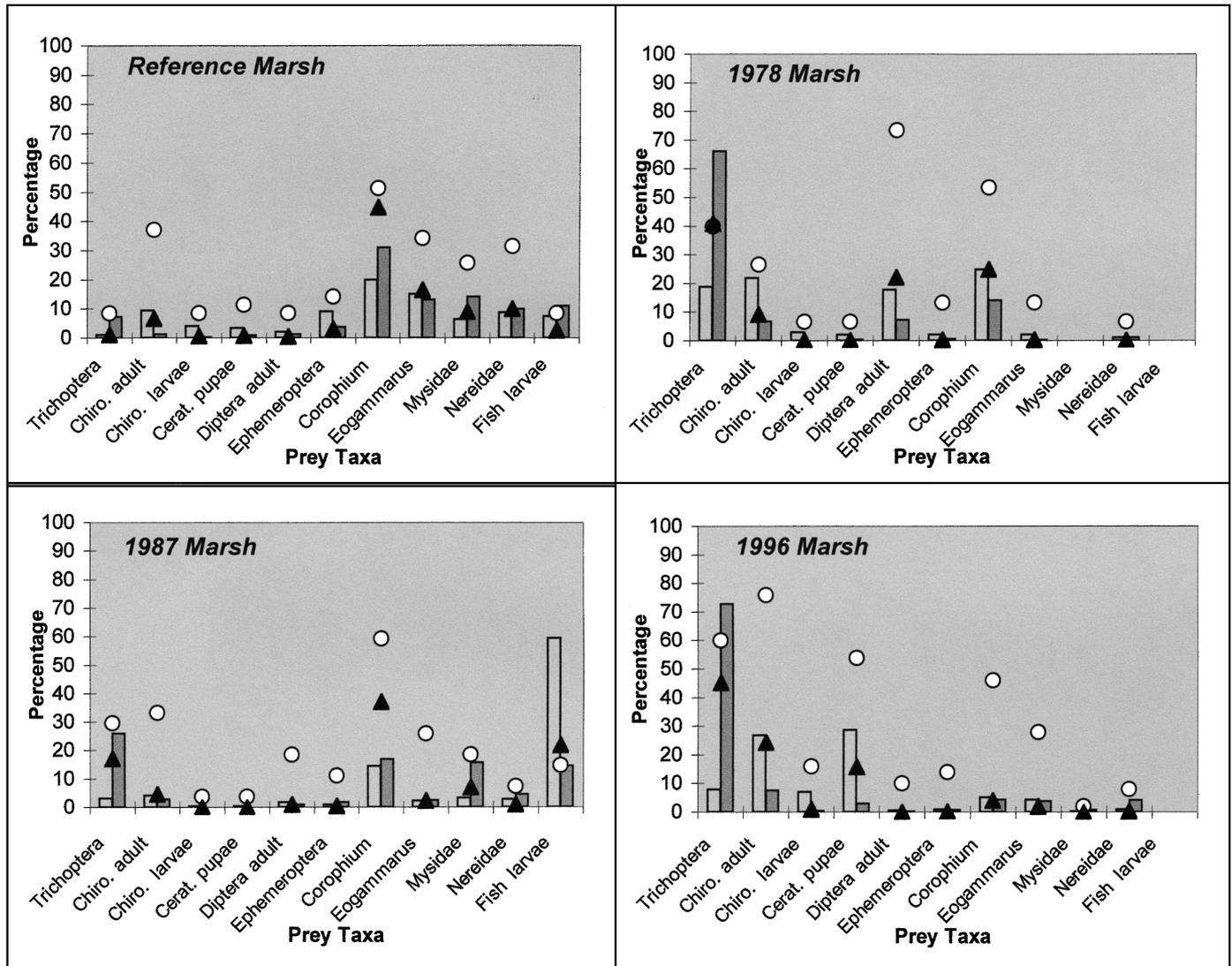
Figure 11. Diet composition as percent numeric abundance (light gray bar), percent gravimetric abundance (dark gray bar), percent frequency of occurrence (white dot), and percent total index of relative importance (black triangle) in the four marshes.

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 sys-

tem. 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

Table 2. Percent similarity of diet composition* between sites.

	Control vs. Restored Sites			Comparison of Restored Sites		
Sampling Date	CTR vs. 78	CTR vs. 87	CTR vs. 96	78 vs. 87	78 vs. 96	87 vs. 96
May 1998	9.34%	—	19.13%	—	23.74%	—
Jun 1998	10.96%	19.25%	7.40%	10.42%	24.49%	5.78%
Apr 1999	34.28%	52.90%	26.06%	70.75%	14.52%	29.61%
Early May 1999	—	67.17%	1.07%	—	—	7.35%
Late May 1999	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.

ecosystem maturity or quality. However, they were also found in large numbers in the 1996 site, suggesting a response to estuarine gradient position 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) 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 (38%) 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

Table 3. Percent similarity index between diets and available prey.

Month	Site	PSI
Apr 1998	1996	48%
May 1998	CTR	27%
	1978	53%
	1996	24%
Jun 1998	CTR	15%
	1978	33%
	1987	9%
	1996	91%
Apr 1999	CTR	23%
	1978	31%
	1987	49%
	1996	45%
May 1999	CTR	38%
	1978	28%
	1987	67%
	1996	31%
Average PSI		38%

CTR, control. Index used to determine overlap between diet composition and available prey.

Table 4. Standardized forage ratios for important prey items.

Prey Item	Apr 1998 1996	May 1998 CTR	1978	1996	Jun 1998 CTR	1978	1987	1996	Apr 1999 CTR	1978	1987	1996	May 1999 CTR	1978	1987	1996
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			

Bold indicates positively selected prey items. CTR, control.
Standardized forage ratios measure fish selectivity for particular prey taxa.

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, after 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.

Acknowledgments

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