THE FLORAL AND FAUNAL RECOVERY OF A RESTORED COASTAL

WETLAND: KUNZ MARSH, SOUTH SLOUGH, COOS BAY, OR.

by

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We certify that we have read this study and that it conforms to acceptable standards of scholarly presentation and is fully acceptable, in scope and quality, as a thesis for the degree of Master of Arts.

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ABSTRACT

Improved understanding of the importance of estuarine wetlands to the function of coastal ecosystems has lead to increased restoration efforts of degraded wetlands. The South Slough National Estuarine Research Reserve (SSNERR) in Coos Bay, Oregon, has taken active steps to restore previously diked tidal wetlands. In addition, they have established monitoring programs that focus on the faunal and floral recovery of restored sites within Kunz Marsh. The overall objective of the present study was to determine the degree of community recovery in restored salt marsh sites relative to control sites. Vascular plants, diatoms, invertebrates, and fish were sampled in spring, summer, and fall of 1998 and 1999. Vegetation cover decreased from high to low elevations, was higher in control than restored sites, and increased between 1998 and 1999. Diatom abundances showed seasonality in most sites and were higher in the restored than the control sites. Relative abundance of invertebrates in the first year was higher in control than in restored sites. More of these animals were found in vegetated than open areas. Fish abundance increased with decreasing elevation and Kunz Marsh sites showed a species composition similar to the adjoining Winchester Creek. In general, community recovery, particularly for plants and invertebrates, occurred more quickly in the higher restored sites. This may be due to the aggressive restoration method used, as well as successional processes that are known to occur more rapidly in the high marsh.

DEDICATION

This thesis is dedicated to my parents, Joachim and Heide Fritz, for their never-ending support in my quest to become a biologist, to my sisters, Susi and Helga, my friends Esther Bühl-Behling, Anita Köhler, and “die Tübinger”.

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PART I - ESTUARINE WETLANDS: A LITERATURE REVIEW

INTRODUCTION

Healthy coastal wetlands are important to humans, as well as to floral and faunal communities. From early settlements to the present, agricultural interests and industries have relied on the nutrient rich soils, easy access to waterways, rich fauna and flora, and water sources of wetlands for their livelihood. However, reckless use and destruction has left one of the world's most important ecosystems in bad shape. Although efforts have been taken to reverse the damaging effects, the results of wetland protection and restoration efforts are questionable. Overall, there is still a net loss of wetland habitats and functions in the United States. Development of a detailed introduction to wetlands, their functions, and spatial and temporal characteristics, should help to obtain an improved understanding of this habitat's complexity. Furthermore, it should aid in the evaluation of past restoration projects and to evaluate the necessity of innovative approaches. Adjusting restoration methods so that they work with natural patterns and community dynamics, should lead to greater success in restoring these important ecosystems.

WETLANDS

Estuarine and freshwater wetlands are complex and multi-functional ecosystems. A wetland is defined by the National Research Council (1995) as

> *…an ecosystem that depends on constant or recurrent, shallow inundation, or saturation at or near the surface of the substrate. The minimum essential characteristics of a wetland are recurrent, sustained inundation, or saturation at or near the surface and the presence of physical, chemical, and biological features, reflective of recurrent, sustained inundation or saturation. Common diagnostic features of wetlands are hydric soils and hydrophytic vegetation. These features will be present except where specific physiochemical, biotic, or anthropogenic factors have removed them or prevented their development*.

Estuarine wetlands are tidal areas that are usually semi-enclosed by land but that have at least partial spatial or temporal access to open ocean waters. They are limited upstream by salinities of less than 5ppt (Cowardin et al. 1979).

Although hydrology is the primary factor in determining characteristics of wetlands, it is different for each wetland. Furthermore, it is not easy to assess hydrological patterns. As organisms found within a wetland are mainly the result of the existing wetness, vegetation has historically been used to characterize wetlands and their zonation patterns (Tiner 1999). However, there is an increasing awareness of the importance of characterizing communities using other attributes. Typically, dominant plants in estuarine wetlands are grasses and herbaceous plants (Tiner 1999). Estuarine wetlands are characterized by bare mud or sand flats in lower elevations, then higher areas with low-growing vegetation, followed by an even higher transition zone to freshwater swamp or swamp forest (Chapman 1977). Diatoms have increasing abundances from lower to higher marshes (Oppenheim 1991), whereas invertebrate and fish numbers follow a tidal gradient with higher numbers in lower elevations. However, changes due to elevation are better observed in species diversity and composition, as some groups of organisms occur only at certain elevations.

In low marshes, that are more regularly flooded and contain a combination of open mudflats and patches of vegetation, large populations of invertebrates occur along with diatoms and small plants (Kozloff 1973). Of these plants, cordgrass (*Spartina foliosa*) is characteristic of regularly flooded marshes throughout California. Glasswort (*Salicornia virginica*) is found as a colonizer in sheltered Californian marshes like Humboldt Bay (Macdonald 1977), and common glasswort (*S. europaea*) and arrowgrass (*Triglochin maritimum*) are dominant in Alaskan low marshes (Tiner 1999). Within low marshes, invertebrate populations can be divided into three major groups: polychaete annelids, bivalve mollusks, and crustaceans. Kozloff (1973) notes that the ghost shrimp *Neotrypaea californiensis*, *Leptochelia dubia*, which is the most common tanaid in salt marshes, as well as several *Corophium* species are especially important. The gammarid *Eogammarus confervicolus* is also a common inhabitant (Batzer and Resh 1992). One of the primary deposit feeders is the sabellid *Manayunkia aestuarina*, which can be found in vegetated areas (Yozzo 1994) as well as in dense algal mats (Furota and

Emmett 1993). In a study by Kneib (1984) nematodes, oligochaetes, and polychaetes such as *Manayunkia aestuarina* and *Streblospio benedicti* were found in high abundances throughout the marsh but mainly at low to mid elevations. Addressing typical salt marsh animals on the west coast, Kozloff (1973) describes the anemone *Nematostella vectensis*, which can be found in shallow pools throughout the marsh, the two strictly salt marsh gastropods *Ovatella myosotis* and *Assiminea california*, and three crabs (*Hemigrapsus oregoniensis*, *H. nudus*, and *Pachygrapsus crassipes*) to be likely inhabitants. Recently the European green crab (*Carcinus maenus*) has been introduced to low estuarine habitats on the Pacific coast (Cohen et al. 1995).

In more irregularly flooded marshes of mid and high elevations, bulrushes (*Scirpus* spp.) and sedges (*Carex* spp.) are dominant plants (Kozloff 1973). *Scirpus robustus* is common in California and Oregon, whereas *S. maritimus* occurs in Washington and British Columbia, Canada. One of the most characteristic plants is the glasswort (*Salicornia virginica*), which is often mixed with fleshy jaumea (*Jaumea carnosa*). Saltgrass (*Distichlis spicata*) is the most prevalent true grass in the mid marsh. However arrowgrass can also be found. Furthermore, tufted hairgrass (*Deschampsia caespitosa*), spiny rush (*Juncus acutus,*), and Lyngby's sedge (*Carex lyngbei*), which is dominant on silty soils, can be added to the list (Tiner 1999). Of the invertebrates in mid marsh areas, special attention is given to the larval and pupal life stages of two dipteran families, Chironomidae and Ceratopogonidae, which are important food sources for juvenile fish (Shreffler et al. 1992). Chironomidae larvae and pupae occurred in higher marshes (mid and high) and were more commonly associated with open rather then vegetated areas, while Ceratopogonidae were more common in vegetated, undisturbed habitats (Shreffler et al. 1992). In an East Coast study, several polychaetes, such as *Nereis succinea*, showed an even distribution along an elevation gradient, while others, such as *Fabricia* sp., were more abundant in higher elevations (Kneib 1984). In addition, the polychaetes *Pygospio elegans* and *Hobsonia florida*, and the amphipods *Corophium salmonis* and *Eohaustoris estuaries* can be found in the lower mid-intertidal (Furota and Emmett 1993).

Gumweed (*Grindelia integrifolia*) and fat hen (*Atriplex patula*) inhabit the high marsh along with species such as sand-spurry (*Spergulary* spp.), *Tetragonia expansa* and *Baccharis pelularis* (Kozloff 1973). In addition to Chironomidae and Ceratopogonidae larvae and pupa, Ephydridae and Dolichopodida are abundant in higher elevations (Furota and Emmett 1993, King and Brazner 1999).

Monaco et al. (1992) call typical fish species found in Pacific coast estuaries the Northern Riverine group, which include the following 15 species: cutthrout trout, chum salmon, Chinook salmon, steelhead, redtail surfperch, Pacific herring, threespine stickleback, English sole, starry flounder, walleye surfperch, Pacific tomcod, northern anchovy, and shiner perch. Based on studies in Humboldt Bay by Chamberlain and Barnhart (1993) topsmelt, jacksmelt, Bay goby, and Arrow goby should be added. Monaco et al. (1990) also include surf and longfin smelt, and striped bass as common species in brackish waters of Coos Bay, Oregon.

Several factors influence these zonation patterns found in wetlands. The decrease of plant cover from higher and drier to lower and wetter areas is assumed to be the result of most plants being unable to withstand high salt concentrations and poor soil aeration due to water inundation (Vogl 1966, Tiner 1999). Furthermore, plant distribution is influenced by competition for limited nutrients (Levine et al. 1998, Van Wijnen and Bakker 1999) and by interspecific competition, which affects the upper limit of a species (Bertness 1991). Several factors, such as desiccation (Hopkins 1964, Meadows and Anderson 1968, McIntire and Overton 1971) and sediment characteristic (Saburova et al. 1995) influence diatom distributions. Larger diatom species are found in vegetated areas and smaller ones are found in open sand and mudflats as sediment movement caused by water motion increases frustule damage (Delago et al. 19991). Diatom interspecific competition is only relevant to distribution patterns on a microscale of 10-1000cm^2 (Saburova et al. 1995). Competition is only a minor factor influencing invertebrate distributions in soft bottom habitats (Peterson 1991). In contrast to rocky intertidal areas, organisms living in mud and sand are not subject to space limitations, or food reduction, as tidal activities keep redistributing nutrients and food particles. The gradual zonation of soft bottom invertebrates is less influenced by biotic factors and depends more on environmental factors such as desiccation and sediment characteristics. Fish distribution is influenced by the accessibility of the marsh areas due to channel availability and tidal levels. Juvenile and smaller fish are searching for protection from predation and fast water flows in channels of mid and high marshes (Shreffler et al. 1992).

In addition to the spatial differences between lower and higher marsh areas, seasonality affects marsh communities in temperate climates. Annual vascular plants complete their lifecycle within one year, achieving maturity usually during late spring and the early summer months. Dense vegetation in higher marshes protects from high evaporation loss in the drier and warmer summer months. Therefore, seasonal fluctuations of percent cover are larger in lower elevations where vegetation patchiness is common. Upper marsh diatoms reach highest numbers in the summer, while in lower open marsh areas overall smaller numbers show no seasonal pattern (Oppenheim 1991). Seasonal changes in invertebrate communities are usually attributed to individual life cycles and extreme conditions such as drought and are less likely due to seasonal environmental fluctuations (Robert and Matta 1984). Changes in fish communities depend on several factors. Some fish are migratory and enter marsh areas only during certain times of the year, and only their early life-history stages forage in marsh channels and less accessible areas (Shreffler et al. 1990, 1992). Finally, year round resident fish can be influenced by seasonality during extreme weather conditions, such as seasonal droughts and hypoxia (Suthers and Gee 1986)

SUCCESSION IN WETLANDS

Successional processes observed in natural marshes can include temporal changes within one zone or the sequential development of one zone into another. Coastal wetlands are dynamic environments progressing from mudflats through several different plant communities; from low submerged vegetation to irregularly flooded brackish water marshes to high coastal communities, which form a sharp boundary with the terrestrial upland shrub vegetation (Chapman 1977). Low marshes can develop into higher marshes, as vascular plants baffle water action, while their root systems trap sediments. Increasing elevation changes the physical environment such as inundation periods and salinities, and therefore select for new communities.

In most areas along the Pacific coast, cordgrass (*Spartina foliosa*) is the primary colonizer of low marsh/mudflat areas as it is the most tolerant halophyte (Macdonald and Barbour 1974). Glasswort is usually found at slightly higher elevations taking over former cordgrass areas. However, in sheltered areas, such as Humboldt Bay, CA, glasswort becomes the primary invader and is then replaced by salt grass, a common mid and high marsh grass (Macdonald 1977). Phosphorus is absent in early successional stages however, further depositing of clay increases the amount available in older marshes (Van Wijnen and Bakker 1999). Furthermore, plants of early succession and lower elevations are more susceptible to herbivory, especially during the winter season, when grazing can have a long-term effect on marsh development (Dormann et al. 2000). Therefore, the combination of herbivory and competition for light between smaller plants and taller grasses determines vegetation patterns.

Invertebrate communities undergo succession as well. The first colonizers, such as tube-forming polychaetes, are small and achieve high densities and productivity, but also experience high mortality (Valiela 1984). They usually feed on suspended or recently deposited sediments. Later colonizers are larger, less productive and suffer lower mortalities. They burrow deeper and are typically deposit feeders. It has been shown that tube-building polychaetes, such as *Manayunkia astuarina*, attract other polychaetes, oligochaetes, and bivalves due to sediment restructuring. Other invertebrates become established with developing habitat and food availability, especially those directly depending on certain plants such as the flies within the family Dolichopodiae (Kraeuter and Wolf 1974). Additional effects of plant communities on invertebrates are protection from predation and provision of a stabilizing platform. As soon as habitat and food sources are provided, fish use salt marshes as nursery areas for their young and as foraging habitats during high tides (Valiela 1984, Shreffler et al. 1990).

IMPORTANCE OF WETLANDS

Estuarine wetlands represent an important link in the ecology of coastal and near-shore habitats. Due to their high primary productivity, they supply estuarine systems with carbon, which is also exported into coastal waters (Hoffnagle 1980). Associations among producers and consumers indicate that inputs from intertidal macroalgae, marsh microalgae, and salt marsh vascular plants provide organic matter to these ecosystems that in turn support invertebrates, fish, and birds (Kwak and Zedler 1997). Furthermore, macro- and microalgae enhance oxygenation of the water column and decrease CO_2 (Browder et al. 1994 in Goldsborough and Robinson 1996).

Submerged and emergent vegetation provides habitat for many different species of invertebrates and vertebrates, including endangered species. Ten of the 94 animal species listed in 1989 as endangered or threatened in California live in coastal areas and all are associated with wetlands (Zedler 1991). These include species such as the California brown pelican, the unarmored threespine stickleback, and the San Francisco garter snake.

In combination with estuarine and watershed systems, tidal wetlands function as important nursery grounds for many fish species (Sogard 1992) and invertebrates, including commercially important species such as salmon (Macdonald et al. 1987), and English sole (Gunderson et al. 1990). Overall they provide spawning and nursery habitats, as well as feeding areas for approximately 80% of North America’a coastal fisheries (Hussey 1994).

Furthermore, wetlands provide migratory stopovers and overwintering grounds for waterfowl, including ducks and brant geese (Zedler 1996a).

Estuarine wetlands play an important role in ground water recharge (Hollands 1985), sediment stabilization, as well as shoreline anchoring, storm protection, and purification of surface waters (Sather and Smith 1984). Some wetlands are specifically used for the treatment of run-off and drainage water (Magmedov et al. 1996). In addition, estuarine wetlands function in flood control (Larson 1985), as they intercept and hold precipitation and store floodwaters (Hey and Philippi 1995). Finally, wetlands have aesthetic, cultural, and educational values (Niering 1985).

WETLAND DEGRADATION

Unfortunately, wetlands along the Pacific Coast have been subject to dredging, diking, and draining for industrial, residential, and agricultural purposes (Maltby and Dugan 1994, Wheeler 1995). The U. S. Fish and Wildlife Service estimates that approximately one-half of wetlands that once existed in the lower 48 states have been dredged, drained, or filled (Kusler and Kentula 1990). Over 30%, 40%, and 90% of all wetlands in Washington, Oregon, and California, respectively, have been eliminated between 1780 and 1980 (Dahl 1990). In the Coos Bay, OR area alone, 85% of marshland habitats have been lost (Reffalt 1985).

In addition to obvious hydrological changes, chronic dredging leads to alterations of the chemical, biological, and physical characteristics of the upper estuarine sediments (Bella and Williamson 1980). Tidal restrictions due to diking cause a reduction in salinity, lowering of the water table, and a drop in marsh surface elevation (Roman et al. 1995) and a decrease in habitat availability. This leads to an inevitable change in species composition and abundance and decrease of detritus export into estuaries (Montague et al. 1987). In addition, several studies indicate that the disturbance of habitat favors the spread of non-native plants, which eventually out-compete the native flora (Taylor 1983), especially during prolonged low salinity and soil saturation periods (Kuhn and Zedler 1997).

WETLAND RESTORATION

Restoration efforts in North America have increased over the past three decades partly due to legislative decisions based on Section 404 of the Clean Water Act and the National Environmental Policy Act. Several approaches are available to comply with the legal requirements concerning wetlands. These include conservation, mitigation, and restoration; the latter can be part of the mitigation process.

The US Army Corps of Engineers and the Environmental Protection Agency share the responsibility of implementing the Clean Water Act (Blumm and Needleman 1992). Mitigation is part of the regulatory program of the Clean Water Act. It allows wetlands to be filled, drained or otherwise degraded, as long as compensation for the lost area and biological function is implemented either through restoration, enhancement, or creation of wetlands in a different location (Reppert 1992). Theoretically, these projects have to prove that no other alternatives are possible to wetland degradation and that the project is water dependent. Furthermore, it has to be shown that wetland impacts have been minimized. Overall, the most important specification is the necessity that mitigation projects ensure no net loss of wetland areas, as well as their ecological functions (National Wetland Policy Forum 1988).

Enhancement is the increase of biological functions within an already existing wetland. Depending on the type of wetland lost and its important features, a specific goal is set, for example to increase fish abundance or more specifically juvenile salmon habitat. Depending on the goals, different techniques can be used, which include increasing water flow, channel complexity, wetland area, or the establishment of certain wetland niches. The creation of wetlands is the change of upland habitat into wetland areas, through interception of groundwater or connecting the chosen site to a source of surface water (Brinson and Rheinhardt 1996). Restoration is the return of an area to its pre-existing wetland condition (Brinson and Rheinhardt 1996). Methods used to achieve wetland status range from partial to complete dike removal, dechannelizations, as well as the planting of native plants, and more aggressive approaches such as grading to appropriate elevations.

Created and restored wetlands offer considerable promise in wastewater and storm water treatment, non-point source pollution control, and flood prevention (Young 1996). Enhancement of tidal action can also aid in the reduction of mosquito populations, which are an increasing problem even in temperate areas (Kramer et al. 1995). Furthermore, created wetlands play an important part in the enhancement of fish populations, as well as other animals and plants. For example, restored wetland and estuarine habitats enhance the detritus-based food chain, and hence chironomid insect populations, which are selected by juvenile salmon as a primary food source (Shreffler et al. 1990). Such food sources supported the temporary residency of juvenile chum and Chinook salmon in a restored wetland system in the Puyallup River estuary, WA. This particular wetland was created to support salmon, waterfowl, shorebirds, raptors and small mammals. In its present state, it shows regular flooding, establishment of mudflats, higher wetland vegetation, and upland grassland (Shreffler et al. 1992). Overall, Zedler et al. (1997) suggest that fish assemblages relate more to channel morphology and hydrology than to type (i.e. natural versus constructed habitat). Therefore, as functions of small creeks differ from large channels, restoration projects must provide the hydrologic variety of a natural wetland to achieve the same performance. Research on small, created wetlands in Wisconsin (Reinartz and Warne 1993) shows that reseeding of wetlands with native seeds can enhance long-term vegetative diversity. Unseeded wetlands in this study needed longer to reach the same diversity levels and did not exhibit a stable community structure. Finally, substrata without plant cover may develop high salinities and become unfavorable for some native high marsh species (Haltiner et al. 1997).

Unfortunately, the ultimate success of restoration projects to date is unknown. Although measures for structural attributes of habitats exist, ecologists are asked to come up with simple, fast, and cheap measures of functions such as primary productivity, nutrient cycling, organic matter accumulation, predator-prey interactions, sustainability, and resistance to exotic invaders (Zedler 1996b). Proper evaluations of restoration efforts are limited by the lack of monitoring programs to measure functional attributes of these projects (Haltiner et al. 1997), and the lack of comparable reference wetlands, or control sites (Simenstad and Thom 1996).

The National Research Council (1992) states that mitigation efforts have not yet duplicated lost wetland functions, nor has it been shown that restored wetlands maintain biodiversity. Presently, compensatory mitigation is favored, rather than avoiding or minimizing wetland impacts (Race and Fonseca 1996). In addition, as opportunities to create new wetlands decrease due to the lack of available space, remodeling existing wetlands to enhance value is favored, but such projects have shown little success to date (Race 1985, Zedler 1996b). Some restoration plans may even threaten biodiversity, including endangered species, as they can result in net losses of wetland habitat (Zedler 1988). Race and Fonseca (1996) stated, “Decisive action must be taken by placing emphasis on improving compliance, generating desired acreage, and maintaining true baseline.” Wetland and restoration terms have to be well defined and project goals clearly stated. Otherwise, restoration success cannot be properly evaluated and appropriate adjustments cannot be taken (see Race 1985, Harvey and Josselyn 1986). There is need for regional co-ordination, as individually viewed projects often do not allow priorities to be set nor do they aid in determining the best areas for future work (Zedler 1988).

SUMMARY

The importance of wetlands has been well established. Since approximately the 1970's international policies such as the Ramsar Convention as well as national policies and laws, including Section 404 of the Clean Water Act, have provided the legal incentive to protect and restore these habitats. However, we are still facing the fact that restoration projects do not fulfill the most basic of goals: to stop the loss of wetland habitat. A better understanding of wetland dynamics, as well as the spatial, seasonal, and successional patterns should enhance the ability to develop appropriate restoration techniques. Updating legal achievements and implementation of new policies such as the Estuary Restoration Act of 2000 (www.estuaries.org, 08/12/2001), is just part of a new approach to ensure the protection of wetlands. Besides legal incentives, innovative and aggressive approaches are needed, along with well planned projects and mandatory observations over ecological time frames of five, ten, or 20 years, and the willingness to re-evaluate and re-adjust protective and restoration measures.

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PART II – RECOVERY OF RESTORED KUNZ MARSH, SOUTH SLOUGH,

COOS BAY, OR

INTRODUCTION

Coastal wetlands are important ecosystems with physical and biotic links between fresh and marine waters. High rates of primary productivity by algae and plants support diverse communities of invertebrates, fish, and birds (Cornell 1978, Kwak and Zedler 1997), some of which are of great economic importance. Wetlands play a major part in the eutrophication of coastal waters (Hoffnagel 1980), flood control (Larson 1985), ground water recharge (Hollands 1985), sediment stabilization, and in the purification of surface waters (Van der Ryn and Cowan 1996).

Unfortunately, estuarine wetlands along the Pacific Coast have been dredged, diked, and drained for industrial, residential, and agricultural purposes (Maltby and Dugan 1994, Wheeler 1995). Tidal restrictions cause changes in salinity, soil subsidence (Roman et al. 1995), as well as changes in species composition, and decreases in detritus export to coastal waters (Montague et al. 1987). Legislative efforts require the restoration and mitigation of lost wetlands but there is still a net loss of habitat acreage and quality (Zedler 1991). In addition, the long-term success of restoration projects is unknown. Limitations that prevent adequate evaluation of restoration efforts include the lack of monitoring programs to measure the ecological performance and functional attributes of the wetlands (Haltiner et al. 1997), and the lack of comparable reference wetlands, or control sites (Simenstad and Thom 1996).

Significant restoration activities have been initiated in the South Slough watershed, Coos Bay, OR. It encompasses 7,810 hectares of an integrated coastal ecosystem that include upland forests, streams, riparian areas, salt marshes, and estuarine mudflats (Rumrill and Cornu 1995). Estuarine tidelands account for approximately 530 hectares (about 7%) of the entire South Slough system. Almost a quarter of the watershed (2,023 hectares) is part of the South Slough National Estuarine Research Reserve (SSNERR). The integrated watershed restoration program of the SNERR includes: 1) abandonment of unused roads in order to control erosion and sedimentation, 2) re-establishment of diverse riparian vegetation and streamside forestation, 3) re-creation of in-stream habitat complexity for juvenile fish, and 4) restoration of coastal fresh and salt water wetlands (Rumrill and Cornu 1995).

These ongoing restoration efforts in the SSNERR provide the opportunity to critically evaluate an extensive wetland restoration project. Long-term monitoring programs have been established since 1996. Within this system, the successional stage of relatively undisturbed areas of the marsh can be compared to recently restored sites. The Kunz Marsh restoration project initiated in 1996 included removal of earthen dikes, re-establishment of tidal circulation, and the construction of experimental sites, each of which is at a different tidal elevation. Planners of this project took the aggressive step of using the soil from the dike to landscape sites at different elevations alongside the main channel of Winchester Creek. This step was taken in order to accelerate the process of community recovery relative to other restoration efforts, which have only breached earthen dikes (Rumrill and Cornu 1995) and passively waited for sediment accumulation and community recovery.

The overall objective of the present study was to determine the degree of community recovery in restored salt marsh sites relative to control sites. The specific objective was to determine the influence of tidal elevation on rates of community recovery. Intertidal community members that received specific attention were the vascular plants, microalgae (e. g. diatoms), invertebrates and fish.

MATERIALS AND METHODS

Study area

Kunz Marsh is a fringing tidal wetland located in the riverine region of the South Slough estuary, SSNERR, Coos Bay, OR (Figures 1 and 2). As mentioned in the introduction above, tidal circulation was restored in 1996. In addition, several experimental sites have been established at different tidal elevations. Geotextile fences have been placed between the sites to prevent soil moving horizontally and to keep the landscaped elevations in place. Sampling was conducted in the 2.2m (high), 1.8m (mid), 1.5m (low), and non-filled 1.1m (passive) marsh sites (Figure 3). The low marsh site was half filled with redistributed dike material, which resulted in the formation of a brackish pond in the upland portion of the site. As this pond did not reflect ecological dynamics of a marsh at an elevation of 1.5m, only the front half was sampled. Control sites were selected based on comparable water levels during high tide to the restored sites. They were located as close as possible to the restored sites for access and efficient sampling within tidal cycles.

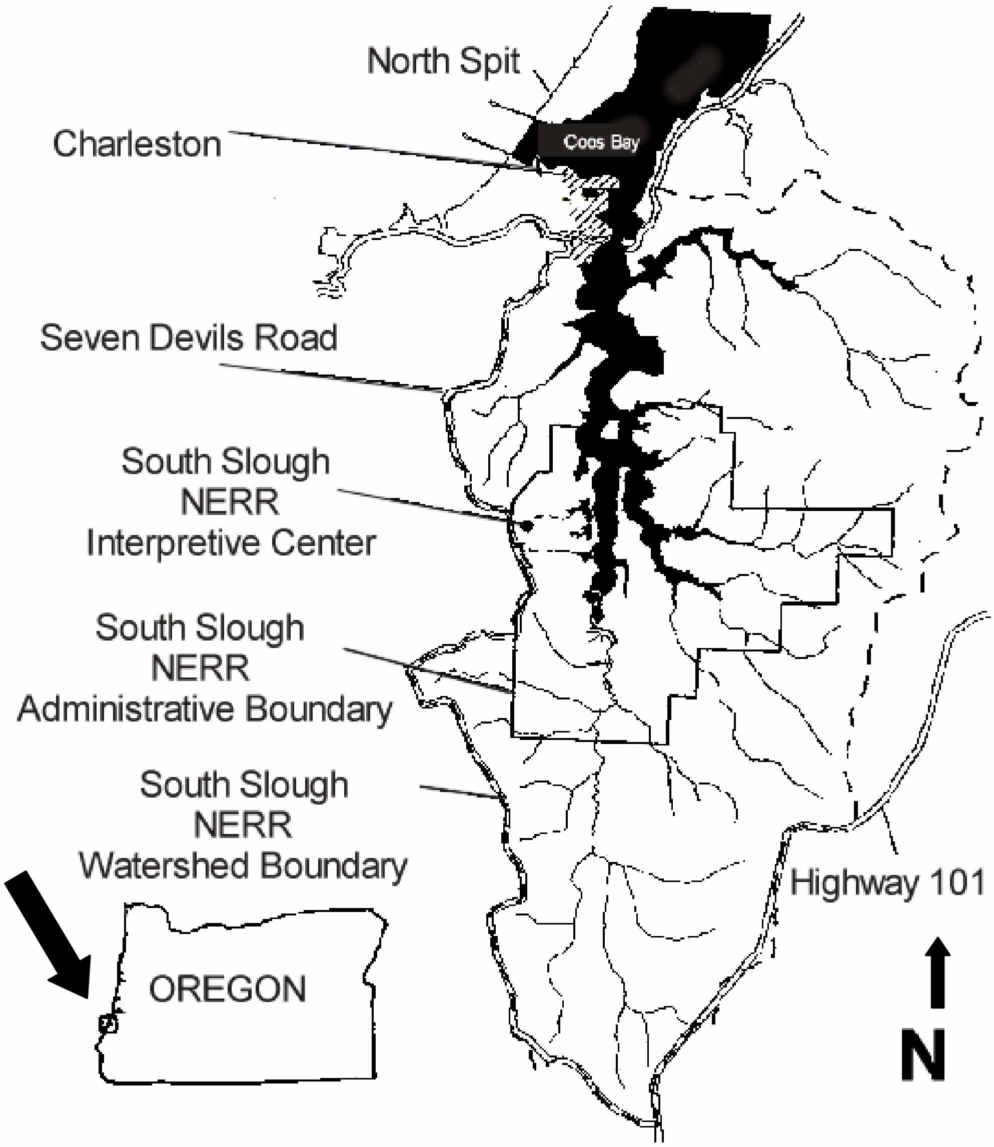

Figure 1. Location of the South Slough National Estuarine Research Reserve (SSNERR), Coos Bay, OR.

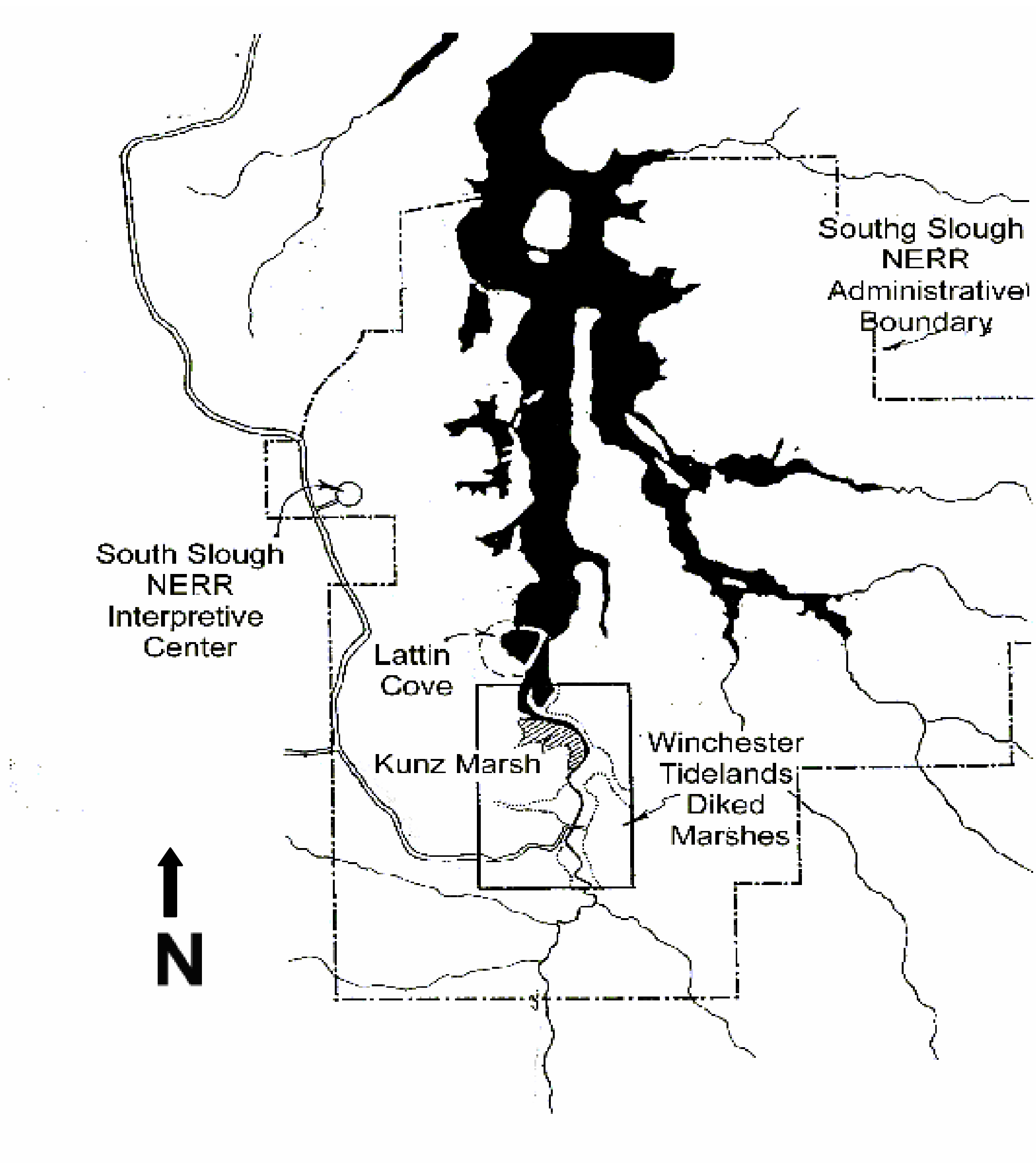

Figure 2. Location of restored Kunz Marsh and nearby control sites within SSNERR boundaries, Coos Bay, OR.

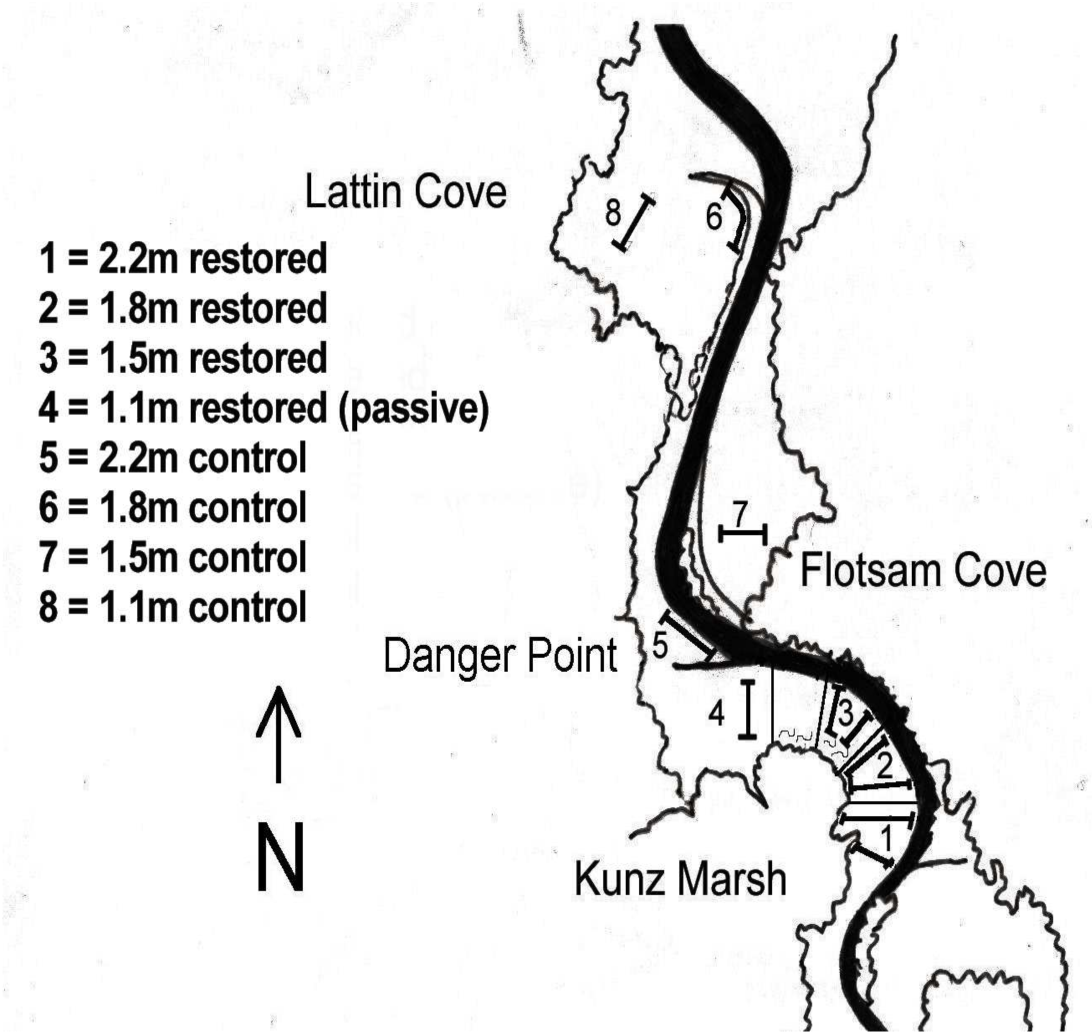

Figure 3. Location of sampling sites and transect lines (|—|) within SSNERR, Coos Bay, OR. Lines between sites 1 – 4 represent geotextile fences, which were established to prevent soil from moving horizontally. Control sites = undisturbed, natural marshes, Restored = dike was removed and soil distributed to achieve specific elevations, Restored (passive) = dike was removed, site left at subsidence level.

Danger Point was a small natural marsh located immediately north of Kunz Marsh. It had an elevation of 2.2m and functioned as control site for the high marsh site. For the low marsh, a small marsh (Flotsam Cove) across the channel of Danger Point was used as a control. The passive site was compared to a marsh area downstream (Lattin Cove), which showed similarities in elevation, restricted tidal flushing, and freshwater influence. A 1.8m mid-marsh framed this low marsh and functioned as control for the mid-marsh restored site. Table 1 shows the sites, their locations, elevations, and tidal heights.

Table 1. Sampling site characteristics of restored Kunz Marsh and control sites, SSNERR, Coos Bay, OR, 1998. NAVD = North American Vertical Datum, Appr. = Approximately, NA = not available

Site	**Location**	**Elevation (m NAVD)**	**Marsh-type**	**Tidal depth (at +2.3m)**	**Area (m^2)**
Restored	Kunz Marsh	2.2	Restored high	0.1	5,664
	Kunz Marsh	1.8	Restored mid	0.5	5,583
	Kunz Marsh	1.5	Restored low	0.8	5,543
	Kunz Marsh	1.1	Non-filled, passive	1.2	24,276
Control	Danger Point	2.2	Natural high	Appr. 0.1	NA
	Lattin Cove	1.8	Natural mid	Appr. 0.5	NA
	Flotsam Point	1.5	Natural low	Appr. 0.8	NA
	Lattin Cove	1.1	Natural lower	Appr. 1.2	NA

Sampling methods

Sampling began in April 1998. We also sampled during July and September of 1998 to include the main periods of the growing season for north temperate wetlands. Sampling was also conducted during the same seasons, in the months of May, July, and October, in 1999. Except for spring 1998, when sampling was completed over two consecutive weekends, all samples were obtained within a few days of one sampling period.

Sampling was conducted along transect lines in each of the eight sites (Figure 3). Two transects were used in the high, mid, and low marsh of the restored sites to cover more habitat. In all other sites, one transect line was used due to the shape and area of the habitat. The position of transect lines within restored sites corresponded with transects used for other projects conducted by the SSNERR to allow for a better comparison between the data collected.

Within sites, two microhabitats were identified. One was open space with bare mud/sand substrate, and the second showed clumps or extended areas of emergent vascular plants and some entangled algae. Therefore, stratified sampling for invertebrates and algae was used to monitor these niches separately. These two strata were located by determining the closest open and nearest vegetated area to the random sampling point on the transect line. In addition to changes within and among sites, this strategy allowed recovery rate data to be collected in relation to existing vegetation data, and it measured the influence of vegetation patterns on algae and invertebrate abundances. Open areas were absent in the control high marsh and were reduced to small patches in the mid control marsh. Therefore, we determined that a 10x10cm area without emergent living vegetation was sufficient to be considered “open”. In the 1.1m restored site, vegetation consisted mainly of dead cattail (*Typha latifolia*), and small patches of living plants. Therefore, cattail stems were considered vegetated areas.

Vascular plants

Percent cover and richness of emergent wetland vascular plants was documented. Thirty random numbers were chosen along transect-lines (Table 2). When two transect lines were used per site, 15 positions per line were sampled. Percent cover was determined by using a 1x1m quadrat. To monitor a wider area of habitat and to avoid clustering of sampling points, samples were taken on the left side of the transect line when an even numbered position was picked, and on the right side for odd numbers. Variables were total percent plant cover, as well as percent cover of annual and perennial plants. All species were identified within each quadrat. Additional species noted outside of quadrats were also recorded and added to the species list. When species could not be identified in the field, specimens were taken into the laboratory to key out by using appropriate literature and a species list collected by SSNERR personnel. If positive identification was still uncertain, Gordon Leppig (Curator of the Vascular Plant Herbarium, Humboldt State University, CA) was consulted. Non-indigenous species were noted.

Table 2. Sampling design for vascular plants, diatoms, and invertebrates in restored Kunz Marsh and control sites, SSNERR, Coos Bay, Oregon. Numbers indicate number of samples taken per site and microhabitat per sampling period (spring, summer, and fall), for the years 1998 and 1999.

Site	**Vascular plants**	**Microhabitat**	**Diatoms**	**Invertebrates**
2.2m restored	30	Open	12	4
(high)		Vegetated	12	4
1.8m restored	30	Open	12	4
(mid)		Vegetated	12	4
1.5m restored	30	Open	12	4
(low)		Vegetated	12	4
1.1m restored	30	Open	12	4
(passive)		Vegetated	12	4
2.2m control	30	No open	-	-
(high)		Vegetated	12	4
1.8m control	30	Open	12	4
(mid)		Vegetated	12	4
1.5m control	30	Open	12	4
(low)		Vegetated	12	4
1.1m control	30	Open	12	4
(as passive)		Vegetated	12	4

Benthic diatoms

Benthic algae that occur on marsh areas consist primarily of diatoms (Bacillariophyceae). For each site, 12 random numbers were chosen along transect-lines. When two transect-lines were used per site, six positions per line were sampled. Again, depending on the random number, the sample was taken either to the left or the right of the transect-line. A 2x2cm quadrat was used to take benthic scrapes. At each random point, the nearest open and vegetated strata were picked resulting in a total of 24 samples per site (Table 2). Samples in vegetated areas were taken as close as possible to a vascular plant. Scrapes were placed in labeled plastic jars and some brackish water added.

Samples were fixed in 10% formalin and then diluted with distilled water to 250 ml. One-milliliter sub-samples were placed in a gridded Sedgewick-Rafter Cell. Diatom cells were counted to 500; a sample size established in previous studies (Amspoker 1977, McIntire 1978). Based on areas scraped and subsequent volume of solution analyzed, calculations result in diatom numbers per cm^2. All diatom samples of 1998 as well as summer samples of 1999 were analyzed.

Benthic invertebrates

Invertebrate samples were taken at the same 12 positions used for diatoms. A 7.5cm corer was used for invertebrate samples. The corer was placed over vegetation or in the open areas. Samples were taken to a depth of 5.0 cm. The cores were placed in numbered plastic bags and filled with brackish water. Benthic samples were kept cool until they could be transported to the laboratory for processing. Invertebrate samples were sieved through a 0.5mm sieve. Samples were fixed in 10% formalin. Individual organisms were picked out off the substrate, sorted and stored in 40% isopropyl alcohol. To ensure that all organisms present were found, each sample was picked twice. Organisms were counted and identified. Due to time constraints, only eight (four per substratum), out of the 24 samples per site were randomly chosen and picked. Samples of spring, summer, and fall of 1998 as well as summer of 1999 were analyzed (Table 2).

Fish

Because SSNERR had established a long-term fish monitoring program, fish data were collected by South Slough staff. Fish abundance and diversity were monitored at each of the restored sites, but also in Winchester Creek, which supplies Kunz Marsh with saltwater during a flood tide.

A modified beach seine was used to sample restored Kunz Marsh sites (Figure 4). Due to slightly different topography between sites, each site required different seine configurations. Sampling occurred monthly between

November 1998 and May 1999 during an ebb tide. Although most fish were trapped immediately in the seine-bag, the entire seine haul was not processed until almost all water had been drained from the sites. Fish were placed in buckets of water to be counted, measured, and identified whereupon they were released. Sub-sampling was used when large amounts of fish were caught.

A standard beach seine was used to sample within Winchester Creek. Based on topographic differences, the creek was divided into three sections: lower, upper, and middle reaches (Figure 4). Each reach had several seining sites for replication. Sampling occurred monthly during high tide from November 1998 thorough June 1999. Fish were handled as described above.

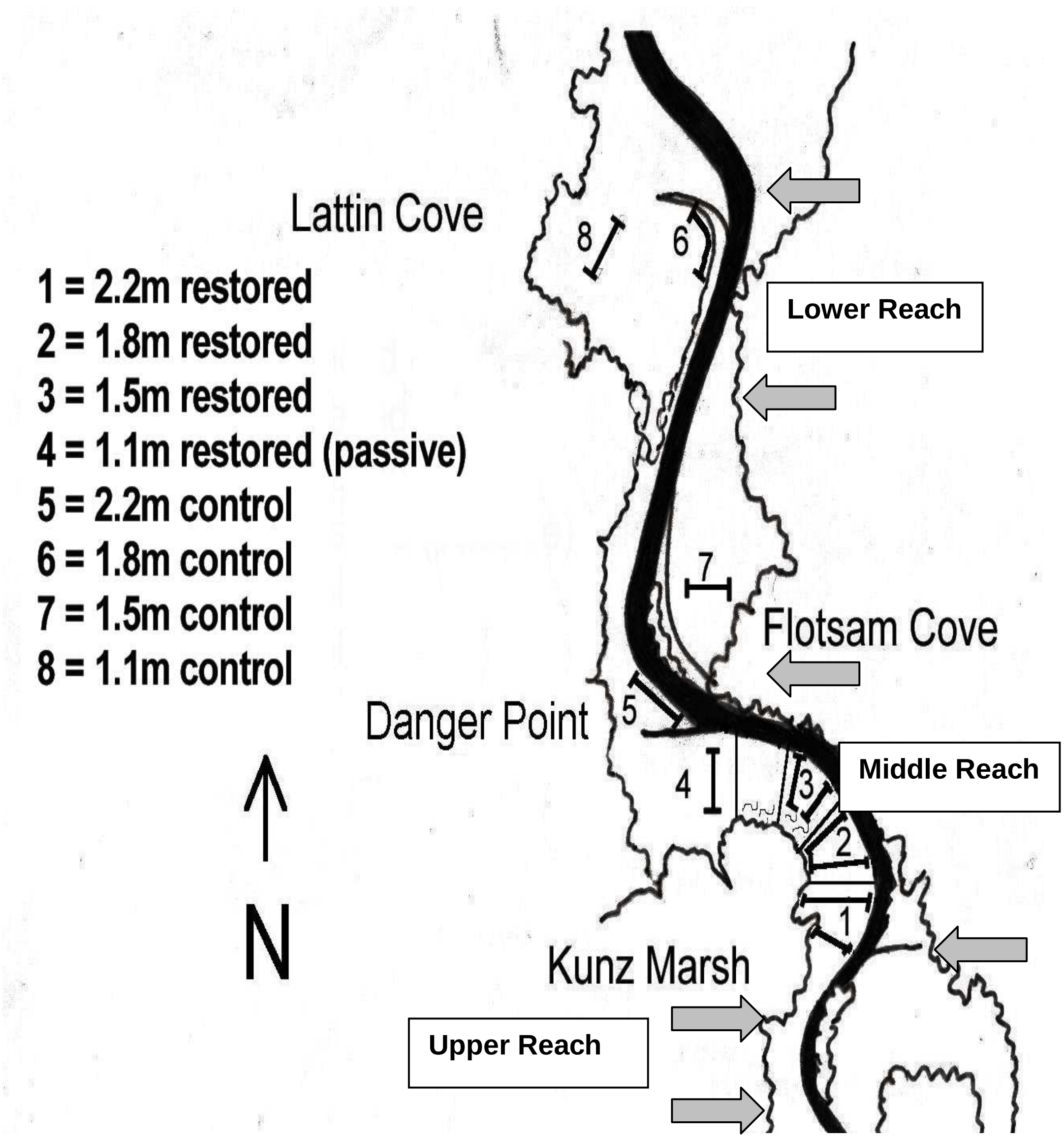

Figure 4. Location of seining sites in Winchester Creek, SSNERR, OR, November 1998 through June 1999. Arrows bracket reaches. Modified beach seines were placed in front of each Kunz Marsh sites. Standard beach seines were used in the creek. (|—|) = transect lines used for vascular plants, diatoms, and invertebrates.

Precipitation and temperature

Monthly temperature and precipitation data were obtained from the Municipal airport in North Bend, OR for 1998 and 1999.

Data analysis

Vascular plant percent cover and richness

In order to determine the effect of treatment and time on plant abundance, an ANCOVA was used to analyze the perennial plant cover. Variables included year, season, and treatment, as well as elevation as covariate. For annual plant cover assumptions of normality and homogeneity of variances were not met. Therefore, no ANCOVA was used.

The abundance analysis was followed by a comparison of plant community composition using a cluster analysis. We used frequency of species found in each site in spring, summer and fall of 1998 and 1999 as input variable. A cladogram for each season was graphed using the Ward's method of linkage and Euclidean distance measure.

A Bray-Curtis ordination analysis was used to determine the variables accounting for the most variation in plant species occurrence (McCune and Mefford, 1999). To decrease noise, all taxa that occurred less than five times were eliminated, creating a data set with 18 instead of 40 taxa. To relieve the zero-truncation problem, data were modified using Beal's Smoothing, which is also called the sociological favorability index and determines the probability that a particular species can be found in a given habitat (Beals 1984).

A list of non-indigenous vascular plant species was compiled. The average percent cover of *Cotula coronopifolia* in 1998 and 1999 was graphically displayed.

Diatom abundance

To determine the effects of season, treatment, and microhabitat on diatom abundance in 1998 an ANCOVA was used. Variables included season, treatment and microhabitat, as well as elevation as covariates. A second ANCOVA with elevation as covariate and year, treatment, and microhabitat as main effects was used to analyze the data of summer 1998 and 1999. Both data sets had to be square root transformed to meet the assumptions of normality and equal variances.

Invertebrate abundance and richness

Relative abundances were used to graphically display invertebrate abundances. They were calculated by using the number of specimens found per unit of interest (site per season or invertebrate taxa) and dividing it by the total number of invertebrates found within this study (84, 367). The total number of invertebrates was used as the denominator in all cases to allow easier comparison between different graphs.

An ANCOVA was used to determine the effects of season, treatment, and elevation on invertebrate abundance in 1998. As data did not meet assumptions of normality and equal variances, they were transformed using $\sin(x_i+1)$. In addition an ANCOVA was used to compare the abundance of invertebrates between summer 1998 and summer 1999, after data were transformed using tan(x_i).

Furthermore, to compare invertebrate community composition a cluster analysis on the frequency of species found in each site in summer of 1998 and summer 1999 was used. A cladogram was graphed using the Ward's method of linkage and the Euclidean distance measure.

A Bray-Curtis ordination was used to determine the variables accounting for most of the invertebrate species occurrence. To decrease noise, all taxa that occurred less than eight times were eliminated, creating a data set with 42 instead of 50 taxa. To relieve the zero-truncation problem, data were modified using Beals Smoothing.

The relative abundance of the ten most common invertebrate taxa were graphed. *Manayunkia aestuarina* are tube-forming polychaetes, which are important for early successional stage marshes. As they were among the top ten most common invertebrates, their average abundance was graphed.

Ceratopogonid and chironomid larvae and pupae are important prey items for juvenile fish. To determine their presence in the restored and control sites, their abundance in spring, summer, and fall 1998 and summer 1999 was graphically displayed. Furthermore, the average abundance of the non-indigenous species *Streblospio benedicti* in spring, summer, and fall 1998 and summer 1999 was graphically displayed.

Fish

Species richness was displayed for Kunz Marsh sites between November 1998 and April 1999, as well as for Winchester Creek between November 1998 and June 1999. In addition, the number of species found in the restored sites and the three reaches of Winchester Creek was graphed.

In order to determine the effect of elevation and season on fish species composition, a Bray-Curtis ordination was used. To decrease noise, all taxa that occurred less than 2 times were eliminated, creating a data set with 9 instead of 11 taxa. In addition, two sites were eliminated, the 2.2m and 1.8m restored site in April 1999, as no fish were found. To relieve the zero-truncation problem, data were modified using Beals Smoothing.

As salmonids are species of special concern, the number of salmonids as well as their mean fork length were graphically displayed.

Precipitation and temperature

Monthly mean, maximum and minimum temperatures as well as monthly total precipitation were graphed for 1998 and 1999.

RESULTS

Vascular plants

Inter-annual, seasonal, treatment, and elevation effects on vascular plants were determined. Annular vascular plants were found in both the 2.2m and the 1.8m restored sites throughout both years except in fall 1998 (Figure 5A). Overall there were more annuals found in 1998, but both years showed a seasonal distribution with peaks in the summer. Highest percent cover was found in the 1.1m control site in spring 1998 (Figure 5B).

In restored sites, perennial percent cover was highest at the 2.2m restored site in 1998 (Figure 6A). However, it decreased the following year at this site, but increased in the 1.8m and 1.5m restored sites. The 2.2m and 1.8m control sites were the only sites that did not show seasonal changes or differences between 1998 and 1999 (Figure 6B). ANCOVA results supported these trends. The covariate elevation and all main effects (year, season, and treatment) significantly contributed to the variation found in percent cover of perennial plants in 1998 and 1999 (Table 3). Cover was higher in 1998 and there was a distinct seasonality with higher cover in the summer than in spring and fall. There was also greater cover in control than restored sites.

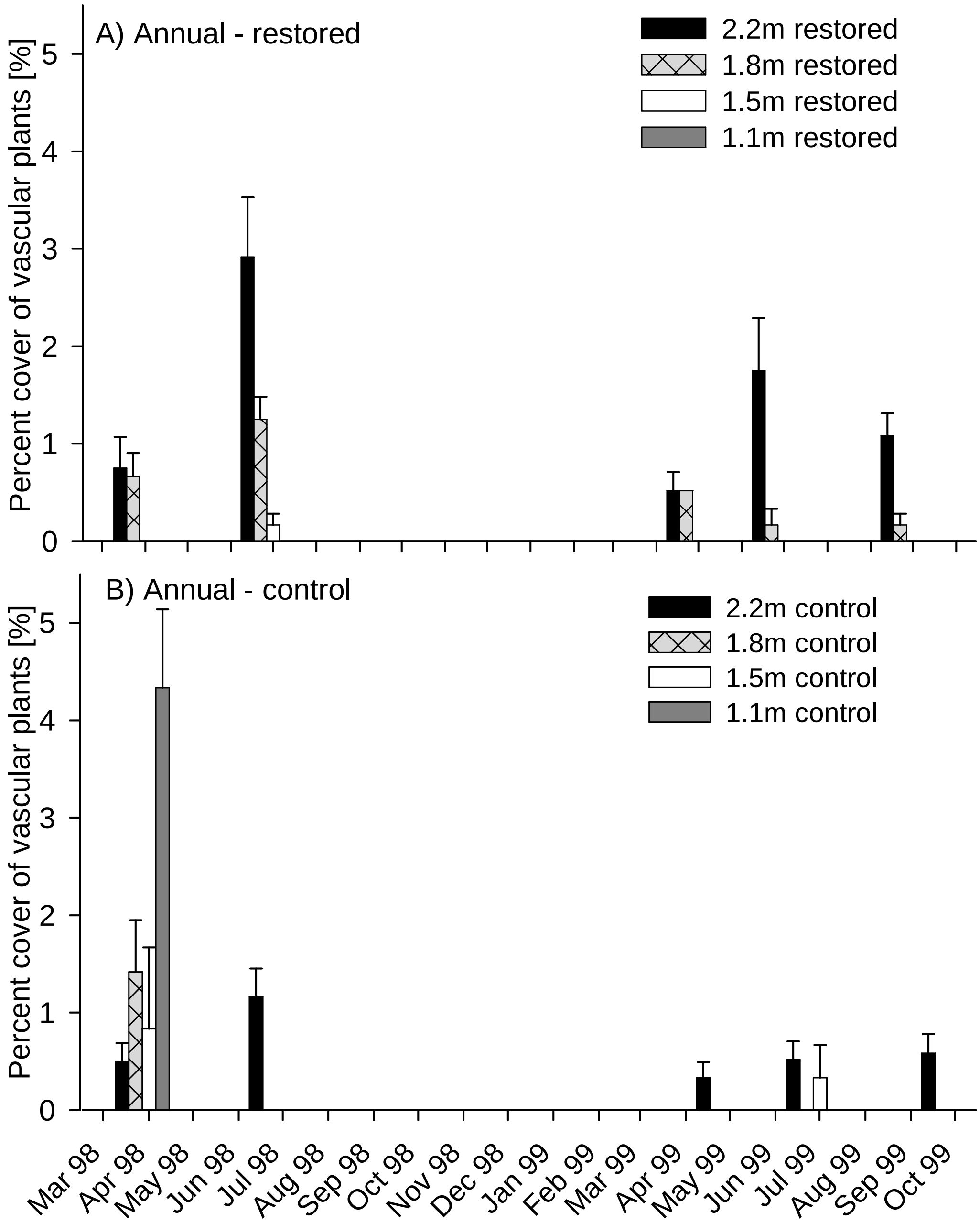

Figure 5. Annual vascular plant cover in A) restored Kunz Marsh and B) control sites in SSNERR, OR during spring, summer, and fall 1998 and 1999. 1.1m restored = passive site. Error bars are ± 1 Standard Error of the Mean.

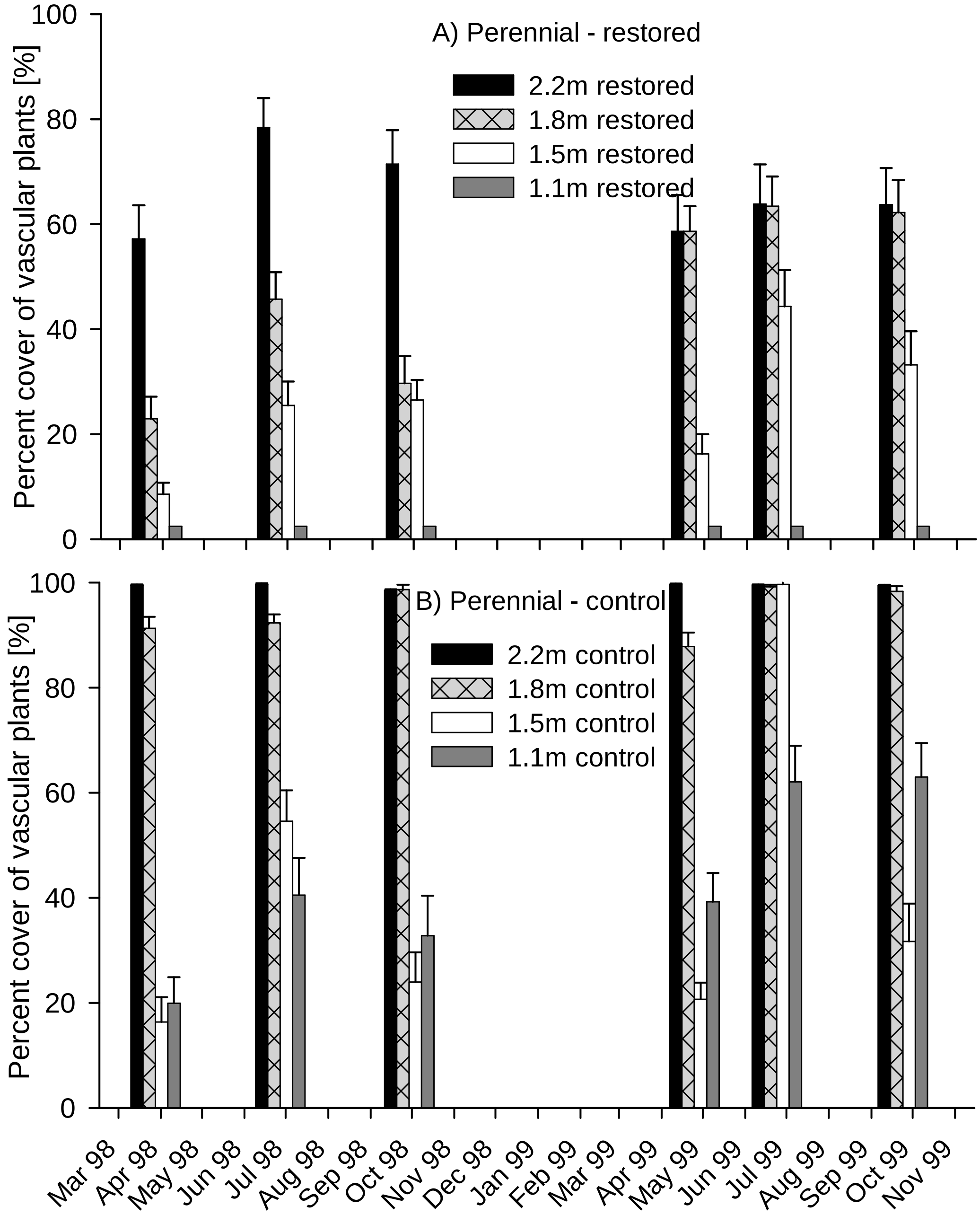

Figure 6. Perennial vascular plant cover in A) restored Kunz Marsh and B) control sites in SSNERR, OR in spring, summer, and fall 1998 and 1999. 1.1m restored = passive site. Error bars are ± 1 Standard Error of the Mean.

Table 3. Analysis of Covariance (ANCOVA) to determine the effects of year (1998, 1999), season (spring, summer, fall) and treatment (restored, control) on the percent cover of perennial vascular plant in restored sites of Kunz Marsh and control sites, SSNERR, OR. Elevation was the covariate. df = degrees of freedom, P-value = Probability value, * = significant at $\alpha = 0.05$

Source of Variation	**df**	**Mean-Square**	**F-ratio**	**P-value**
Year	1	25640.54	32.13	0.000*
Season	2	42140.67	52.81	0.000*
Treatment	1	452210.61	566.72	0.000*
Year x Season	2	1575.32	1.97	0.139
Year x Treatment	1	2376.79	2.98	0.085
Season x Treatment	2	2182.07	2.74	0.065
Year x Season x Treatment	2	1336.37	1.68	0.188
Elevation	1	803719.16	1007.24	0.000*
Error	1425	797.95		

In addition, comparing plant community compositions showed that higher and lower sites were different. The 1.1m restored, passive site and the low control sites did not cluster with any site from the opposite treatment (Figure7). However, the high restored site in both years had similar communities as the high control site in 1998. The mid restored marsh was similar to the high restored and control sites in 1998, but in 1999 it showed more similarity with the control site at the same elevation. Cladograms for the spring and fall samples exhibited similar patterns and are not presented here.

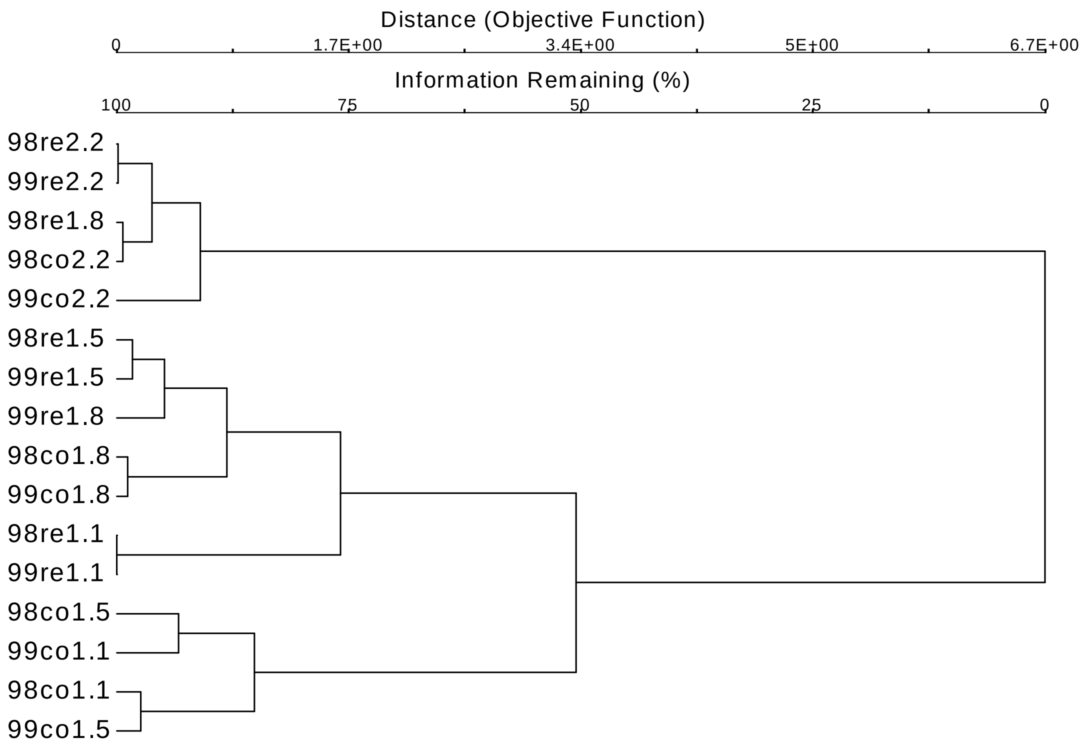

Figure 7. A comparison of vascular plant species composition between restored Kunz Marsh and control sites, SSNERR, OR, in the summer of 1998 and the summer of 1999. A cluster analysis was performed based upon the frequency of species occurrence in each site. Distance (Objective Function) = measure of information loss as agglomeration proceeds. When clusters are fused, information is lost therefore the remaining information is given in percent. Elevations in meters, treatment: re = restored sites, co = control.

Elevation and treatment explained most of the variation seen in vascular plant species distribution. Tidal elevation accounted for 68.32% of the variability (r^2 = 0.809), whereas treatment explained an additional 24.58% (r^2 = 0.178). Several plant taxa had a strong correlation with higher elevations (Table 4), and only a few, such as *Zostera japonica* were associated with low elevations. For most plant taxa, correlations within treatment were small. Exceptions were *Triglochin maritimum* and *Salicornia virginica*, which were highly associated with control sites, and *Cotula coronopifolia* was strongly associated with restored sites.

Table 4. Results from a Bray-Curtis ordination to determine the effects of year, season, elevation, and treatment on the distribution of vascular plant species during spring, summer, fall 1998 and summer 1999. The independent variable was the frequency of species occurrence in each restored Kunz Marsh and control sites, SSNERR, OR, during 1998 and 1999. Shown are 18 out of 40 species left after removing those with an occurrence in less than 5 sites. (r = Pearson correlation coefficient).

	Axis1 Elevation			Axis2 Treatment	
Taxa	**r**		**Taxa**	**r**	
Deschampsia caespitosa	- .975	High	*Cotula coronopifolia*	.822	Restored
Agrostis alba	-.949		*Eleocharis parvula*	.656	
Holcus lanatus	-.938		*Trifolium wormskjoldii*	-.007	
Trifolium wormskjoldii	-.928		*Agrostia alba*	-.076	
Atriplex patula	-.922		*Holcus lanatus*	-.166	
Potentilla pacifica	-.891		*Distichlis spicata*	-.168	
Distichlis spicata	-.832		*Deschampsia caespitosa*	-.170	
Grindelia integrifolia	-.808		*Juncus bufonius*	-.277	
Grass	-.777		*Zostera japonica*	-.295	
Carex lyngbyei	-.765		*Atriplex patula*	-.368	
Jaumea carnosa	-.469		*Potentilla pacifica*	-.378	
Alopecurus aequalis	-.227		*Grindelia integrifolia*	-.461	
Cotula coronopifolia	-.469		Grass	-.466	
Juncus bufonius	-.227		*Jaumea carnosa*	-.538	
Triglochin maritimum	.230		*Alopecurus aequalis*	-.588	
Eleocharis parvula	.273		*Carex lynbyei*	-.578	
Salicornia virginica	.511		*Salicornia virginica*	-.797	
Zostera japonica	.906	Low	*Triglochin maritimum*	-.871	Control

Overall, six non-indigenous species of vascular plants were found within the restored Kunz marsh and the control sites, SSNERR, OR (Table 5). The brass button (*Cotula coronopifolia*) was the most abundant exotic (Figure 8), which was present in all restored sites throughout both years. In spring 1998 and summer 1999, it was also found in the 1.1m control site. Its percent cover decreased in the high, mid, and low restored sites from 1998 to 1999, but increased in the control site.

Table 5. Non-indigenous vascular plant species found in the restored sites of Kunz Marsh and control sites, SSNERR, OR during 1998 and 1999.

Species	**Description**	**Sites found**
Agrostis alba	Perennial turf grass from Europe	High and mid restored, high control sites
Cotula coronopifolia	Perennial, somewhat succulent herb from South America	All restored sites and 1.1m control site
Holcus lanatus	Perennial, velvety grass from Europe	High and mid restored, high control
Lolium perenne	Short-lived perennial, tufted grass from Europe	High restored site
Trifolium repens	Perennial herb from Europe	High restored site
Zostera japonica	Annual or short lived perennial	Low restored and control site

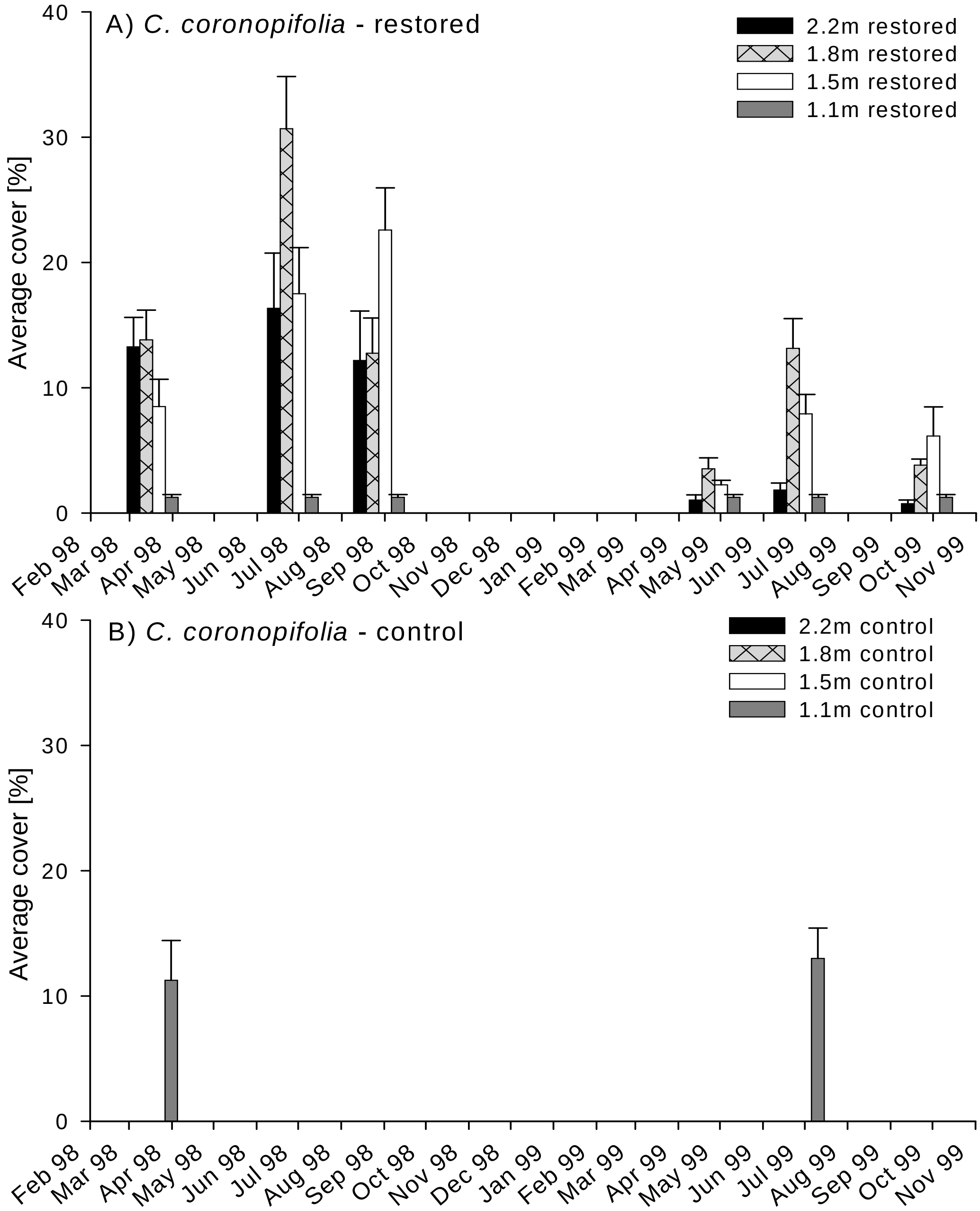

Figure 8. Percent cover of the exotic vascular plant *Cotula coronopifolia* within Kunz Marsh and control sites, SSNERR, OR during spring, summer, and fall, 1998 and 1999. 1.1m restored = passive site. Error bars are ± 1 Standard Error of the Mean.

Benthic diatoms

The highest diatom densities occurred during the summer of 1998 in the 1.8m and 1.1m control sites, and year was a significant effect in the ANCOVA (Figure 9, Table 6). The treatment effect was also significant (Table 6) and control sites showed greater seasonality than restored sites (Figure 9). Although densities were at times higher in the vegetated than open areas, microhabitats, as an effect, were not significant (Table 6).

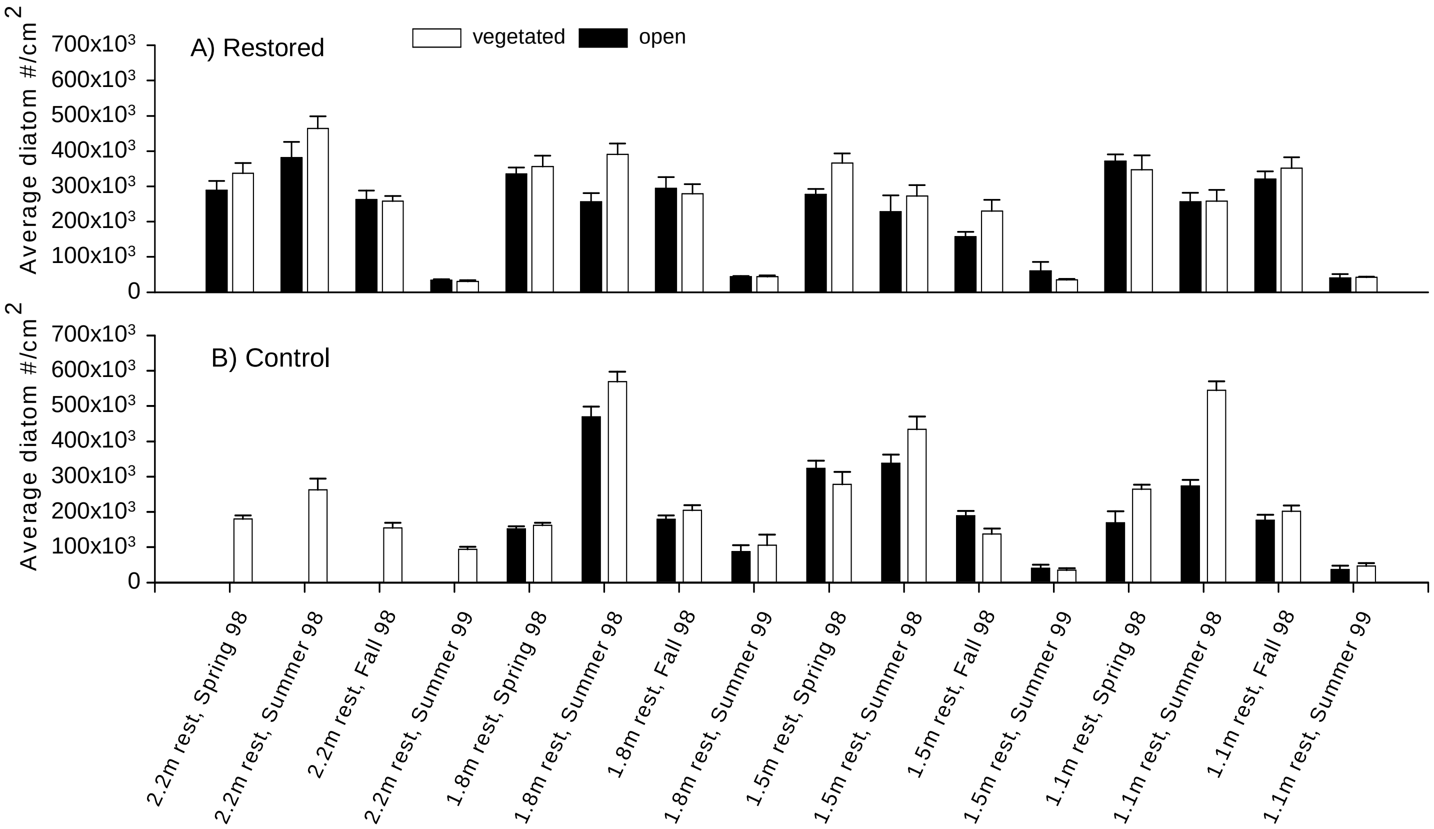

Figure 9. Diatom abundance in A) restored Kunz Marsh and B) control sites in SSNERR, OR in spring (Sp), summer (Sum), and fall 1998 and summer 1999. 1.1m restored = passive site . Error bars are ± 1 Standard Error of the Mean.

Table 6. Analysis of Covariance (ANCOVA) results to determine the effects of Year (summer 1998, summer 1999), season (spring, summer, fall) and treatment (restored, control) on abundance of diatoms found in restored sites of Kunz Marsh and control sites, SSNERR, OR. Elevation was the covariate. A) spring, summer and fall 1998 and B) summer 1998 and 1999. To meet assumptions of normality and equal variances data were transformed using square root. df = degrees of freedom, p-value = probability value, * = significant with α = 0.05

A) Source of Variation	**df**	**Mean-Square**	**F-ratio**	**P-value**
Season	2	624385.64	55.98	0.000*
Treatment	1	238260.35	21.36	0.000*
Microhabitat	1	103632.44	9.29	0.002*
Season x Treatment	2	533663.33	47.85	0.000*
Season x Microhabitat	2	39889.45	3.58	0.029*
Treatment x Microhabitat	1	4674.52	0.42	0.518
Season x Treatment x Microhabitat	2	9537.24	0.86	0.426
Elevation	1	233.28	0.02	0.889
Error	525	11152.88		

B) Source of Variation	**df**	**Mean-Square**	**F-ratio**	**P-value**
Year	1	2888755.00	185.63	0.000*
Treatment	1	65075.71	4.18	0.042*
Microhabitat	1	19583.42	1.26	0.263
Year X Treatment	1	38841.26	2.50	0.116
Year X Microhabitat	1	18513.25	1.19	0.277
Treatment X Microhabitat	1	1924.59	0.12	0.725
Year X Treatment X Microhabitat	1	807.94	0.05	0.820
Elevation	1	208026.20	13.37	0.000*
Error	210	15561.61		

Invertebrates

Invertebrate abundance was influenced by year, treatment, microhabitat, and by elevation. Within restored sites, vegetated areas had a slightly higher abundance of invertebrates than open areas (Figure 10, Table 7A). Within open microhabitats, most invertebrates were found at the 1.1m restored site, except in fall 1998 when highest abundances occurred in the 1.5m restored site. In vegetated microhabitats, highest abundances were found in the 1.5m restored site except in July 1999.

Control sites had significantly higher invertebrate abundances than restored sites (Figure 11, Table 7A). Highest abundances occurred in open microhabitats of the 1.8m control site, which showed a steady increase of invertebrates from April through September 1998. In summer 1999 lower numbers were found than in summer 1998, and highest numbers were at the 1.5m restored site. The highest invertebrate abundances were found in the vegetated areas of control sites (Figure 11). The 2.2m restored site had the highest abundances with a distinct seasonal distribution of increasing numbers in the summer and a decrease in the fall. Again, overall invertebrate abundance was lower in summer 1999 than summer 1998.

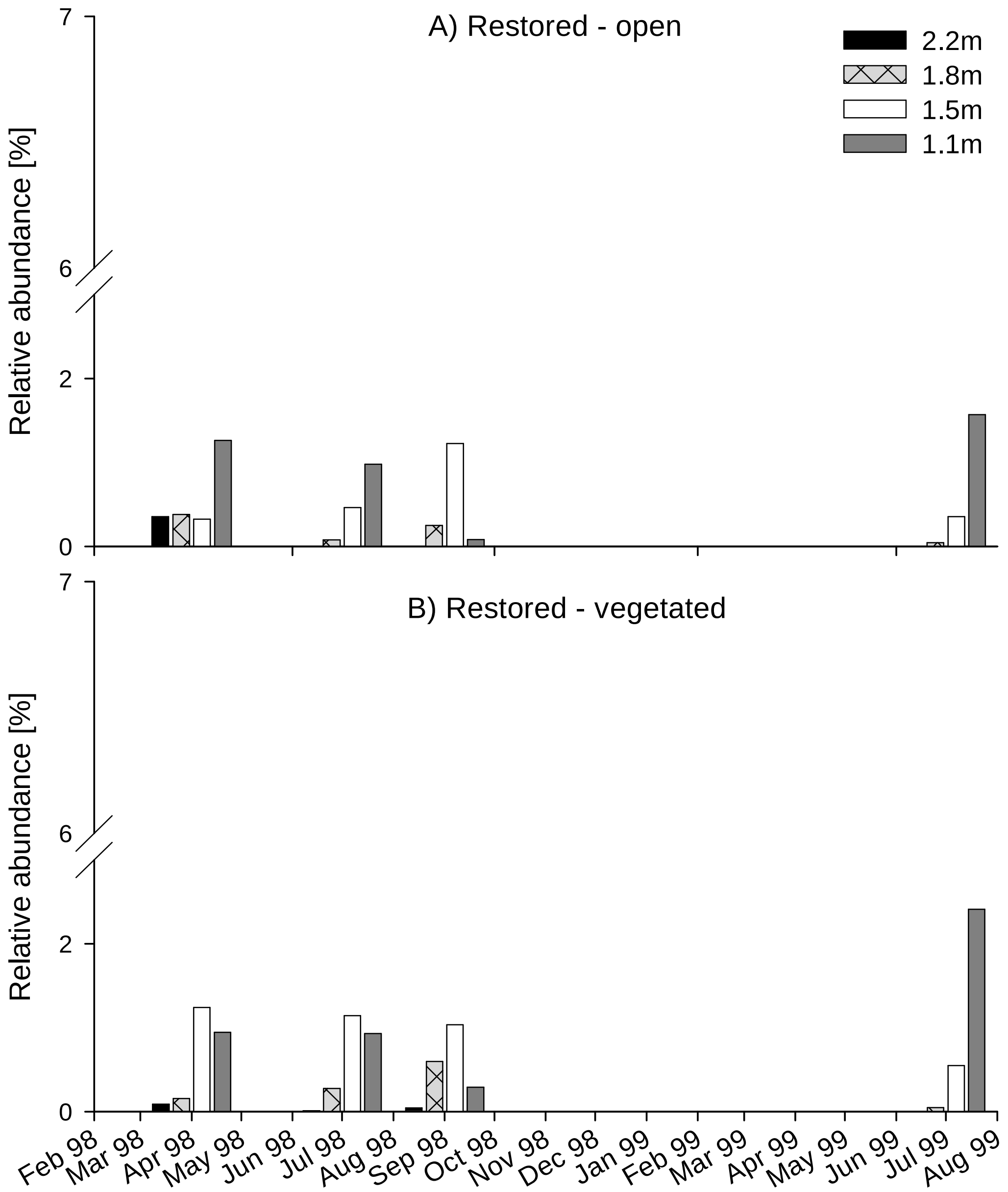

Figure 10. Relative abundances of all invertebrates found in A) open and B) vegetated microhabitats in restored Kunz Marsh sites, SSNERR, OR, during spring, summer, and fall 1998 and summer 1999. 1.1m restored = passive site. Relative abundance = # individuals found per site and time/ total # invertebrates found in entire study (84, 367).

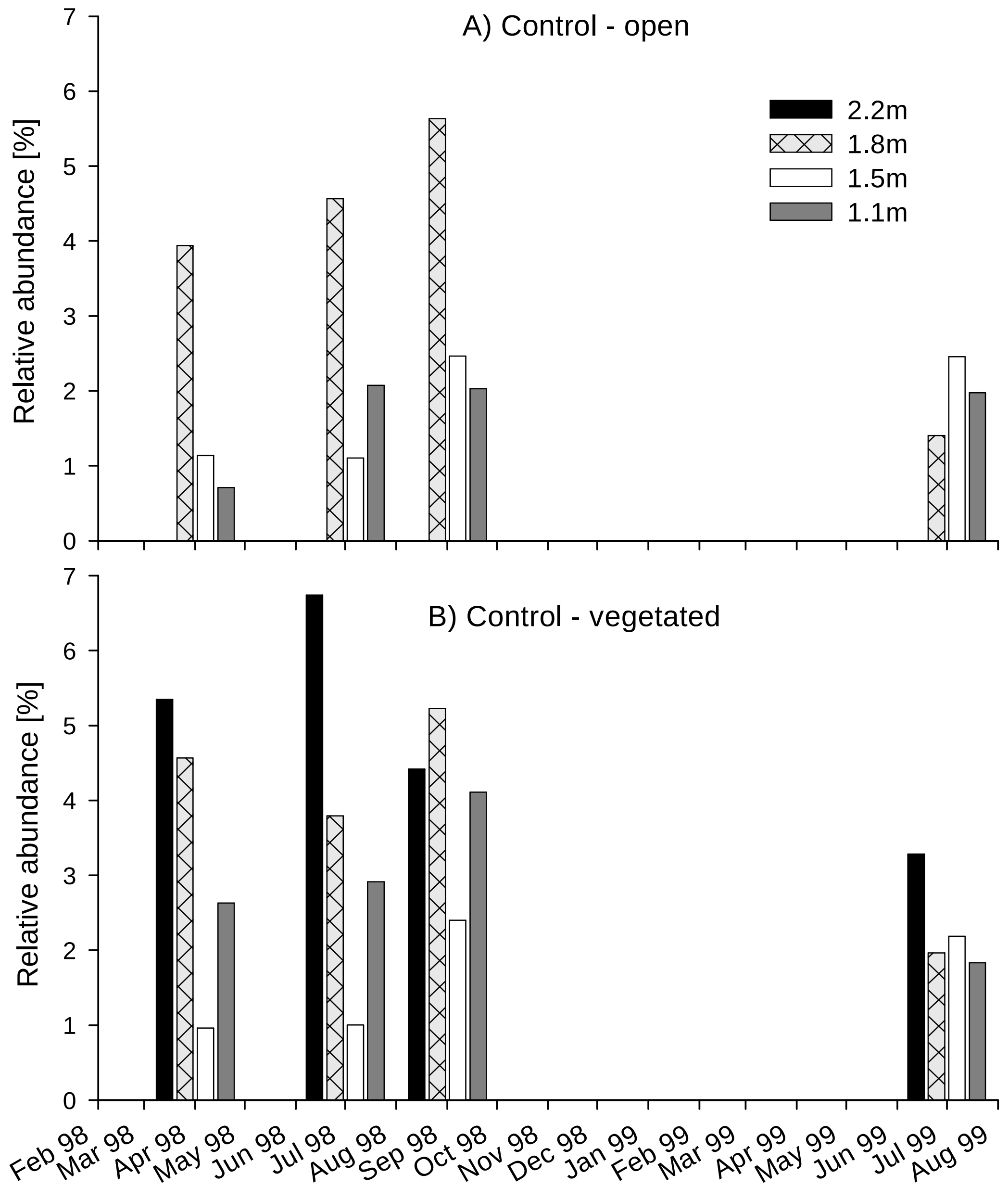

Figure 11. Relative abundance of all invertebrates found in A) open and B) vegetated microhabitats in control sites, SSNERR, OR, during spring, summer, and fall 1998 and summer 1999. Relative abundance = # individuals found per site and time/ total # invertebrates found in entire study (84, 367).

Table 7. Analysis of Covariance (ANCOVA) results to determine the effects of year (summer 1998, summer 1999), season (spring, summer, fall), and treatment (restored, open) on total invertebrate abundance found in restored Kunz Marsh and control sites, SSNERR, OR. Elevation was the covariate. A) spring, summer and fall 1998 and B) summer 1998 and 1999. As assumptions of normality and equal variances were not met, data were transformed using A) sinus (x_i+1) and B) tan(x_i). df = degrees of freedom, P-value= probability value, * = significant at α = 0.05.

A) Source of Variation	**df**	**Mean-Square**	**F-ratio**	**P-value**
Season	2	50.51	0.45	0.636
Treatment	1	22252.43	199.77	0.000*
Microhabitat	1	776.03	6.97	0.010*
Season x Treatment	2	329.48	2.96	0.055
Season x Microhabitat	2	0.91	0.01	0.992
Treatment x Microhabitat	1	147.55	1.32	0.251
Season x Treatment x Microhabitat	2	128.23	1.15	0.319
Elevation	1	870.81	7.82	0.006*
Error	179			
B) Source of Variation	**df**	**Mean-Square**	**F-ratio**	**P-value**
Year	1	78.75	1.55	0.216
Treatment	1	31.63	0.62	0.432
Microhabitat	1	2.02	0.38	0.541
Year x Treatment	1	19.08	0.04	0.842
Year x Microhabitat	1	252.31	4.97	0.028*
Treatment x Microhabitat	1	40.71	0.80	0.372
Year x Treatment x Microhabitat	1	51.59	1.02	0.316
Elevation	1	9.33	0.18	0.669
Error	111	50.77		

Of the ten most common invertebrate species, most occurred in the control sites (Figure 12). Three taxa, foraminifera, acari and *Streblospio benedicti*, were almost completely absent from the restored sites. Higher abundances were found in vegetated microhabitats for foraminifera, the polychaete *Manayunkia aesturarina*, oligochaetes, and acari, whereas abundances for the remaining taxa were higher in the open microhabitat.

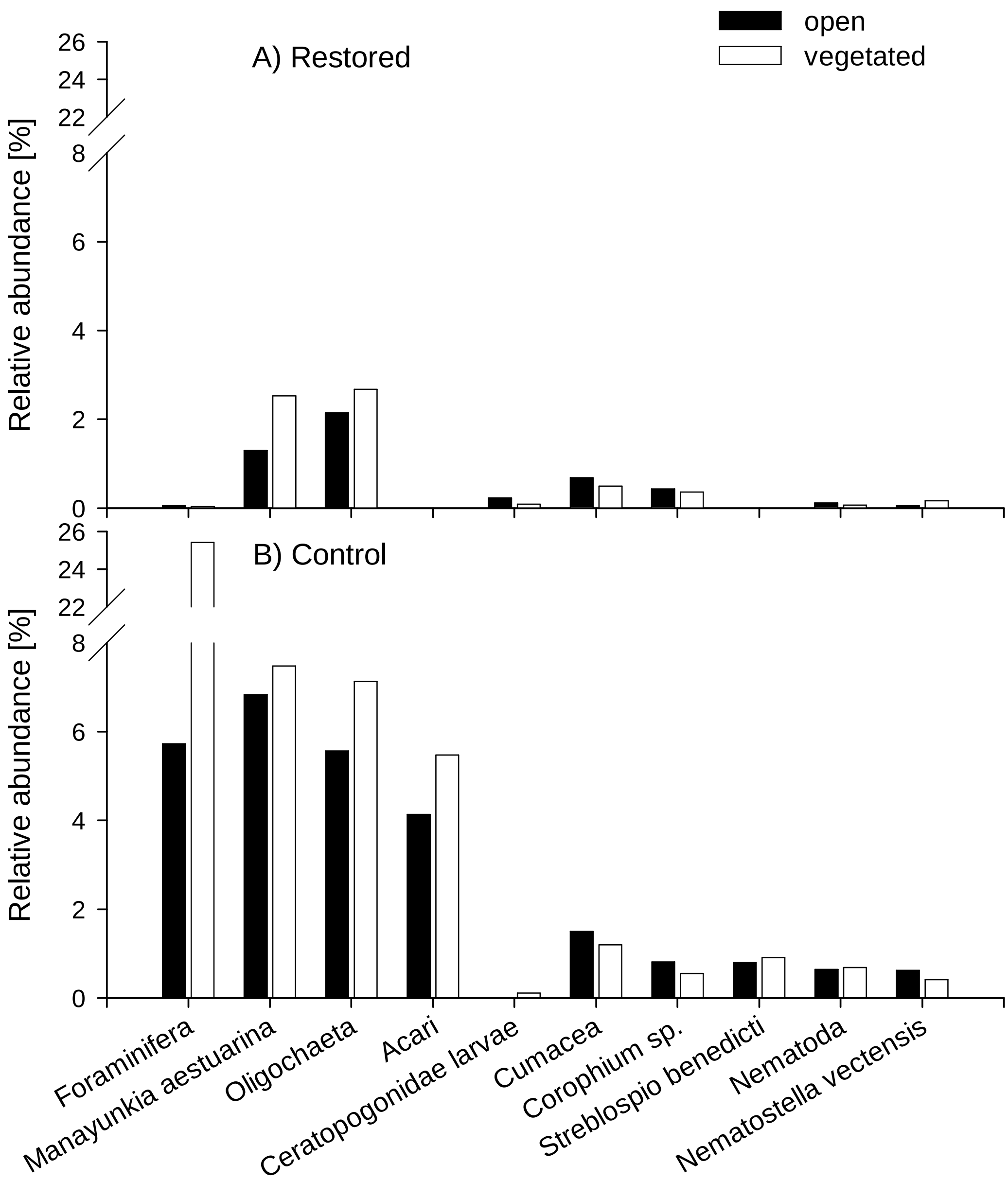

Figure 12. Relative abundance of the ten most abundant invertebrate taxa found in open and vegetated microhabitats in A) restored Kunz Marsh and B) control sites, SSNERR, OR during spring, summer, and fall 1998 and summer 1999. Relative abundance = # individuals per taxa found in entire study/ total # invertebrates found in entire study (84, 367).

Based upon their composition and abundance invertebrate communities from high restored and control sites formed a large cluster and then split up according to microhabitat (Figure 13). Another large invertebrate community located in the middle of the cladogram is comprised of mostly low and mid elevation restored sites, and some low and mid elevation control sites, most of which are vegetated. The last major invertebrate community, located at the bottom of the cladogram, is comprised entirely of animals found in the mid elevation control sites (Figure 13).

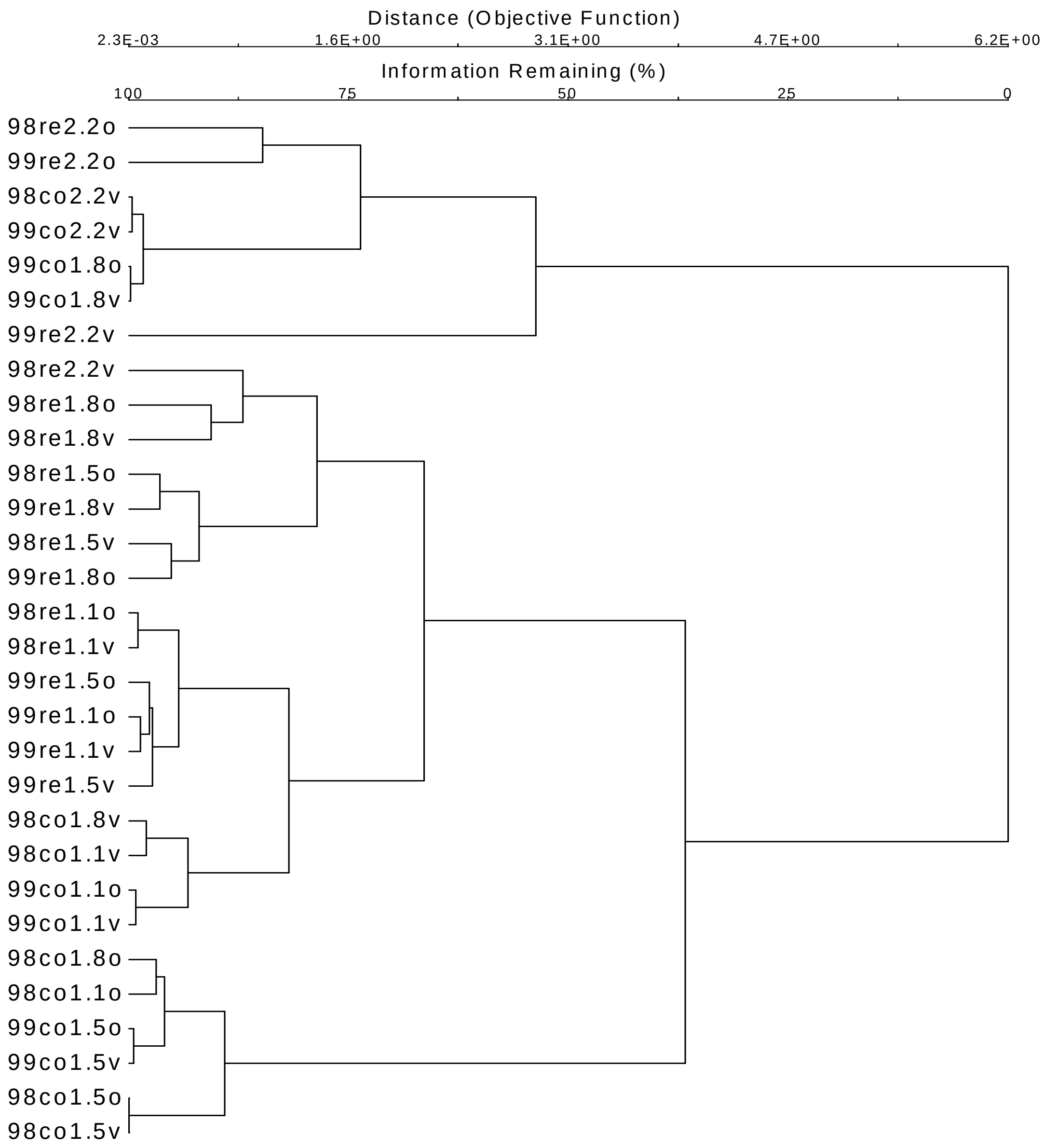

Figure 13. A comparison of invertebrate taxa composition between restored and control sites in the summer of 1998 (98) and 1999 (99). A cluster analysis was performed based upon invertebrate abundances in restored Kunz Marsh and control sites, SSNERR, OR. Distance (Objective Function) = measure of information loss as agglomeration proceeds. When clusters are fused, information is lost therefore the remaining information is given in percent. 1.1m restored = passive. re = restored, co = control, elevations given in meters.

Elevation explained 61.49% (r^2 = 0.854) of the variation observed in the distribution and abundance of invertebrate taxa. Treatment and microhabitat added an additional 18.91% (r^2 = 0.109) and 7.93% (r^2 = 0.014), respectively. Several taxa were associated with low elevations such as several crustacea, the hydroid *Nematostella vectensis*, and the dipteran Ceratopogonidae larvae (Table 8). Furthermore, most taxa were strongly correlated with control sites including four annelid taxa. The only taxa associated with restored sites were dipteran Dolichopodidae larvae. Dipteran Sciomyzidae larvae were associated with open microhabitat, but most taxa, including *Corophium spp.* and Ceratopogonidae larvae, were strongly correlated with vegetated areas.

Table 8. Results from a Bray-Curtis ordination to determine the effects of year, season, treatment, microhabitat, and elevation on the distribution of invertebrate taxa found in restored Kunz Marsh and control sites, SSNERR, OR, during spring, summer, fall 1998 and summer 1999. Forty-two out of 60 taxa were left after removing those with an occurrence in less than 8 sites. Shown are r > |0.700|. Elevation: + r = high, treatment: + r = restored, microhabitat: + r = vegetated. (r = Pearson correlation coefficient)

Axis1 Elevation		Axis2 Treatment		Axis3 Microhabitat	
Taxa	**r**	**Taxa**	**r**	**Taxa**	**r**
SpeciesF	.903	Dolichopodidae	.727	*Corophium spp*	.901
spider	.886	*Pygospio elegans*	-.726	*E. confervicolus**	.883
Acari2	.841	Polychaeta	-.734	*Neanthes limnicola*	.871
Gastropoda	.813	Nematoda	-.816	*Nematostella vectensis*	.859
Sciomyzidae	.801	Nemertea	-.831	*Manayunkia aestuarina*	.857
Unknown Insect larvae	.786	Gammerid E	-.839	*Hobsonia florida*	.823
*G. insulare***	-.716			Ceratopogonidae larva	.809
Leptochelia dubia	-.749			*Sinelobus stanfordii*	.799
Streblospio benedicti	-.768			Cumacea	.785
Manayunkia aestuarina	-.818			Annelid B	.753
Hobsonia florida	-.877			*Amphitoe lacertosa*	.738
Sinelobus stanfordii	-.897			Crustacea D	.732
Isoptera	-.908			Isoptera	-.702
E. confervicolus	-.909			Coleoptera larvae	-.757
Ceratopogonidae larvae	-.939			spider	-.837
Neanthes limnicola	-.943			Sciomyzidae	-.845
Corophium spp	-.957				
Nematostella vectensis	-.974			* *E.=Eogammarus*	
Cumacea	-.987			**G.=Gnorimosphaeroma	

Average abundances of the tube-forming polychaete *Manayunkia aestuarina* are, in general, greater in open than vegetated areas except at the 1.5m restored site (Figure 14). Highest numbers occurred in the 1.8m control site, and overall abundance decreased from 1998 to 1999.

Ceratopogonidae larvae and pupae, which are important prey for juvenile salmonid fish, were spatially and temporally variable. Ceratopogonidae larvae increased in numbers between 1998 and 1999, with highest abundances found at the 1.1m restored, passive site in July 1999 (Figures 16). Overall there were more larvae and pupae in vegetated areas (Figures 16 and 17). In contrast, Chironomidae larvae were only conspicuous in the high, restored marshes during spring 1998 (Figure 17), and pupae were not found or were in low abundance (<5 individuals) in spring and summer 1998 (data not presented).

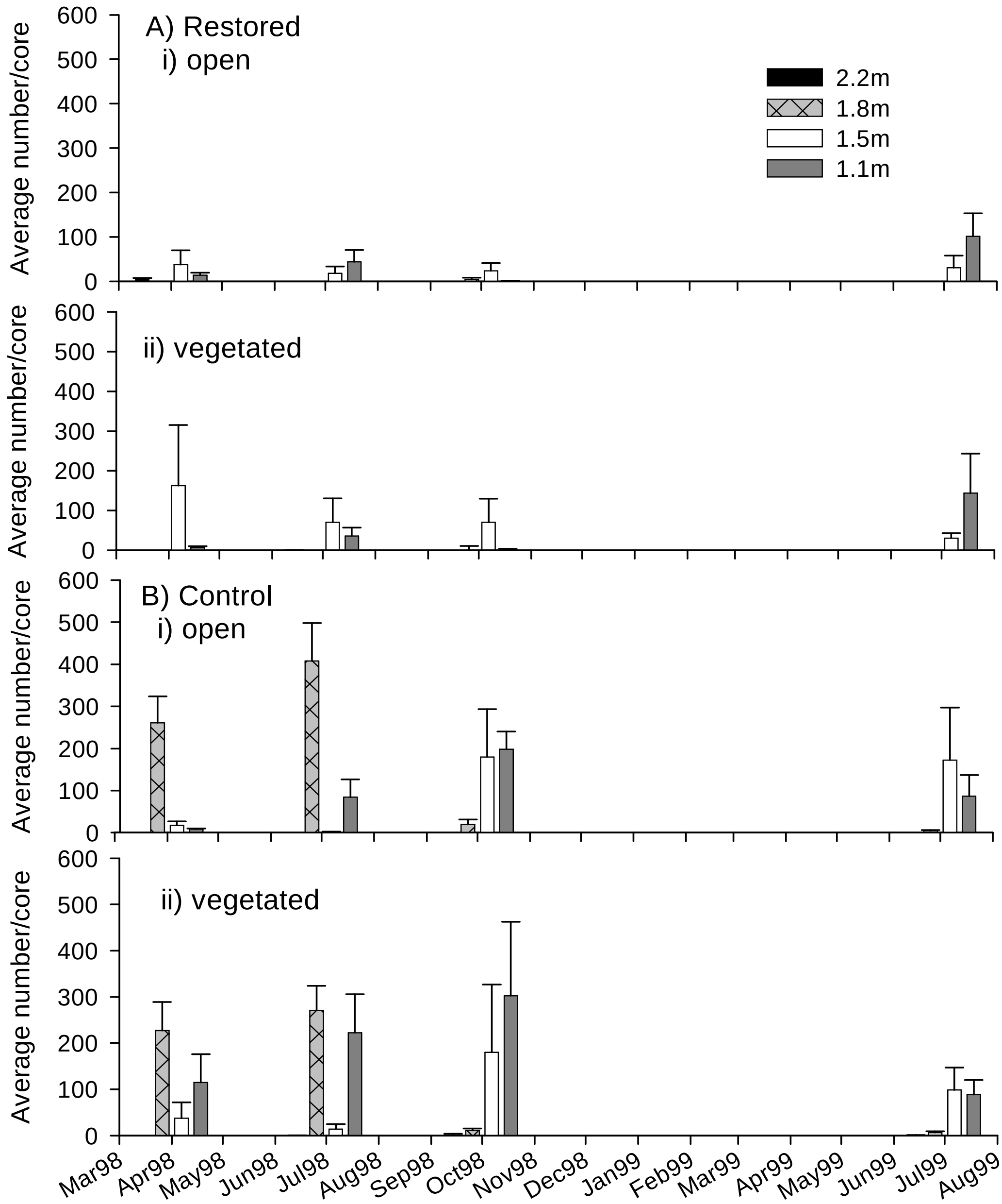

Figure 14. Average abundance of the tube-forming polychaete *Manayunkia aestuarina* in sediment cores taken from A) open and B) vegetated microhabitats in restored Kunz Marsh and control sites in SSNERR, OR, during spring, summer, fall 1998 and summer 1999. Error bars are ± 1 Standard Error of the Sample Mean.

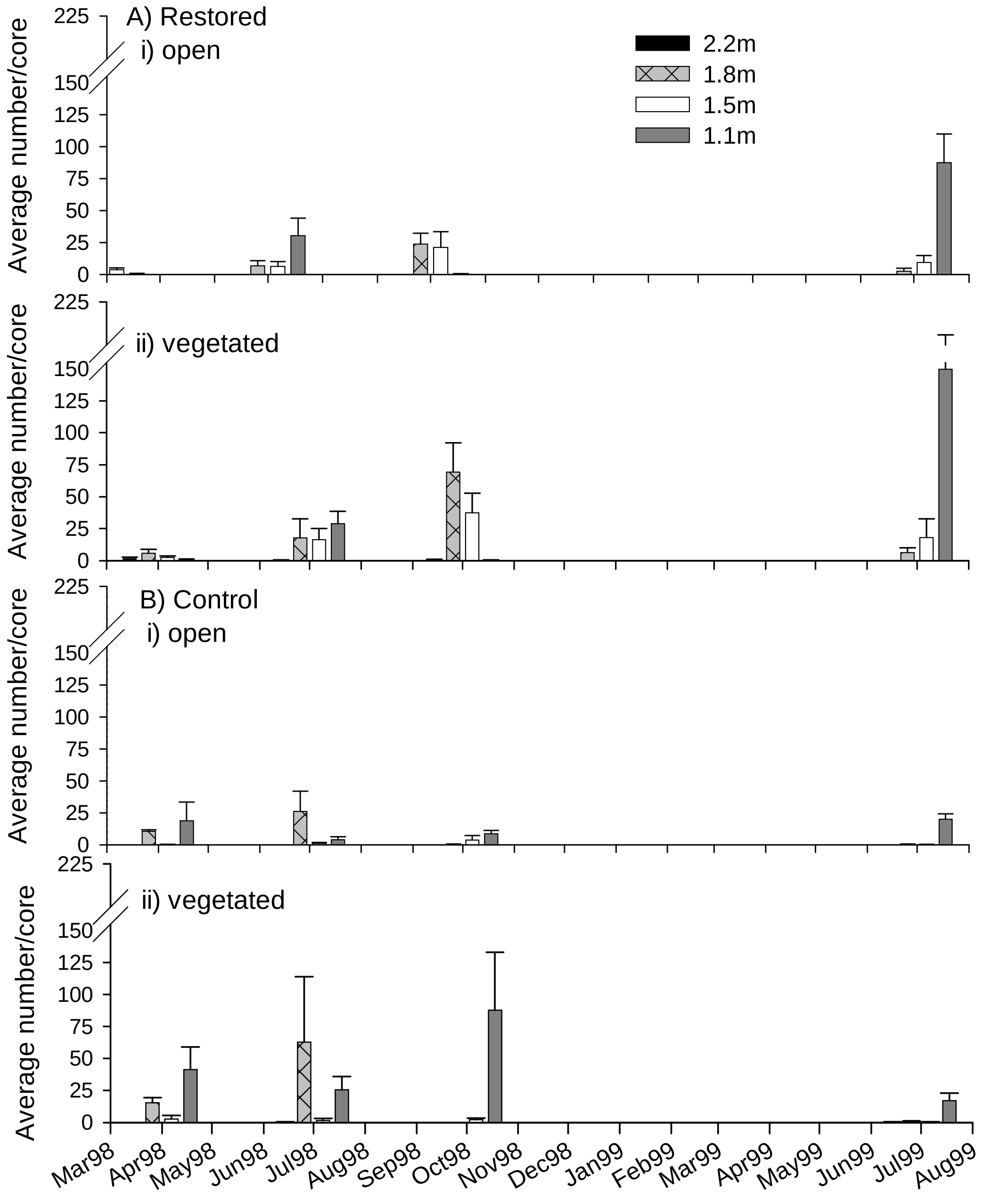

Figure 15. Average abundance of dipteran family Ceratopogonidae larvae in i) open and ii) vegetated microhabitats in A) restored Kunz Marsh and B) control sites in SSNERR, OR, during spring, summer, and fall 1998 and summer 1999. 1.1m restored = passive site. Error bars are ± 1 Standard Error of the Sample Mean.

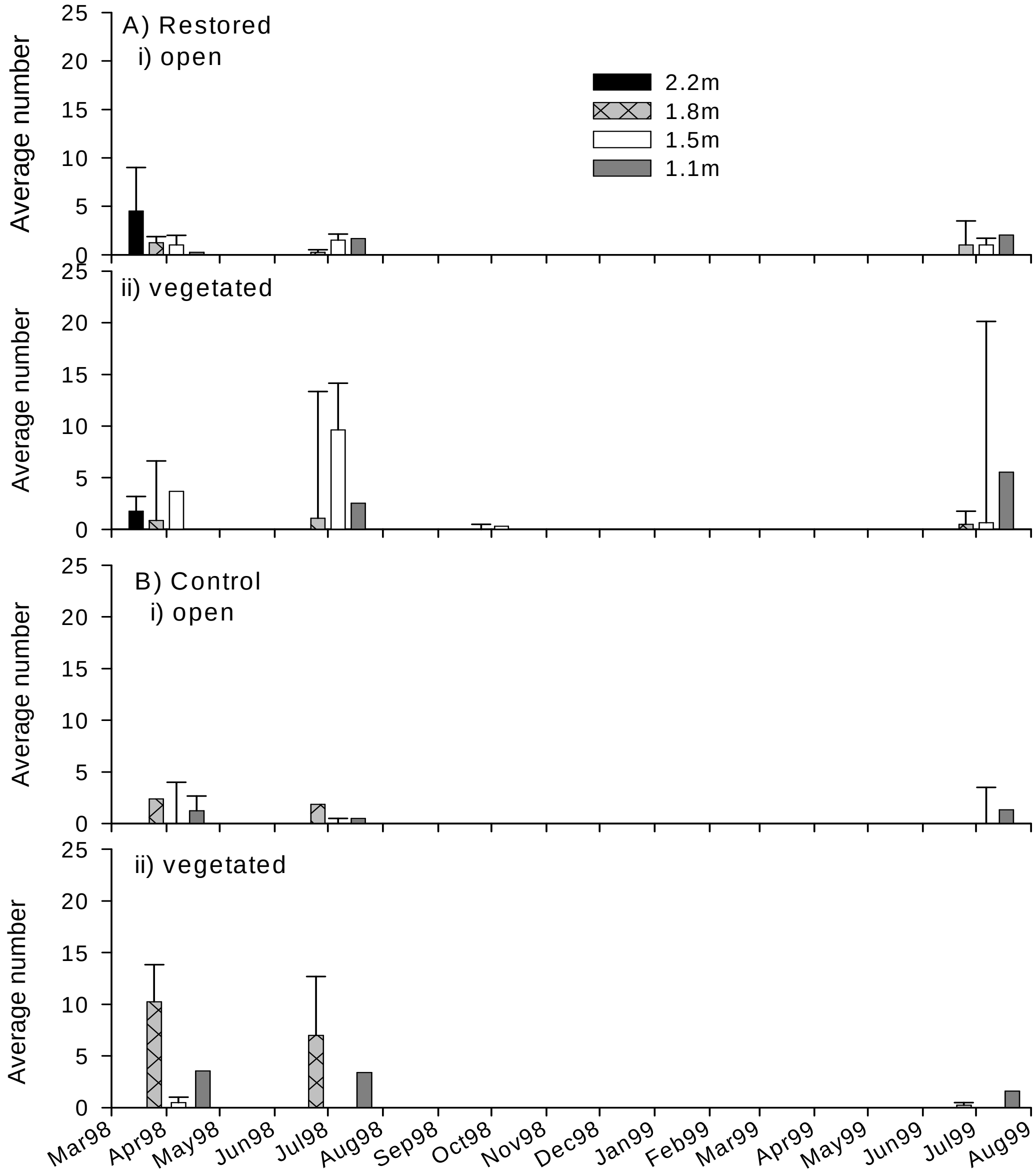

Figure 16. Average abundance of dipteran family Ceratopogonidae pupae in i) open and ii) vegetated microhabitats in A) restored Kunz Marsh and B) control sites in SSNERR, OR, during spring, summer, and fall 1998 and summer 1999. 1.1m restored = passive site. Error bar are ± 1 Standard Error of the Sample Mean.

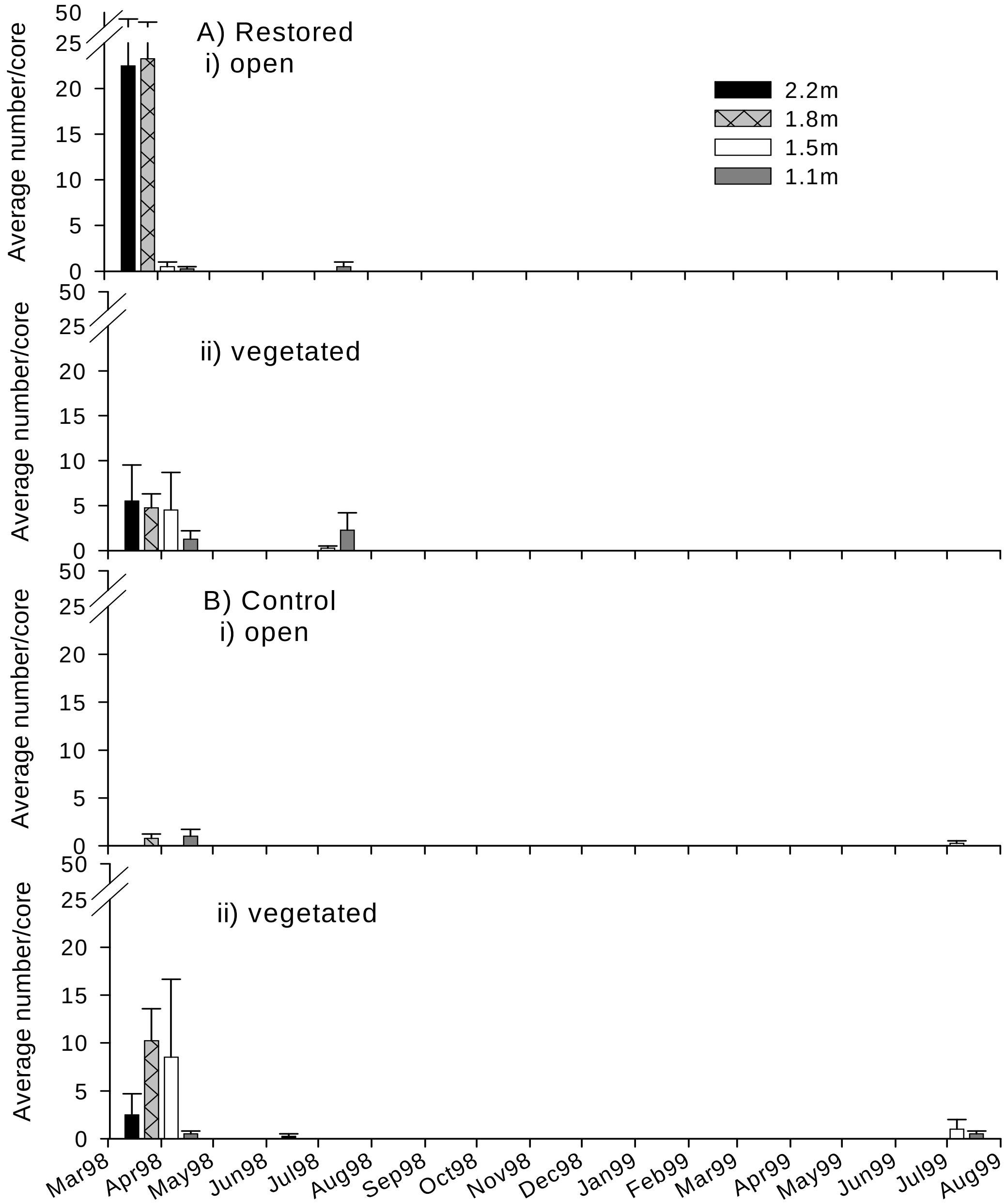

Figure 17. Average abundance of dipteran family Chironomidae larvae in i) open and ii) vegetated microhabitats in A) restored Kunz Marsh and B) control sites in SSNERR, OR, during spring, summer, and fall 1998 and summer 1999. 1.1m restored = passive site. Error bar are ± 1 Standard Error of the Sample Mean.

Of all the invertebrate taxa found, only two species, *Pseudopolydora kempi* and *Streblospio benedicti* (Polychaeta), were identified as non-indigenous. *Pseudopolydor kempi* was rare and *S. benedicti*, a taxon potentially important during early succession, occurred most frequently in control sites (Figure 18). *Streblospio benedicti* appeared in open and vegetated patches at the same time, with peak abundance occurring in the 1.5m control site during fall 1998, with slightly lower numbers in summer 1999.

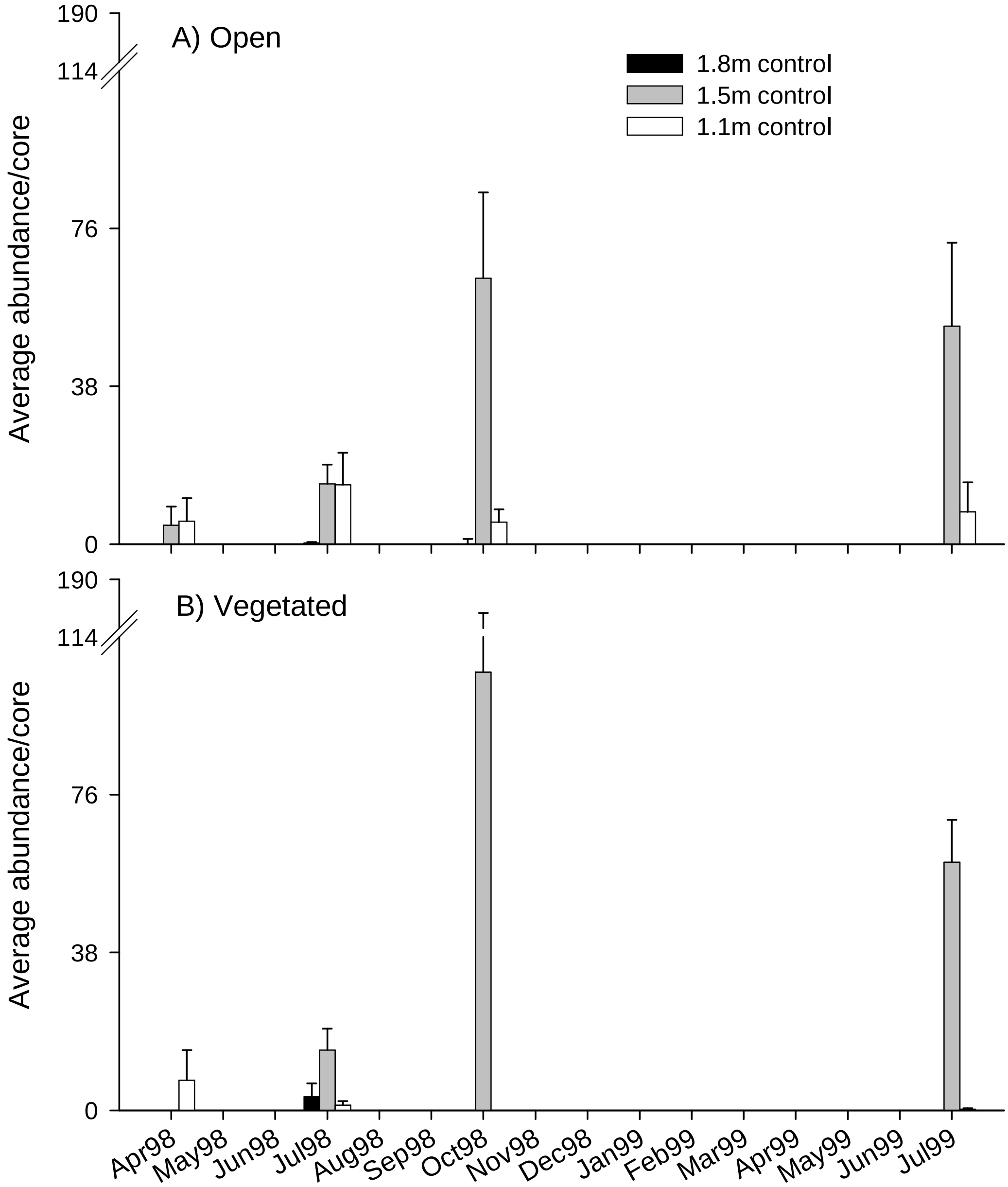

Figure 18. Average abundance of the non-indigenous, tube-forming polychaete *Streblospio benedicti* in sediment cores from control sites, SSNERR, OR, during spring, summer and fall 1998 and summer 1999. Error bars are ± 1 Standard Error of the Mean.

Fish

The passive 1.1m restored site showed the highest fish species richness of the Kunz Marsh restored areas (Figure 19). Richness in Winchester creek and the restored Kunz Marsh sites showed a general increase over the sampling period with seasonal peaks in December 1998 and April 1999. However, these peaks were more pronounced in the restored Kunz Marsh sites. There was a distinct increase in fish richness with decreasing elevation in restored sites of Kunz Marsh (Figure 20). Species and their abundances per site per month are shown in Table14 and in Appendix B.

An ordination, based upon the abundance of each fish species, revealed that elevation explained 84.26 % (r^2 = 0.901) of the variation in their distribution. As a variable, the month of sampling contributed another 12.22% (r^2 = 0.091). Most species were associated with low elevations and were seasonal (Table 9). However, topsmelts were slightly correlated with higher elevations and do not show seasonality.

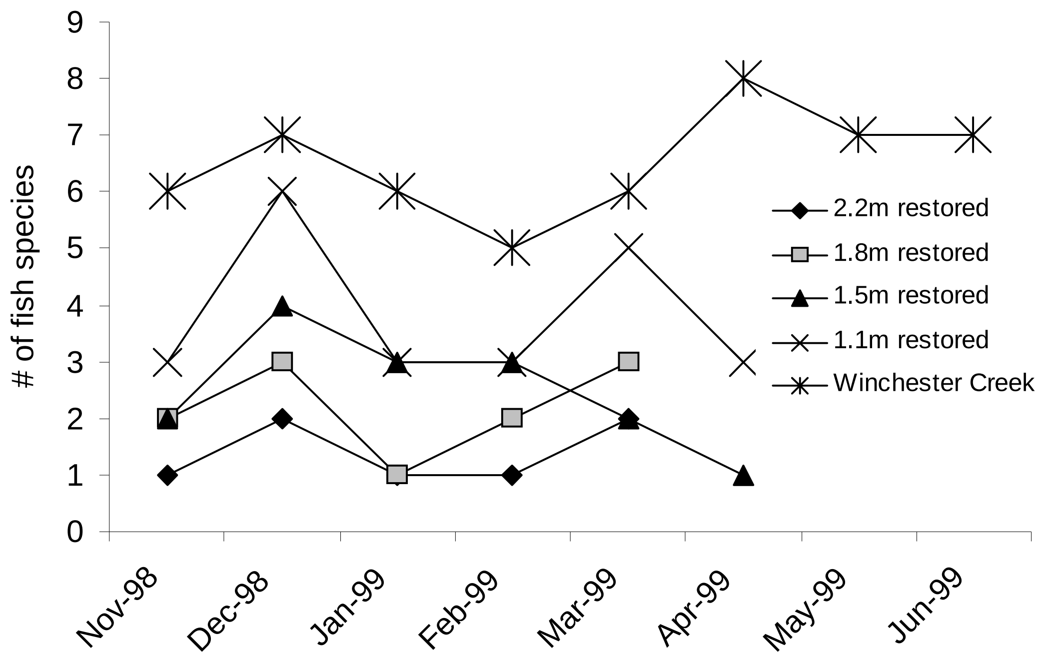

Figure 19. Number of fish species found in restored Kunz Marsh sites, SSNERR, OR between November 1998 and April 1999 and in Winchester Creek, SSNERR, OR between November 1998 and June 1999. 1.1m restored = passive site.

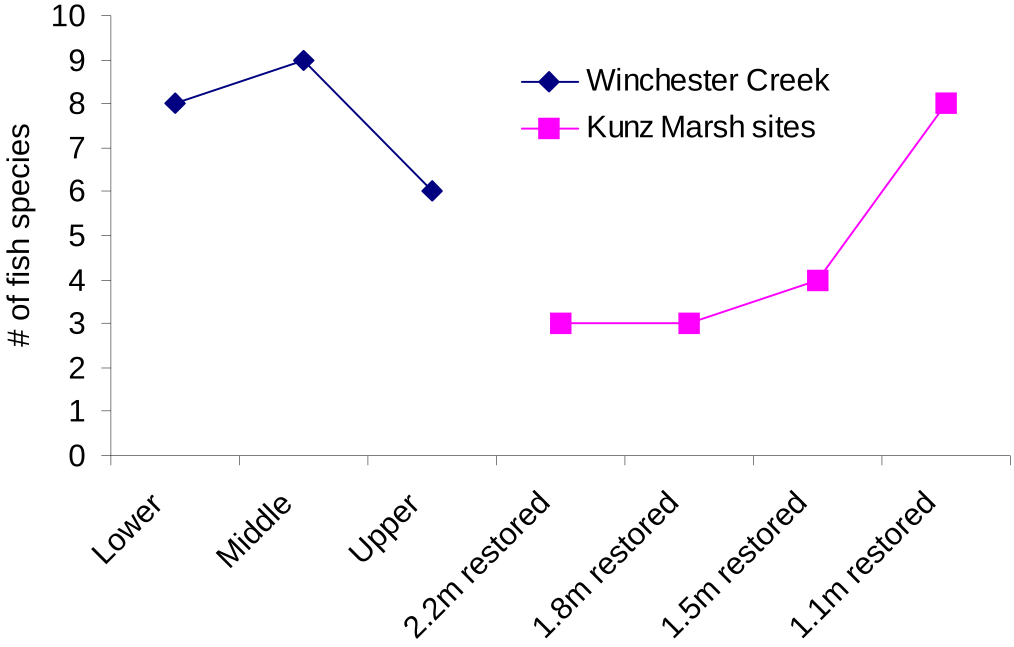

Figure 20. Total number of fish species found in restored Kunz Marsh sites and three reaches of Winchester Creek (see Figure 4), SSNERR, OR between November 1998 and April 1999 and June 1999, respectively. The middle reach was located along the Kunz Marsh sites.

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Table 9. Results from a Bray-Curtis ordination to determine the effects of elevation and month on the distribution of fish species in restored Kunz Marsh sites and Winchester Creek, between November 1998 and June 1999. Shown are nine out of eleven species left after removing those with an occurrence in less than 2. (r = Pearson correlation coefficient.)

Axis 1 Elevation			Axis 2 Month	
Taxa	**r**		**Taxa**	**r**
Prickly sculpin	-.984	Low	Cutthroat trout	-.935
Cutthroat trout	-.962		Starry flounder	-.935
Starry flounder	-.951		Prickly sculpin	-.911
Coho salmon	-.922		Pacific Herring	-.834
Threespine stickleback	-.915		Threespine stickleback	-.814
Pacific Herring	-.716		Coho salmon	-.763
Northern Anchovy	-.027		Northern Anchovy	.258
Topsmelt	.674	High	Topsmelt	.511

<u>Salmonids</u>

As salmonids are threatened or even endangered, special attention has been given to salmonids found within restored Kunz Marsh sites and Winchester Creek. The highest numbers of *Onchorynchus kisutch* were found in January, April and May 1999 (Figure 21). Overall, more *Salmo clarki* were present, which had a peak abundance in December 1998/January 1999 and in April/ May 1999. No salmonids were found in the sampled restored Kunz Marsh sites, including the passive restored site.

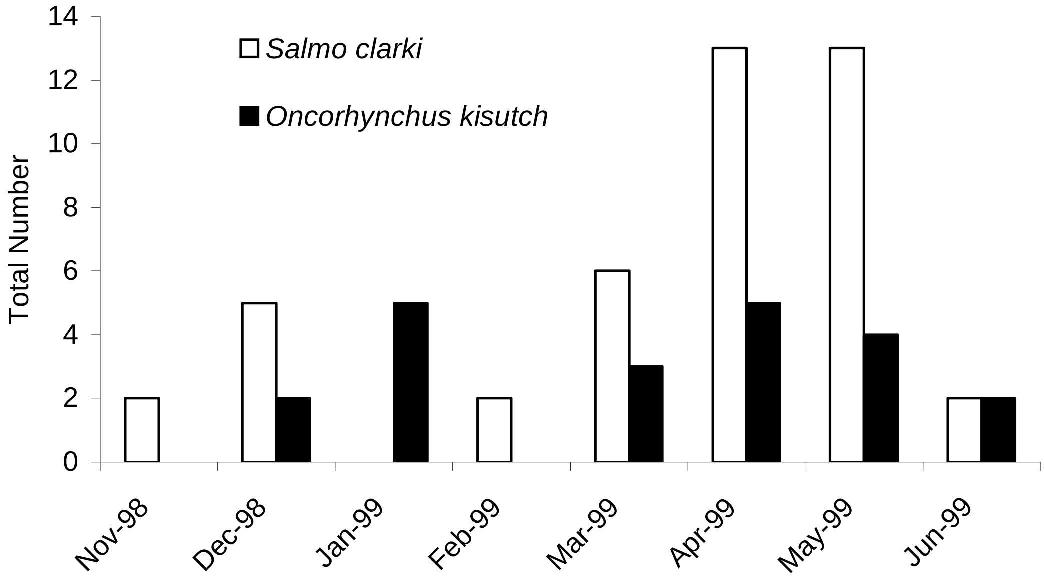

Figure 21. Number of salmonids found in Winchester Creek, SSNERR, Coos Bay, OR between November 1998 and June 1999.

Precipitation and temperature

There was almost no precipitation during the summer months of 1998 and 1999 (Figure 22), but rainfall increased in the fall months of both years with peaks in November and December. Yearly rainfall was 32.5cm in 1998, and 23.14cm in 1999. Temperatures generally were highest in the summer months, but there were extreme high temperatures in October 1998 and April 1999, as well as extreme low temperatures in December 1998.

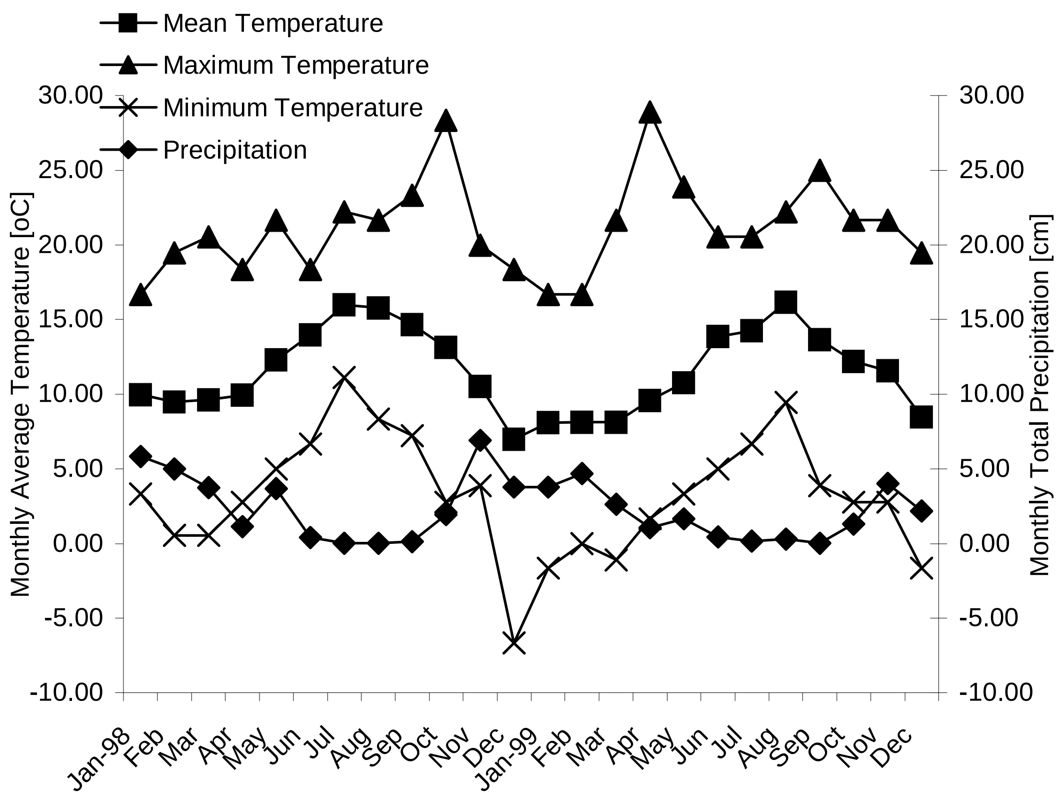

Figure 22. Average monthly mean, maximum and minimum temperatures and monthly total precipitation for 1998 and 1999 as recorded at the Municipal Airport in North Bend, OR, (~5 miles north of SSNERR).

DISCUSSION

Restoration efforts initiate community succession as substrate and space are made available to colonizers (Valiela 1984). The aggressive restoration methods used in Kunz Marsh, SSNERR, OR, which involved the landscaping of different intertidal elevations following the removal of a dike, initiated a series of successional events in 1996 that will proceed until these elevations develop community characteristics similar to sites in the South Slough that were never diked. As is the case following any disturbance to a community, the trajectory of recovery, or succession, in Kunz Marsh depends on if the disturbance was severe enough to remove the established bank of spores and seeds, and how the timing of the disturbance affects the composition and abundance of newly arriving seeds, algal spores, as well as eggs and larvae of fish and invertebrates. Thus, for Kunz Marsh, the starting community composition within restored sites was determined by the propagules and plant fragments already in the dike sediments, and the production of propagules from the surrounding terrestrial, freshwater, and marine habitats during the summer of 1996. The present study, which began in spring 1998 and finished during fall 1999, therefore started two years after restoration activities were initiated in Kunz Marsh. During this interval, the abundance and composition of plants and invertebrates from restored sites became similar to control sites more quickly within high versus low elevation sites. However, diatom abundance was more affected by season than elevation, and fish followed an elevational gradient with more fish found in lower elevations with a species composition similar to adjacent Winchester Creek.

Vegetative cover and species composition are the most obvious characters of a coastal wetland. Plant cover generally decreases from higher and drier to lower and wetter areas (Vogl 1966, Chapman 1977). Similar trends were found in the restored sites of Kunz Marsh, which had overall less cover than the control marshes. The higher restored sites had higher cover than the lower control sites. Both patterns were expected, as the restoration removed all vascular plants in the restored sites in 1996 and recovery had to start from substrate without standing vegetation (Cornu 1998). Furthermore, as mentioned above, lower elevations are known for less vegetation than higher tidal marshes (Chapman 1977), due to the inability of many plant species to withstand high salt concentrations and poor soil aeration due to water inundation (Vogl 1966).

Cover decreased in the 2.2m restored site between 1998 and 1999, was stagnant in the passive site, but increased in all other restored sites. This could have been due to the relatively low temperatures in winter 1998 and the high, early temperatures in spring 1999 that could have damaged plants. Although phosphorus levels were not monitored, previous studies have shown that young marshes with low organic matter are limited by phosphorus (Van Wijnen and Bakker 1999). In addition to annual changes in plant cover, seasonal variation was observed in plant cover in all restored sites except at the lowest, passive site. However, within the control sites of this study, seasonal changes were observed in cover during both years, except at the high marsh site where dense cover occurred during all seasons. This seasonal pattern of abundance is typical for temperate coastal wetlands (Kozloff 1973, Tiner 1999). This seasonal difference between restored and control sites is probably due to the density of vegetation. The high restored site still showed seasonality as the vegetation was not as dense as in the control site, where seasonal changes in biomass occur even though percent cover remains at 100 percent year round. The passive restored site did not show seasonal patterns as vegetation cover was lower that five percent at all times.

Three years after restoration, in 1999, the composition of the plant community within high and mid restored sites had also become similar to control sites at the same elevations. The higher marsh sites were dominated by the expected grasses, sedges, and herbaceous plants, such as clover (*Trifolium* sp.) and gumweed (Kozloff 1973). Although Kozloff (1973) mentioned *Scirpus* sp. as the prevailing mid marsh grass, the mid marsh control site was dominated by a different sedge, Lyngby's sedge (*Carex lyngbei*), which was mixed with glasswort (*Salicornia virginica*), arrowgrass (*Triglochin maritimum*), and toad rush (*Juncus bufonious*). But some species like *Scirpus robustus* were absent. However, the 1.8 m marsh restored site had relatively high richness with several grass species present and was similar to the high restored site in 1998 but obtained a similar species composition as its control site in 1999. In the lower control sites a variety of small plants dominated: Japanese eelgrass (*Zostera japonica*), glasswort, arrow-grass, and brass buttons (*Cotula coronopifolia*). However, in the lower restored site only the brass buttons was present. This site was the so-called passive site as its dike was removed but no filling with dike material occurred. While diked, a freshwater pond had formed behind the dike due to upland creeks, and a large population of cattail (*Typha latifolia*) covered the area. Cattail is a mesohaline to freshwater species (Tiner 1999) and died off due to dike removal and increased salinity. Decomposition left most of the site with anaerobic soil except for the top few centimeters. An increase of sediment accumulation is probably necessary to allow the establishment of a plant species composition in the passive restored site similar to its control site. Overall it seems that the vascular plant species composition of the 2.2m restored marsh was similar to its control in 1998 as well as 1999, although abundances were still lower. The 1.8m restored marsh had similarities to its control site in vascular plant species composition, whereas the lower restored sites, especially the passive site, were still different in percent cover as well as species composition from natural lower salt marsh areas.

The trajectory of succession in a restoration project has the potential to be altered by the invasion of exotic species (Zedler 1996a). Due to the introduction of exotic species during the past centuries, it is unusual to find marshes with only native species. In the Kunz Marsh restored sites a large proportion of plant species were non-native, or were not normally found in mature salt marshes as they are considered to be non-persistent colonizers, or temporarily persistent freshwater marsh or pasture vegetation (Cornu 1998). In the high, mid, low and passive sites, 56%, 38%, 30% and 68% of the plant species respectively observed in 1996/97 are normally not found in salt marshes (Cornu 1998). In contrast, the high control marsh site was dominated by a rich community of native salt-tolerant perennials with a few exceptions (e.g. *Agrostis alba*, a non-native, and *Atriplex patula*, an annual). This pattern was still notable in 1998. One species, the exotic *Cotula coronopifolia,* was present in all restored sites but was found only in the 1.1m control site. *Cotula coronopifolia* (brass buttons) is a non-indigenous, perennial species that was introduced from South Africa. It is widely distributed throughout the world and is especially found in areas of disturbance such as restoration sites (Pojar and MacKinnon 1994). It was the only plant species found, besides dying cattail, in the 1.1m passive restored marsh site and it persisted throughout the year. In addition, it was the species with the highest frequency in the 1.5m and 1.8m restored sites. In both of these sites, *C. coronopifolia* occurred more frequently during Fall 1998. Cornu (1998) reported similar results, as *C. coronopifolia* was present in more than 50% of the sample quadrats in the mid and low restored sites in his study between 1996 and 1997. However, it decreased in the high, mid, and low restored sites during 1999, but increased in the restored and control 1.1m sites. It is likely that the restoration of Kunz

Marsh allowed the exotic to establish itself and spread into the healthy marsh areas of SSNERR. *Cotula coronopifolia* is reportedly not a persistent species and seems to be out-competed by native grasses in the higher marshes (Cornu pers. communication). As there are only smaller plants and tall grasses in the lower marshes, it is possible that the brass buttons could persist and change species composition as well as ecological functions.

Like plant abundance and composition, invertebrate abundances and composition from high restored sites were more similar to high control sites. Invertebrate recovery in the low restored sites was slower. Within these sites, invertebrate numbers were highest in the low passive site, whereas for control sites, they were the highest in the mid marsh. Several factors could explain this pattern. Vegetative cover was lower in the restored sites than in the equivalent control sites. Therefore, invertebrates were more exposed to desiccation, especially in the higher sites. Furthermore, the soil in the higher restored marshes was harder, due to less organic content, and seemed to be less moist than in the control sites. This could have made it more difficult for invertebrates to burrow during low tides and escape desiccation. The mid control site, however, had very dense vegetation and rich organic and lose soil, which may have given invertebrates protection from desiccation as well as the possibility to escape low tides by burrowing.

Annual changes were observed in invertebrate abundance with overall fewer specimens found in summer 1999 than summer 1998 with the exception of the passive restored site. This could be due to the extreme temperatures observed in winter 1998 and spring 1999. Again, the loose soil in the passive restored site probably allowed invertebrates to burrow and escape temperature fluctuations. Unlike the plants, total invertebrate abundance was not affected by seasonality. In other studies seasonal changes have been observed in individual taxa (Robert, Jr. and Matta 1984), but these patterns could get lost by combing taxa with different seasonal life cycles as was done in this study.

With respect to invertebrate composition, the restored sites were inhabited by nearly all of the invertebrate species found in the control sites. Oligochaetes, which make up to 50% of the invertebrate community in natural marshes (Levin et al. 1996), were dominant in our restored as well as control sites. The sabellid *Manayunkia aestuarina* was also common, with its highest abundances in the mid control site. This taxon is known as a dominant or sub-dominant species in benthic marsh assemblages (Bell 1982), and has been noted to be more common in vegetated areas (Yozzo 1994). Chironomidae larvae and pupae occurred, as expected, in higher numbers in open areas of the higher marsh sites, which was probably the reason why more Chironomidae larvae were found in restored than in control sites.

Although the same species generally inhabited the restored and control sites, there were three exceptions. The abundant temperate marsh polychaete *Hobsonia florida*, known as a common mid-low intertidal species

(Furota and Emmett 1993), was found mainly in the lower restored sites. The dipteran Dolichopodidae, that were mainly found in the high restored sites, are known to become established as marshes mature and exhibit different plant communities than lower and younger marshes (Kraeuter and Wolf 1974). On the other hand, watermites that were abundant in the control sites did not occur in the restored sites, which could be due to the reduced abundance of tall grasses and sedges.

Although invertebrate numbers were high in the passive restored site, species composition differed from its low control site. As mentioned above, this could be due to the fact that the passive restored site is supported by a detritus based microbial food web supplied by decaying cattails. However, the presence of tube-forming polychaetes indicated a typical community in an early successional stage (Valiela 1984). Although *Streblospio benedicti* is a polychaete characteristic of developing marshes (Levin et al. 1996), it was not present in any of the restored sites, but might have been replaced by a polychaete with similar ecological functions such as *Hobsonia florida*.

Microhabitats did significantly influence species distribution in the lower restored and control marsh sites. Dipteran Ceratopogonidae larvae and pupae, for example, preferred vegetated areas, as did several crustacean taxa and the hydroid *Nematostella vectensis*, which is known as a typical salt marsh animal (Kozloff 1973). Although several invertebrate taxa followed distribution patterns described in previous studies, the lower restored sites did not show many similarities to their control sites.

Overall, the high restored marsh showed a similar species distribution to the higher control sites, but the abundances were much lower. Microhabitat differences also influenced this observed pattern. The open areas of the high restored sites were similar to the respective controls in 1998, but the vegetated areas did not show similarities until 1999. The species distribution of the mid restored marsh site, on the other hand, was similar to the low restored sites in both years. The invertebrate community of the passive restored site has not reached the same composition as its control. Elevation as well as the availability of open or vegetated areas influenced the distribution of invertebrate species in these restored marsh sites.

The pattern of diatom abundance differed from the spatial and temporal patterns of plant and invertebrate abundance since diatoms did not follow an elevational gradient and exhibited different seasonal patterns depending on elevation and microhabitat. The lack of pattern in the distribution of diatoms from higher to lower elevations was surprising, as elevation determines exposure time, which is a primary factor influencing diatom abundance (Saburova et al. 1995). Diatoms are often more abundant in lower wetland elevations (Meadows and Anderson 1966) since desiccation and exposure are more intense at higher elevations (McIntire and Overton 1971, McIntire 1978, Saburova et al. 1995). However, these previous studies observed diatom distribution along an elevational gradient within one habitat (e.g. on one piling at different elevations or within one mudflat). The elevational gradient in this study was through several different habitats including mudflats with vegetation patches, mid marsh areas, and higher salt marsh sites with dense vegetation. Therefore, a possible explanation for the distribution pattern observed in this study could be that the overall higher vegetative cover in higher marsh sites protected diatoms from desiccation.

Within a particular site, patterns of diatom abundance were affected by microhabitat. Substrate movement can cause frustule damage (Delago et al. 1991), and desiccation is an important factor in diatom distribution (Saburova et al. 1995). Vegetative cover protects diatoms from both influences as it buffers the water movement and provides shade. Therefore it was expected that even on a small scale, vegetated microhabitats within a site would make a difference in diatom abundance. However, this was not the case, as microhabitat (open and vegetated areas) was not significant for the distribution patterns found in diatoms. It could be that the chosen microhabitat areas were too small or in close proximity of each other, which could possibly not provide the expected protection for diatoms.

Diatom abundances did follow a seasonal trend with highest numbers in the summer and a decrease in the fall in almost all sites, except the 1.5m and 1.1m restored, passive site. Although they were expected to exhibit a linear decline after the spring bloom due to overgrowth and shading by vascular plants (Valiela 1984), changes in temperature as well as light intensity can cause seasonal cycles in benthic diatoms (Oppenheim 1991). Due to longer inundation periods and turbidity in the lower elevations, there is usually a lack of seasonal growth (Oppenheim 1991) and this is what occurred in the low as well as the passive restored site in our study. As not all sites within this watershed follow the same seasonal pattern it is possible that due to differences in microhabitat and elevation, a combination of shading, temperature changes, and inundation time influenced seasonal changes in the diatom assemblages in this study.

Richness and abundance of fish species were measured in the restored sites and adjacent Winchester Creek. Therefore, the creek was considered the source population for colonizers entering restored sites and was used, together with published studies, to evaluate the recovery success of these sites. As expected, more fish species were found in Winchester Creek than in the restored marsh sites. This is due to the permanent water availability in the creek, even during low tide. The 1.1m passive restored site had the highest numbers of species found throughout the November 1998 to April 1999 sampling period. Prolonged flooding is probably the reason for this pattern. In low marsh areas, accessibility directly increases species abundance. The higher restored marsh sites took longer to establish drainage channels, which can be used by fish to enter and exit the marsh flats. Consequently, the higher richness recorded at the 1.1m passive restored marsh site may be due to the earlier formation of drainage channels and longer periods of flooding. Although staghorn sculpin and topsmelt were among the top three species (including threespine stickleback) found in the restored sites as well as in Winchester Creek, they were not included in survey lists of the most common species inhabiting other Pacific coast wetlands (Monaco et al. 1990, Monaco et al. 1992). However, Chamberlain and Barnhart (1993) pointed out that they were common in a created marsh due to their euryhaline characteristic. On the other hand, species listed as common in other studies, such as the arrow goby and bay goby, were not found at all, possibly due to specific habitat differences between the wetlands.

Evaluation of Restoration Methods

Although the need for wetland restoration has long been recognized, mitigation and restoration efforts to date have not duplicated wetland functions nor have they shown that restored wetlands maintain biodiversity (National Research Council 1992). Previous salt marsh restoration projects have typically included dike removal and short-term monitoring of plant communities. However, some restored wetlands require more than seven years to gain the functional attributes of natural habitats (Williams et al. 1988 in Haltiner et al. 1997). Establishing mandatory long term monitoring programs in specific estuaries and wetlands allows us to develop better restoration and mitigation plans, as well as provide comparable reference wetlands for other projects (Shreffler et al. 1992, Zedler and Powell 1993).

Dike removal alone does not account for soil subsidence and compaction. Establishing certain elevations by landscaping rather than leaving passive areas subject to tidal flushing has distinct advantages. In the Kunz Marsh project, soil subsided up to 70cm over a 100-year period (Rumrill and Cornu 1995). It would take hundreds of years for natural sedimentation to fill the sites to a historical level, where high marsh vegetation could form a mature community. Furthermore, the invasion of exotic plants can be reduced due to establishing physical regimes favored by native organisms (Zedler 1996a). As seen in Kunz Marsh, the exotic species *C. coronopifolia* did well in lower elevations but was out-competed by native plants in the higher elevations.

If the goal is to establish fish populations rapidly, the passive restoration method might appear favorable. However, it will not be able to support itself over time and is certainly not a stable environment. Lack of vegetation does not only decrease primary productivity but also reduces the protection of organisms from sun exposure, soil and water movement, and predation.

Restoration efforts should include a landscape level approach (Zedler 1996c) that focuses on diverse habitats and their floral and faunal communities as well as consider “wetland restoration for its own sake” (Zedler and Powell 1993). Although expectations should still be defined for successful restoration projects, such an approach might help to establish a complex network of marsh habitats with different physical and biological characteristics as observed in natural systems. Including different elevations in a restoration project is part of such an approach.

Conclusions

Based on the succession observed in natural salt marshes, it is possible to evaluate the successional stages of the restored Kunz Marsh sites. It is important to point out that the restoration of Kunz Marsh was completed in the summer of 1996. Because this is the time of flower and seed production for most vascular plants, it is likely that the natural marsh vegetation provided a seed bank for the newly restored sites. Furthermore, seeds and roots of plants found in the earthen dike were redistributed throughout the sites during establishment of elevations. Tidal activity allowed dispersal of invertebrates into the sites from nearby natural marsh habitats.

Two years after the restoration was completed, when I began the present study, Kunz Marsh sites had reached different levels of maturity. The high marsh site was in a more mature successional stage. Glasswort was still present but dominant vascular plants, such a salt grass, were already present. The invertebrate community in the 2.2m site reflected a high marsh stage showing the absence of polychaetes but the prevalence of insect communities associated with grasses. The restored mid marsh site had a significant salt grass cover but the dominant plant species was the exotic *Cotula coronopifolia*. More sedges and grasses were expected in the mid marsh than were actually found, while glasswort should be more common in the low site than it was (Kozloff 1973). The 1.1m passive site did not reflect any of the expected low marsh vascular plant components, such as glasswort and arrowgrass, but it was dominated by the introduced brass button. Although this plant community was not as expected for a low marsh, the invertebrate species found, mainly tube-building polychaetes, are common to early successional stages. Finally, fish species diversity was greatest in this site. In addition to direct access diversity may be high here due to feeding opportunities on crustaceans, such as Cumacea and *Gnorimospharoma insulare*, and polychaetes, such as *Manayunkia aestuarina*, supported by the high amounts of nutrients.

Data collected from spring 1998 until fall 1999 indicated that succession has proceeded at a faster rate in the higher marshes than in the lower marshes. Vascular plant communities became established early in the high marsh site, and an increase in percent cover occurred more rapidly. The invertebrate communities followed the same pattern. This seems to indicate that the establishment of different elevations and, therefore, the elimination of subsidence appears to enhance marsh recovery. Other studies have shown that the re-establishment of natural elevations allows for faster recovery of wetland communities (Roman et al. 1995) and a more precise establishment of target communities (Zedler 1991; 1996b). Our vascular plant data suggest that restored and control sites behave differently from each other, with more similarities in percent cover and species composition recorded between higher marsh communities than the lower marsh sites and their controls. Over time, the lower restored marsh areas are expected to develop into more natural, mature habitats with vascular plant communities similar to those found in the lower control areas.

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APPENDIX A – EVALUATION OF METHODS

THE 1.5M RESTORED SITE

As mentioned in the Materials and Methods section, the 1.5m restored site was not completely filled, as dike material was unavailable. Consequently, the lower back of the site filled with water and formed a brackish pond that existed for a few years and then slowly drained. The question arose if the creation of such pools should be avoided in future restoration projects, as they do not reflect the targeted marsh elevation, and therefore, marsh community. However, brackish ponds do exist in naturally diverse marsh habitats and are part of a complex ecosystem. Tidal fluctuations, flow variations, and sediment transport will either stabilize this pool or eliminate it. The pond area showed a distinct plant community, and small fish species such as sculpins and topsmelt used it as a refuge. Finally, a drainage ditch has formed allowing excess water to flow into Winchester Creek, minimizing the entrapment of fishes, increasing access of fish during high tide, and progressively draining the pond.

EVALUATION OF METHODS USED FOR DIATOMS

Quadrat size and aliquot precision

The quadrat size of 2cm x 2cm was chosen arbitrarily. To determine whether this size was appropriate a second set of 24 samples using a 4cm x 4cm quadrat was taken in the 2.2m restored marsh during fall 1998. Sampling variation was about the same for both quadrat sizes (Figure 23), however, numbers of frustules were distinctively lower in the larger quadrat.

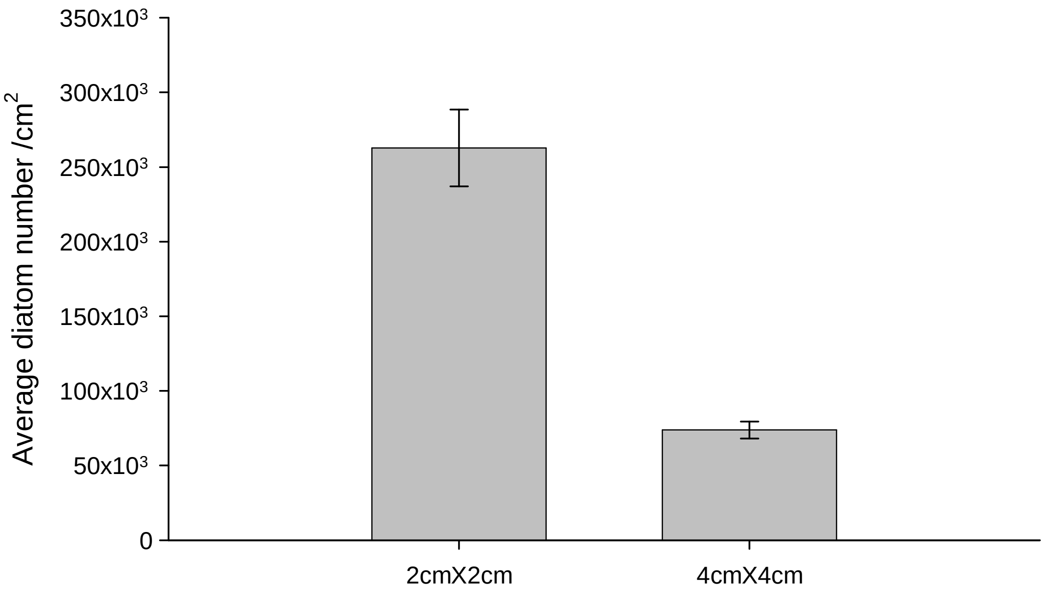

Figure 23. Comparing the use of different quadrat sizes for diatom samples taken on the 2.2m restored marsh during fall 1998. For each quadrat size, 24 samples have been analyzed. Error bars are ± 1 Standard Error of the Mean.

Although the increased quadrat size might decrease variation in diatom distribution, several other problems arise. With a larger quadrat, the actual samples varied in depth and hence amount of substrate taken. The smaller quadrat allowed for more standardized sample volume. Due to the increased volume, more sediment obscured the diatom frustules. They were harder to see and count. It would be still be interesting to see if it is possible to standardize the taken volume of a larger quadrat, eliminate more sediment, and compare sampling variation. As Guarini et al. (1998) showed in their study, sampling error decreased with increasing sample size while average chlorophyll increased.

As sediments obscured the sight of diatom frustules, it was questionable that counting a sub-sample once would give an accurate estimate of the abundance in the sample quadrat. To determine aliquot precision, one sub-sample was counted 5 times to determine standard error of the mean and was compared to the average number of frustules found in 12 samples of the same site (Figure 24). Although the number was larger than the average, it showed that the sample error of the mean of one sample counted several times was smaller than of 12 samples counted once each. This reassured me that the aliquot precision was appropriate.

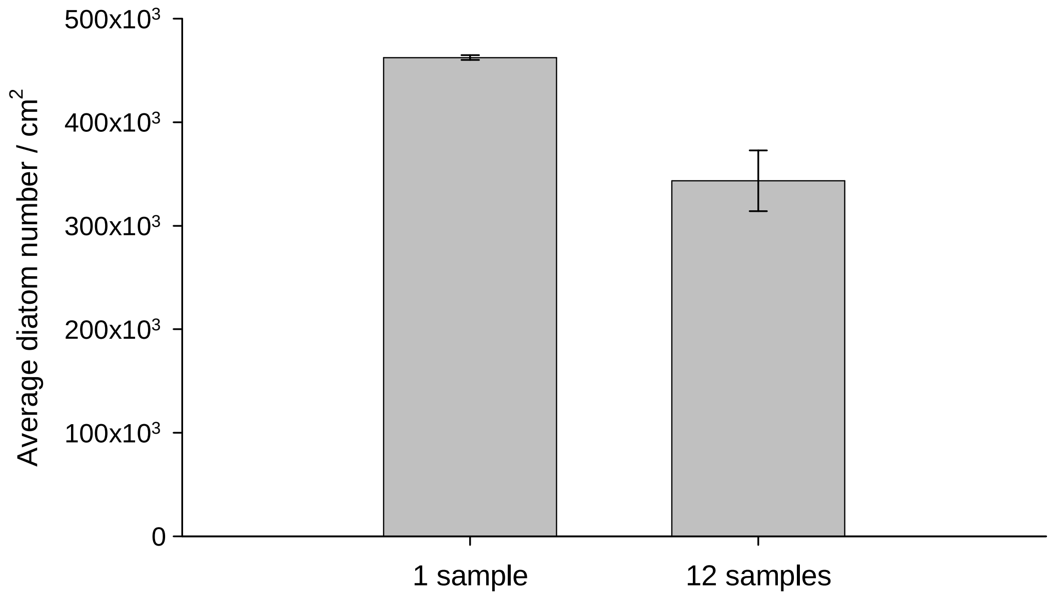

Figure 24. Determination of aliquot precision for diatom samples. 1sample = Average number of diatoms found within 1 sample counted 5 times, 12 samples = Average number of diatoms found within 12 sampled of the same site counted once each. Error bars are ± 1 Standard Error of the Mean.

Comparison of chlorophyll analysis versus frustule counting

Microalgae are an important aspect of this project due to their role in primary productivity and as a food source to many organisms. Because diatoms represent the majority of algae in tidal wetlands, they merit specific attention. After the first year of sampling, it became clear that they also require an incredible amount of time and knowledge to be processed. Counting diatom frustules allows for determining precise abundance estimates as well as community structures based on species present. There are other methods to evaluate the primary productivity of algae, and some are more time efficient, but have other deficiencies. We tested chlorophyll ‘a’ analysis in the study of Mad River slough, Humboldt Bay, CA and compared the results with the Kunz Marsh results. Chlorophyll ‘a’ can be used as an indirect measure of standing stock. It was expected that this method would show less variation than the counting method. However, this was not true (Figure 25).

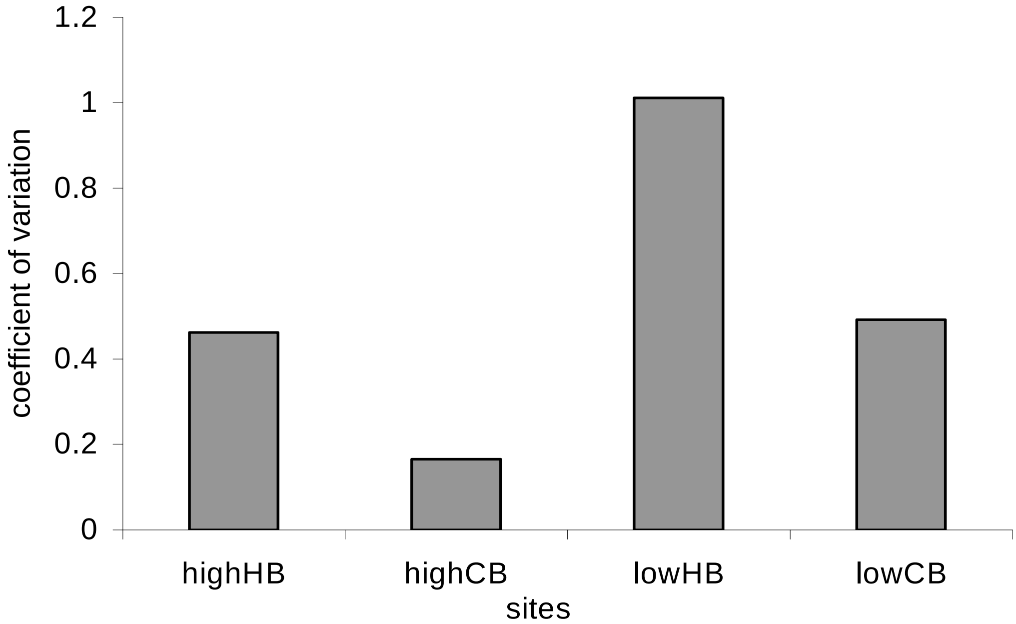

Figure 25. Comparison of chlorophyll ‘a’ analysis to counting diatoms. Coefficient of variation is obtained by dividing Standard deviation by the mean. HB = Humboldt Bay, site of chlorophyll analysis. CB = Coos Bay, site of the counting method. High = high marsh with approximately 2.2m (NAVD) and low = low marsh with approximately 1.5m (NAVD).

Chlorophyll analysis has the advantage that it is relatively quick; however, collected samples must be processed within a few days. As both methods have their advantages, but their use depends on the specific interest of a given study. If the main focus is on standing stock it is highly recommended to use chlorophyll analysis. Unfortunately, this does not allow for a description of the algal community and demographic changes. Additional samples would have to be taken to identify species, and hence, these could not be correlated with the measured amount of chlorophyll. In summary, a method should be chosen depending on how many samples need to be taken and how many people can be involved in the analysis. The counting method is very time and work intensive. Therefore this method is recommended only if the number of samples is kept relatively small and/or there is assistance in the processing.

EVALUATION OF METHODS USED FOR VASCULAR PLANTS

In constructing species lists, certain weaknesses must be addressed. In our study, species identification in spring and fall has been difficult due to the absence of seeds and flowers on many species, especially weeds and grasses. Furthermore, some perennial plants lose their foliage and only the subsurface parts survive the winter, making the spring identification difficult. However, the importance of species compositions must be emphasized. First of all, many wetland classification schemes are based on vascular plant communities. Secondly, complete evaluations of habitat function and health require information on species composition as well as abundance. Trends within systems might be more visible in community structure than overall percent cover. Both percent cover and species composition differed between restored and control sites. In the high and mid restored sites, cover changed within a few months. Species compositions were only similar to corresponding controls at the high restored site. Lower marshes are naturally patchy in cover; therefore changes in the low restored sites might not be obvious. However, the species compositions were extremely different between the mid and low restored sites and their controls. If these marshes are comparable and are intended to have similar ecological functions, it is expected that these differences in species composition will decline over time.

EVALUATION OF METHODS USED FOR FISH

Pacific salmon are commercially important fish and several species and stocks are listed as endangered under the Endangered Species Act. Therefore, special attention is given to salmonids in marsh projects, as juvenile salmon use estuarine marshes as rearing habitats before they enter the ocean. Although some salmon were found within restored Kunz Marsh and adjacent Winchester Creek, the low numbers were somewhat surprising. However, seining was done monthly during day-light from November 1998 until April 1999. However, previous studies indicated that fish do not migrate into the higher marsh areas until night. In addition to more fish within the habitat in question, fishing at night decreases their ability to avoid fishing gear. After April 1999, Steve Sadro, SSNER, kept sampling the 1.5m restored Kunz Marsh site for a different project focusing on juvenile coho salmon (Miller and Sadro 1999 unpubl.). This particular Kunz Marsh site was now sampled once or twice a month during the nighttime spring tides. Fish caught in Kunz Marsh were marked with India ink. A total of 51 juvenile Coho were marked in Kunz Marsh, but fish that had been marked upstream in different areas of South Slough were also recaptured within this site. This study showed that salmon used the restored Kunz Marsh sites. In addition, Miller and Sadro (1999 unpubl.) also found that fish recaptured in tidal channel and in Kunz Marsh had higher growth rates, with lowest growth rates observed in fish that were reared in upstream habitat. Furthermore, the condition factors were determined. The lowest were recorded in fish caught midstream, whereas the highest factors observed were from fish caught in the upper watershed and again, in Kunz Marsh and adjacent channels.

Although Coho salmon were not assumed to use estuaries as rearing habitat, this study showed that they do make use of the food availability and protection found within the marsh areas.

APPENDIX B – SPECIES LISTS

Table 10. Vascular plant species found in restored (rest) Kunz Marsh and control (con) sites in SSNERR, OR in spring (S), summer (Su), and fall (F) 1998. X = found, p = present but not within sampling

Species	2.2m rest			1.8m rest			1.5m rest			1.1m rest			2.2m con			1.8m con			1.5m con			1.1m con		
	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F
Atriplex patula [a]	X	X		X	X			X	P				X	X	X									
Juncus bufoniuous [a]		X			X											X	P	P	X			X	P	P
Agrostis alba		X			X								X	X										
Alopeurus aequalis		X			X									X										
Aster subspicatus		X																						
Carex lyngbei	X	X			X		X	X					X	X		X	X	X	X					X
Cotula coronopifolia	X	X		X	X		X	X		X	X											X		
Deschampsia	P	X			X			X					P	X			X							
Distichlis spriata	X	X		X	X		X	X					X	X			X							
Eleocharis parvula			X			X		X													X			X
Glaux maritima														X										
Grindelia integrifolia	X	X			X								X	X										
Holcus lanatus	P	X		X										X										
Jamea carbosa	X			X									X											
Lathyrus sp.																								
Lolium perenne		X												X										

Table 10 continued

Species	2.2m rest			1.8m rest			1.5m rest			1.1m rest			2.2m con			1.8m con			1.5m con			1.1m con		
	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F
Lotus sp.																								
Oeanthe sarmentosa																								
Potentilla pacifica	**X**	**X**		**X**	**X**								**X**	**X**										
Rumex sp.		**X**			**X**									**X**										
Salicornia virginica					**X**								**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Scirpus sp.		**X**												**X**										
Solidago sp.																								
Trifoium repens																								
Trifolium wormskjoldii																								
Triglochin maritium		**X**		**X**	**X**		**X**	**X**					**X**	**P**	**P**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Typha latifolia										**X**	**X**	**X**												
Zostera japonica																			**X**	**X**	**X**			

[a] annual plant

Table 11. Vascular plant species found in restored (rest) Kunz Marsh and control (con) sites in SSNERR, OR in spring (S), summer (Su), and fall (F) 1999. X = found, p = present but not within sampling quadrats

Species	2.2m rest			1.8m rest			1.5m rest			1.1m rest			2.2m con			1.8m con			1.5m con			1.1m con		
	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F
Atriplex patula [a]	X	X	X			X							X	X	X									
Juncus bufoniuous [a]	X	X			X	X														X			P	
Agrostis alba	X	X	X	X	X	X		X																
Alopeurus aequalis			X			X								X	X									
Aster subspicatus			P			P																		
Carex lyngbei	X	X	X	X	X	X	X	X	X				X	X	X	X	X	X			X			
Cotula coronopifolia	X	X	X	X	X	X	X	X	X	X	X	X											X	
Deschampsia cespitosa	X	X	X	X	X	X	X		X				X	X	X									
Distichlis spriata	X		X		X		X		X						X	X	X	X						
Eleocharis parvula	X			X	X	X	X	X	X							X						X	X	X
Glaux maritima																								
Grindelia integrifolia		X	X										X	X				X						
Holcus lanatus	X	X		X																				
Jamea carbosa													X	X	X									
Lathyrus sp.	X																							
Lolium perenne		X																						

Table 11 continued

Species	2.2m rest			1.8m rest			1.5m rest			1.1m rest			2.2m con			1.8m con			1.5m con			1.1m con		
	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F	S	Su	F
Lotus sp.		**X**																						
Oeanthe sarmentosa	**X**																							
Potentilla pacifica	**X**	**X**	**X**	**X**	**X**	**P**							**X**	**X**	**X**									
Rumex sp.	**X**	**P**																						
Salicornia virginica					**X**	**X**							**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Scirpus sp.	**X**	**X**				**X**		**X**																
Solidago sp.			**X**																				**X**	
Trifoium repens	**X**																							
Trifolium wormskjoldii	**X**	**X**		**X**	**X**																			
Triglochin maritium	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**				**X**	**X**	**X**	**P**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**P**
Typha latifolia										**X**	**X**	**X**												
Zostera japonica							**P**												**X**	**X**	**X**	**X**	**X**	

[a] annual plant

Table 12. Invertebrate taxa found in restored (rest) Kunz Marsh, SSNERR, Coos Bay, OR, in spring (Sp), summer (S), and fall (F) 1998, as well as summer 1999 (S2). Elevations given in NAVD.

Invertebrate Taxa	**2.2m rest**				**1.8m rest**				**1.5m rest**				**1.1m rest**			
	Sp	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**
ANNELIDA																
Hobsonia florida					x		x		x	x	x	x	x	x	x	x
Manayunkia aestuarina	x					x	x		x	x	x	x	x	x	x	x
Juv. *Neanthes* sp.	X								x				x	x		
Neanthes brandti												x	x			x
Neanthes limnicola		x										x	x	x	x	x
Nematoda	x				x		x	x	x	x	x	x	x	x		x
Nemertea																
Oligochaeta	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Polydora nuchalis											x		x		x	
Pseudodora kempi										x	x					
Pygospio elegans						x					x		x			
Streblospio benedicti		x							x		x			x		
Unknown polychaeta						x	x				x	x				
Annelid B											x					
ARACHNIDA																
Araneae	x	x		x		x										
CRUSTACEA																
Allorchestes angusta												x				
Amphitoe lacertosa											x					x
Corophium sp						x	x	x	x	x	x	x	x	x	x	
Cumacea sp	x				x		x		x	x	x	x	x	x	x	
Detonella papillicornis																
Eogammarus confervicolus		x					x									
Gnorimosphaeroma insulare					x		x		x	x	x	x	x			
Hemigrapsus oregoniensis											x					
Leptochelia dubia										x	x					x
Sinelobus stanfordii									x		x	x	x	x	x	x
Traskorchestia georgiana	x		x													
Crustacean D			x								x					
INSECTA																
Acari	x		x	x	x	x			x		x	x				x
Acari2	x					x	x			x			x			x
Ceratopogonidae larva	x	x	x		x	x	x	x	x	x	x	x	x	x	x	x
Ceratopogonidae pupa	x				x	x		x	x	x	x	x	x	x		x
Chironomidae larva	x				x				x	x			x	x		
Chironomidae pupa	x				x				x				x	x		

Table 12 continued

Invertebrate Taxa	**2.2m rest**				**1.8m rest**				**1.5m rest**				**1.1m rest**			
	Sp	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**
Colembolla	**x**									**x**						
Coleoptera adult							**x**			**x**		**x**	**x**			
Coleoptera larva		**x**		**x**		**x**				**x**		**x**				
Dolichopodidae						**x**		**x**	**x**	**x**				**x**		
Ephydridae	**x**					**x**								**x**		
Isoptera		**x**		**x**												
Salididae	**x**	**x**			**x**	**x**				**x**						
Sciomyzidae	**x**			**x**	**x**	**x**				**x**				**x**		
Tipulidae	**x**		**x**		**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**		**x**	**x**	**x**
Insect adult	**x**	**x**				**x**					**x**					
Insect pupa	**x**				**x**	**x**					**x**			**x**		
Insect larva						**x**	**x**				**x**					
MOLLUSCA																
Bivalves (juv)	**x**															
Gastropoda							**x**									
Nudibranchs						**x**				**x**	**x**	**x**		**x**		**x**
PROTOZOA																
Foraminifera	**x**		**x**	**x**	**x**		**x**	**x**	**x**	**x**	**x**					**x**
MISC																
Eggs and egg cases	**x**	**x**	**x**		**x**	**x**	**x**			**x**	**x**	**x**	**x**	**x**		**x**

Table 13. Invertebrate taxa found in control sites (contr), SSNERR, Coos Bay, OR, in spring (Sp), summer (S), and fall (F) 1998, as well as summer 1999 (S2). Elevations given in NAVD.

Invertebrate Taxa	**2.2m contr**				**1.8m contr**				**1.5m contr**				**1.1m contr**			
	Sp	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**
ANNELIDA																
Hobsonia florida					x	x				x	x	x		x	x	
Manayunkia aestuarina		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Juv. *Neanthes* sp.										x			x			
Neanthes brandti																
Neanthes limnicola						x		x	x	x	x	x	x	x	x	
Nematoda	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Nemertea			x		x	x			x	x	x	x	x	x	x	
Oligochaeta	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Pygospio elegans					x	x		x	x	x	x	x	x	x	x	x
Streblospio benedicti						x			x	x	x	x	x	x	x	x
Unknown polychaeta					x	x			x	x	x	x	x	x	x	
Annelid B									x	x	x	x	x			
ARACHNIDA																
Araneae	x	x			x	x	x	x						x		
CRUSTACEA																
Corophium sp						x		x	x	x	x	x	x	x	x	
Cumacea sp					x	x			x	x	x	x	x	x	x	x
Eogammarus confervicolus					x	x			x	x	x	x	x	x		
Gnorimosphaeroma insulare						x										x
Hemigrapsus oregoniensis									x	x						
Leptochelia dubia												x		x	x	x
Sinelobus stanfordii				x	x	x			x	x	x	x	x	x	x	x
INSECTA																
Acari	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Acari2	x	x	x		x	x	x	x						x		x
Ceratopogonidae larva		x		x	x	x	x	x	x	x	x	x	x	x	x	x
Ceratopogonidae pupa					x	x		x	x				x	x	x	x
Chironomidae larva	x	x			x				x			x	x			x
Chironomidae pupa					x	x			x							
Colembolla			x										x			
Coleoptera adult		x		x												x
Coleoptera larva					x	x										x
Dolichopodidae	x					x										x
Ephydridae					x								x		x	x
Isoptera	x	x	x	x	x		x	x			x	x	x	x		x

Table 13 conitnued

Invertebrate Taxa	**2.2m contr**				**1.8m contr**				**1.5m contr**				**1.1m contr**			
	Sp	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**	**Sp**	**S**	**F**	**S2**
Salididae						**x**										
Sciomyzidae	**x**		**x**	**x**	**x**		**x**	**x**	**x**				**x**	**x**	**x**	**x**
Tipulidae	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
Insect adult										**x**			**x**			
Insect pupa					**x**										**x**	
Insect larva	**x**	**x**					**x**								**x**	
MOLLUSCA																
Bivalves (juv)						**x**						**x**	**x**	**x**	**x**	**x**
Gastropoda	**x**	**x**	**x**	**x**		**x**	**x**	**x**	**x**		**x**				**x**	
Nudibranchs		**x**				**x**								**x**		**x**
PROTOZOA																
Foraminifera	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**
MISC																
Eggs and egg cases	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**	**x**

Table 14. Invertebrate taxa sampled in restored Kunz Marsh and control sites, SSNERR, OR, during 1998 and 1999. Vouch id: initials of person vouchering and serial number. Sample id: three numbers = month and year sampled (MYY), Restored C1-C4 =high, mid, low and passive sites, Control C5-C8 = high, mid, low, and as passive site, E = emergent vegetation, O = open patch, last digit= plot sampled.

Voucher id	**Sample id**	**Invertebrate taxa**
GF6	598 C8E8	*Streblospio benedicti* (Polychaeta)
GF7	598 C3E8	*Hobsonia florida* (Polychaeta)
GF8	598 C8E8	Salididae (Insecta, Diptera)
GF9	598 C6O8	*Manayunkia aestuarina* (Polychaeta)
GF10	598 C6O8	Ceratopogonidae larva (Insecta, Diptera)
GF11	598 C6E5	Acari –1 (Chelicerata, Acarina)
GF12	798 C7O4	*Nematostella vectensis* (Actinaria)
GF13	798 C4E4	Ceratopogonidae pupa (Insecta, Diptera)
GF14	598 C4O3	*Neanthes limnicola* (Polychaeta)
GF15	798 C3E5	*Gnorimosphaeroma insulare* (Crustacea, Isopoda)
GF16	798 C3E5	Tipulidae pupa (Insecta, Diptera)
GF17	798 C6E5	*Sinelobus stanfordii* (Crustacea,
GF18	798 C6O5	*Pygospio elegans* (Polychaeta
GF19	799 C8O7	Juvenile bivalve
GF20	799 C8E11	Dolichopodidae (Insecta, Diptera)
GF21	799 C5E4	Isoptera (Insecta)
GF22	799 C5E12	Acari- 2 (Chelicerata, Acarina)
GF23	799 C2E10	Nudibranch
GF24	1098 C3E6	*Hemigrapsus oregoniensis* (Crustacea,
GF25	798 C3E2	Acari – 3 (Chelicerata, Acarina)
GF26	798 C3E2	Coleoptera larva C (Insecta, Coleoptera)
GF27	799 C1O4	Coleoptera larva A (Insecta, Coleoptera)
GF28	598 C8E8	Isoptera 2
GF29	598 C5E3	Acari –5 (Chelicerata, Acarina)
GF30	798 C5E8	*Detonella papilicornis* (Crustacea, Isopoda)
GF31	798 C5E10	Coleoptera adult-1 (Insecta, Coleoptera)
GF32	798 C5E11	Insect larva D
GF33	798 C5E4	Insect larva E
GF34	1098 C3E3	Insect larva F
GF35	798 C3O5	Coleoptera adult-2 (Insecta, Coleoptera)
GF36	798 C3E7	Colembolla-1
GF37	798 C3O11	Sciomyzidae (Insecta, Diptera)
GF38	798 C1O1	Isoptera –3
GF39	798 C1E5	Insect-1
GF40	598 C1012	Insect pupa
GF41	1098 C2E11	Insect larva G
GF42	798 C6E11	Coleoptera larva B
GF43	1098C6E6	Insect larva 1
GF44	1098C6E11	Insect larva 2
GF45	1098C8O1	Poly 1
GF46	799C3O2	Isopod
GF47	1098C4E1	*Polydora nuchalis*

Table 15. Species list of fishes found in the restored sites of Kunz Marsh and in Winchester Creek, SSNERR, Coos Bay, OR in November and December 1998 and January, through April 1999.

Site	Month	Species	Total #
2.2m restored	November 98	Staghorn sculpin	2
	December	Staghorn sculpin	18
		Threespine stickleback	1
	January 99	Staghorn sculpin	34
	February	Staghorn sculpin	2
	March	Topsmelt	94
		Staghorn sculpin	4
	April	-	-
1.8m restored	November	Staghorn sculpin	39
		Topsmelt	6
	December	Staghorn sculpin	87
		Topsmelt	107
		Threespine stickleback	1
	January	Staghorn sculpin	83
	February	Topsmelt	28
		Staghorn sculpin	78
	March	Topsmelt	19
		Staghorn sculpin	176
		Threespine stickleback	1
	April	-	-
1.5m restored	November	Staghorn sculpin	28
		Topsmelt	44
	December	Staghorn sculpin	99
		Threespine stickleback	2
		Prickly sculpin	1
		Topsmelt	185

Table 15 continued

	January	Topsmelt	141
		Staghorn sculpin	95
		Threespine stickleback	2
	February	Topsmelt	310
		Staghorn sculpin	55
		Threespine stickleback	4
	March	Topsmelt	214
		Staghorn sculpin	112
	April	Staghorn sculpin	36
1.1m restored	November	Staghorn sculpin	21
		Topsmelt	465
		Northern anchovy	70
	December	Staghorn sculpin	78
		Threespine stickleback	10
		Pacific herring	7
		Prickly sculpin	2
		Northern anchovy	1
	January	Topsmelt	164
		Staghorn sculpin	15
		Threespine stickleback	3
	February	Topsmelt	457
		Staghorn sulpin	20
		Threespine stickleback	3
	March	Topsmelt	591
		Staghorn sculpin	47
		Threespine stickleback	3
		Cutthroat trout	2
		Striped bass	1
	April	Staghorn sculpin	100
		Topsmelt	32
		Threespine stickleback	2
Winchester Creek	November 98	Staghorn sculpin	500
		Threespine stickleback	12
		Starry flounder	6
		Cutthroat trout	2
		Prickly sculpin	4

Table 15 continued

	December	Staghorn sculpin	571
		Prickly sculpin	6
		Threespine stickleback	14
		Starry flounder	6
		Coho salmon	2
		Topsmelt	1
		Cutthroat trout	5
	January 99	Staghorn sculpin	383
		Threespine stickleback	16
		Coho salmon	5
		Prickly sculpin	3
		Starry flounder	1
		Topsmelt	1
	February	Staghorn sculpin	445
		Prickly sculpin	2
		Threespine stickleback	9
		Cutthroat trout	2
		Saddlebag gunnel	1
	March	Staghorn sculpin	644
		Threespine stickleback	11
		Prickly sculpin	2
		Starry flounder	3
		Cutthroat trout	6
		Coho salmon	3
	April	Staghorn sculpin	927
		Threespine stickleback	3
		Prickly sculpin	11
		Starry flounder	7
		Cutthroat trout	13
		Coho salmon	5
		Topsmelt	1
		Pacific herring	1
	May	Staghorn sculpin	679
		Threespine stickleback	25
		Prickly sculpin	16
		Starry flounder	9
		Cutthroat trout	13
		Coho salmon	4
		Shiner surfperch	21
	June	Staghorn sculpin	667
		Threespine stickleback	22
		Starry flounder	5
		Cutthroat trout	2
		Prickly sculpin	14
		Coho salmon	1
		Shiner surfperch	48