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# A Survey of inbenthic macrofauna at a South San Francisco Bay salt marsh

Brian T. Pittman  
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A SURVEY OF INBENTHIC MACROFAUNA AT A SOUTH SAN FRANCISCO  
BAY SALT MARSH

A Thesis  
Presented to the Faculty of the  
Department of Geography And Environmental Studies  
San Jose State University

In Partial Fulfillment  
of the Requirements of the Degree  
Master of Science

By  
Brian T. Pittman  
December 1996

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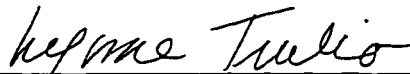
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Dr. Lynne Trulio, Assistant Professor of Environmental  
Studies, Committee Chairperson



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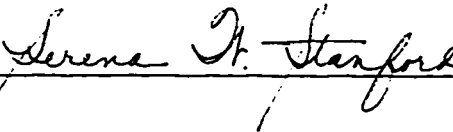
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ABSTRACT  
A SURVEY OF INBENTHIC MACROFAUNA AT A SOUTH SAN FRANCISCO  
BAY SALT MARSH

by Brian T. Pittman

At Charleston Slough, Mountain View, macroinvertebrates and physical data were compared between an undisturbed reference marsh and a salt pond mudflat twelve years following partial tidal restoration. While distributional studies of salt marsh invertebrates are common, invertebrate responses to muted tidal regimes and colonization of salt evaporation ponds have not been thoroughly examined. Infaunal abundance was compared with soil redox depth and width, organic matter content, and grain size using MANOVA, and canonical correlation analysis. MANOVA comparisons detected differences in Polychaeta and Bivalvia distribution between slough regions, with differences in species richness, evenness, soil organic matter content, soil redox depth and width, and the soil sand fraction also observed. Preliminary findings suggest that high soil organic matter resulting from persistent root systems impede functional inbenthic recovery at this salt pond. A positive correlation was found between equilibrium species presence and soil redox depth. Implications of these findings to functional ecosystem recovery and salt marsh monitoring programs are discussed.

Key words: salt marsh; redox; organic matter; Polychaeta; Bivalvia; San Francisco Bay.



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## SELECTED TERMINOLOGY

**Enhancement** - to improve the ecological functions of a site which already exhibits some ecosystem functions.

**Restoration** - re-establishment of predisturbance ecosystem functions and related physical, chemical and biological characteristics (National Research Council 1992).

**Inbenthic** - designating conditions found within the soil; literally "in the bottom."

**Epibenthic** - designating conditions found above the soil; literally "upon the bottom."

**Bioindicator** - a biological index of environmental quality based upon the relative abundance of organisms with unequal tolerances to environmental pollution; a pollution sensitive or insensitive species.

**Macrofauna** - relatively large animals; in this study macrofauna include all soil invertebrates retained by a 512 micron screen.

**Meiofauna** - relatively smaller organisms; organisms generally not retained by a 500 micron screen.

**Muted tidal regime** - exhibiting an abbreviated tidal fluctuation range usually due to the presence of a tidal regulating structure, such as a culvert or tide gate, which limits water flow.

## INTRODUCTION

This study was undertaken to determine the extent of inbenthic recolonization at Inner Charleston Slough, Mountain View, California, following an unsuccessful 1984 restoration attempt. Relevant environmental and faunal variables were measured at Inner Charleston Slough and compared with a model site, Outer Charleston Slough. Parameters studied were abundance of major infaunal groups, species richness, species evenness, redox potential discontinuity (depth and width), soil organic matter content, and sediment composition. The primary goals of the study were to quantitatively assess the similarity of the benthic infauna and physical factors of Inner Charleston Slough compared with a neighboring undisturbed salt marsh, analyze the interrelationships of animal and environmental parameters, and to develop recommendations for restoration of salt evaporation pond inbenthic communities. Studying the Charleston Slough inbenthic community should yield new data on the role of the inbenthic marsh community in restoration attempts and on the need to enhance overall marsh inbenthic diversity. These data will provide a baseline for present restoration efforts at Inner Charleston Slough.

### Background

Charleston Slough is located in Mountain View, California, about 15 miles north of the City of San Jose (Figure 1). The Slough consists of two regions: the 103-acre Inner Charleston Slough which is surrounded by levees and receives limited

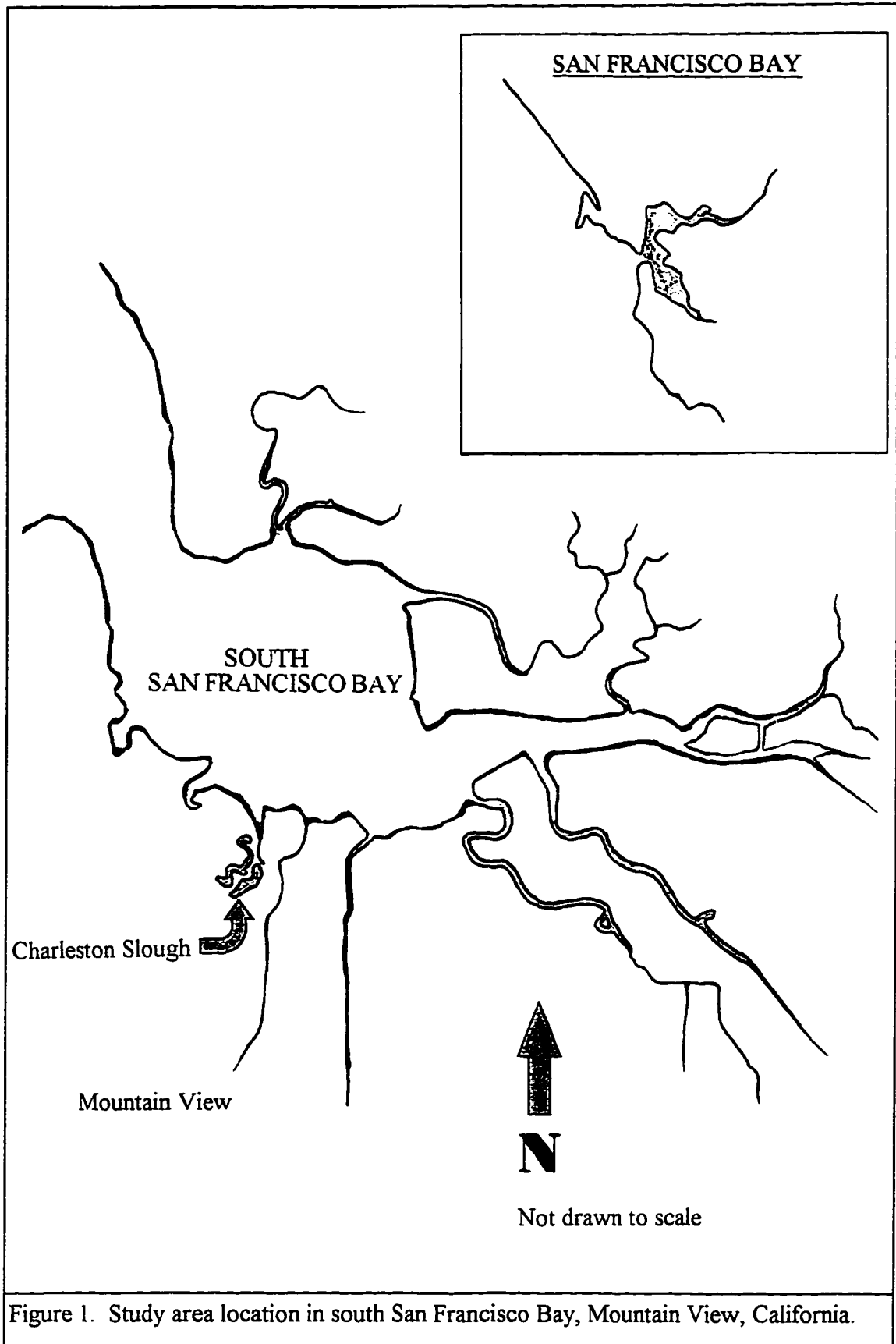


Figure 1. Study area location in south San Francisco Bay, Mountain View, California.

tidal circulation through a single tide gate, and the natural Outer Charleston Slough which is continuous with San Francisco Bay and receives full tidal action. Inner Charleston Slough is bordered to the west by the Palo Alto Baylands flood basin, to the north by Outer Charleston Slough, and to the east by Cargill salt evaporation pond A1 (Figure 2).

Inner Charleston Slough was diked between 1920 and 1950 by the Leslie Salt Company for use as a salt evaporation pond. Before it was diked, the Inner Slough was part of the extensive South Bay tidal marsh ecosystem whose dominant vegetation was pickleweed (Salicornia virginica) and cordgrass (Spartina foliosa). When it was diked, the slough became a shallow water impoundment. However, the slough was connected to San Francisco Bay by a culvert and supported between 40 and 60 acres of tidal marsh vegetation (Trulio 1995a). The dominant pre-1975 vegetation likely consisted of pickleweed and cordgrass, with alkali heath (Frankenia grandifolia) and salt grass (Distichlis spicata) occurring at higher elevations. In 1975, Leslie Salt Company modified the Inner Slough tidal regime by installing a smaller culvert positioned higher in the levee. Subsequent ponding reduced tidal exchange to 0.1' to 0.2' per tidal cycle and drowned most existing vegetation. Ownership and restoration responsibility were transferred to the City of Mountain View in June 1980 under a modified San Francisco Bay Conservation and Development Commission permit that authorized the City to remove water from the Inner Slough for their 50 acre Shoreline small boat lake (Trulio 1995a).

In July 1984 a larger culvert was installed by the City of Mountain View in an effort to encourage the growth of 30 acres of tidal marsh vegetation



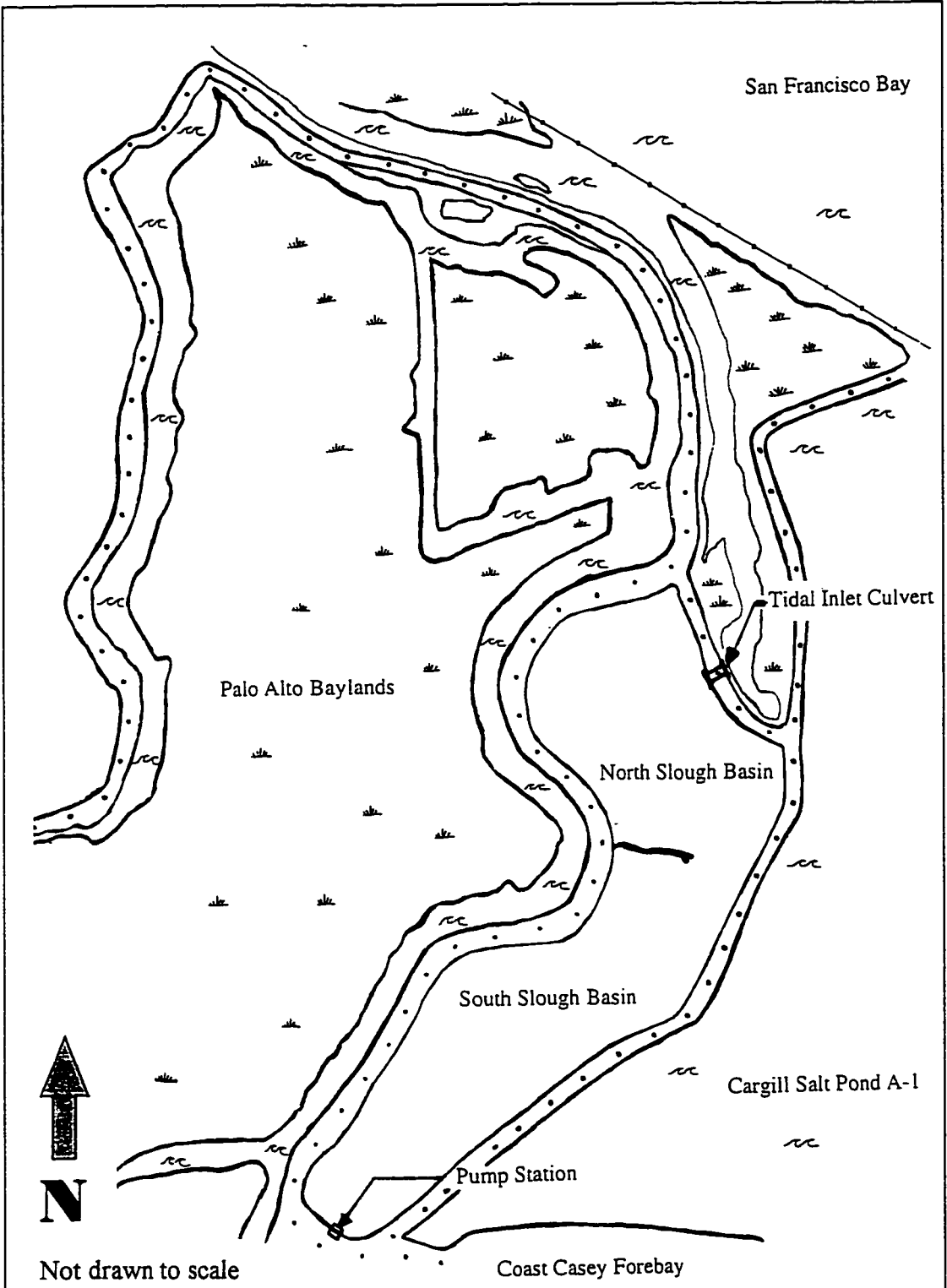


Figure 2. Diagram of Inner and Outer Charleston Sloughs showing the location of the north and south slough basins, outer slough, and surrounding land uses. The Palo Alto Baylands lie west of the Slough, Cargill Salt Pond A-1 to the east, Outer Charleston Slough and San Francisco Bay to the north, and the Coast Casey Forebay (City of Mountain View) to the south. Positions are indicated for the tidal inlet structure, pump station.

(Trulio 1995a). This action allowed partial drainage of the 103-acre mudflat, but a tidal range between 0.1' and 0.2' did not aid in slough sedimentation or vegetative colonization (Phillip Williams & Associates 1989; Trulio 1995a). The Slough remained a shallow water pond. The City plans to implement another restoration project designed to renew 53 acres of tidal marsh in the Inner Slough. The initial phases of this project are scheduled to begin in Fall 1996. The creation of 53 acres of vegetated pickleweed-cordgrass tidal marsh is expected within 3.5 to 13.5 years (Phillip Williams & Associates 1993).

### Related Research

Theoretical setting. The study of natural community changes over time was popularized by Clement's (1916) classic treatise on plant community organization and by Gleason's (1964) individualistic hypothesis, but has only recently been expanded to include estuarine macroinvertebrate communities (Yingst and Rhoads 1980). The role of inbenthic organisms in modifying and enhancing the physical and chemical properties of their environment following disturbance has been well documented (Simon and Dauer 1977; Rhoads et al. 1977). Much of the theoretical work on the succession of inbenthic communities has focused upon marine environments, but similar soil types and organism characteristics found in brackish salt marsh environments reveal trends that apply to each community type (Ricketts et al. 1985).

Until recently, studies of benthic communities were considered from a "steady-state" paradigm (Rhoads and Boyer 1982). This hypothesis of inbenthic community organization recognizes discrete assemblages of particular characteristic species. According to this hypothesis, inbenthic communities are classified by species that are spatially and temporally prevalent, dominate in terms of biomass, and are long-lived (Rhoads and Boyer 1982). The merits of classifying faunal associations based upon "dominant" organisms is attractive in its simplicity, and can be practical, as Ricketts et al. (1985) advise, when organismal arrangements are used for suggestive rather than definitive purposes.

The colonization of marine environments typically follows organized re-establishment patterns much like the upland vegetative succession counterpart. Pioneer inbenthic organisms alter the chemical and physical properties of the soils and surrounding water thus preparing the site for colonization of later successional stages. Salt marsh pioneer species are typically deposit-feeding polychaetes and mollusks (Rhoads and Young 1970; Rhoads et al. 1977; Simon and Dauer 1977; Trapani 1987).

The predominant salt marsh organisms, grouped by feeding type, are deposit-feeders and suspension-feeders, which depend upon deposited sediment or suspended sediment, respectively, as a primary food source. Although there is a potentially unrestricted supply of suspended sediment available in estuarine systems, suspension feeders (excepting many bivalves) generally have limited distribution within salt marshes. This occurs because unstable substrate tends to impede water

filtration structures, bury larvae, disturb larval settlement, and prevent many filter-feeders from attaching to the substratum (Rhoads and Young 1970). As a result, many suspension feeders are generally confined to sandy or firm mud bottoms, while deposit feeders are the dominant inbenthics found in muddy salt marsh substrates.

The rapid dispersal capabilities of pioneering deposit-feeders such as the tube dwelling polychaete worms Streblospio benedicti, Capitella capitata, and Ampelisca spp. (gammaridean amphipods), combined with their ability to survive in compacted soils with a shallow (< 2 cm) redox potential discontinuity layer (the boundary layer between aerobic and anaerobic soil layers; hereafter referred to as the RPD layer) allow them to colonize and rework upper soil layers quickly. Under normal conditions, McCall (1977) found colonies of small, opportunistic tube-dwelling polychaetes at densities of  $10^5 / \text{m}^2$  within ten days following a disturbance. Tubicolous amphipods have been noted to appear shortly after the initial polychaete expansion (Rhoads and Boyer 1982).

The sedimentary effects of deposit-feeding pioneer species include:

- 1) construction of dense tubal aggregations which may affect surface texture and microtopography;
- 2) fluid bioturbation (the pumping of water through vertically oriented tubes);
- 3) fecal pelletization of soils due to surface deposit-feeding; and,
- 4) an RPD layer limited to within a few centimeters of the surface (Rhoads and Boyer 1982). Constant physical reworking of soils occurs vertically, as soil and fecal materials are transported upward during feeding, and horizontally, through tube construction and burrowing. These activities alter soil reaction and solute diffusion

characteristics, creating a heterogenic assortment of biochemical microenvironments (Aller 1982).

Mucus secreted by inbenthic organisms serves to provide structural support to the tube network, and encourages bacterial growth, an important food source for many deposit feeders (Yingst and Rhoads 1980). Inbenthic organisms enhance soil conditions through the processes of irrigation, pelletization, tube construction, and mucus secretion, which affect water flow and erodibility of surface sediments, and by burrowing and feeding activities that rework soils (Rhoads and Boyer 1982; Kennish 1986). Cumulative changes in the habitat from these modifications brings an associated transformation in the faunal community as the site becomes better suited for secondary colonizers (Rhoads and Boyer 1982; Taghon et al. 1984).

Higher-order successional assemblages of the San Francisco Bay, which may include Nereis spp., Eteone spp., Glycera spp. (all polychaete worms), Urechis caupo (an echiuroid worm), and Macoma spp. (bivalves), are found where disturbances are less common. These "climax" communities typically occur in areas that are not subject to intense pollution, and where the bottom is not exposed to intense tidal action or storm perturbation. The sedimentary effects of equilibrium species in shallow water environments include: 1) vertical transfer of water and soil particles over distances of 10-20 cm; 2) intensive particle mixing, which tends to homogenize soils; 3) feeding from the bottom of tubes, which creates water filled voids; and, 4) an RPD layer located at depths greater than 2 cm from the surface and commonly to depths of 10-20 cm (Rhoads and Boyer 1982).

At any given site, the inbenthic community commonly appears as a temporal and spatial patchwork of different inbenthic species assemblages, each in different successional stages (Rhoads and Boyer 1982). If a physical soil disturbance affects only the upper few centimeters, pioneer species and equilibrium species can coexist in the same sediments. In contrast, competitive exclusion mechanisms such as trophic group amensalism, or "biological bulldozing," whereby pioneer species are plowed under or consumed by larger deposit-feeders, may exclude pioneer organisms from equilibrium species assemblages (Rhoads and Boyer 1982).

Equilibrium-type animal associations bring increased alterations to the physical environment as secondary colonizers work deeper soil regions and cycle soil materials toward the surface (Taghon et al. 1984). Laboratory and field studies have produced an impressive list of essential estuarine infaunal roles in the processes of soil transformation and ecological functioning. Burrowing activities have been shown to increase organic matter decomposition rates and encourage nutrient cycling (Aller and Yingst 1978). Locomotion and feeding activities affect soil texture through particle aggregation, particle mixing and mucus secretion (Rhoads and Boyer 1982). The production of surface fecal tube mats alters surface topography, and by physically binding and stabilizing bottom sediments, may decrease surface soil erodability and reduce turbidity (Rhoads and Young 1970; Rhoads 1974). The mixing and transport of particles has been shown to affect soil drainage characteristics and enhance interstitial gas and water circulation (Rhoads 1974; Aller 1982; PERL 1990). Dense aggregations of sedentary deposit-feeders, primarily

head-down feeding polychaetes, transfer enormous amounts of sediment upward to the soil-water interface which, importantly, can decrease compaction of surface sediments (Rhoads and Young 1970). The activities of inbenthic invertebrates also affect the rates and spatial patterns of solute diffusion, creating biogeochemical microhabitats rather than vertically stratified distributions (Aller 1982).

Invertebrate bioindicators. Because inbenthic invertebrates share such an intimate relationship with their physical environment we can readily estimate environmental quality using invertebrate bioindicators. The varied tolerance levels of different invertebrate species to environmental conditions and aquatic pollution (Filice 1954; Filice 1958; Henriksson 1969; Rhoads et al. 1977; Shaw et al. 1983; Douglas-Hill and Nelson 1992; Coull and Chandler 1992; Ward 1992; Papathanassiou and Zenetos 1993) makes them ideal for evaluating the long term health of estuaries and useful for identifying problem regions within salt marsh restoration projects.

A realistic illustration of the power of using invertebrate bioindicators in relation to more traditional monitoring methods can be made by considering the scope of many present salt marsh restoration monitoring programs. The most commonly monitored water quality parameters at salt marsh restoration projects do not attain the broad temporal and multivariate resolution available through inbenthic studies. For example, dissolved oxygen, water temperature, and turbidity are frequently studied environmental quality parameters that are vital to proper

ecological functioning (PERL 1990; San Francisco Foundation 1990). Unless monitored continuously however, these factors can only give a partial representation of actual estuarine health. Moreover, unmonitored parameters such as soil organics or soil compaction could escape detection and unknowingly limit ecosystem recovery. This is less likely, however, when the above studies are coupled with quantitative analysis of soil invertebrates. The presence of higher order invertebrate assemblages attests to overall marsh health, and provides an indirect means of overseeing unmonitored parameters such as soil organic matter content, soil pH, interstitial water salinity, dissolved oxygen, and sediment loading. Once an inbenthic support function deficiency has been confirmed, sampling of physical parameters may be necessary to identify the source of disturbance. Distributional studies of marsh invertebrates can also be used to identify localized problem areas within larger projects.

In a similar manner, the functional goal of establishing vegetation at a restoration project as a measure of ecosystem health does not provide as comprehensive a view of marsh health as invertebrate studies. Halophytic plants respond slowly to sporadic changes in environmental quality that readily affect faunal ecosystem components, and are thus poorer indicators of episodic environmental fluctuations. Because soil invertebrates are highly responsive to variations in dissolved oxygen, interstitial water salinity, desiccation, pH, temperature, and water turbidity, species composition analyses better reflect overall environmental quality compared with vegetative parameters. It follows that



vegetation acreage goals common to many restoration projects may erroneously indicate satisfactory ecosystem functioning. In addition, the sole use of vegetation recovery as a functional restoration goal can downplay the importance of mudflats and mudpans in healthy salt marsh ecosystems.

The use of invertebrates to identify episodic water quality fluctuations provides for "continuous" monitoring of parameters that influence faunal organizations. An abundance of sessile, long-lived equilibrium-type organisms at a site (Nereis, Eteone, Urechis, Macoma, etc.) implies a high quality environment and well-functioning ecosystem. Whereas, the presence of pollution indicators such as C. capitata, S. benedicti, and Nematodes, particularly in small numbers and in the absence of equilibrium species, should raise questions as to overall marsh health (Shaw et al. 1983). C. capitata thrive under conditions of pollution that few other animals can tolerate, and have thus been used widely in pollution analysis (Ricketts et al. 1985).

Previous researchers have found that vegetated marsh flats support infaunal species assemblages distinct from non-vegetated areas (Ricketts et al. 1985). Thus, comparisons of like habitat types should be practiced whenever possible.

### Study Parameters

In addition to directly observing soil invertebrates we can readily measure their effects upon subsurface environments using redox potential discontinuity depth, soil organic matter and other parameters as suggestive measures of inbenthic health.

This study compares several environmental parameters against invertebrate distribution to aid in understanding faunal interactions with their environment. Variables for this analysis were chosen because they relate to the functional goals of providing habitat and food chain support within the marsh community and because they convey the progress of the Inner Slough restoration project. It was also important that the selected parameters were relatively inexpensive to study, easily quantifiable in the field, and easy to process in the laboratory. The physical parameters selected for comparison with invertebrate characteristics were redox potential discontinuity depth and width, soil organic matter content, and soil texture.

Redox potential discontinuity depth was chosen for analysis because it provides a three dimensional view of habitat usage by soil invertebrates. Measured as a color change in silty-clay soils, RPD depth has been shown to be associated with successional assemblages, oxygen penetration, and the general quality of subsoil habitats (Rhoads et al. 1982). One widespread observation of the RPD-faunal relationship holds that macroinbenthic animals utilize upper oxygen-rich layer soils, and only venture occasionally to the anoxic regions below the RPD layer.

Evidence presented in this study and by previous workers indicate that RPD depth can be a quick and inexpensive indicator of the presence of secondary colonizing species. Rhoads and Boyer (1982) note that RPD depths greater than 2 cm from the surface and commonly to depths of 10-20 cm typically coincide with the presence of larger secondary colonizing species. RPD depth measurement is an indirect measure of soil oxygen penetration, which is a function of soil composition.

soil texture, soil bioturbation (biological mixing activity), and the quality of overlying waters.

In addition to the use of RPD measurements as indicators of community health, this study examined their usefulness as measures of restoration progress. The application of redox measurements to salt marsh restoration studies have usually focused upon measurements taken with a redox probe to analyze the biogeochemical cycles of nitrogen and sulfur, and mobility of heavy metals (PERL 1990). The relative simplicity of RPD depth measurement, and potential applications to restoration monitoring make this technique worth evaluating. The study of RPD depth as an indirect measure of subsoil habitat quality has been based largely upon theoretical discussions of organism-sediment relations and infaunal succession (Rhoads and Boyer 1982), with limited practical application to restoration monitoring. If shown to be an effective measure of community health, the relative ease and speed of RPD width and depth measurement, compared to other parameters such as organic content, invertebrate composition, and redox measurements, make this measure ideal for rapid assessment of wetland inbenthic conditions.

Another easily measurable variable in silt-clay salt marsh soils is RPD width, which may be indicative of faunal activity at a site. RPD width has not received the extensive literature attention afforded to RPD depth, but nonetheless may be important as it adds dimension to our understanding of invertebrate habitat use. This study examines the usefulness of this measure in context with salt marsh restoration.

Organic matter content was selected for study because of the ubiquitous presence of remnant pickleweed roots at Inner Charleston Slough, and their unknown effects upon estuarine restoration progress. Percent weight loss on ignition (LOI) can be used as a gross indicator of soil organic matter content provided the soils being compared are of similar origin and composition (Lewis 1994a). Soils with high organic matter have been shown to develop low redox potentials which result in low soil pH and low soil oxygen, and inhibited growth of some marsh plants. Low organic levels can result in limited nutrient availability and limited plant growth (PERL 1990). The history of Charleston Slough as a drowned pickleweed salt marsh makes the question of soil organic matter content interesting and relevant to understanding current slough conditions.

Sample weight loss during ignition may occur due to a multitude of reasons and is intended as only a rough guide to the amount of soil organics present. It is assumed that weight loss occurring due to external factors other than organic matter combustion are consistent throughout the sample series, and that the combined effects of loss of lattice water ( $H_2O$ ), loss of  $CO_2$  from carbonates, oxidation of ferrous iron to ferric iron (a weight gain), potential loss of sulfur, and the loss of other minor volatile components (Lewis 1984a) is equivalent, or the weight loss due to these factors between successive samples is small enough to be non-significant.

Lewis (1984a) notes that there is little practical use for determinations of LOI because the contributory causes of sample weight loss cannot be accurately quantified. However, this procedure seems very useful as a suggestive index for

comparing samples that differ little in origin or mineral composition. While the researcher may not know the absolute value of organic matter loss, or the extent that unknown factors have contributed to the final value, when used with discretion, LOI analysis can be a useful index to organic matter with practical applications in identifying field sites that warrant further attention.

Sediment grain size distribution was analyzed to develop a particle size distribution profile for the natural and restored marshes and to examine invertebrate use of different soil textural classes. Particle size has been shown to be an important influence on soil functioning, soil nutrient flux, sediment transport processes, and composition of inbenthic organisms (Rhoads 1974; Rhoads et al. 1977; Rhoads et al. 1982; Taghon et al. 1984; PERL 1990). Grain size distribution is also useful in examining hydrologic conditions and identifying problematic marsh regions.

Pipette size analysis is the traditional method for determining the size distributions of fine sediments ( $< 62 \mu\text{m}$ ), or in this case, the relative fractions of sand, silt, and clay (Folk 1968; McManus 1988; Anderson 1996), although other methods such as hydrometer analysis are sometimes preferred (PERL 1990). For the purposes of this thesis, only the comparative fractions of sand (particles larger than  $4\phi$  in size)<sup>1</sup>, silt (particles between  $4\phi$  and  $8\phi$  in size), and clay (particles  $8\phi$  and smaller) were desired (Krumbein 1934; Krumbein 1936).

Macroinvertebrate studies were chosen for this analysis because, unlike many other parameters that only provide a quick snapshot of habitat conditions at

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<sup>1</sup> Phi ( $\phi$ ) units are logarithmic expressions of particle diameter.  $\phi = \log_2 d$ , where  $d$  is the grain diameter in millimeters (Krumbein 1934).

Charleston Slough, invertebrate surveys incorporate a temporal element that illustrates a long-term measurement of community health. Judging from the distribution, abundance and composition of pollution sensitive long-lived animals such as Eteone and Nereis, and rapidly colonizing pollution indicator species like C. capitata and S. benedicti, a biotic index can be created that follows the marsh stability over time. The study of faunal parameters in conjunction with the previously described environmental factors is an effective means for characterizing the health of Inner Charleston Slough, understanding restoration difficulties, and enhancing the marsh in the future.

#### Common South San Francisco Bay Infauna

Although the focus of this thesis is not the taxonomy or ecology of estuarine species, a short list of representative salt marsh invertebrates is provided to familiarize the reader with the most common inbenthic species. This list is by no means exhaustive. The Annotated Systematic Index and General Bibliography in Ricketts et al. (1985), and Smith and Carlton (1975) are excellent sources for more detailed discussions of invertebrate ecology and natural history. The benthic infauna of Inner Charleston Slough have not been thoroughly studied, but many similar studies exist on native and introduced species in San Francisco Bay (Filice 1954; Filice 1958; Smith and Carlton 1975; Allen 1971; Ricketts et al. 1985; Trapani 1987).

Most marine worms found in the soft salt marsh substrate belong to the annelid class Polychaeta, and are "errant," or free-living forms (Hartman 1968).

These errant polychaetes move about freely on the mudflat surface or through the muddy substrates consuming detritus and deposited sediment. However, some species are active carnivores. The large predatory neriid worm, Nereis procera, and the blood worm, Glycera robusta, are common to Bay mudflats. Each has powerful chitinous jaws and a protrusible pharynx and relies upon a diet consisting of smaller worms and other invertebrates (Ricketts et al. 1985). Native deposit feeders (sediment eaters) are represented by several species of Polydora, the lugworms Abarenicola spp. and Arenicola spp., the ubiquitous C. capitata, Eteone spp., and tubificid (sedentary annelid) worms. Arenicola creates a U-shaped burrow through which it induces water currents by peristaltic body contractions. Through mud burrowing processes, soil invertebrates such as Arenicola escape detection from predators, avoid desiccation and wave action, and are virtually surrounded by their food supply (Ricketts et al. 1985). Many introduced deposit-feeding annelids, including Polydora ligni, S. benedicti, and Neanthes succinea, which apparently arrived in the 1800's, now exhibit cosmopolitan distribution (Carlton 1975).

Intertidal bivalves are chiefly represented by clams and mussels in San Francisco Bay mud bottomed salt marshes. In the soft sandy mud, clams bury themselves between a few centimeters deep (as does the minute Atlantic gem clam, Gemma gemma) or burrow up to 1.3 meters (for example the geoduck, Panopea generosa) (Ricketts et al. 1985). Diversity in digging ability and siphon length, ability to withstand different types of pollution, tolerance for water turbidity and desiccation, and soil texture preferences, among other factors, dictate the distribution

of bivalves within the marsh. These animals are generally more limited in distribution than annelids, occupying the low to middle intertidal zones, below approximately 2.0 feet National Geodetic Vertical Datum (NGVD).

Other incidental mudflat bivalves include the native jackknife clam, Tagelus californianus, the gaper clam Tresus nuttallii, the Washington clam, Saxidomus nuttalli, and the bay mussel, Mytilus edulis (typically found on rocky upper intertidal shores). The non-native Atlantic gem clam, Gemma gemma, measuring only a few millimeters in length, exhibits a shallow yet catholic distribution in both healthy and disturbed Bay mudflats. The introduced Atlantic soft shelled clam, Mya arenaria, is a larger, commercially-important species that thrives in polluted estuary mud (Ricketts et al. 1985). Another east coast denizen, the Atlantic ribbed mussel, Ishadium demisum, clings in masses by means of byssal threads to marsh vegetation and exposed roots along embankments of the upper intertidal. Ricketts et al. (1985) reports that the endangered clapper rail is fond of this new food source, however, a mussel occasionally grasps an unlucky bird, which founders in the incoming tide. There are even rare accounts of rails choking upon the unique V-shaped shells of I. demisum.

Perhaps the most common San Francisco Bay clam, Macoma balthica is evidently native to the west coast from Puget Sound northward, and was brought to California by prehistoric man (Carlton 1975). This introduction conceivably displaced M. balthica's closest Bay Area relatives: the bent-nosed clam, Macoma nasuta, and the white sand clam, Macoma secta. Considerable study of ecologically



similar native and introduced species occupying overlapping geographic regions supports the theory that exotic species may contribute to the decline or competitive displacement of native fauna (McDowall 1968; Moyle and Nichols 1974; Moyle 1981; Race 1982), particularly in the presence of environmental disturbance (Orians 1984). Limited studies have identified hundreds of introduced invertebrate species, but the total extent of these biological invasions and associated ecological consequences upon native species is not known.

Ecological studies of the introduced gastropod Ilyanassa obsoleta have revealed dramatic changes in resultant native species distributions. One study of habitat utilization indicated that the native mud snail, Cerithidea californica, is limited to only a portion of its former range in San Francisco Bay by the introduced ecological equivalent, I. obsoleta (Race 1982). Native to tidal creeks, tidal pools and mudflats of San Francisco Bay, the tall-spired horn snail, C. californica, is presently restricted to tidal pools (Race 1982). I. obsoleta further discourages its competitor by eating the eggs and juveniles of indigenous species (Ricketts et al. 1985). The effects of another ecologically similar non-native gastropod, Batillaria attramentaria, upon C. californica populations are more poorly understood. Other notable gastropod introductions include the carnivorous east coast oyster drill, Urosalpinx cinerea, and the small pulmonate snail Ovatella myosotis (Carlton 1975).

Beginning primarily with 19th century maritime activities associated with the Gold Rush, the last 200 years have ushered in an age of invertebrate invasions as ships en route from Asia and the Atlantic coast via South American waters streamed

toward the California coast. Maritime activities were a great source of organism influx as incoming vessels dumped water ballast tanks teeming with planktonic invertebrate larvae into San Francisco Bay (Carlton 1975). Starting in the 1860s, the commercial oyster industry was another principal source of invertebrate introductions from the Atlantic Coast and Japan into the San Francisco Bay (Carlton 1975). Unfortunately, studies of species introductions indicate that, all too often, the presence of non-native fauna leads to geographical limitations of native species throughout their habitats (Race 1982; Orians 1984).

#### Problem statement

Comparisons of ecological functioning between artificial and natural salt marshes at the Sweetwater Marsh National Wildlife Refuge, San Diego, CA, show that after five years, replacement ecosystems are only marginally effective at approximating natural systems (Zedler et al. 1991). Zedler et al. (1991) found functional deficiencies in 10 out of 11 marsh parameters studied including epibenthic (above ground) invertebrate abundance (36% of normal), epibenthic invertebrate species diversity (78% of normal), and soil organic matter content (51% of normal). Inbenthic species were not addressed in Zedler's studies, but her findings suggest impaired nutrient cycling and biological functioning that would affect ground dwelling communities.

In a similar study, Sacco et al. (1994) found that inbenthic organisms and trophic group proportions were comparable between 1 to 17 year old artificially

established North Carolina salt marshes and natural marshes. However, they noted lower total organism and trophic group densities within the artificial marshes. Organic matter content at the oldest artificial marshes was approximately half that of the normal marshes (Sacco et al. 1994). Charleston Slough is not an artificially established salt marsh per se, but its prior usage as a salt evaporation pond and modified hydrologic conditions warrant investigation into invertebrate support functions and other functional parameters.

It is imperative to understand the physical and biological consequences of utilizing single culvert "pocket marsh" hydrologic designs and muted tidal regimes, as future Bay Area wetland gains through restoration of salt ponds and diked areas are likely to incorporate these procedures. Pocket salt marshes are defined here as linear hydrologic systems where water influx occurs through a single opening, usually a tidal control structure. Tide gates typically control the rate of flow into the pocket marsh and fresh water inputs are limited, except for precipitation. Examples of such restored salt evaporation pond pocket marshes can be found throughout the South Bay Area and include New Chicago Marsh, Alviso, Charleston Slough and the Mountain View Tidal Marsh, Mountain View, and the Tract 102 Marsh, Newark (Albertson 1996).

Several interrelated complications associated with restoring full tidal regimes in salt evaporation ponds often necessitate creation of muted tidal hydrologic systems. Ground subsidence, poor internal levee integrity, and flooding liability are substantial barriers to restoring natural tidal regimes. Because salt evaporation

ponds and surrounding developed lands have often subsided to below sea level. ponding and potential for flooding of surrounding urban communities are recurrent problems.

New Chicago Marsh (NCM). Alviso exemplifies the extreme environmental conditions typically found at many salt evaporation pond restoration sites. The 200-acre salt marsh is connected to south San Francisco Bay by a mile-long channel. Limited salt water influx occurs during high tide through the single 24-inch Waterman control gate. Thermally polluted, sediment-free water trickles slowly to the marsh, which is located at 3-5 feet below sea level; normal water levels at NCM average -5.0 feet NGVD. pH is less than 5 in some regions, and excessive salinity is a problem (Nuttall 1986). Attempts to raise marsh elevations at NCM through flooding and natural sediment buildup are not feasible because they conflict with federal and state mandates for protecting the endangered salt marsh harvest mouse, Reithrodontomys raviventris. Flooding could drown the pickleweed vegetation and make the marsh uninhabitable for the mice. If tidal inundation were an option, the integrity of NCM levees would be an issue, as the neighboring City of Alviso is also below sea level.

To further complicate matters, NCM lies within the City of San Jose's flood basin, making full marsh inundation and emergency pumping a regular event. City managers are uneasy with inherent liability issues that arise from full tidal regimes, and the inordinate costs associated with raising levees. Because similar conditions are found at salt evaporation ponds throughout San Francisco Bay it seems safe to

project that many forthcoming salt marsh restoration projects will be allowed only partial tidal regimes.

As illustrated by NCM and Charleston Slough, many salt pond restoration projects combine the effects of single culvert hydrology, linear marsh design, and lack of freshwater inputs (except during extreme flood events) creating distinct physical conditions generally not encountered at natural salt marshes. The effects of partial tidal regimes and tidal trickling upon functional salt marsh recovery is a poorly understood subject, demanding much closer examination. The present study provides one way to investigate the effects of linear tidal circulation and muted tidal regimes upon macrobenthic invertebrate populations, and thus serves as a means for understanding the overall health of these systems.

Evaluating the functional capacity of a wetland following a restoration project is key to understanding its processes. Assessment of a project not only allows fine tuning of variables involved during different phases of restoration, it also provides valuable hindsight which is essential to the creation and refinement of future designs.

Because conditions such as the inundation period, tidal levels, sediment composition, soil temperature, organic matter content, and soil moisture content of the Inner Slough likely vary substantially from Outer Slough conditions (Filice 1954; Filice 1958; Rhoads and Young 1970; Aller and Yingst 1978; Kristensen and Andersen 1992), it is reasonable to expect distinct biotic associations at each locale.

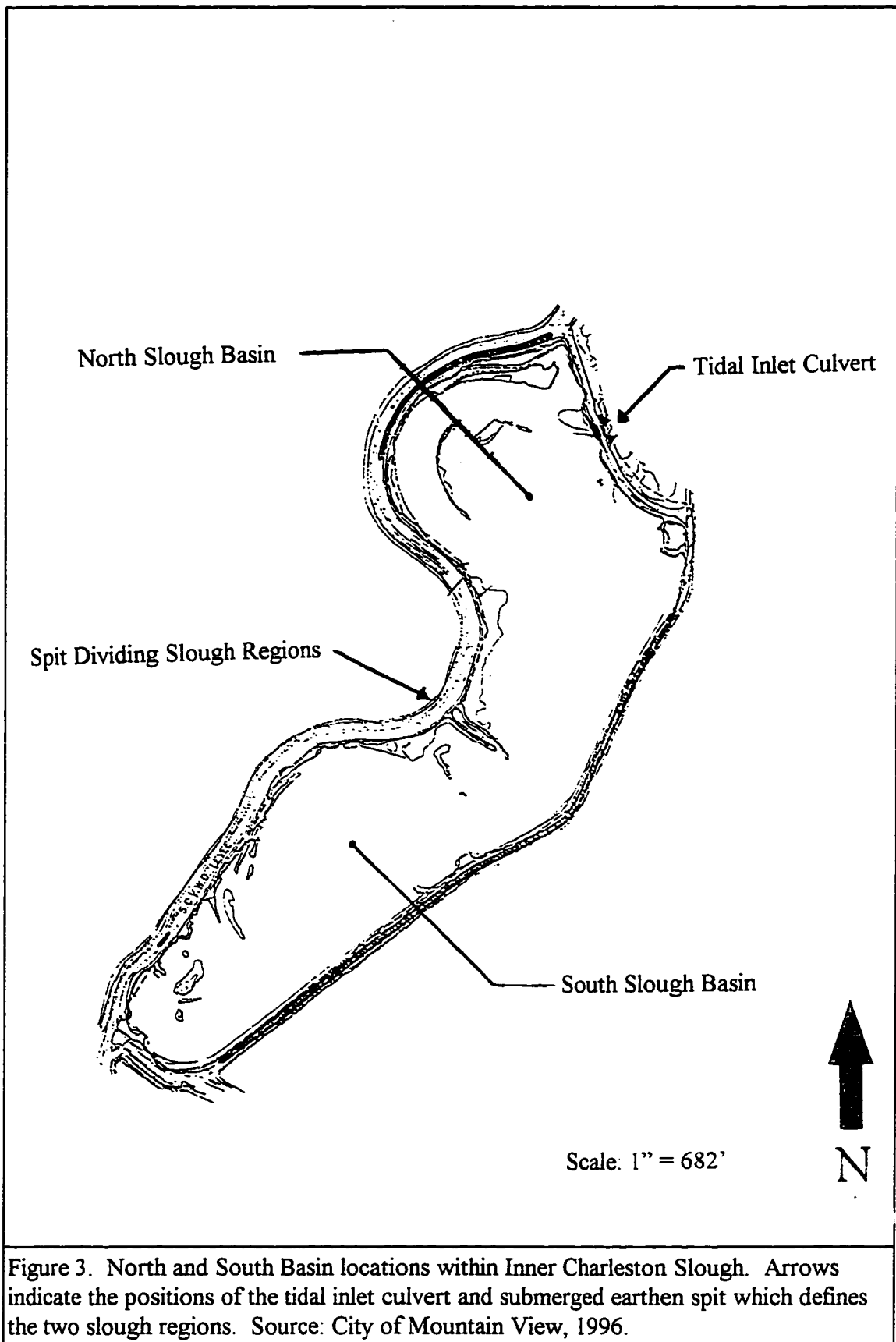
### Research Questions

This study was performed to examine the spring season infaunal communities associated with an impounded water wetland in the South San Francisco Bay. There were four primary objectives: (1) to examine and quantify diversity, abundance and distribution of the spring season infaunal communities associated with the mudflats and mudpans of Inner and Outer Charleston Sloughs, impound water and natural tidal salt marshes, respectively; (2) to determine to what extent the inbenthic macrofauna of Inner Charleston Slough resembles that of the Outer Slough and examine regional differences in substrate composition, soil organic matter, RPD depth, RPD width, species composition, richness and evenness; (3) to investigate relationships between species composition and physical factors; and, (4) to determine whether inbenthic bioindicators and the selected physical variables, RPD depth and RPD width, are feasible indicators of marsh restoration progress.

This analysis examines the effectiveness of the 1984 Inner Charleston Slough restoration project, and provides a baseline for future studies of infaunal succession at this site. This study also provides data and a framework for analyses of muted tidal marsh enhancement programs, and their effectiveness in supporting inbenthic communities.

Because Inner Charleston Slough is a linear pocket marsh with varying conditions found throughout, it was hypothesized that inbenthic organizations differed between the northern and southern portions of the Slough. Conveniently, a submerged earthen spit partially bisects the Inner Slough, creating two comparably

sized regions which will be referred to here as the North and South Slough Basins. During summer, the South Basin experiences occasionally anoxic conditions due to algal growth, and higher soil and water temperatures compared with the North Basin (Trulio 1995a). It was postulated that poor water quality condition the South Basin limited infaunal associations to shallow pioneer species, with few unevenly distributed equilibrium type species. Conversely, the North Basin, being closer to oxygen rich Bay waters, was postulated to support more equilibrium-type assemblages. To test this hypothesis the Inner Slough was analyzed as two separate regions to identify potential differences in the physical environment and faunal distribution. Figure 3 displays the two sample regions.





## METHODS

### Description of the Study Areas

The research site is a large restoration project at Charleston Slough, Mountain View, California. Inner and Outer Charleston Sloughs are brackish-water salt marshes that support halophytic vegetation exclusively. The Inner Slough is surrounded by levees ranging between 4.0' to 9.0' NGVD, and is approximately 5%-10% vegetated with perennial pickleweed (Trulio 1995a). Termed a "pocket marsh," this linear hydrologic system receives no watershed inflow, a characteristic component of many natural South Bay salt marshes. The tidal inlet culvert located in the North Basin is the sole water source, except for occasional rainfall.

Inner Charleston Slough is a shallow salt pond with a surface elevation between about -0.5 to 0.0 NGVD. The ground surface is nearly level, with shallower reaches occurring in the North Basin, along marsh fringes, and in the vicinity the submerged spit that bisects the Inner Slough. Deeper zones occur near the water inlet structure, in the northeast slough elbow, and near the pump station.

Water removal from the South Basin varies seasonally, and presently occurs at a rate of about 3-5 million gallons during a 6-12 hour operational day (Gasser 1992; Trulio 1995b). This removal is required to maintain water quality in the City of Mountain View's small boat lake. Pumping necessitates permanent flooding of some slough areas to maintain flow to the pump station. Circadian water levels in the Slough typically fluctuate between about 0.1' and 0.2' (Trulio 1995a).

### Selection of Study Sites

Inner Charleston Slough was selected for this study for several reasons:

(1) this area presents a unique opportunity to examine environmental and biotic conditions at a restoration site with muted tidal action and impounded water conditions; (2) the Slough is a representative South San Francisco Bay salt evaporation pond which can serve as a model site for future salt pond restorations; (3) analysis of this site will aid in the understanding of this unique marsh, and enhance opportunities for future scientific study of muted hydrologic systems and impounded water salt marshes; and, (4) due to the scheduled fall 1996 Inner Charleston Slough restoration project, present faunal and environmental conditions portray the most advanced state of the 1984 Charleston Slough enhancement project, and the final opportunity for its study.

Criteria used in selection of an appropriate natural tidal marsh for comparison with Inner Charleston Slough included immediacy to the Inner Slough, relative size of the natural marsh, degree of natural functioning still intact, and similarity of hydrologic characteristics such as the lack of freshwater inputs. Although it is somewhat smaller than the Inner Slough, Outer Charleston Slough was selected because it fulfills the above requirements, and because the Outer Slough is the only source for water and organismal influx at the Inner Slough.

### Sample Collection

Four environmental parameters were quantified at each field site for

comparison with infaunal characteristics: (1) redox potential discontinuity depth (RPD depth); (2) redox potential discontinuity width (RPD width); (3) organic matter fraction using LOI analysis; and (4) analysis of sand/ silt/ clay ratios using pipette analysis.

Selection of field sites at the Outer Slough control region, North Basin, and South Basin was done using a completely randomized design. Twenty-five samples were taken to characterize each slough region, for a total of 75 samples. Samples were grouped for analysis of regional differences.

Sampling occurred over a 24-day period according to the following schedule:

<u>Inner Charleston Slough:</u>	
South Basin:	March 22, 24, 1996
North Basin:	March 25-27, 1996
<u>Outer Charleston Slough:</u>	March 30-31, April 12-14, 1996

There were no storm events during the 24-day sampling period that would have caused invertebrate mortality, and no apparent evidence of organism migrations. Conceivably, a favorable tide, or the full moon event on 3 April 1996 could have reduced the abundance of sexually mature Nereids or other polychaete groups. This is unlikely however, because all polychaete groups, excepting S. benedicti, were more common at the Outer Slough, which was the last region sampled. Such reductions would positively contribute to already higher infaunal abundance found at the Outer Slough.

### Sampling Restrictions

Although there was an abundance of cordgrass (*S. foliosa*) and pickleweed (*S. virginica*) vegetation in the Outer Slough, this study only considered non-vegetated areas. The presence of vegetation has been demonstrated to affect infaunal species assemblages (Ricketts et al. 1985). Consequently, this study only attempted faunal comparisons between corresponding habitat types: namely mudpans, mudflats, and within unvegetated channels.

Two faunal groups encountered at Charleston Slough were not included in this study: Foraminifora and Gastropoda. Foraminiferans were excluded because of their small size, extreme numbers at some sites, and because living organisms could not be distinguished from empty shells. Epibenthic gastropods (mostly Nassariidae) were not included because of the inbenthic emphasis of this study. No attempt was made to determine shrimp identifications beyond the subphylum Crustacea (except for Ostracoda and Amphipoda).

Desiccation is a major factor influencing the zonal distribution of invertebrates. Thus, for comparable results with Inner Slough samples only Outer Slough mudflats between 0.0'-3.0' NGVD were sampled.

Field transportation was carried out by two means: boat and wooden plank. Because the Inner Slough was ponded during the course of this study, movement within this area was possible only by boat. Water depths within the Inner Slough ranged between 15 cm and 70 cm, with the shallowest areas reachable only during high tides. Restricted boat maneuverability along marsh fringes made some areas

inaccessible for study. Waters less than 10 cm in depth occurred approximately 5-25 m from shore and were not accessible for sampling. Sampling near islands presented no such problems.

Samples were taken in Outer Charleston Slough only during low tide periods. Movement in this area was accomplished using wooden sheets. The northernmost sampling boundary in Outer Charleston Slough was determined by safety and mobility considerations. The larger mudflats (more than 25 meters across) found near the bay could not be traversed safely using wooden sheets and are thus excluded from this study. Similarly, due to mobility considerations, only the Outer Slough west bank was sampled. These relatively minor restrictions did not prevent the collection of representative and random samples.

### Study Parameters

Redox potential discontinuity depth and width. RPD depth and width samples were collected using a one-inch diameter transparent plastic soil sample liner available from the Ben Meadows Company, Atlanta, Georgia. The liner was inserted into the substrate to a depth of 25 cm, the top plugged, and the soil column removed for analysis. The redox potential transition was read as a change in soil color, generally manifested as an abrupt transition between a lighter brown oxygen-rich soil layer overlaying a darker sulfurous layer.

RPD depth was interpreted as the distance from the soil surface to the upper portion of the RPD band. The top of the RPD band was chosen for the depth

measurement because it is totally independent of the RPD width measurement (Figure 4). Measurements were recorded to the nearest 0.1 cm. RPD width was determined by measuring the region of overlap between light and dark soil layers to the nearest 0.1 cm. At each sampling station four RPD depth and width measurements were made within a 1.0 m<sup>2</sup> area, which were averaged to create a mean value for each site.

In addition to the 25 RPD depth and width samples taken in each region for statistical analysis, an additional 25 randomly selected sites were sampled within each region to develop a distributional profile of potential marsh problem areas.

Soil compaction during RPD depth and width core recovery was not accounted for in this study. Compaction was assumed equal among successive samples. Lewis and McConchie (1994a) cite a 13%-20% rate of soil compaction during core sampling; the equivalent of 2-3 cm compaction/ 15 cm depth sample.

Soil organics fraction. One sample was taken at each of the 75 collection sites for soil weight loss on ignition (LOI) analysis. For each sample, the 1" diameter soil sample liner was inserted into the substrate to a depth of 15 cm, the normal range of invertebrate intrusion, and the soil column removed. Samples were sealed in Ziplock storage bags, homogenized, and stored for analysis.

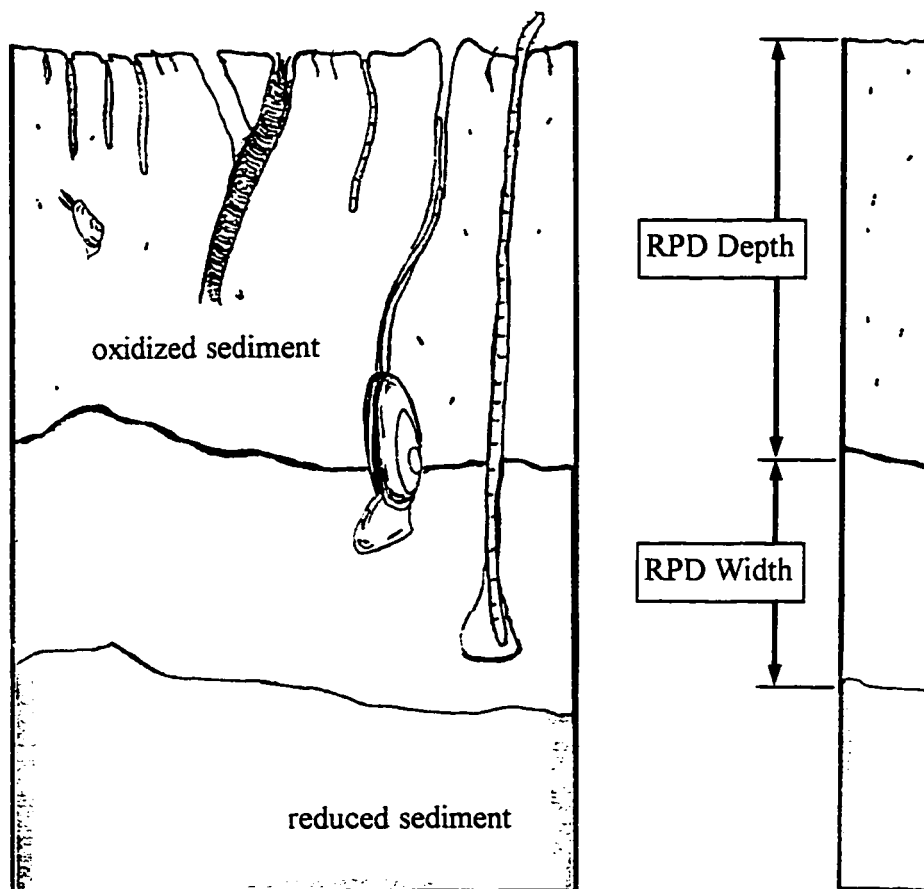


Figure 4. Illustration of redox potential discontinuity depth and width measurement. RPD depth was interpreted as the distance from the soil surface to the upper portion of the RPD transition layer. RPD width was determined by measuring width of the transitional layer between light aerobic soils above and dark anoxic soils below. The left drawing shows a hypothetical soil profile, with a corresponding sample core on the right.

Weight loss on ignition was determined by monitoring weight loss upon sample combustion in a 450° C furnace (Lewis and McConchie 1994a). Furnace temperatures higher than 450° C were avoided to prevent driving bound water off of clay particles, and safeguard against mineral degradation (Andersen 1996). High temperatures have been reported to degrade calcium carbonate, resulting in over-reporting of the sand/shell fraction (PERL 1990).

For each sample, five grams of wet substrate was spread in a thin layer upon a preweighed ceramic crucible and heated overnight, approximately 12 hours, in a 105° C drying oven to evaporate to dryness (Lewis and McConchie 1994). To avoid problems associated with initial water absorption, samples were cooled to room temperature and allowed to equilibrate two hours to reach ambient moisture levels (Folk 1968). Samples were weighed to 0.001 g in the same order as they were removed from the oven.

To ash organic material, samples were placed in a room temperature furnace and heated quickly to 450° C over a 30 minute period. The samples were baked at 450° C for 5 hours. Cooled samples were allowed to equilibrate to ambient moisture levels, weighed, and the percent weight lost recorded.

Soil texture analysis. The relative fractions of sand, silt and clay were determined for each site by pipette analysis based upon Folk (1968). For this technique, five grams of wet soil were placed in a beaker and covered with 25 ml of 30% hydrogen peroxide to digest organic material. Larger root particles were



cleaned with deionized water and removed. Oxidation was complete within two weeks, at which time the samples were baked at 95° C for 12 hours to remove any remaining oxidant. Soil samples were transferred to a Waring blender where 20 ml of 0.309 N sodium hexametaphosphate dispersing agent and 100 ml of deionized water were added, and the mixture blended at low speed for 10 seconds.

Sand size particles were collected from the mixture using a 63 micrometer (0.025 inch) soil sieve and deionized water. The sand fraction was placed in a preweighed beaker, dried overnight, allowed to stabilize to room moisture levels for two hours, and weighed to 0.001 g.

The remaining solution was placed in a 1000 ml graduated cylinder, brought to 900 ml with deionized water, and observed overnight for sediment flocculation. The solution was then brought to 1000 ml and thoroughly mixed with a piston-type stirring rod. Upon setting 20 seconds, a 20 ml pipette aliquot was drawn at 20 cm depth to determine the combined fraction of silt and clay. A second aliquot was taken at 1 hour 51 minutes at 10 cm depth to determine the fraction of clay in the sample. Relative proportions of sand, silt and clay were calculated as described in Folk (1968), correcting for the dispersant weight.

Invertebrate collection and identification. One sample was collected at each of the 75 field sites for macrobenthic invertebrate analysis. For each sample a 0.035 m<sup>2</sup> (6 inch diameter) hand held PVC coring device was used to a depth of 40 cm. Twenty-five invertebrate cores were taken in each of the three sample areas for a

total area of 0.875 m<sup>2</sup> per area (2.635 m<sup>2</sup> total area). Samples were bagged in the field and screened within 24 hours using a 0.512 mm No. 30 standard sieve. This mesh size is consistent with Reish's (1959) observations which indicate that quantitative determinations of infaunal composition should employ a fine resolution screen. The chosen mesh size should yield samples with close to 100% of macroinbenthic species and over 95% of macrofaunal biomass (Dauer and Simon 1976). For a thorough discussion of the importance of screen size in sample washing, and taxonomic sufficiency as it relates to mesh size see Reish (1959) and Ferraro and Cole (1992). Organisms were preserved using a 70% isopropyl alcohol solution and stained with rose bengal.

For laboratory analysis an initial infaunal screening was performed through a 4.0 mm sieve to remove large animals and debris that could not be easily pipetted. The material that washed through the screen was subsampled to determine species composition and estimate abundance of smaller organisms. For this procedure, samples were placed in a graduated cylinder, brought to total volume of 1000 ml, and stirred to evenly distribute suspended particles. While continually mixing, a 15.0 ml aliquot was drawn with a large-mouthed pipette. In this manner, five additional subsamples were drawn for a total volume of 90.0 ml.

Species composition and abundance data from the six subsamples were combined within each site, and extrapolated to a total volume of 1000 ml (multiplying by a factor of 11.11). Species were separated, identified to the lowest possible taxon and counted. Species data from the pipette analysis were summed

with results from the 4.0 mm sieve for a final estimation of infaunal characteristics. Taxonomic classification of infaunal groups was based upon Ricketts et al. (1985), Smith and Carlton (1975), Hartman (1968, 1969), and Gosner (1971). Species group identifications were confirmed by W. E. Bros and V. C. Kenk, San Jose State University, Biology Dept.

Evenness was determined by the Shannon-Weiner diversity index:

$$\text{Evenness, } J' = H' / H'_{\max}, \text{ whereby } H' = - \sum_{i=1}^k p_i \log_2 p_i$$

and  $H'_{\max} = \log_2 k$ . In this equation  $k$  is the number of categories,  $p_i$  is the proportion of observations in category  $i$  (Zar 1996).

The Shannon-Weiner evenness index is a measure of taxonomic heterogeneity within a site and resource sharing among organisms. An evenness value of 1.0 indicates an equal proportion of organisms at a particular site, with values decreasing toward zero as dominant organisms emerge. The index is especially responsive to changes in species of moderate abundance, and changes in importance of rare species (Westman 1985).

### Statistical Analysis

Soil characteristics and infaunal species data were grouped within sample regions to facilitate comparison of conditions between the Inner and Outer Slough regions, and between the North Basin and South Basin regions.

To reduce variance associated with non-normal data distributions, species abundance data were transformed prior to multivariate analysis using a  $\text{Log}_{10}$

function. All fractional data (sand, silt, clay, and organics) were transformed prior to analysis using an arcsin function.

Pearson Product-Moment correlation analysis (Zar 1996) was performed to examine whether the presence or absence of at least one equilibrium-type faunal group, Macoma balthica, Nereis procera, or Arenicola spp., was correlated with RPD depth or RPD width.

Pearson analysis was performed to examine relationships between RPD depth, RPD width, soil organics fraction, percent sand, silt and clay, species richness (the number of species groups per site), and evenness, and to reduce multicollinearity between variables used in multivariate analyses. A 60% correlation level was considered a strong relationship. The criteria used to eliminate the variable "percent silt" and retain "percent clay" in multivariate analyses were, first, biological significance, and second, highest coefficient of dispersion value. A one-way multivariate analysis of variance (MANOVA) was used to examine regional differences in the dependent variables: species richness, species evenness, abundance of C. capitata, S. benedicti, Nematoda, and G. gemma, RPD depth, RPD width, soil organic matter fraction, and clay and sand fractions (Tabachnick and Fidell 1996). The significance levels for all statistical tests was  $\alpha = 0.025$ .

Canonical correlation was used to determine the best possible relationships between faunal attributes and environmental variables. Two separate canonical correlation test series were performed. The first analysis examined relationships between environmental variables (RPD depth, RPD width, soil organics fraction, soil

clay and sand fractions) and species evenness and richness. The second analysis studied the interplay of environmental variables with the four numerically dominant species groups: C. capitata, Nematoda, S. benedicti, and G. gemma (Thompson 1984).

## RESULTS

The taxonomic composition of Charleston Slough infaunal macrofauna was very similar between the three study regions. An extrapolated total of 60,095 organisms representing 14 taxonomic groups was enumerated from the 75 samples taken at Charleston Slough. Mean infaunal density ( $\pm 2$  SE) of all three sites combined was 26,065 ( $\pm 155$ ) organisms /m<sup>2</sup> (n=75). Among the sampling regions the greatest infaunal abundance occurred at the natural Outer Slough with 36,562 ( $\pm 559$ ) organisms /m<sup>2</sup>. Mean infaunal density was 18,510 ( $\pm 328$ ) organisms /m<sup>2</sup> at the South Basin, and 13,608 ( $\pm 124$ ) organisms /m<sup>2</sup> at the North Basin. The mean Inner Slough abundance was 16,059 ( $\pm 226$ ) organisms /m<sup>2</sup>. Though only 14 taxonomic groups were enumerated in the study, the composition of Charleston Slough invertebrate taxa was comparable to findings of other San Francisco Bay and Pacific coast estuarine studies (Appendix 1)(PERL 1990; Josselyn et al. 1984; Allen 1971; Jones 1961; Filice 1958).

Between the three slough regions, four prominent taxonomic groups comprised 95.8% of total infaunal abundance. The most numerically prevalent taxa encountered among the three slough regions were the polychaete Capitella capitata, the group Nematoda, the polychaete Streblospio benedicti, and the bivalve Gemma gemma. These four invertebrate groups comprised 57%, 21%, 15 %, and 2%, respectively, of total infaunal density for all species groups encountered. Abundance data for individual taxa grouped by region are presented in Appendix 1.

Pearson Product-Moment correlation analysis between faunal and environmental variables detected one relationship with a strong correlation: as expected, silt and clay fractions were negatively correlated ( $p = -0.68$ ,  $n = 75$ ).

A multivariate analysis of variance (MANOVA) identified significant regional differences in species richness, evenness, abundance of the four prominent species groups, and environmental variables.

MANOVA and *a priori* results for species richness and evenness are presented in Table 1. The Inner Slough showed no difference in richness compared with the Outer Slough, however, evenness values were higher in the Inner Slough relative to the control region (Figure 5). Comparisons of Inner Slough conditions detected greater richness and evenness values in the North Basin compared with the South Basin (Figure 6).

TABLE 1. 1-Way ANOVA Univariate F Tests and *a priori* comparisons for testing differences in species richness and evenness among locations: Outer Slough (OS), Inner Slough (NB+SB), North Basin (NB), and South Basin (SB):  $\alpha = 0.025$ . Significant findings are marked with an asterisk.

DEPENDENT: RICHNESS					
VARIABLE	SS	DF	MS	F	P
LOCATION	46.107	2	23.053	8.714	< 0.001*
OS VS NB+SB	1.927	1	1.927	0.728	0.398
NB VS SB	44.180	1	44.180	16.700	< 0.001*
ERROR	190.480	72	2.646		

DEPENDENT: EVENNESS					
VARIABLE	SS	DF	MS	F	P
LOCATION	2.576	2	1.288	48.718	< 0.001*
OS VS NB+SB	2.304	1	2.304	87.158	< 0.001*
NB VS SB	0.272	1	0.272	10.278	0.002*
ERROR	1.904	72	0.026		

MANOVA RESULTS: HOTELLING-LAWLEY TRACE					
SOURCE	HOTELLING-LAWLEY TRACE	F-STATISTIC	DF	PROB	
LOCATION	1.571	27.500	4, 140	<0.001*	
OS VS NB+SB	1.285	45.614	2, 71	<0.001*	
NB VS SB	0.287	10.172	2, 71	<0.001*	



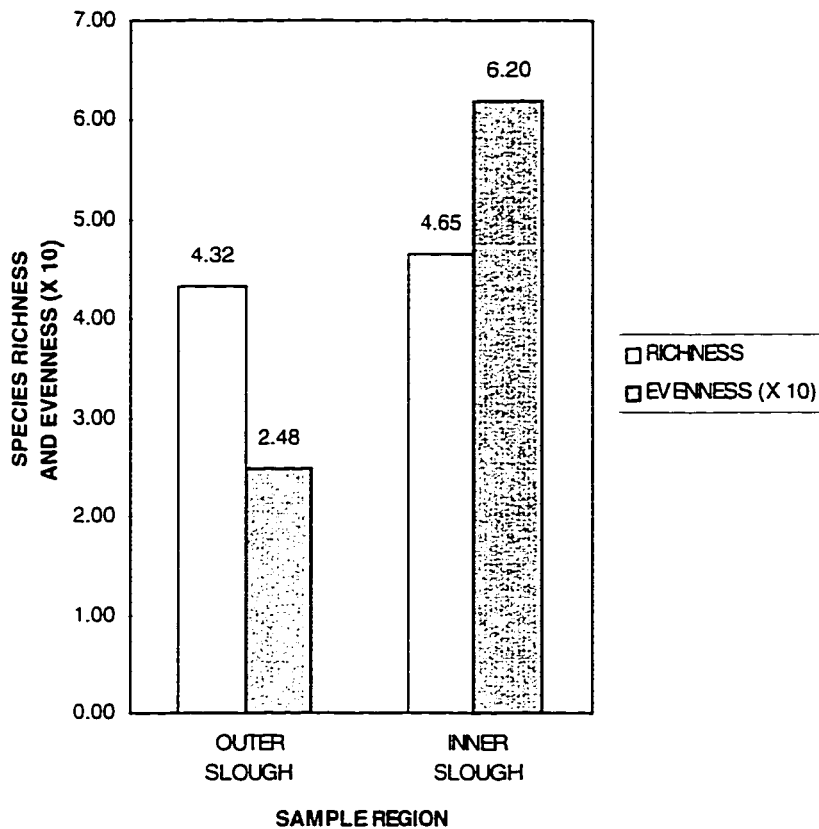


FIGURE 5. Mean species richness and evenness (x 10) values between Inner Charleston Slough and Outer Charleston Slough. MANOVA analysis found significant regional differences in species evenness between the Inner Slough and Outer Slough control region. Differences in richness were not detected. Mean values for each variable are provided, with evenness values adjusted by a factor of 10. Richness is expressed as the mean number of organisms detected per site. Evenness was determined by the Shannon-Weiner Diversity Index (Zar 1996).

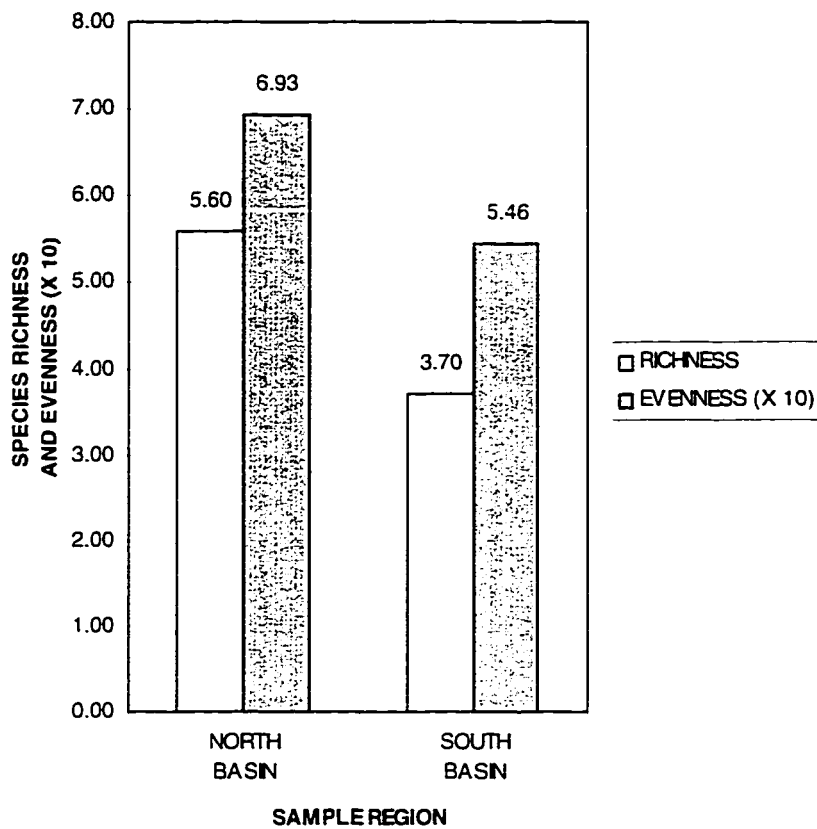


FIGURE 6. Mean species richness and evenness (x 10) values between the North Basin and South Basin. MANOVA analysis found significant regional differences in species evenness between the North Basin and South Basin region. Mean values for each variable are provided, with evenness values adjusted by a factor of 10. Richness is expressed as the mean number of organisms detected per site. Evenness was determined by the Shannon-Weiner Diversity Index (Zar 1996).

MANOVA and *a priori* comparisons of prominent species groups identified significant differences in the abundance and distribution of C. capitata, S. benedicti, and G. gemma among locations: Outer Slough, Inner Slough, North Basin, and South Basin (Table 2). The Outer Slough displayed greater abundance of C. capitata and lower abundance of S. benedicti compared with the Inner Slough, with no differences detected in either Nematoda or G. gemma distribution (Figure 7). The North Basin versus South Basin regional comparison found that C. capitata and G. gemma were more abundant in the North Basin, and S. benedicti was more abundant in the South Basin, with no differences detected in Nematoda distribution (Figure 8).

TABLE 2. 1-Way ANOVA Univariate F Tests and *a priori* comparisons for testing differences in the abundance of Capitella capitata, Nematodes, Streblospio benedicti, and Gemma gemma among locations: Outer Slough (OS), Inner Slough (NB+SB), North Basin (NB), and South Basin (SB);  $\alpha = 0.025$ . Significant findings are marked with an asterisk.

DEPENDENT: Capitella capitata

VARIABLE	SS	DF	MS	F	P
LOCATION	291.675	2	145.837	58.584	< 0.001*
OS VS NB+SB	220.271	1	220.271	88.485	< 0.001*
NB VS SB	71.404	1	71.404	28.684	< 0.001*
ERROR	179.233	72	2.489		

DEPENDENT: Nematoda

VARIABLE	SS	DF	MS	F	P
LOCATION	28.237	2	14.118	3.353	0.041
OS VS NB+SB	9.819	1	9.819	2.332	0.131
NB VS SB	18.418	1	18.418	4.374	0.040
ERROR	303.200	72	4.211		

DEPENDENT: Streblospio benedicti

VARIABLE	SS	DF	MS	F	P
LOCATION	194.034	2	97.017	33.362	< 0.001*
OS VS NB+SB	176.710	1	176.710	60.766	< 0.001*
NB VS SB	17.325	1	17.325	5.958	0.017*
ERROR	209.377	72	2.908		

DEPENDENT: Gemma gemma

VARIABLE	SS	DF	MS	F	P
LOCATION	44.921	2	22.461	9.390	< 0.001*
OS VS NB+SB	6.122	1	6.122	2.559	0.114
NB VS SB	38.799	1	38.799	16.221	< 0.001*
ERROR	172.216	72	2.392		

MANOVA RESULTS: HOTELLING-LAWLEY TRACE

SOURCE	HOTELLING-LAWLEY TRACE	F-STATISTIC	DF	PROB
LOCATION	4.049	34.416	8, 136	<0.001*
OS VS NB+SB	3.137	54.121	4, 69	<0.001*
NB VS SB	0.912	15.724	4, 69	<0.001*

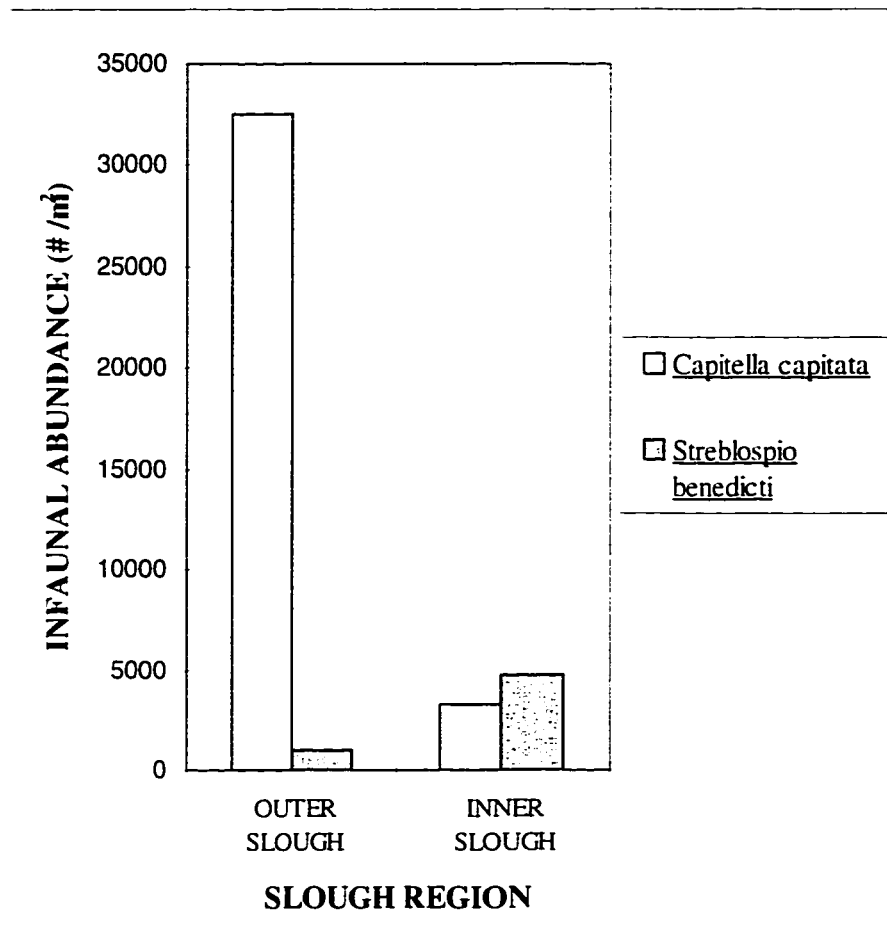


Figure 7. Results of MANOVA analysis, showing differences in relative invertebrate abundance (organisms /m<sup>2</sup>) between Inner Charleston Slough and Outer Charleston Slough. The polychaetes *C. capitata* and *S. benedicti* were most abundant in the Outer Slough and Inner Slough, respectively. Regional differences in *G. gemma* and Nematoda abundance were not detected. The community appears as a spatial continuum with *C. capitata* diminishing in relative importance between the Outer Slough and Inner Slough region, and *S. benedicti* increasing over the same range.

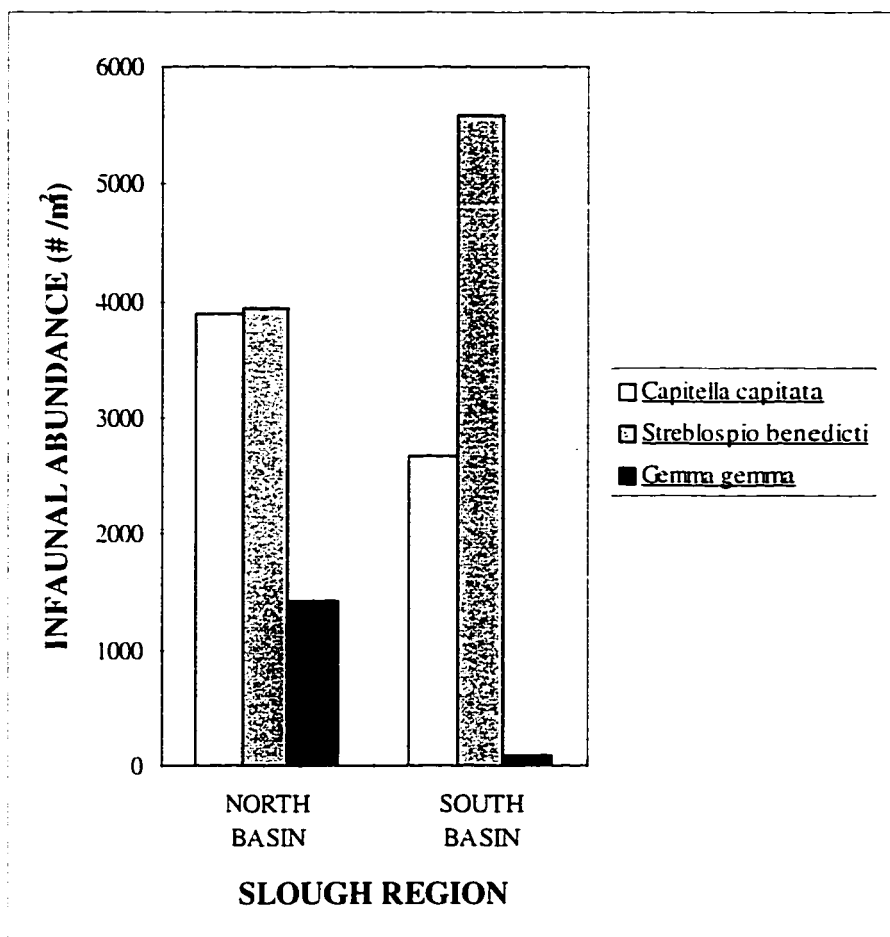


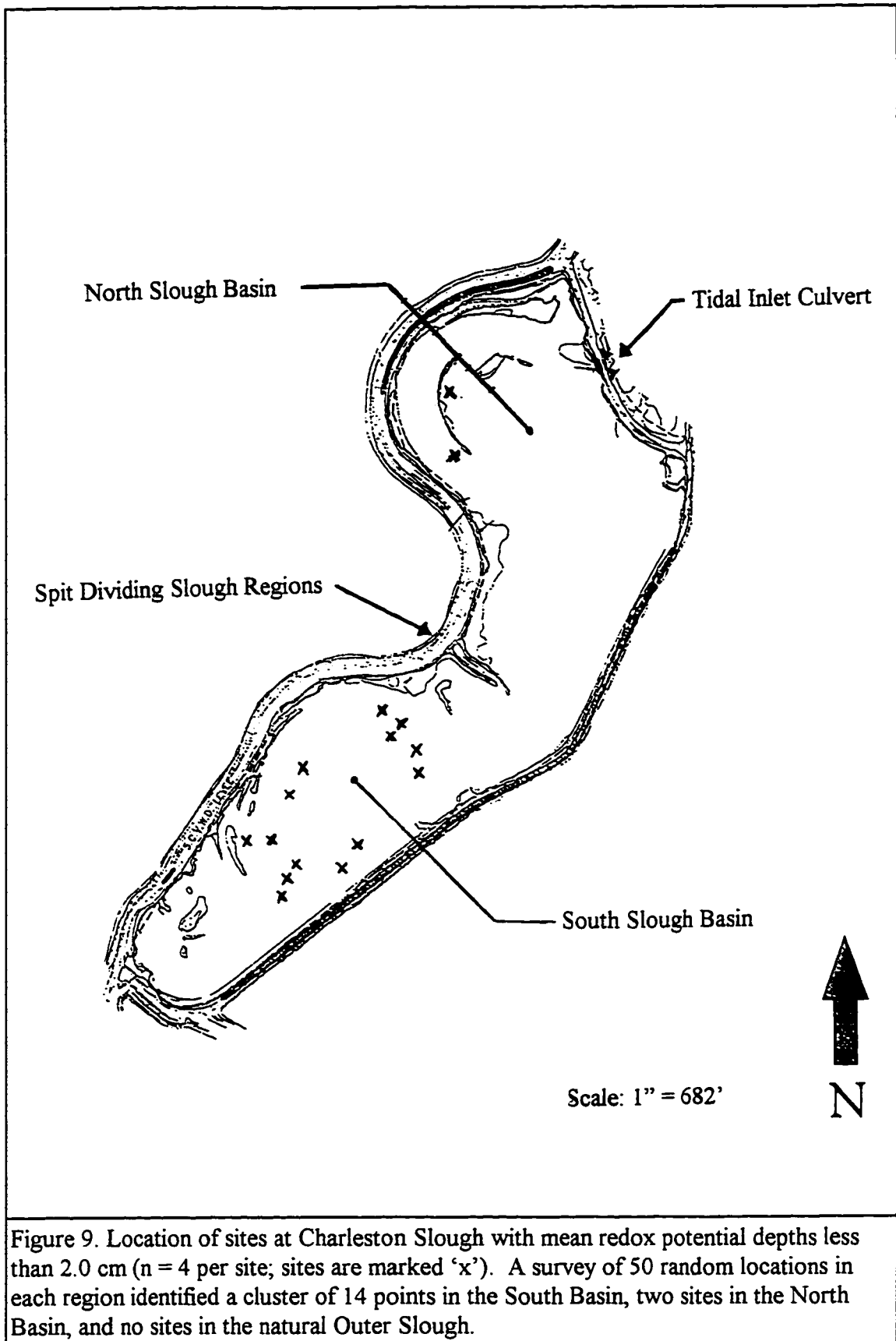
Figure 8. Results of MANOVA analysis, showing differences in relative invertebrate abundance (organisms/ m<sup>2</sup>) between the North Basin and South Basin. *C. capitata* and *G. gemma* were more abundant in the North Basin, and *S. benedicti* was more abundant in the South Basin, with no differences detected in Nematoda distribution.

MANOVA and *a priori* comparisons of environmental variables detected significant regional differences in RPD depth, RPD width, the soil organic fraction, and the soil sand fraction (Table 3). Differences were not detected in the soil clay fraction. The variable 'silt' was eliminated from analysis to reduce bias associated with multicollinearity. Outer Slough sites exhibited greater RPD depth values (9.0 cm vs. 6.1 cm), lower soil organic matter content (6.91% vs. 8.10%), and lower sand fractions (1.1% vs. 2.9%) compared with the Inner Slough (Appendix 2). The North Basin region showed higher RPD depth (8.8 cm vs. 3.4 cm) and RPD width values (4.1 cm vs. 1.6 cm), and lower soil organic matter content (7.5 % vs. 8.7%) compared with South Basin locations. Mean values for all physical parameters are presented in Appendix 2. Analysis of RPD depth distribution in the South Basin identified two geographic clusters (14 sites out of 50) with mean RPD depths less than 2.0 cm (Figure 9). In comparison, only 2 out of 50 North Basin sites had mean depths less than 2.0 cm.

TABLE 3. 1-Way ANOVA Univariate F Tests and *a priori* comparisons for testing differences in redox potential discontinuity depth, redox potential discontinuity width, soil sand fractions, soil clay fractions, and soil organic fractions among locations: Outer Slough (OS), Inner Slough (NB+SB), North Basin (NB), and South Basin (SB);  $\alpha = 0.025$ . Significant findings are marked with an asterisk.

DEPENDENT: Redox Potential Discontinuity Depth					
VARIABLE	SS	DF	MS	F	P
LOCATION	505.672	2	252.836	30.530	< 0.001*
OS VS NB+SB	139.009	1	139.009	16.785	< 0.001*
NB VS SB	366.663	1	366.663	44.275	< 0.001*
ERROR	596.270	72	8.282		
DEPENDENT: Redox Potential Discontinuity Width					
VARIABLE	SS	DF	MS	F	P
LOCATION	79.184	2	39.592	16.627	< 0.001*
OS VS NB+SB	0.056	1	0.056	0.024	0.878
NB VS SB	79.128	1	79.128	33.231	< 0.001*
ERROR	171.442	72	2.381		
DEPENDENT: Soil Sand Fraction					
VARIABLE	SS	DF	MS	F	P
LOCATION	0.059	2	0.029	8.038	0.001*
OS VS NB+SB	0.057	1	0.057	15.651	< 0.001*
NB VS SB	0.002	1	0.002	0.425	0.517
ERROR	0.264	72	0.004		
DEPENDENT: Soil Clay Fraction					
VARIABLE	SS	DF	MS	F	P
LOCATION	0.053	2	0.026	1.785	0.175
OS VS NB+SB	0.000	1	0.000	0.002	0.966
NB VS SB	0.053	1	0.053	3.568	0.063
ERROR	1.067	72	0.015		
DEPENDENT: Soil Organic Matter Content					
VARIABLE	SS	DF	MS	F	P
LOCATION	0.014	2	0.007	17.342	< 0.001*
OS VS NB+SB	0.008	1	0.008	19.556	< 0.001*
NB VS SB	0.006	1	0.006	15.127	< 0.001*
ERROR	0.029	72	0.000		
MANOVA RESULTS: HOTELLING-LAWLEY TRACE					
SOURCE	HOTELLING-LAWLEY TRACE	F-STATISTIC	DF	PROB	
LOCATION	1.594	10.678	10, 134	<0.001*	
OS VS NB+SB	0.618	8.409	5, 68	<0.001*	
NB VS SB	0.975	13.266	5, 68	<0.001*	





Pearson analysis of the clam Macoma balthica, and the polychaete worms Nereis procera, and Arenicola spp. detected a strong positive correlation ( $p = 0.000$ ,  $n = 75$ ) between the presence of at least one organism and RPD depth. Mean RPD depth ( $\pm 2$  SE) was  $9.1 \pm 0.1$  cm in the presence of at least one equilibrium-type organism, and  $3.9 \pm 0.2$  cm in the absence of these groups. No such correlation was detected with RPD width. Primary colonizing species (C. capitata, S. benedicti, etc.) presence was not correlated to RPD depth or width.

The canonical correlation comparing species richness and evenness with environmental variables resulted in one significant root pair with a negative correlation of 50.3% ( $p = 0.002$ ). The faunal component of the first canonical root pair was a positive function of species richness (mean number of infaunal groups per sample). The environmental portion of the root pair was a negative function of RPD depth and RPD width, and a positive function of percent organic matter. Therefore, as RPD depth and width decreased, and organic carbon increased, the number of infaunal groups tended to increase.

A table of variables included in the first canonical association and their respective loadings is presented in Table 4. A graph of the results is shown in Figure 10.

TABLE 4. Results of the canonical correlation comparing species richness and evenness against the environmental variables redox potential discontinuity depth, redox potential discontinuity width, soil organics fraction, soil clay fraction, and soil sand fraction. The first canonical association has negative correlation of 50.3%. The faunal component of the first canonical root pair was a positive function of species richness. The environmental portion of the root pair was a negative function of RPD depth, RPD width and a positive function of percent organics. Therefore, as RPD depth and width increased, and percent organics decreased, the number of infaunal groups increased. Variables with canonical loadings below |0.3| (marked with asterisks) were considered unimportant to this analysis.

First Canonical Pair	
<u>Faunal Variables</u>	
Richness	0.884
Evenness	-0.117*
<u>Environmental Variables</u>	
RPD Depth	-0.987
RPD Width	-0.536
Percent Organics	0.595
Percent Sand	0.293*
Percent Clay	0.111*

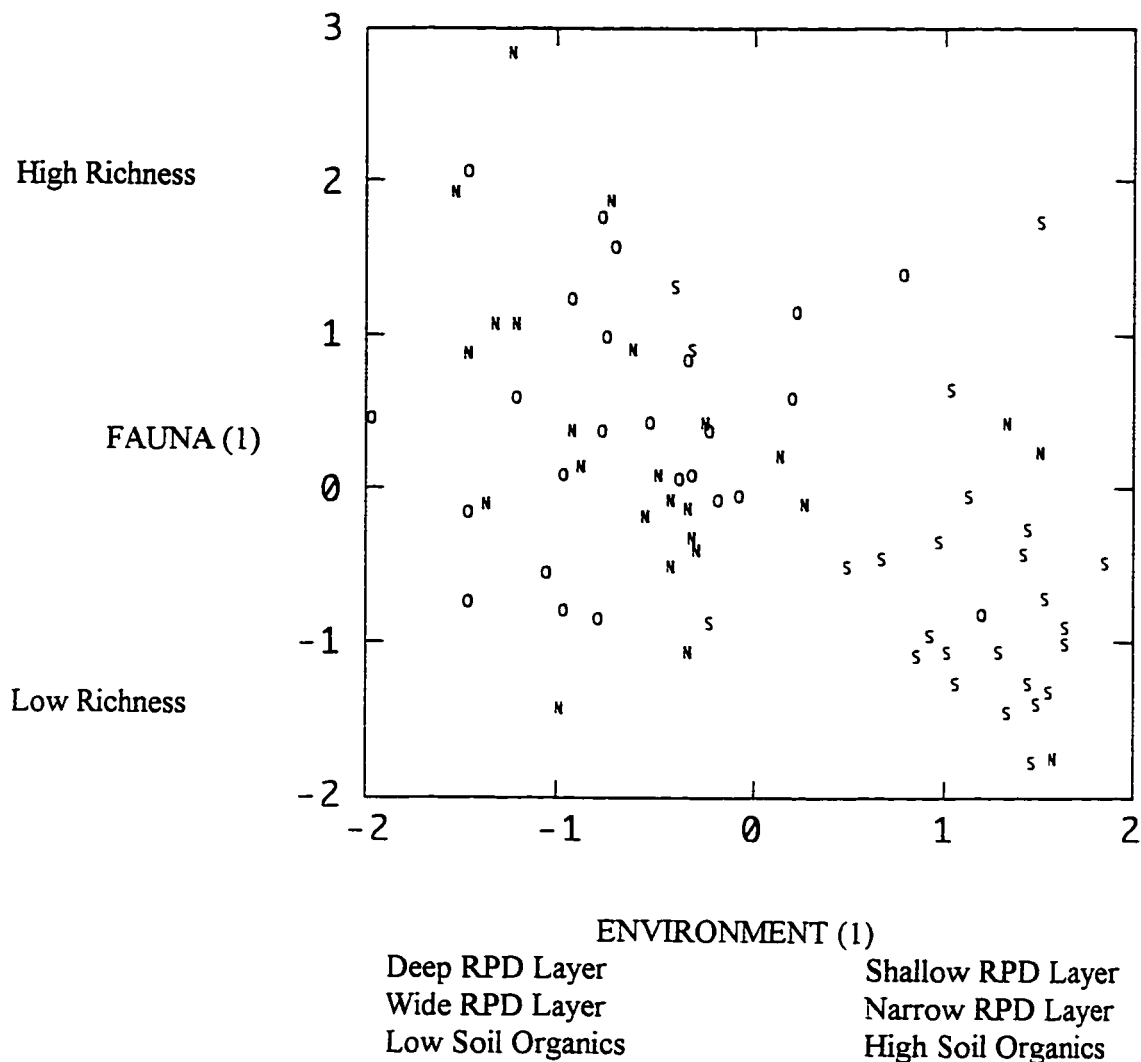


FIGURE 10. Graph of the first canonical root pair examining relationships between species richness, evenness, and the environmental variables redox potential discontinuity (RPD) depth, redox potential discontinuity width, soil organics fraction, soil clay fraction, and soil sand fraction; canonical correlation equals 51.2% ( $p = 0.002$ ). The faunal component is a positive function of species richness (mean number of species groups identified per sample). The environmental member is a positive function of RPD depth and width, and a negative function of percent organic matter. Therefore, as RPD depth and width increase, and percent organics decrease, the number of infaunal groups also increase. The symbols (O, N, and S) represent the factor scores of single data collection sites, with the axial numeric values indicating the ranking relative to other sites.

The canonical correlation comparing the distribution of C. capitata, Nematoda, S. benedicti, and G. gemma with respect to the environmental variables (RPD depth, RPD width, soil organics fraction, soil clay fraction, and soil sand fraction) resulted in one significant residual root pair with a positive correlation of 73.4% ( $p < 0.001$ ). The faunal component was a positive function of C. capitata and G. gemma abundance, and a negative function of S. benedicti and Nematoda abundance. The environmental component was a positive function of RPD depth and width, and a negative function of percent organic matter. Therefore, as RPD depth and width increased, and percent organics decreased, the infaunal groups C. capitata and G. gemma tended to increase, while S. benedicti and Nematoda decreased. Table 5 displays the constituent variables and canonical loadings of the first root pair, with the results presented graphically in Figure 11.

TABLE 5. Canonical correlation comparing the four prominent species groups: Capitella capitata, Nematoda, Streblospio benedicti, and Gemma gemma against the environmental variables RPD Depth, RPD width, soil organics fraction, percent sand and percent clay. The first canonical association has a positive correlation of 73.4% ( $p < 0.001$ ). The faunal component of the first canonical root pair is a positive function of C. capitata and G. gemma, and a negative function of Nematoda and S. benedicti. The environmental portion of the root pair is a positive function of RPD depth, and RPD width and negative function of organic matter content and percent sand. Therefore, as RPD depth and width increased, and percent organics and sand decreased, the abundance of C. capitata and G. gemma tended to increase while Nematoda and Streblospio generally decreased. Variables with canonical loadings below  $|0.3|$  (marked with asterisks) were considered unimportant to this analysis.

Canonical Loadings	
<u>Faunal Groups</u>	
<u>Capitella capitata</u>	0.681
<u>Gemma gemma</u>	0.671
<u>Streblospio benedicti</u>	-0.559
Nematoda	-0.420
<u>Environmental Variables</u>	
RPD Depth	0.978
RPD Width	0.613
Percent Organics	-0.622
Percent Sand	-0.297*
Percent Clay	-0.007*

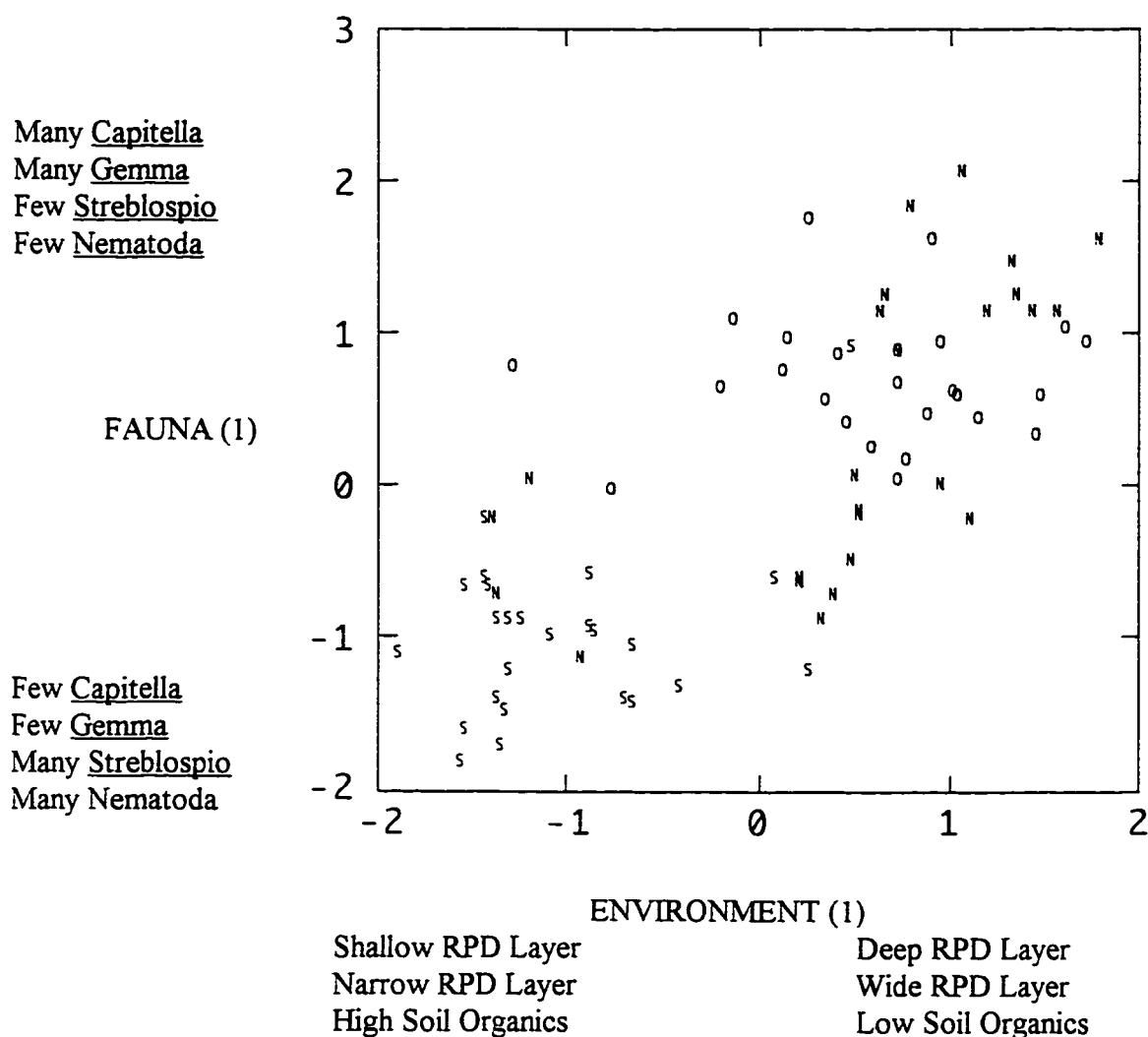


FIGURE 11. Graph of the first canonical root pair examining relationships between the four prominent infaunal groups: Capitella capitata, Nematoda, Streblospio benedicti, and Gemma gemma vs. the environmental variables redox potential discontinuity depth, redox potential discontinuity width, soil organics fraction, soil clay fraction, and soil sand fraction; canonical correlation equals 73.4% ( $p < 0.001$ ). The symbols (O, N, and S) represent the factor scores of single data collection sites, with the axial numeric values indicating the ranking relative to other sites. A distinct South Basin data cluster is evident in the lower left graph corner, corresponding to positive relationship between S. benedicti and nematode presence and environmental variables.

## DISCUSSION

The decline in overall species abundance between the Outer Slough (32.837 animals /m<sup>2</sup>) and the Inner Slough (16.059 animals /m<sup>2</sup>) is attributable to a reduction in the primary component organism: C. capitata. The cause of the decline in C. capitata density between slough regions is not readily clear, however, over this same range the abundance of S. benedicti increased significantly. While low inbenthic macroinvertebrate density and distributional differences have sometimes been attributed to the presence of dominant natant predators such as fish (Kneib and Stiven 1982; Wenner and Beatty 1988), physical conditions are suspected of controlling organism density and spatial arrangements at Inner and Outer Charleston Sloughs. Understanding the varied responses of the two main indicator groups to environmental quality parameters is key to describing Slough environmental conditions.

Interphyletic abundance comparisons are intended as only rough estimates of community health because they function poorly when comparing different sized animals such as nematodes and Nereis procera. High South Basin mean densities (18.510 organisms /m<sup>2</sup>) relative to the North Basin densities (13.608 organisms /m<sup>2</sup>) are explained by the cumulative effects of nematode and S. benedicti abundance in the South Basin. The profusion of freeliving nematodes in the South Basin is consistent with accounts of large densities occurring in highly organic marine mud (Pearse et al. 1987) and their notorious tolerance for pollution extremes (Neilson et al. 1996). Notwithstanding high density values found in the South Basin, the



S. benedicti-Nematoda dominated assemblages of this region are less desirable than faunal assemblages found in the North Basin and Outer Slough. S. benedicti and nematodes are small, shallow-burrowing opportunists that indicate conditions of environmental pollution (DeWolf 1983; Llansó 1991; Neilson et al. 1996).

However, it is the relative scarcity of equilibrium species in the South Basin relative to the North Basin and Outer Slough that warrants greatest concern (DeWolf 1982).

Low abundance of M. balthica, Arenicola spp., Eteone spp., and N. procera in the South Basin relative to the reference marsh attest to poorer environmental quality in this region. The greater presence of the large polychaete N. procera in the South Basin relative to the North Basin is an interesting, but not surprising, exception as these animals are tolerant of anoxic conditions (Llansó 1991).

Despite findings of similar richness values in the restored and natural marshes, faunal composition and relative abundance data indicate lower abundance of equilibrium species and more stressful conditions in the Inner Slough compared with the Outer Slough. Inner Slough richness values resulted from the presence of largely pioneer-type species assemblages with many small, shallow-burrowing organisms such as ostracods, amphipods, copepods, shrimp, and G. gemma. In contrast, Outer Slough richness values resulted from the common occurrence of temporally persistent species such as M. balthica, N. procera, and Eteone spp., with fewer opportunistic species detected in this region. Common opportunistic taxa found at the Inner Slough such as Amphipoda, Ostracoda, and Gemma gemma, displayed limited distribution at the Outer Slough. Invertebrates observed at Inner

Charleston Slough agreed with Wenner and Beatty's (1988) findings of pioneer community establishment within salt water impoundments.

Higher richness values in the North Basin compared with the South Basin indicate greater taxonomic variation in the former region, while evenness differences between the Inner Slough and Outer Slough resulted primarily from lower species equitability at the Outer Slough, with richness contributing to a lesser extent (Westman 1985). Moderate evenness values at the Inner Slough resulted from the presence of three dominant, comparably abundant taxa, while low Outer Slough values were the result of dominance from a single species, C. capitata. Inner Slough species assemblages consisted of three dominant taxa: C. capitata, Nematoda, and S. benedicti comprising 20.5%, 37.7%, and 29.7% of total species density, respectively, with an equitable presence among lesser pioneer groups and few equilibrium species. In contrast, the polychaete C. capitata comprised 89.9% of the total Outer Slough species complement, with the remaining fraction occupied primarily by nematodes (5.9%), S. benedicti (2.7%), and larger equilibrium species. The dissimilar subsoil community assemblages found between the Outer Slough and Inner Slough attest to varying selective physical pressures and inbenthic resource distribution between the two areas.

Higher evenness scores in the North Basin compared with the South Basin resulted from greater species equitability and the distribution of dominance between several species at the North Basin. Nematode dominance at the South Basin contributed greatly to the low evenness scores in this region.

Relative composition and density results from the four prominent species groups were consistent with prior observations of disturbed estuarine environments. The Inner and Outer Slough inbenthic communities generally appear as a composite of C. capitata dominated regions, and Nematoda/ S. benedicti dominated areas, with Outer Slough C. capitata dominated areas diminishing in importance through the North Basin, and further diminishing in the South Basin. Conversely, the Nematoda/ S. benedicti community increase relative importance over the same range, with both taxa being more common in the restored marsh. The observed spatial patterns suggest the presence of pollution-induced disturbances affecting subsoil habitat quality in the Inner Slough (Shaw et al. 1983; Neilson et al. 1996; Bridges 1996).

Both C. capitata and S. benedicti have been described as highly tolerant of hypoxic (low oxygen) conditions, however, the rapid colonization and productive life history strategies of these species may also describe their distributions within the Inner Slough. Both species have been shown extensively to endure and even thrive under various conditions of environmental pollution (Grassle and Grassle 1976; Warren 1976), and take advantage of perturbed habitats through rapid colonization (Llansó 1991). Bridges (1996), for example, demonstrated that juvenile C. capitata doubled in size when sediments were fortified with organic sewage.

Findings that S. benedicti have a lower tolerance than C. capitata to anoxia, and moderate tolerance to hydrogen sulfide concentrations (Llansó 1991) do not account for observed faunal distributions seen at Charleston Slough. If oxygen availability was the primary factor controlling organism distribution, higher C.

capitata abundance would be expected in the South Basin. The synergistic effects of anoxia, hydrogen sulfide concentrations, soil organics, soil pH, and interspecific competition are suspected of contributing to observed faunal patterns. While the source of the diametric pattern between C. capitata and S. benedicti abundance between the North Basin and South Basin is not apparent, a test of parameters influencing the distributions of these species is presented in the canonical correlation discussion.

It is uncertain whether the profusion of C. capitata at the Outer Slough denotes poor environmental conditions relative to other natural sloughs. It is conceivable that poor quality outflow from Inner Charleston Slough promotes C. capitata dominated assemblages at the Outer Slough. However, the numerous presence of M. balthica, N. procera, and Arenicola spp. in this area attest to a relatively stable inbenthic community with at least moderate soil and water quality

The strong Pearson correlation between equilibrium species presence and RPD depth supports previous qualitative studies of organism-soil interactions (Rhoads and Boyer 1982). Although equilibrium species were found at sites with RPD depths as shallow as 1.5 cm, the average RPD depth of 9.1 cm in the presence of larger secondary fauna backs previous organism-sediment observations that shallow RPD depths may indicate equilibrium-fauna absence, and deeper RPD layers show their presence.

Observed Inner Slough RPD depth values ( $\mu = 6.1$  cm), by Rhoads and Boyer's (1982) 2.0 cm criteria, indicate soils deep enough to support larger soil

invertebrates. As expected, RPD depth was greatest in the Outer Slough compared with the Inner Slough, and higher in the North Basin compared with the South Basin. Fourteen problem areas (mean RPD depth < 2.0 cm) were detected in the South Basin and at two sites in the North Basin. The shallow RPD depth cluster in the South Basin raises questions about the ability of present tidal conditions to promote functional ecosystem recovery in such remote slough regions, and as to what factors, environmental or otherwise, prohibit the affected regions from developing deeper RPD depths. Shallow RPD depths at these sites corresponded to the nearly complete absence of equilibrium species (Arenicola spp. was detected at one site). These preliminary findings maintain the usefulness of RPD depth as a rapid and cost effective means of judging subsoil habitat quality and monitoring inbenthic community recovery at chronically polluted salt marsh mudflats. This technique may be particularly useful in serial monitoring programs, studying the development of soil conditions which support advanced inbenthic assemblages, and expanding the scope of vegetation-based restoration monitoring programs. If shown to be effective, the simplicity of field sampling and ease of interpretation make RPD depth assessment ideal for inclusion within salt marsh restoration baseline monitoring programs.

The usefulness of RPD width as a measure of invertebrate distribution, or as an indicator of salt marsh restoration progress was not established by this study. One notable exception to this conclusion with respect to canonical correlation results is the finding that the environmental parameters RPD depth, RPD width, and organic

matter content were related to richness and relative abundance of prominent species groups.

Soil organic matter was higher in the Inner Slough compared with the Outer Slough, and higher in the South Basin compared with the North Basin. These findings are significant because little data exists on organic sediment characteristics of human-made wetlands or of comparisons between these areas and natural controls (Race and Christie 1982), and even less information is available on comparisons with salt water impoundments. In contrast to high organic fractions found at Inner Charleston Slough, low soil organic levels are the norm at salt marsh creation projects (Lindau and Hossner 1981; D'Avanzo 1990; PERL 1990). Thus, active and passive organic matter amendment techniques practiced at artificial salt marshes, such as hay enrichment and vegetative replanting, are not transferable to organic level amelioration at Inner Charleston Slough. While findings here suggest high organic levels in the Inner Slough and particularly the South Basin, relative to the control region, precise organic matter fractions and the potential effects of high organics upon reduced soil oxygen availability and plant growth inhibition are not known.

Both canonical analyses detected profound faunal responses to the cumulative effects of RPD depth, RPD width, and percent organics. In the first analysis, species richness was found to positively correspond to increases in environmental components. It is apparent from Figure 10 that higher quality environmental conditions, manifested by the combination of a deep, wide RPD layer

and relatively moderate soil organics, generally supported more species. Sites with shallow and narrow RPD layers, and high soil organics were common in the South Basin, which supported some of the lowest species richness values detected. Sites with high species richness generally occurred in the North Basin and Outer Slough.

The second canonical analysis, describing the four prominent species groups, showed that the same environmental variables influenced relative species abundance. Sites with poor environmental conditions, evidenced by the presence of shallow and narrow RPD layers and high soil organics, displayed high abundance of S. benedicti and nematodes, and low abundance of C. capitata and G. gemma. Analysis of Figure 11 shows that the poorest environmental quality sites were almost exclusively located in the South Basin, while moderate to high quality sites occurred in the Outer Slough and North Basin. Thus, S. benedicti and Nematoda were more tolerant of stressful South Basin environmental conditions compared with C. capitata and G. gemma.

Interpretation of these results may be facilitated by the qualitative field observation that most of the samples taken in the South Basin were laden with cohesive masses of residual root matter and peat, which occurred infrequently at the other slough regions. Thus, one possible explanation of conditions found within the Inner Slough is that residual root presence influenced RPD depth, RPD width and percent organics, and that the canonical environmental components in both analyses represent the contrast between remnant vegetation presence and absence. That is, the canonical environment values for a given site were arrived at by summing variables

that were typically higher in the absence of remnant root systems (RPD depth and width) and subtracting variables that were higher in the presence of roots (soil organic fraction). According to this observation, high species richness values typically occurred at field sites with relatively few remnant roots. These conditions also favored high abundance of C. capitata and G. gemma, and relatively fewer S. benedicti and Nematoda. Discretion must be used in interpreting these data; while instructive, conclusions that physical and faunal differences between the South Basin and other regions resulted from the presence of remnant root systems should be regarded as suggestive and preliminary.

This finding provides some of the first evidence suggesting that inhibition of infaunal recovery at salt evaporation ponds may occur due to the presence of remnant root systems. It is suspected that high soil organics from persistent root systems may be pervasive in salt evaporation ponds with drowned vegetation stands, and that these circumstances may potentially inhibit invertebrate support functions. Conditions at the South Basin may indicate a cyclical relationship between remnant vegetation decomposition and low soil oxygen, with low soil organics slowing the decomposition of vegetative material, and root decomposition contributing to anoxic soil conditions. Such conditions would influence infaunal species assemblages, resulting in a reduction of food resources for higher trophic levels. This factor may be common at salt evaporation ponds where rapid vegetation dieoff has been described (in this case due to drowning), and should be considered within baseline monitoring programs of similar projects.



Invertebrate preferences for substrate type have been well demonstrated (Thorson 1950; Filice 1958; Rhoads and Young 1970; Ricketts et al. 1985; Ward 1992), however, neither canonical correlation analysis detected preferential use of different particle size classes. Findings that the clay fraction (thus, the correlated silt fraction) did not differ between slough regions may have contributed to these results, as sediment composition may have been similar enough between successive samples that faunal substrate preferences could not be distinguished.

This study found suggestive evidence that subsoil quality at Inner Charleston Slough was highest in the slough region nearest the tidal inlet culvert, which could have resulted from linear, pocket-marsh slough design. Whether the observed decline in environmental quality was related to distance from the tidal inlet is unclear, however, distribution of observed parameters (soil organic matter, redox potential discontinuity depth, redox potential discontinuity width, and invertebrate composition) support this conclusion. Convergence of Inner Slough subsoil conditions, including RPD depth and soil organic content, with conditions found in the control region is expected to occur in the North Basin prior to the South Basin. Twelve years following the 1984 slough enhancement project, the South Basin exhibited physical conditions that were not conducive to the development of species assemblages found in the control region.

There has been much work on natural colonization of inbenthics at disturbed sites (McCall 1977; Rhoads et al. 1977; Trapani 1987). These studies principally document the rates of inbenthic colonization, noting changes in species composition

and faunal densities over time. Studies of benthic-sediment interactions have yielded insight on larval and adult colonization mechanisms, and site-specific chemical cues that promote settlement (Rhoads 1970; Aller 1982; Rhoads and Boyer 1982; Pawlik 1992), but unfortunately little has been done in terms of enhancing restoration techniques.

Salt marsh restoration projects typically focus upon restoring very visible floral and faunal species, assuming other aspects of the ecosystem such as the inbenthic community will colonize quickly and completely. However, this study and others have identified significant deficiencies in invertebrate support functions in restored tidal wetlands (Dauer and Simon 1976; Zedler et al. 1991; Sacco et al. 1994), suggesting that salt marsh restoration and enhancement projects should take steps to quantify the success of inbenthic restoration and actively restore the invertebrate wildlife component, if necessary.

Optimal recovery of restored salt marsh ecosystems is especially important when environmental restoration is used to compensate for the loss or damage to natural wetlands. The replacement of numerous natural marshes with equivalent acreage of functionally deficient salt marshes will contribute to a significant overall loss of valuable salt marsh habitat.

## RECOMMENDATIONS

Soil testing should be an integral component of restoration planning phases to ensure optimal functional recovery and early recognition of barriers to salt marsh recovery. Soil conditions have a direct and significant impact upon the quality and types of vegetation and animals found at site. It is recommended that monitoring of the following parameters should be included in baseline and seasonal studies for the 1996 Inner Charleston Slough enhancement project: soil organic matter content, soil pH, interstitial water salinity, and nitrogen dynamics. Invertebrate colonization studies are also recommended to quantify biotic responses to present restoration. Results of this study indicate that future monitoring programs should be initiated in advance of salt marsh restoration planning to anticipate restoration obstacles. Recognizing and avoiding restoration obstacles early in project planning phases promotes project cost-effectiveness and optimal ecosystem recovery.

The most fundamental approach to improvement of low invertebrate abundance, dominance by less desirable pioneering species, high soil organics, and shallow RPD depths at Inner Charleston Slough and similar restorations is to increase tidal inundation and observe slough siltation rates. This method is presently being pursued at Inner Charleston Slough by the City of Mountain View. Evidence from this study suggests that high organics may still hamper functional South Basin recovery until a substantial layer of sediment is deposited by tidal action. It is recommended at Inner Charleston Slough and similar projects that sediment deposition and RPD depth should be periodically monitored. In marshes with

remnant organic layers, target sediment deposition goals should be at least 10-20 cm, with RPD depths between 5-10 cm. It is important that a layer of sediment at least 10 cm deep be deposited over the organic layer.

Salt pond restorations with high organics which will not produce substantial sediment deposition may require other techniques. If excessive soil organics are detected then remediation of slough conditions may involve the technique of increasing soil aeration through the physical reworking of sediments, or removing the peaty soils and replacing them with imported material such as dredge spoils. Although both methods are widely practiced, the comparative effectiveness of these methods, and their ability to approximate natural conditions have not been fully evaluated. Soil aeration has been accomplished by methods of soil raking, or on a larger scale, soil disking to a depth of approximately 0.3 m. Disking was employed in December 1988 at a portion of the 150-acre Newark Tract 102 Marsh, a tidally restored salt evaporation pond, but invertebrate responses to these activities have not been quantified (Albertson 1996).

Soil pH is another parameter that should be monitored in salt pond restorations to assess soil quality, particularly if elevated soil organics and acid sulfate soils are suspected. This measure should not be confused with water pH. Low soil pH values occur in high clay soils under impeded tidal conditions as a result of the oxidation of sulfides to sulfates and sulfuric acids (PERL 1990). Broome (1990) found that vegetation did not survive in soils with pH below 3.0.

Highly acidic soils also negatively affect soil invertebrates communities. Both lime additions and tidal flooding have been used effectively to raise soil pH.

Other significant soil parameters contributing to salt marsh functional recovery are interstitial water salinity, nitrogen dynamics, and redox potential. Seasonal marsh drainage or clay capped saline soils may result in areas with high residual salt concentrations. High salinity negatively affects seed germination, seedling establishment, and soil invertebrate associations (Broome 1990; PERL 1990). Due to evaporative concentration during summer draw-down periods, attention should be paid to depressions that remain continually flooded. Low redox potentials (PERL 1990) and poor nitrogen dynamics have been shown to inhibit pickleweed (Broome 1990) and cordgrass (Zedler 1991) growth. Analysis of these parameters should be included in pre-restoration soil monitoring programs.

This study has shown the importance of monitoring inbenthic species diversity and abundance as a bioindicator of marsh health. In addition, RPD depth is suggested as an easy and inexpensive monitoring tool for marsh restorations. This method shows marsh health and progression toward a mature marsh with equilibrium organisms. Such monitoring tools are critical for evaluating restoration success.

## SUMMARY

At Charleston Slough, Mountain View, 25 stations were set up at three slough locations at which invertebrate bottom samples and physical data were taken. The two experimental regions each comprised approximately half of a 103-acre salt evaporation pond, with the northernmost region receiving water inflow from the control region through a single culvert and a daily tidal range of between 0.1'-0.2'. Inbenthic bottom samples and physical data were taken from mudflats at each of the three slough regions. On the basis of these data the following conclusions were drawn regarding the state of the estuary:

- 1) Among the three regions a strong correlation between redox potential discontinuity (RPD) depth and equilibrium species presence support the use of RPD depth measurements as rapid and cost effective means for assessing infaunal community health at this site, and identifying problematic slough regions.
- 2) Lower species richness values and fewer equilibrium species in the South Basin compared with the North Basin indicate lower quality inbenthic habitats at the South Basin. Altered redox conditions with low oxygen and high hydrogen sulfide, or high soil organics may have contributed to observed faunal conditions.
- 3) High evenness values in the Inner Slough compared with the control region resulted from the overwhelming dominance of the single polychaete species Capitella capitata in the Outer Slough, and sharing of dominance between the opportunistic species C. capitata, Nematoda, Streblospio benedicti, and Gemma

gemma in the Inner Slough. Greater resource sharing between species was also observed in the North Basin relative to the South Basin.

4) Although C. capitata is considered the classic indicator of poor marsh environmental quality, the high abundance of S. benedicti in the South Basin is believed to correspond with the poorest environmental conditions among the regions studied. Parameters that support this conclusion are soil organic matter content, RPD depth, and RPD width.

5) It is hypothesized that high soil organics in the South Basin due to the presence of peat and remnant vegetation roots inhibit the development of normal redox conditions. It is postulated that species assemblages in this region have been limited to organic tolerant nematodes and S. benedicti, with other opportunistic species performing relatively minor community roles. It is suspected that high soil organics from persistent root systems may be pervasive at denuded salt evaporation ponds, and that these circumstances may potentially inhibit invertebrate support functions. It is well recognized that high soil organics reduce available soil oxygen. Conditions at the South Basin may indicate a cyclical relationship between remnant vegetation decomposition and low soil oxygen, with low soil organics slowing the decomposition of vegetative material, and root decomposition contributing to anoxic soil conditions. Such conditions would influence infaunal species assemblages, resulting in a reduction of food resources for higher trophic levels.

While results are still preliminary, observations of redox layer and soil organic matter characteristics, and inbenthic fauna support the conclusion that

remnant vegetation roots at sites with drowned vegetation may hamper natural salt marsh recovery. Restoration planning of similar sites should include the characterization of soil organic matter content in baseline studies, and if necessary, in periodic monitoring programs.

6) Convergence of Inner Slough subsoil conditions, including RPD depth and soil organic content, with conditions found in the control region is expected to occur in the North Basin prior to the South Basin.

7) Species richness was lowest in the presence of a shallow and narrow RPD layer, and high soil organic fraction. Species richness was high when soil conditions showed a deep and wide RPD layer, and low soil organics. While this observation demands further study, differences in species richness are hypothesized to be a function of the quantity of persistent root material present at a site.

8) Canonical correlation found that the species C. capitata and G. gemma were positively correlated with RPD depth and width, and negatively correlated with the soil organics fraction, and the groups Nematoda and S. Benedicti were negatively correlated with RPD depth and width and positively correlated with soil organics.

The observed distributions of these species at Inner Charleston Slough are hypothesized to be a function the quantity of persistent root material present at a site, with Nematoda and S. benedicti responding positively to dense root conditions.



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APPENDIX 1. Mean invertebrate species abundance /m<sup>2</sup> (+/- 2 SE) for each of the 14 taxa identified at Charleston Slough. Mean density /m<sup>2</sup> is adjusted from a total sampling area of 0.875 m<sup>2</sup> per sampling region (2.635 /m<sup>2</sup> for computation of total mean density). Density totals for the restored Inner Slough were calculated by combining results from the North and South Slough basins. Soil cores were taken to a depth of 40 cm, and screened through a 512 micron mesh. The four numerically prevalent species groups are highlighted in gray.

MEAN INVERTEBRATE DENSITY (# Organisms /m <sup>2</sup> )					
TAXON	SAMPLE REGION				
	Outer Slough (n = 25)	North Basin (n = 25)	South Basin (n = 25)	Inner Slough (n = 50)	TOTAL (n = 75)
TOTAL	36,562 (559)	13,608 (124)	18,510 (328)	16,059 (226)	26,065 (155)
<u>Capitella capitata</u>	32,837 (497)	3,897 (59)	2,678 (153)	3,288 (106)	14,957 (142)
Nematoda	2,144 (46)	2,691 (56)	9,423 (235)	6,057 (146)	5,411 (57)
<u>Streblospio benedicti</u>	1,002 (71)	3,949 (74)	5,594 (72)	4772 (73)	4,002 (29)
<u>Gemma gemma</u>	75 (3)	1,426 (39)	89 (7)	758 (23)	603 (10)
Shrimp, unid.	75 (4)	619 (13)	531 (11)	575 (12)	465 (4)
Ostracoda	25 (2)	683 (13)	38 (2)	361 (8)	283 (3)
<u>Arenicola spp.</u>	197 (6)	118 (2)	25 (1)	72 (2)	129 (2)
Amphipoda	0	165 (5)	75 (4)	120 (5)	91 (1)
<u>Heteromastus spp.</u>	63 (3)	13 (1)	13 (1)	13 (1)	34 (1)
<u>Nereis procera</u>	42 (1)	16 (1)	27 (1)	22 (1)	32 (1)
<u>Macoma balthica</u>	50 (2)	6 (1)	3 (1)	5 (1)	22 (1)
<u>Eteone spp.</u>	39 (2)	13 (1)	0	7 (1)	20 (1)
Halacaridae	13 (1)	0	13 (1)	7 (1)	10 (1)
Copepoda	0	13 (1)	0	7 (1)	5 (1)

APPENDIX 2. Mean values of physical parameters tested at Charleston Slough, grouped by region. Findings of significant differences between Outer Slough and Inner Slough, and North and South Basin sites as determined by multivariate analysis of variance are indicated with an asterisk next to the higher value.

REGION	ENVIRONMENTAL PARAMETER					
	RPD DEPTH	RPD WIDTH	SAND	SILT	CLAY	ORGANICS
OUTER SLOUGH	9.0 cm *	2.9 cm	1.1%	42.1%	56.5%	6.9%
INNER SLOUGH	6.1 cm	2.9 cm	2.9% *	39.0%	57.9%	8.1% *
NORTH BASIN	8.8 cm *	4.1 cm *	2.8%	41.2%	55.7%	7.5%
SOUTH BASIN	3.4 cm	1.6 cm	3.1%	38.6%	60.1%	8.7% *