Spring 5-31-2001

Benthic communities in Spartina alterniflora and Phragmites australis dominated salt marshes

Catherine E. Yuhas
New Jersey Institute of Technology

ABSTRACT

BENTHIC COMMUNITIES IN *SPARTINA ALTERNIFLORA* AND *PHRAGMITES AUSTRALIS* DOMINATED SALT MARSHES

by
Catherine E. Yuhas

The benthic communities were investigated in *Phragmites australis* and *Spartina alterniflora* salt marshes, conducted in natural and mitigated salt marshes located in a highly urbanized area, the Hackensack Meadowlands, New Jersey. Benthic samples were taken with a 5-centimeter core at two habitats, the creek bank and the edge of the vegetation in the low marsh zone. Salinity levels and textural and structural sediment characteristics were also collected at each site. A recolonization experiment that utilized sediment from an undisturbed and uncontaminated salt marsh was conducted to determine if substrate is important in benthic colonization. The results suggest that there were differences in abundance, taxa richness, diversity, and composition in the benthic communities found among the different types of grasses as well as the mitigated and natural marshes. The *Phragmites australis* marsh had a more diverse benthic community than the natural and mitigated *Spartina alterniflora* marshes. The mitigated marshes had a greater abundance and lower diversity than the natural marshes. However, there were differences in salinity levels (oligohaline to polyhaline) between the mitigated and natural marshes that could result in different types of benthic communities. Substrate and contamination did not seem to be a factor in the recolonization of benthic communities. The principles of opportunism were responsible for shaping the benthic communities in the recolonization experiment. This study shows that *Phragmites australis* supports a healthy benthic community. The benthic community of the mitigated marsh did not resemble the natural marshes after 12-years of establishment.

BENTHIC COMMUNITIES IN *SPARTINA ALTERNIFLORA* AND *PHRAGMITES AUSTRALIS* DOMINATED SALT MARSHES

by
Catherine E. Yuhas

A Master's Thesis
Submitted to the Faculty of
New Jersey Institute of Technology and
Rutgers, The State University of New Jersey-Newark
In Partial Fulfillment of the Requirements for the Degree of
Master of Science in Environmental Science

Department of Chemical Engineering, Chemistry, and Environmental Science

May 2001

Blank Page

APPROVAL PAGE

BENTHIC COMMUNITIES IN *SPARTINA ALTERNIFLORA* AND *PHRAGMITES AUSTRALIS* DOMINATED SALT MARSHES

Catherine E. Yuhas

Dr. Judith Weis, Thesis Advisor — Date
Professor of Biology, Rutgers University, Newark, New Jersey

Dr. Jean Marie Hartman, Committee Member — Date
Associate Professor of Landscape Architecture, Rutgers University, New Brunswick, New Jersey

Dr. Richard Trattner, Committee Member — Date
Professor of Chemical Engineering, Chemistry and Environmental Science, New Jersey Institute of Technology

BIOGRAPHICAL SKETCH

Author: Catherine E. Yuhas

Degree: Master of Science in Environmental Science

Date: May 2001

Undergraduate and Graduate Education:

- Master of Science in Environmental Science,
  New Jersey Institute of Technology and
  Rutgers, The State University of New Jersey,
  Newark, New Jersey, 2001

- Bachelor of Science in Biology,
  Fairfield University, Fairfield, Connecticut, 1998

Major: Environmental Science

Presentations and Publications:

Yuhas, Catherine E., Hartman, Jean Marie, and Weis, Judith S. "Benthic Communities in *Spartina alterniflora* and *Phragmites australis* Dominated Salt Marshes." *The Third Annual Wetland Regulatory Workshop*, Atlantic City, New Jersey, October 30, 2000-November 3, 2000.

This thesis is dedicated to
the salt marshes of New Jersey that they are managed correctly and wisely while being preserved endearingly for future generations to enjoy and learn from.

ACKNOWLEDGEMENT

I would like to express my most sincere thank you to my Advisor, Dr. Judith S. Weis, and my Committee member, Dr. Jean Marie Hartman, for their guidance, advice, and support. I am very appreciative for all of their time and energy. I would also like to thank Dr. Hartman for the use of her facilities. I would like to thank my other Committee member, Dr. Richard Trattner for all his assistance as well. I would like to acknowledge Ed Konsevick and Brett Bragin for taking me out in the field, and Lisa Windham for all her help in the field. I would also like to express my gratitude to the Hartman Lab for all of their assistance and advice. This research was funded under a grant from the Hackensack Meadowlands Development Commission under the direction of Dr. Hartman.

TABLE OF CONTENTS

Chapter **Page**

1 INTRODUCTION 1

1.1 Background Information 1

1.1.1 Salt Marshes 3

1.1.2 *Phragmites australis* Invasion 6

1.1.3 Benthic Invertebrates 9

1.1.4 Restored versus Natural Salt Marshes 16

1.2 Objective 20

2 METHODS 21

2.1 Study Sites 21

2.2 Materials and Methods 28

2.2.1 Benthic Sampling 28

2.2.2 Biota 29

2.2.3 Sediments 30

2.2.4 Recolonization Experiment 31

2.2.5 Statistical Analysis 32

3 RESULTS 34

3.1 Environmental Parameters 34

3.1.1 Organic Matter 34

3.1.2 Percentage of Silt, Sand, and Clay 35

3.1.3 Salinity 39

TABLE OF CONTENTS
(Continued)

Chapter	Page
3.2 Benthic Samples	40
3.2.1 Abundance	42
3.2.2 Taxa Richness	44
3.2.3 Taxa Diversity Index	46
3.2.5 Composition	49
3.3 Recolonization Experiment	56
3.3.1 Abundance	57
3.3.2 Taxa Richness	58
3.3.3 Taxa Diversity Index	59
3.3.4 Composition	61
4 DISCUSSION	68
4.1 Benthic Invertebrates	68
4.2 Recolonization Experiment	73
5 CONCLUSIONS	76
APPENDIX SUMMARY OF HEAVY METALS CONTAMINATION	78
REFERENCES	79

LIST OF TABLES

Table		Page
3.1	Principal component analysis results of sediment characteristics	37
3.2	Mean Abundance, Taxa Richness, and Shannon-Wiener and Simpson's Taxa Diversity Index	41
3.3	Shannon-Wiener Taxa Diversity Index	48
3.4	Simpson's Taxa Diversity Index	48
3.5	Taxa present and absent	49
3.6	Principal components analysis results of benthic sampling	53
3.7	Principal component analysis results of recolonized and ambient samples	65

LIST OF FIGURES

Figure		Page
2.1	Site map of Hackensack Meadowlands District, New Jersey	24
2.2	Site map of Mill Creek, Secaucus, New Jersey	25
2.3	Site map of Sawmill Creek, Lyndhurst, New Jersey	26
3.1	Mean percentage of organic matter	34
3.2	Mean percentage of silt, sand, and clay at the creek bank habitat	36
3.3	Mean percentage of silt, sand, and clay at the edge of the vegetation habitat	36
3.4	Principal component analysis for sediment characteristics	38
3.5	Salinity levels	39
3.6	Mean abundance of benthic invertebrates at the creek bank habitat	43
3.7	Mean abundance of benthic invertebrates at the edge of the vegetation habitat	43
3.8	Mean taxa richness at the creek bank habitat	45
3.9	Mean taxa richness at the edge of the vegetation habitat	45
3.10	Percent composition of the common taxa at Sawmill Creek Sites	52
3.11	Percent composition of the common taxa at Mill Creek Sites	52
3.12	Principal component analysis results of benthic sampling	55
3.13	Mean abundance of benthic invertebrates found in the recolonized and ambient samples	57

LIST OF FIGURES
(Continued)

Figure **Page**

3.14 Mean Taxa richness of benthic invertebrates found in the recolonized and ambient samples 58

3.15 Mean Shannon-Wiener Taxa Diversity Index for Recolonization Experiment 60

3.16 Mean Simpson's Taxa Diversity Index for Recolonization Experiment 60

3.17 Percent composition of the common taxa found in recolonized and ambient samples at Sawmill Creek 62

3.18 Percent composition of the common taxa found in the recolonized and ambient samples at Mill Creek 63

3.19 Principal component analysis results of recolonized and ambient samples ... 66

CHAPTER 1

INTRODUCTION

1.1 Background Information

Salt marshes are tidal wetlands that are a part of the tidal marsh-estuarine ecosystem. *Spartina alterniflora* (salt marsh cordgrass) is abundant on tidal creeks, and it dominates the low marsh (Teal 1962; Bertness 1991). *S. alterniflora* provides a habitat for salt marsh species as well as a source of food for benthic invertebrates and insects (Teal 1962; Healy and Walters 1994; Van Dolah 1978; Able and Hagan 2000; Graca et al. 2000; Kneib et al. 1997). Another type of salt marsh grass is *Phragmites australis*. *P. australis* is an invasive reed grass in the Atlantic coastal salt marshes, and its rate of expansion is 1-6% per year (Fell et al. 1998; Angradi et al. 2001; Windham and Lathrop 1999; Weinstein and Balletto 1999). *P. australis* colonizes into low marshes and replaces *S. alterniflora*, but it is usually found on high marshes.

P. australis changes the marsh physically, hydrologically, and chemically (Angradi et al 2001; Windham and Lathrop 1999). These changes can affect the utilization of the marsh by fish, birds, and other animals. Even though there is some knowledge on *P. australis*, its role in salt marsh ecology is poorly understood (Angradi et al. 2001; Fell et al. 1998; Meyer et al. 2001) especially its effect on marsh macroinvertebrates (Fell et al. 1998; Angradi et al. 2001). Recent studies on *P. australis* have found it to be an ecologically functional habitat for salt marsh inhabitants such as nekton (fishes and swimming decapod crustaceans) and benthic invertebrates (Rilling et al. 1998; Meyer et al. 2001). There have also been recent studies that have found it to be a food source for fish (Wainright et al. 2000; Weinstein et al. 2000). However, there have also been studies that have found *P. australis* to be a poor spawning and nursery habitat for *Fundulus heteroclitus* (Able and Hagan 2000; Raichel 2001).

Benthic invertebrates are also important to a functioning salt marsh ecosystem. These organisms are a source of food for salt marsh inhabitants (Kneib 1998; Fell et al. 1988; Sarda et al. 1995), and they are at the base of the estuarine food web (Ishikawa 1989). Benthic communities may also be affected by the invasion of *P. australis*, and there have been recent studies that have looked into this. One study found a *P. australis* marsh to be functionally equivalent to a non-*P. australis* marsh in which both marshes supported benthic invertebrates (Fell et al. 1998) while another study found that a *Spartina* marsh had more primary and secondary production than a *Phragmites* marsh (Angradi et al. 2001).

Benthic invertebrates are being used to assess whether or not salt marsh restorations and/or creations are functioning ecosystems. In order to create a functioning ecosystem a salt marsh restoration should involve reintroducing vegetation while duplicating the nekton and benthos along with other environmental factors of the marsh (Packard and Stiverson 1976; Allen et al. 1994; Sacco et al. 1994). There have been studies that have compared natural versus created or restored marshes, and similarities and differences have been found between the two with regard to fauna (Minello and Webb 1997; Minton 1999; Craft et al. 1998; Havens et al. 1995; LaSalle et al. 1991; Sacco et al. 1994; Moy and Levin 1991). However, the number of years needed for a benthic community in restored or created marshes to be comparable to natural marshes is uncertain.

1.1.1 Salt Marshes

Salt marshes are found in the mid- to high latitudes in the temperate regions of the Northern and Southern Hemispheres. They are found along the coastlines in all parts of the world, and they are an interface between terrestrial and marine habitats (Mitsch and Gosselink 1993). According to Chapman (1977), there are seven characteristics that determine the distribution of salt marshes, which are air temperature, protected coastlines, shallow shores, currents, salinity, tidal range, and substrate.

Salt water or salinity is essential to a salt marsh as well as the tidal range. The distribution of salt marsh vegetation is effected by salinity levels, tidal inundation and elevation of the marsh (Chapman 1960). Tidal flooding divides salt marshes into two sections, which are the upper and lower (intertidal zone) marsh (Mitsch and Gosselink 1993). The upper marsh is flooded irregularly (<1 submergence per day in day light per <360 days per year) while the lower marsh is flooded daily (>1.2 submergence per day in day light per >360 days per year) (Mitsch and Gosselink 1993).

Salt marshes are important to the tidal marsh-estuarine ecosystem. They serve several different important functions. Some of these functions are being involved in nutrient cycling within estuaries, and serving in pollution filtration, sediment accretion as well as erosion control (Niering and Warren 1980). Salt marshes are also home to many resident nekton species as well as a temporary home to transient species from nearby waters (Hettler 1989; Kneib and Knowlton 1995).

Salt marshes are divided into vegetation zones from the low marsh to the terrestrial zone, and the zones are formed by marsh elevation, daily tidal inundation, and interspecific competition (Bertness 1991; Niering and Warren 1977; Niering and Warren

1980; Mitsch and Gosselink 1993). Along creeks and bay fronts, *S. alterniflora* dominates the low marsh while *Spartina patens* (salt meadow cordgrass) mixed with the spike grass *Distichlis spicata* dominates the high marsh (Bertness 1991; Niering and Warren 1977; Niering and Warren 1980; Mitsch and Gosselink 1993). The terrestrial zone is divided into a lower and upper border in which the black grass *Juncus gerardi* (a rush) dominates the lower border while the switch grass *Panicum virgatum* dominates the upper border (Bertness 1991; Niering and Warren 1977).

The zone of importance to this study is the low marsh zone where *S. alterniflora* dominates. It dominates the low marsh zone because of its ability to tolerate the daily tidal inundation since it has a well-developed gas transport system (Bertness 1991). The roots and stems of *S. alterniflora* provide oxygen to its rhizosphere. It also provides oxygen to the sediments and therefore to the benthic invertebrates that are living around the roots in the sediment (Teal and Kanwisher 1966; Teal and Wieser 1966; Healy and Walters 1994). For example, oligochaetes can be found living in the stem sheaths and on the stems to about 5-cm below the sediment (Healy and Walters 1994).

S. alterniflora is a habitat for salt marsh inhabitants such as benthic invertebrates, nekton, and fish. The amphipod *Gammarus palustris* lives in close association with *S. alterniflora* around the culms, and the density of this species decreases with the decrease in the density of *Spartina* (Van Dolah 1978). Fiddler crabs live on the marsh surface, where the root mat density of the vegetation affects their burrowing (Ringold 1979). Crabs are also found at their greatest density at the creek bank habitat where there is vegetation present (Kneib and Weeks 1990). Nekton access the marsh surface during tidal inundation, and utilize the surface where they are protected from predators and where there is an abundant supply of food (Kneib 1993). The foraging area in the intertidal zone of a marsh that is accessible to nekton depends on the tide, and Kneib and Wagner (1994) found that nekton were not evenly distributed on the marsh at high tide, but were more abundant in the creek bank habitat.

Fish and nekton utilize *S. alterniflora* as a spawning and nursery habitat. *F. heteroclitus* places its eggs in *Spartina* stems, and spawns on *Spartina* marshes at high tide in the intertidal zone (Able and Hagan 2000; Kneib 1993). The grass shrimp *Palaemonetes pugio* forage on salt marshes, and feed on benthic invertebrates and plant matter (Kneib and Knowlton 1995). The larvae of *P. pugio* are released into the estuary, and the post-larvae and juveniles utilize the marsh surface in the intertidal zone as a nursery (Kneib and Knowlton 1995).

S. alterniflora is also a good source of food for benthic invertebrates (Teal 1962). Graca et al. (2000) found that the marsh periwinkle *Littoraria irrorata*, the salt marsh coffee-bean snail *Melampus bidentatus*, and the talitrid amphipod *Uhlorchestia spartinophila* feed upon the organic biomass from decomposing *S. alterniflora* leaf blades more so than leaf sheaths. *U. spartinophila* lives in close association with the leaves of *S. alterniflora*, and the decomposing leaves or fungi associated with it are a major part of its diet, which will enable them to grow and reproduce (Kneib et al. 1997).

1.1.2 *Phragmites australis* Invasion

P. australis has been replacing native vegetation (Weinstein and Balletto 1999; Fell at al. 1998; Angradi et al. 2001; Windham and Lathrop 1999) in the Atlantic and the Gulf coasts since the 1900s. Marsh managers have been trying to decrease the presence of *P. australis* on salt marshes. Studies on *P. australis* and its effects on salt marshes are very few. Nevertheless, restoration/mitigation efforts have been occurring to eradicate the unwanted grass and replace it with *Spartina*. Some view *P. australis*-dominated salt marshes as degraded systems, but others view them as an important habitat for wildlife (Fell et al. 1998).

According to Fell et al. (1998), *P. australis* is known to decrease plant species diversity; change the physical habitat of the marsh; change the production of detritus and therefore the consumption of detritus by marsh fauna. These changes can effect the utilization of the marsh by fish, birds, and other animals. *P. australis* increases the elevation of the marsh surface and decreases the microtopography of the surface. Nekton could have a difficult time accessing the marsh surface due to the decreased microtopography (Weinstein and Balletto 1999).

Recent studies seem to indicate that *P. australlis* marshes do provide a habitat to support nekton similarly to *S. alterniflora* (Rilling et al. 1998; Meyer et al. 2001), and that *P. australis* detritus enters food webs similar to *S. alterniflora* detritus (Wainright et al. 2000). A study conducted in *P. australis*-dominated and restored high marshes on the Connecticut River found that *F. heteroclitus*, the most abundant fish species, was captured in both marsh types and were similar in total number and biomass (Rilling et al. 1998), but the mean total length of the fish was longer in the *P. australis* marsh. The

American eel (*Anguilla rostrata*) was also found in both types of marshes with no significant difference in total number or biomass. Species diversity of fish was low for both types of marshes, but it was higher in the *P. australis* marsh. *P. australis* marsh provides a habitat for foraging by fish since they can move around the marsh easily due to the separated stems of *P. australis*. Rilling et al. (1998) established that *P. australis* and the restored marshes were both ecologically functional because both marshes had an abundance of fish and benthic invertebrates, which are a source of food for other animals.

Meyer et al. (2001) also found that nekton species are not significantly different in abundance or biomass between *P. australis* and *S. alterniflora* marshes in the Chesapeake Bay, Maryland. There was also no significant difference in abundances or biomass of the shrimp *P. pugio*, and the three crab species collected in the two types of marshes. Thus, both *P. australis* and *S. alterniflora* support a population of nekton species.

Furthermore, both marsh types can contribute trophically to the nekton species. Isotopic studies have been conducted on *F. heteroclitus* in *Spartina spp.* marshes to determine their source of carbon in their diets, and it has been found to come from a combination of *Spartina spp.*, benthic microalgae, and phytoplankton (Wainright et al. 2000). *P. australis* detritus is a major source of sulphur (S) and carbon (C) for *F. heteroclitus* along with benthic microalgae and phytoplankton in *Phragmites* marshes. Benthic microalgae are not as essential as macrophyte detritus in the trophic food chain in *P. australis* marshes as in *Spartina spp.* marshes. The stable isotope, $\delta^{34}S$, is derived mainly from *Spartina spp.* and *P. australis* as well as other organic sources, and without this source of $\delta^{34}S$ the consumers would suffer from this loss of sulphur in their diets. Another study in the Delaware Bay conducted by Weinstein et al. (2000) looked at the sources of carbon, nitrogen, and sulfur from *Spartina spp.*, *P. australis*, benthic microalgae, and particulate organic matter for the bay anchovy *Anchoa mitchilli* and the white perch *Morone americana*. *P. australis* was found to be a major source of food for these two fish species in the upper estuary of the Delaware Bay while *S. alterniflora* was a dominant food source in the lower estuary where *P. australis* is not a dominant plant. *P. australis* and *Spartina spp.* marshes are therefore both important in the food chain.

However, the two plant species may not be equal in supporting a nursery and spawning habitat for fauna. In two studies, one in southern New Jersey (Able and Hagan 2000) and the other conducted at Mill Creek along the Hackensack River in New Jersey (Raichel 2001), both established that *F. heteroclitus* larvae and small juveniles collected in pit traps were far more abundant in *Spartina* marshes than *P. australis* marshes, probably due to the flatter microtopography of the *P. australis* marshes. However, the size, composition, and abundance of larger fish and abundance of decapods, which were collected in flume nets (Able and Hagan 2000) and the average catch per unit of large juvenile and adult fish collected in minnow traps (Raichel 2001), during the same sampling period for both studies, were similar in both marsh types.

Even though there have been some recent studies conducted on *P. australis* and its affect on salt marshes, there are still areas that need to be further investigated. The utilization of *P.* australis salt marshes by fauna, such as fish and nekton (Rilling et al. 1998; Meyer et al. 2001) and the use of the marsh surface as a nursery habitat by juvenile and fish larvae (Able and Hagan 2000) needs to be further researched. More studies should also address *P. australis'* effect on benthic invertebrates (Angradi et al. 2001).

The functional value of a *P. australis* marsh (Meyer et al. 2001) needs to be assessed as well as whether or not existing ecological functions of a marsh are altered by the invasion of *P. australis* (Rilling et al. 1998).

1.1.3 Benthic Invertebrates

Benthic invertebrates are important components of shallow water estuarine and coastal marine systems (Hopkinson et al. 1999) including salt marshes, because they are a source of food for shorebirds, waterfowl, and the nekton community, especially *F. heteroclitus* and *P. pugio* (Kneib 1988; Fell et al. 1998; Sarda et al. 1995). They play a role in nutrient cycling of the sediments (Ishikawa 1989), and are a link between the primary production of the salt marsh and the nearby body of water (Fell et al. 1998; Sarda et al. 1995). Benthic invertebrates can also be indicators of environmental pollution or disturbance on salt marshes because they are sedentary (Sarda et al. 1995; Fell et al. 1998).

Benthic invertebrates are classified as macrofauna and meiofauna. Macrofauna (large benthic invertebrates) do not pass through a 0.5-millimeter (mm) sieve, while meiofauna are benthic invertebrates that will go through a 0.5-mm sieve, but will not pass through a 0.063-mm sieve (Kneib 1984). Some macrofauna can be considered meiofauna as juveniles, and some meiofauna can be retained on a 0.5-mm sieve as adults (Kneib 1984). Meiofauna include certain taxa, which are nematodes, ostracods, copepods, and foraminifera (Levin et al. 1996), and consist of various feeding types as well (Teal 1962).

There are two major feeding types for large benthic invertebrates, which are filter feeders and deposit feeders (Mitsch and Gosselink 1993). Filter feeders can be found in the hard substrate while deposit feeders dominate the soft substrate. Deposit feeders can be broken down into groups, such as surface-deposit feeders and subsurface-deposit feeders, which are burrowers (Gaston and Young 1992). The deposit feeders that live on the sediment surface, and consume algae, detritus, and meiofauna that include polychaetes, gastropods, crustaceans, and amphipods, while the filter feeders that obtain their food from the water column (plankton) include ribbed mussels and oysters (Mitsch and Gosselink 1993). Teal (1962) categorized annelids, nematodes, oligochaetes, fiddler crabs, and *Littorina* as deposit feeders but all of these are a part of the detritus-algae feeding group. Many benthic invertebrates feed upon detritus (Fell at al. 1998; Sarda et al. 1992), and it is known that they feed upon *S. alterniflora* detritus as well as benthic algae (Teal 1962; Sarda et al. 1992).

Benthic invertebrates are sensitive to their environment. Intertidal benthic invertebrates live in extreme conditions being subject to changes in salinity, temperature, weather, and exposure due to the ebb and flood of the tide (Teal 1962; Sarda et al. 1995). They not only have to contend with the physical factors of salt marshes, but benthic invertebrates also face biological interactions, such as predation, competition, adult-larval interactions, and recruitment (Gaston et al. 1988; Osenga and Coull 1983; Sarda et al. 1995; Kneib and Stiven 1982; Kneib 1984; Sarda et al. 1992; Flynn et al. 1998). Sediment chemistry and physics (Gaston et al. 1988) and substrate type and characteristics (Sarda et al. 1995; Ishikawa 1989) have an effect on the benthos as well. The activities of benthic invertebrates can modify their environment through bioturbation of the sediments, which changes the nutrients in the sediment (Flint and Kalke 1983). Benthic invertebrates also have to deal with the physical structure of plants, such as roots and culms (LaSalle and Rozas 1991). *Spartina* culms affect benthic invertebrate communities since the abundance of individuals, such as macrofauna, juvenile macrofauna, and meiofauna, were greater in samples with culms than those without (Rader 1984). Levin and Talley (2000) reviewed the literature for the effects of vegetation and abiotic environmental factors such as marsh age, salinity, elevation, oxygen, hydrology, organic matter, particle grain size, and faunal biogenic structures on benthic invertebrates in salt marshes, and found that these factors affect the distribution, composition, and diversity of benthic communities.

Salinity also affects the composition, abundances, and diversity of benthic communities (Boesch 1972; West and Ambrose 1992; Sarda et al. 1995; Levin and Talley 2000). Boesch (1972) sampled benthic invertebrates along a salinity gradient from polyhaline (18-30 parts per thousand (ppt)) to oligohaline (0.5-5-ppt) sections of the estuary. The diversity of benthic communities in the Chesapeake-York-Pamunkey estuary was higher in the polyhaline section of the estuary and diversity decreased as the salinity decreased to the oligohaline parts of the estuary (Boesch 1972). West and Ambrose (1992) investigated oligohaline tributaries of the Pamlico River Estuary in North Carolina, and found the diversity of the benthic community to be low and certain species, such as *Streblospio benedicti* were less abundant in these low salinity tributaries. Some species, such as surface-deposit feeders can withstand non-optimal salinity levels (Gaston and Young 1992).

The predators in salt marshes affect the benthic community as well. The aquatic predators of benthic invertebrates in the intertidal zone of salt marshes are fish species (e.g., *F. heteroclitus*), crustaceans, epibenthic invertebrates, and annelids. Kneib and Stiven (1982) found that the size of *F. heteroclitus* determines their foraging behavior in which smaller *F. heteroclitus* prey upon benthic infauna more than the medium to large *F. heteroclitus.* The abundance of annelids, mollusks, and ostracods are affected by fish size.

Benthic communities are not only affected by predators, but also by pollution and the contaminants in their environment. Contaminated sediments directly impact benthic invertebrates (Flynn et al. 1998). The benthic community structure changes with contamination because the organisms cannot move away and reside within the contaminated sediments (Gray et al. 1990; Pocklington and Wells 1992).

There are certain responses that benthic communities have to a stressed, contaminated environment. Some of these responses are a decrease in species diversity, an increase in abundance of certain species, the occurrence of opportunistic species, and changes in the presence or absence of rare species (Gray et al. 1990). The presence or absence of certain species of polychaetes that can or cannot withstand pollution indicate whether or not a benthic community is healthy (Pocklington and Wells 1992). Benthic invertebrates also respond to the pollution in their environment by changing the structure of their community as well by changing the types of feeding groups that are present or absent (Gaston and Young 1992). Opportunistic species also take advantage of a disturbed environment, and some of these species are the spionidae polychaetes *Polydora ligni* and *Streblospio benedicti*, the amphipod *Ampelisca abdita,* and the polychaetes

Capitella spp. These species are considered opportunistic because they have certain characteristics, such as a fast rate of development, high rate of reproduction per year, high recruitment, and high death rate (Zajac and Whitlatch 1982; McCall 1977).

A major source of contaminants that alters benthic communities is heavy metals. Heavy metals in the sediment are affected by environmental processes that can change their bioavailability, such as binding to organic matter and sediment and competing for uptake sites in benthic invertebrates. Gaston and Young (1992) found that the abundance of benthic invertebrates and the number of species were inversely proportional to the metal concentration. The tubificid and naidid oligochaetes (subsurface-deposit feeders) are tolerant to concentration of metals in the sediment (Gaston and Young 1992). In a study in Saudi Arabia, Whaley et al. (1989) found that the abundance, the taxa richness, and the diversity of benthic invertebrates decreased in the outfall channel of discharge water from a petrochemical complex. Maltby (1999) discussed the effects of metal contamination on freshwater benthic invertebrates, and found that some of them can adapt physiologically to reduce their susceptibility to metal stress. Benthic invertebrates that had been residing in a metal contaminated environment could withstand hypoxia, and their respiratory system had not been impaired.

The invasibility of certain plant species may also disturb the environment in which benthic invertebrates reside, and this may also alter the abundance, diversity, and composition of benthic communities. The effect on benthic communities by *P. australis*, the invasive plant species along the Atlantic Coast, is largely unknown. There are only two studies that directly look at the benthic communities found in *P. australis* and non-*P. australis* salt marshes. Fell et al. (1998) conducted a study along the Connecticut River, and found that *P. australis* salt marshes were functionally equivalent to non-*P. australis* salt marshes. Four high marsh macroinvertebrates (*Orchestia grillus*, *Philoscia vittata*, *Melampus bidentatus*, and *Succinea sp.*) were the focus of Fell et al.'s (1998) study since *F. heteroclitus* prey upon these macroinvertebrates. These species were found in both marsh types. Adult *F. heteroclitus* were found on both marsh types on the high marsh, and their gut contents revealed that they were feeding on benthic invertebrates in both marshes. Fell et al. (1998) concluded that both types of salt marshes supported benthic invertebrates, and supported foraging by mummichogs on the marsh surface.

The other study that looked at benthic invertebrates in *Phragmites* and non-*Phragmites* marshes was conducted by Angradi et al. (2001). Benthic invertebrates were found in *S. alterniflora* and *P. australis* salt marshes in southern New Jersey, and Angradi et al. (2001) suggest that the *Spartina* marsh has greater primary and secondary production than the *Phragmites* marsh. Overall abundance of benthic invertebrates was higher in *Spartina* marsh than *Phragmites* marsh, especially at two of the three sampling positions, which were 0.5-meters (m), 5.0-m, and 9.5-m from the edge of the marsh. The abundance of benthos was significantly different between marsh types at the 5.0-m and 9.5-m positions because of a decrease in abundance of nematodes and polychaetes in the *Phragmites* marsh. The taxa richness was significantly higher in the *Spartina* marsh than the *Phragmites* marsh at all sampling positions and dates (Angradi et al. 2001). Dominance by the three most abundant taxa (Oligochaeta, Nematoda, and *Manayunkia aestuarina*) was higher in the *Phragmites* marsh than the *Spartina* marsh, except for the 0.5-m sampling position in which the type of marsh did not affect the results. Dominance was always >85%.

The 12 most abundant taxa in the Angradi et al. (2001) study were Oligochaeta, Ceratopogonidae, Chironomidae, Acarina, small crustaceans, Gastropoda, Isopoda, Polychaeta, Nematoda, Collembola, and Amphipoda. Twelve taxa were noted as the most abundant, eight of which were more abundant in the *Spartina* marsh, while only three of the taxa were more abundant in the *Phragmites* marsh (Angradi et al. 2001). Angradi et al. (2001) also looked at the relationship between benthic invertebrates and the root or detritus biomass found at each marsh type, and discovered that the number of taxa was significantly associated with the root or detritus biomass at the *Phragmites* marsh, but not in the *Spartina* marsh.

A litter bag study was also conducted to examine the abundance, taxa richness and composition of benthic invertebrates in litter bags containing *Spartina*, *Phragmites*, or plastic straws (Angradi et al. 2001). The litter bags with plastic straws seem to have the opposite results (i.e., less or more of certain species) than the *Spartina* and *Phragmites* filled litter bags. This study revealed that the abundance was greater in the *Spartina* than the *Phragmites* litter bags, but there was no difference in taxa richness and composition between the two grass types.

Angradi et al. (2001) found that benthic invertebrates in the *Spartina* marsh were more abundant and diverse than in the *Phragmites* marsh. However, dominance of the three most abundant taxa was higher in the *Phragmites* marsh. These two studies (Fell et al. 1998; Angradi et al. 2001) established that both *P. australis*-dominated and *P. australis*-free marshes are important in the trophic link between the marshes and the estuary.

1.1.4 Restored versus Natural Salt Marshes

Salt marsh restorations/creations along with preservation and protection have become one of the solutions to coastal wetland loss, improving degrading systems (i.e., *P. australis* invaded or diked marshes), and preserving ecological functions of wetlands (Weinstein et al. 1997; Minton 1999). Minton (1999) states that the Rhode Island Coastal Resource Management Council (RICRMC) defines wetland restorations as "...the re-establishment of a wetland which has been degraded to such an extent that the site performs little or none of its original wetland functions", and he states that the NICRMC and the National Research Council (NRC) defines wetland creations as "...the construction of a coastal wetland where one did not previously exist".

Successful restorations should follow three items under the performance criteria developed for regulatory agencies: (1) the percent coverage of favorable vegetation, such as *S. alterniflora* on the marsh (95% of the total vegetated area), (2) the decrease in *P. australis* coverage (≤4% of the total vegetated area), and (3) the percent of open water (≤20% of the total marsh area) (Weinstein et al. 1997). The ecological criteria is concerned with restoring the tidal flow to the marsh that will allow for the growth of *S. alterniflora* and other plants, and sustaining a functioning fish habitat. Weinstein et al. (1997) conducted a study on the Delaware Bay, New Jersey, and projected that *S. alterniflora* should recolonize the marsh at a rate of 9% per year while *P. australis* should decrease in coverage by 10% per year. Minton (1999) discussed a study that was conducted at the Galilee salt marsh in Waltham, Massachusetts conducted by the United States Army Corps of Engineers where the elimination of *P. australis* is estimated to be in approximately ten years after the restoration and increased tidal flooding.

A standard criteria for determining the functionality of restored or created salt marshes in comparison to natural marshes does not exist, and this makes it difficult to assess their quality and to guarantee a "no net loss" of wetlands (Zedler 1993; Posey et al. 1997). The U.S. Army Corps of Engineers has stated an established "no net loss" policy for wetlands, which includes mitigation (creation) as a necessity if any damage is done to a protected habitat (Zedler 1996). According to Zedler (1996), the Office of Technology Assessment defines mitigation "as the avoidance, minimization, rectification, and reduction or elimination of negative impacts or compensation by replacement or substitution". Replacing or substituting the current system is not always effective in restoring ecosystem function because most mitigations involve reintroducing a species, particularly a plant species, without knowing whether or not the existing system can withstand or provide for that plant species (Zedler 1996). The reintroduced plant species may not be able to survive because the necessities for its survival may be absent in the restored ecosystem. Zedler (1996) also raises the question of whether the mitigation should replace the existing ecosystem or reestablish the functioning system that currently exists. "Providing a habitat that is functionally equivalent to the one that will be lost" while maintaining biodiversity through knowledge of the present and historic conditions of the area is the answer (Zedler 1996).

Reintroducing vegetation alone without duplicating the nekton and benthos along with other environmental factors of the marsh does not indicate that the marsh will be a functioning ecosystem (Packard and Stiverson 1976; Allen et al. 1994; Sacco et al. 1994). There have been a number of studies that look at restored or created marshes compared with natural marshes. The abundance of nekton and macroinvertebrates are not similar between created or restored marshes and natural marshes (Minello and Webb 1997; Minton 1999). Minton (1999) points out that the number of years needed for fauna of a restored salt marsh to be equivalent to that of a natural marsh is unknown, and it may be a very long time before they achieve the functionality of a natural marsh. In comparing the benthic communities found in created versus natural marshes, the benthic community in the created marsh was greater in density and species richness than the natural marsh, but the establishment of this benthic community took 25-years (Craft et al. 1998).

There are many studies that have documented differences between natural and restored or created marshes. Since Posey et al.'s (1997) study of the benthic communities found in created marshes of various ages, there has been some new information on the colonization and succession of benthic communities in created marshes. Created marshes are inhabitated by opportunistic benthic species, which will develop according to the sediment, hydrodynamics, and vegetation of the marsh (Posey et al. 1997). Packard and Stiverson (1976) suggest that a created *Spartina* marsh on dredge spoil will eventually sustain a detritus-based community, which is a major food source for benthic invertebrates.

In a few studies, however, salt marsh restorations have been deemed to have similar characteristics as natural marshes (Posey et al. 1997). The more recently created marshes of the Posey et al. (1997) study had more polychaetes than the older created marshes, which had more oligochaetes and amphipods. Some studies indicate that the benthic communities found in natural and restored or created marshes were similar. Havens et al. (1995) established that constructed marshes and natural marshes were similar in benthic communities, but blue crabs and fish abundance were lower in the constructed marsh than the natural marsh. This difference may be due to the lack of microhabitat, marsh morphology, and microtopography of the constructed marsh. Benthic communities as well as fish and shellfish of a naturally developed marsh on dredged material were similar in species composition and abundance to natural marshes (LaSalle et al. 1991). The differences between a 4-year and 8-year developed marsh were investigated and both marshes appeared to be developing.

Minello and Webb (1997) found that the amount of macro-organic matter, the abundance of fish, decapod crustaceans, and benthic invertebrates and overall species richness of the benthic invertebrates were higher in natural marshes than created marshes, but the differences were not associated with the age of the created marshes. Levin and Talley (2000) looked at studies that focused on created versus natural marshes to determine if marsh age was a factor, and these studies have shown that marsh age may or may not affect macrofauna. In Moy and Levin's (1991) study, planted and natural marshes were compared, and established that higher organic matter was found in the natural marshes than the planted marsh. However, gut contents of *Fundulus* were similar or richer in the planted marsh than in the natural marshes. There were fewer juvenile *Fundulus* found on the planted marsh, and Moy and Levin (1991) speculate that this may be due to the smooth, hard surface of the planted marsh, reduced *Spartina* densities, and the lack of mussel and clam shells on the planted marsh, that are spawning sites utilized by *Fundulus*. Benthic communities were structurally similar in the natural and artificially established marshes, but the artificially established marshes had lower densities, which may be due to the lower amount of organic matter and lower stem density (Sacco et al.

1994). Allen et al (1994) found that the there was a difference in the foraging ability of *F. heteroclitus* in the restored impounded and natural marshes, and suggest that a restored impounded marsh was not equivalent to a preimpounded marsh.

1.2 Objective

The first objective of this study was to compare the benthic communities in *S. alterniflora, P. australis* and restored *S. alterniflora* marshes. The (1) abundance of individuals, (2) taxa richness and species diversity (3) species composition, and (4) substrate characteristics were investigated. The second objective of this study was to conduct a recolonization experiment in highly contaminated and disturbed salt marshes using sediment from an uncontaminated and undisturbed salt marsh in order to determine if sediment characteristics are important in benthic colonization. This study was focused on restored and natural salt marshes in a highly urbanized area, the Hackensack Meadowlands District, New Jersey.

CHAPTER 2

METHODS

2.1 Study Sites

Chapman (1960) categorized the salt marshes of the world into nine regions. New Jersey salt marshes, which are the focus of this study, are considered New England salt marshes, which run from Maine to New Jersey and are part of the Western Atlantic Group (Chapman 1977; Niering and Warren 1980). The Hackensack Meadowlands, New Jersey is located in the Piedmont Plateau geologic region of New Jersey (Waksman et al. 1943). New Jersey salt marshes were formed behind barrier beaches and in front of coastlines, and they were composed of salt marsh grasses, which decomposed into fine organic mud (Chapman 1940; Waksman et al. 1943).

The Hackensack River Basin was formed during the Wisconsin Glaciation, which created Glacial Lake Hackensack when the glaciers melted (Heusser 1949; TAMS 1990). A terminal moraine enclosed Glacial Lake Hackensack, and the wetland came about when the terminal moraine was breached by rising sea levels, which allowed the Glacial Lake Hackensack to drain. Once the area was drained, it became a black ash swamp, which was eventually replaced by black spruce and larch due to rising sea level and water table and peat accumulation (Heusser 1949). White cedar (*Chamaecyparis thyiodes*) swamp forest became the predominant vegetation and almost disappeared by 1919. In 1935, white cedar trees were found in the Secaucus bog, and the last remnants of the white cedar swamp forest could still be seen in 1949 in a bog at Moonachie, which is located on the Hackensack River (Heusser 1949). The white cedar swamp was affected by human disturbances such as logging, fires, ditching, and pollution (Heusser 1949). The area then became a tidal salt marsh where *P. asutralis* is dominant.

However, it is uncertain when *P. australis* first appeared in the Hackensack Meadowlands District, but there have been a few published sightings of *P. australis* in the 1800's in New Jersey. Willis (1877) put together a list of the plants that were found in New Jersey and *Phragmites communis* was found in meadows located in Ocean County and the Hackensack Meadows. *P. communis* is the older nomenclature for *P. australis* (Gleason and Cronquist 1991). Britton (1889) also found *Phragmites vulgaris* in swamps and on the borders of ponds throughout New Jersey. *P. vulgaris* is the older nomenclature for *P. communis* (Britton 1889). Harshberger and Burns (1919) cite the presence of *P. communis* as a dominant and common plant species in the Hackensack Meadowlands, and there are photographs of *P. communis* along the Hackensack River that date back to 1916.

The Hackensack Meadowlands also had to contend with a lot of human disturbances since the area first became settled during the 17th century. Some historical disturbances are salt hay farming, diking and draining for farmland and mosquito control. Some recent disturbances are the construction of highways, pipelines, urbanization, the construction of the Oradell Dam, industrial and residential wastewater discharges, heavy metal wastes, and development (TAMS 1990). All of these forms of urbanization are responsible for the altered wetlands and tidal flow in the area.

The Hackensack Meadowlands District is 32-square miles located in Bergen and Hudson counties of New Jersey (Figure 2.1). The study sites are located in Sawmill Creek (SMC) and Mill Creek (MC). MC is located in Secaucus, New Jersey (Figure 2.2) and SMC is in Lyndhurst, New Jersey (Figure 2.3). Sawmill Creek is found along the southern section of the Hackensack River. It is a natural salt marsh that was formed in 1950 by a tide gate breach (Kraus and Kraus 1988). The known salinity range at this site is 6.9 to 15.7-ppt, and the tidal range is approximately 1.5-meters (m). *P. australis* is the dominant vegetation at SMC, but on extensive marshes *S. alterniflora* can be found there as well.

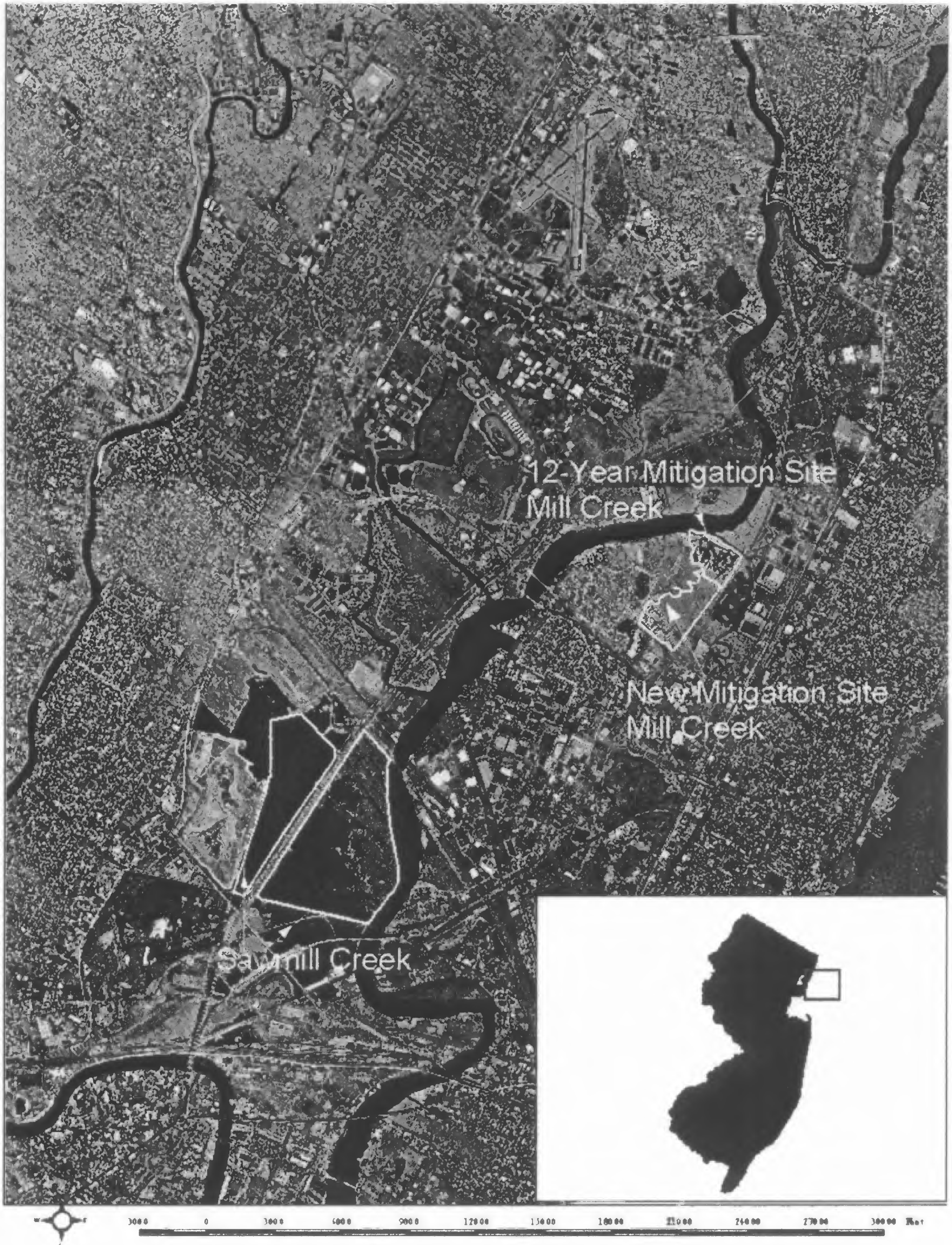

Figure 2.1 Site map of Hackensack Meadowlands District, New Jersey

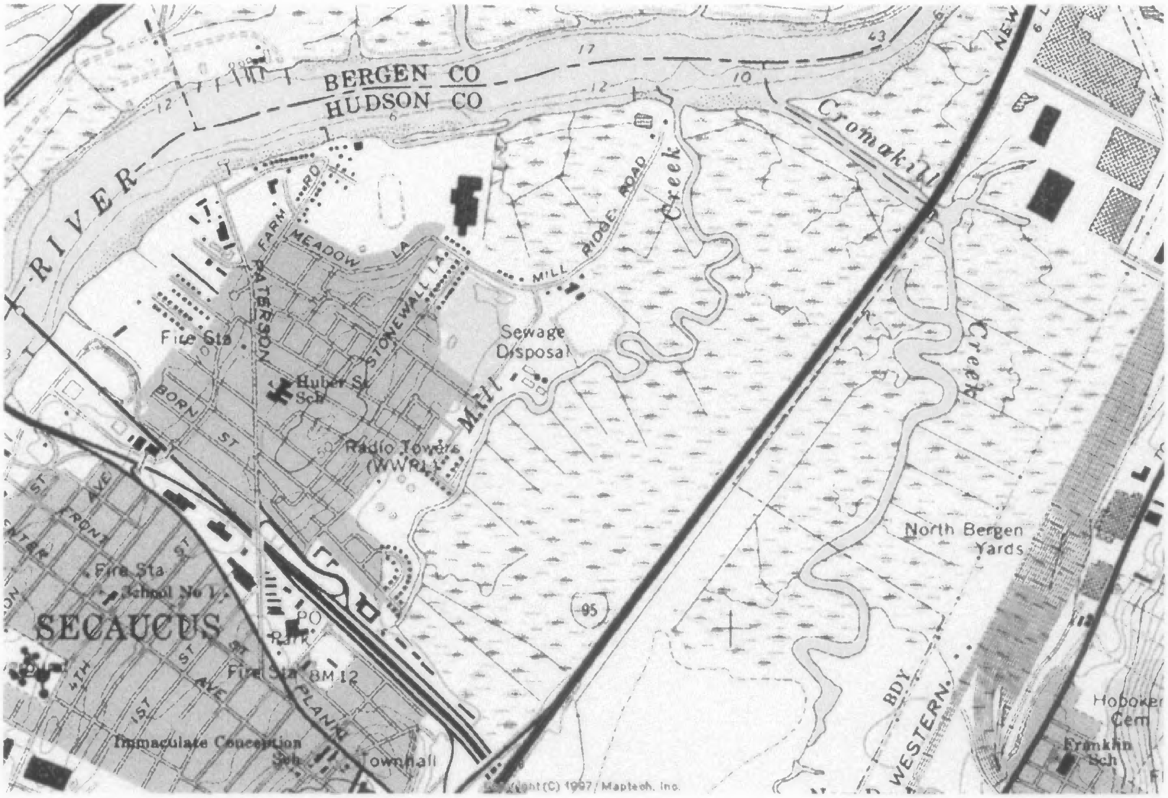

Figure 2.2 Site map of Mill Creek, Secaucus, New Jersey.

Map Name: Weehawken. USGS Reference Code: 40074-G1-TF-024. Year Created: 1967. Revised/Inspected: 1981.

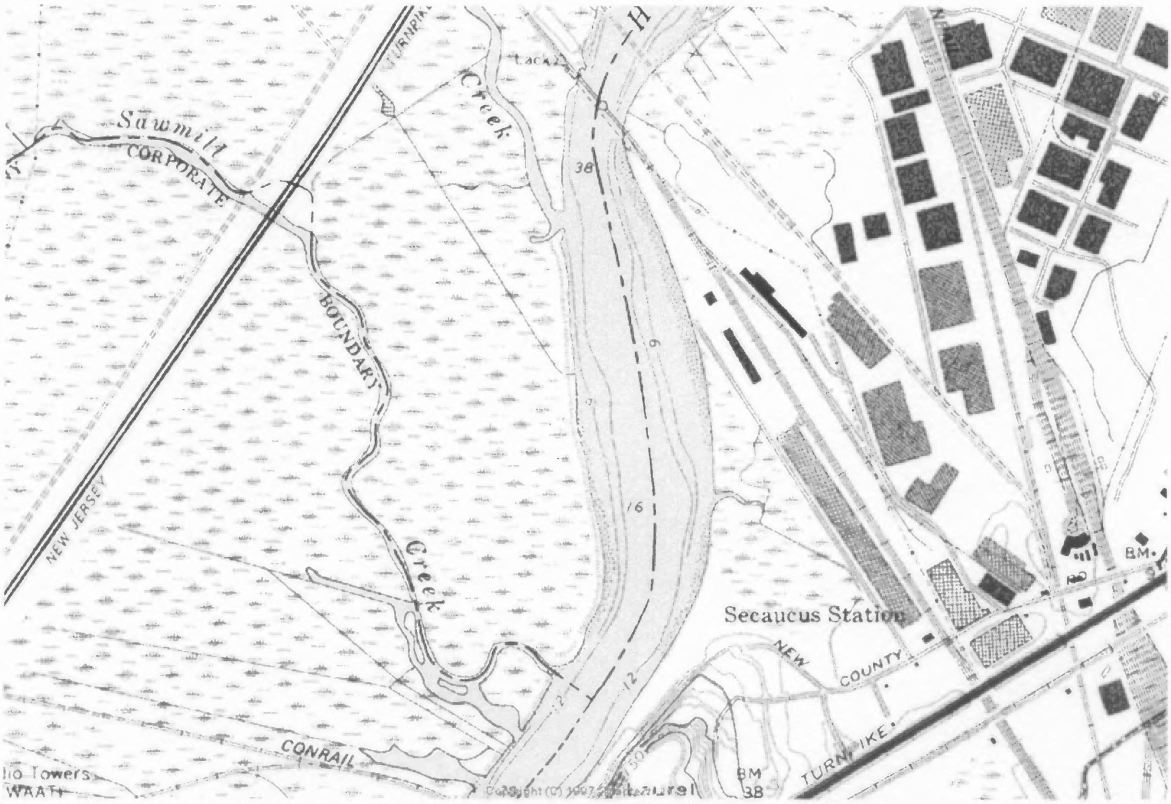

Figure 2.3 Site map of Sawmill Creek, Lyndhurst, New Jersey.
Map Name: Weehawken. USGS Reference Code: 40074-G1-TF-024. Year Created: 1967. Revised/Inspected: 1981.

The other two sites of this study are mitigated marshes located on Mill Creek. One of the marshes was restored 12-years ago while the other was undergoing the beginning stages of the restoration at the time of the sampling. The Hartz Mountain Company mitigated the 12-year Mitigation Site (M-12), and it is found on the western brackish marsh section adjacent to Mill and Cromakill Creeks (HMDC 2000) along the northern part of the Hackensack River. The Hartz Mountain Company in 1985-1988 removed *P. australis*; decreased the site elevations therefore increasing daily tidal inundation; and planted *S. alterniflora*, and monitored the site (TAMS 1990). The site is approximately 63 acres, surrounded by industry, highways, and urbanization (TAMS

1990; HMDC 2000). The tidal range at M-12 is approximately 1.5-m, and the salinity range at this site is 0.5 to 7.0-ppt (Kraus and Kraus 1988). The rest of MC is dominated by *P. australis*.

In 1998 another restoration project took place in MC. The New Mitigation Site (M-0) is a 140-acre site, and the site grading was underway during the time of sampling in 1999. The newly mitigated section of Mill Creek is adjacent to the 12-year Mitigation Site. The area was dominated by monoculture stands of *P. australis* before the mitigation, which was conducted to eliminate *P. australis*, to increase tidal inundation, and to create a low marsh system (HMDC 2000).

Kraus and Kraus (1988) reported that the Hackensack Meadowlands Development Commission (HMDC) in 1982 found heavy metals in sediments of SMC. Kraus and Kraus (1988) conducted their own heavy metal analysis on the sediment. The water quality at SMC is better than at MC because of lower concentration of heavy metals at SMC than MC (Kraus and Kraus 1988). The heavy metal concentrations found in the sediment at MC were lower than SMC because the surface sediment was graded at MC, and this removed the layer of sediment with the highest concentration of metals (Kraus and Kraus 1988). It is also known that skeet shooting is a part of Mill Creek's history, which is responsible for the high concentrations of lead (Kraus and Kraus 1988). Currently, a sewage treatment plant discharges into MC near the newly mitigated section (Figure 2.2).

The mitigated areas of MC are also contaminated by heavy metals. Kraus and Kraus (1988) reported historic findings from the General Plan, Environmental Impact Assessment Report conducted by TAMS in 1985 of the presence of heavy metals. Kraus and Kraus (1988) conducted their own heavy metal analysis of the sediment at MC in 1986 at (1) the non-mitigated section of the marsh, which is adjacent to the mitigation site, and (2) the mitigation site. The site is also considered to be degraded due to the types of organisms found there, such as oligochaetes, hydrobiidae snails, and mummichogs, which are known to be pollution tolerant (TAMS 1990). Some other reasons that the area is known to be degraded are the low dissolved oxygen levels (<4.0-ppm in the summer months) and high biological oxygen demand (>5-ppm) (TAMS 1990).

2.2 Materials and Methods

2.2.1 Benthic Sampling

The area sampled at SMC consists of adjacent stands of *P. australis* (P) and *S. alterniflora* (S) that are separated by a tidal creek. At MC, the samples were collected at the 12-year old mitigated section of *S. alterniflora* (M-12) and the newly mitigated section (M-0). At the time of sampling at the M-0 site, the mitigation was underway, and all that remained was a bare substrate, some *P. australis*, and *Pluchea purpurascens,* an annual salt marsh fleabane that flowers between late summer and fall (Newcomb 1977). The samples were collected at two habitats on the low marsh, which are (1) the creek bank and (2) the edge of each type of vegetation (*P. australis* and *S. alterniflora)* on the marsh plain. At both SMC sites (S and P), the area sampled is a steep creek bank coming off of the vegetated marsh surface that goes into an emergent area in the intertidal zone. At both MC sites (M-12 and M-0), the creek bank is a gently sloping emergent area in the intertidal zone coming off of the vegetated (M-12) or non-vegetated (M-0) marsh surface.

Benthic samples were taken with a PVC core sampler with a diameter of 3.9-centimeters (cm). The cores were taken to a depth of 5-cm at each sampling station. Three replicates were collected at each sampling station. Samples were collected once a month at low tide from June 1999 to September 1999 from the creek bank adjacent to each marsh. The edge of the vegetation on the marsh plain was sampled from July 1999 to September 1999. Salinity at each site was measured with a refractometer.

2.2.2 Biota

In the field, the samples were preserved in 20% Formalin. After approximately one or two weeks, the samples were transferred to 70% Ethanol with Rose Bengal dye for staining of the benthic invertebrates. The samples were sieved in a 0.3-mm sieve, sorted, and identified to the lowest possible taxonomic level (Weiss 1995) to determine taxa richness, overall abundance, and species composition. Some organisms were not identified to species, such as nematodes and copepods while others were identified to species (i.e., *Streblospio benedicti*), and this was determined because of the functionality of the organisms. For example, nematodes and copepods are both small enough in size to be prey for larval fish (Raichel 2001) while *S. benedicti* may be involved in infaunal interference interactions (Levin 1982). The number of nematodes was estimated for those sites that had a count of ≥ 100 individuals in a petri dish. When this was the case, the total number of petri dishes utilized for the sorting procedure was counted, and this was multiplied by the mean number of nematodes counted in the first two petri dishes sorted for that particular site.

2.2.3 Sediments

In September 1999, sediment samples were also taken with the PVC corer for determining sediment characteristics. The samples were taken at a depth of 0-5 cm at each sampling station, and three replicates were taken at each sampling station. Prior to lab work up of samples, they were kept in a cold room. The samples were initially sieved through a 4.75-mm sieve to collect any pebbles, stones, and large pieces of organic material. The sediment characteristics that were determined are percentage of organic matter and the percentage of silt, sand, and clay.

2.2.3.1 Percentage of Organic Matter Five-grams (g) of sediment were placed in a ceramic crucible, weighed, and placed in a 105°C oven for 16 hours to dry. They were then reweighed to determine the dry weight. The organic matter content was then determined. Any pieces that were greater than 1/8-inch (in) diameter were removed from the dry sample. The samples were reweighed and then placed in a 440°C muffle furnace in a hood for 16 hours. The samples were then cooled in the hood in order to be reweighed. This weight is the ashed weight. The percent organic content then can be calculated through this equation:

$$(\text{Dry Weight} - \text{Ashed Weight})/\text{Dry Weight} \times 100$$

2.2.3.2 Percentage of Silt, Sand, and Clay The sample was air dried on a piece of paper in a greenhouse. Once it was dry, pieces that were greater than 1/8-in diameter were removed from the sediment after it was crushed with a mortar and pestal. The sediment was analyzed using the LaMotte Soil Texture Unit® (LaMotte 1999) which provides a volumetric calibration of sand, silt, and clay through sedimentation in an aqueous solution.

2.2.4 Recolonization Experiment

A recolonization experiment was conducted at the study sites to determine if sediment characteristics, such as contaminants, are important in benthic colonization. Sediment was collected on May 25, 1999 at a relatively uncontaminated and undisturbed salt marsh at the Rutgers University Marine Field Station in Tuckerton, New Jersey. The sediment was taken at low tide in the creek bank. It was placed in a bucket and sealed until June 7, 1999. It was left in anoxic and greenhouse heated conditions, in order to defaunate the sediment by depleting the oxygen and killing the ambient organisms (Zajac and Whitlatch 1982). The sediment was then sieved through a 2-mm sieve. The sieved defaunated sediment was placed in 118-milliliter (ml) Rubbermaid® storage containers 5.8-cm diameter and 5.9-cm height. After examining them microscopically to see that no living organisms remained, they were placed in the creek bank at the two Mill Creek and two Sawmill Creek sites in June 1999. The containers were flush with the ground (Smith and Brumsickle 1989; Zajac and Whitlatch 1982). Three replicates were collected during each sampling event in July and August 1999. The samples collected were processed in the same manner as the ambient benthic samples. Sediment samples were collected on

March 3, 2001 to determine the percentage of silt, sand, and clay for the Tuckerton sediment. Sediment samples were also collected on April 7, 2001 to determine the percentage of organic matter for the Tuckerton sediment. The sediment samples were collected in the same manner as the Hackensack Meadowlands sediment.

2.2.5 Statistical Analysis

All of the benthic samples were analyzed on Statistical Analysis System (SAS) Version 8 (SAS 1999-2000). Multiple analysis of variance (MANOVA) along with the Student Newman Keuls (SNK) test was utilized to analyze the monthly benthic samples. The significance level used was $P<0.05$. The percentage of organic matter and the percentage of silt, sand, and clay were analyzed by running analysis of variance (ANOVA) on the data. All data were tested for normality and all of the abundance data were not normally distributed, and so were log transformed. All of the other data collected were normally distributed. The Shannon-Wiener and Simpson's Taxa Diversity Indices were calculated on the Multivariate Statistical Package (MVSP) (MVSP 1985-2000). Shannon-Wiener diversity index is a Type I index, which is affected by the rare species in the community (Krebs 1999). Simpson's diversity index is a Type II index, which is affected by the more abundant species in the community (Krebs 1999). The range for Simpson's index is from zero to one. Principal components analysis (PCA) was conducted on the ten most abundant taxa in the ambient benthic sampling. PCA was also conducted on the 11 most abundant taxa from the ambient and recolonized samples.

These taxa were chosen to be analyzed because they were greater than or equal to 1.0% of the total abundance. PCAs were also conducted on the percentage of silt, sand, clay, and organic matter. PCAs were conducted on SAS as well.

CHAPTER 3

RESULTS

3.1 Environmental Parameters

3.1.1 Organic Matter

In the creek bank samples, there was a significant difference ($P<0.05$) between S and the other three sites in that S had more organic matter present (Figure 3.1). There was also a significant difference between the percentage of organic matter at Tuckerton and the Hackensack Meadowlands sites except for M-12. Tuckerton had less organic matter present. There was no significant difference at the edge of the vegetation amongst the sites. However the mean percentage of organic matter was significantly different from the creek bank and the edge of the vegetation at M-12, M-0, and S. At M-12 and M-0, the mean percentage of organic matter was significantly higher at the edge of the vegetation habitat, and at S, the mean percentage of organic matter was significantly higher on the creek bank.

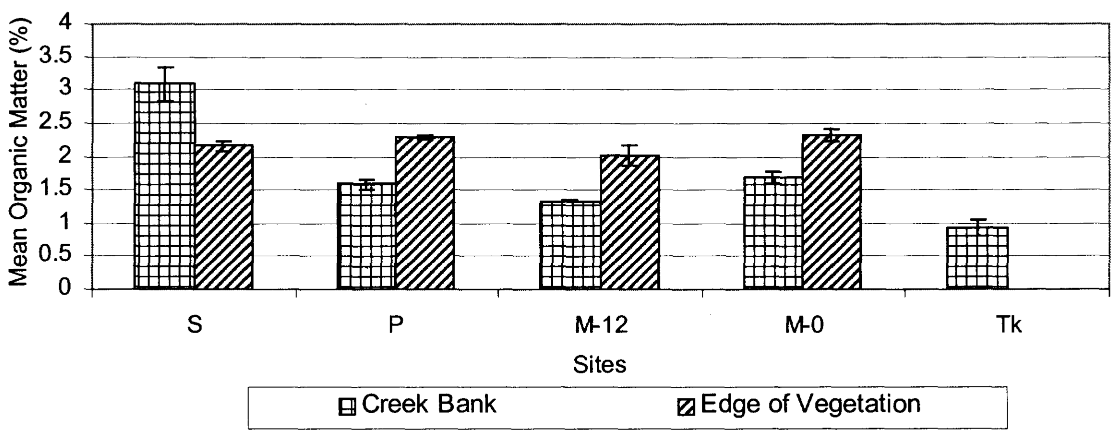

Figure 3.1 Mean percentage of organic matter at the creek bank and the edge of the vegetation habitats. Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) during September 1999. Tuckerton (Tk) samples were taken at the creek bank in April 2001.

3.1.2 Percentage of Silt, Sand, and Clay

The percentage of silt, sand, and clay was determined for both habitats at all sites during the September sampling (Figure 3.2 and Figure 3.3). There was a significant difference found amongst the creek bank sites for the percentage of silt. P and S had a significantly ($P<0.05$) more silt content than M-12. S was significantly coarser than P on the creek bank since S had more sand present. S had fewer fine particles (less clay content) than the other three sites at the creek bank. P was also significantly less coarse than M-12 and M-0 since P had less sand content in its sediments. There was no significant difference of percentage of silt, sand, and clay at the edge of the vegetation.

The sediments were classified by their basic textural classes by using a textural triangle (USDA et al., no year listed). P at both habitats was considered to have clay sediments as well as S and M-0 at the edge of the vegetation habitat. S at the creek bank was considered a loam while Tuckerton at the creek bank was a sandy loam. The sediments at M-12 at both habitats and M-0 at the creek bank were a clay loam.

PCA was conducted on the mean percentage of silt, sand, clay and organic matter. The first principal component (PC1) consisted of 74.8% of the total variation, and the second principal component (PC2) had 25.1% of the total variation (Table 2.1). The correlation coefficients of each variable (sediment characteristic) from PC1 and PC2 that had a significant ($P<0.05$) affect on the principal components were taken into consideration, and these were sand and clay for PC1 and silt for PC2 (Table 2.1). Sc (c-creek bank) and Se (e-edge of vegetation) were different from each other, and Sc had more sand than Se while Se had more clay (Figure 3.4). Pc and Pe were different from each other as well, in which Pc had more silt than Pe (Figure 3.4).

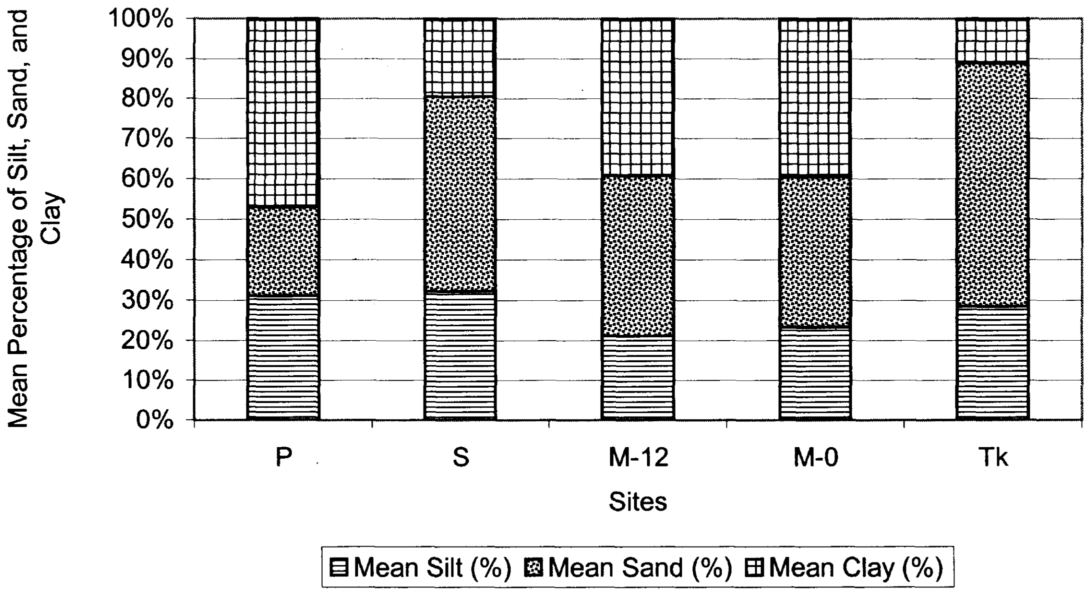

Figure 3.2 Mean percentage of silt, sand, and clay at the creek bank habitat.

Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek, (M-12) and New Mitigation Site Mill Creek (M-0) during September 1999. Tuckerton (Tk) samples were taken at the creek bank in March 2001.

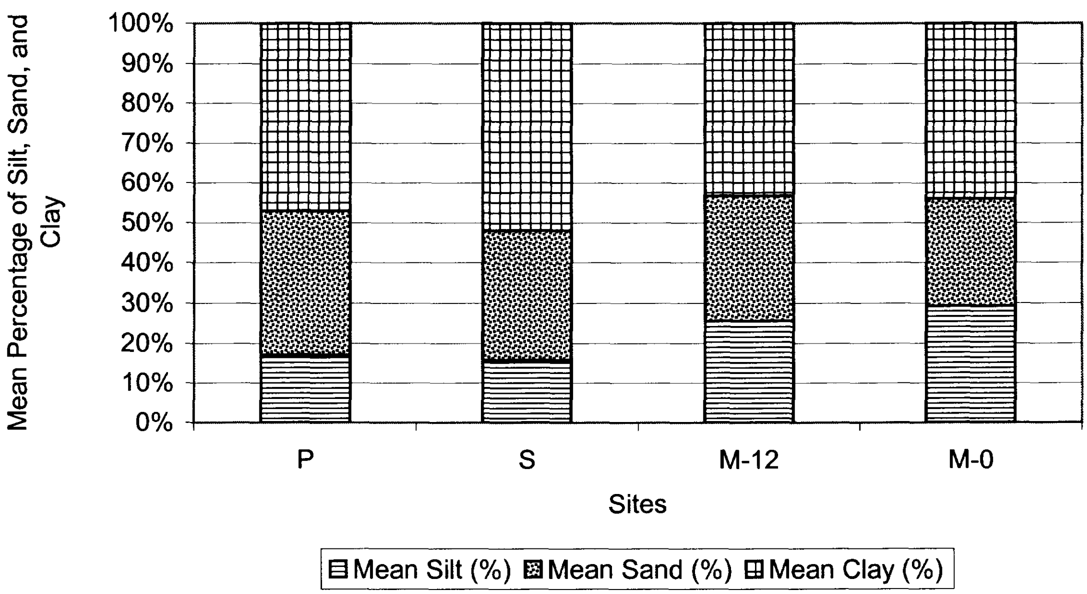

Figure 3.3 Mean percentage of silt, sand, and clay at the edge of the vegetation habitat.

Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) during September 1999. Tuckerton (Tk) samples were taken at the creek bank in March 2001.

Table 3.1 Principal component analysis results of sediment characteristics.

Principal component loadings, eigenvalues, and percentage of variation of each principal component, and the cumulative percentage of variation at the creek bank and edge of the vegetation habitats at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) sampled during September 1999. c=creek bank. e=edge of the vegetation.

Site ID	Site	PC1	PC2
Pc	P-Creek Bank	-9.9298	11.3115
Sc	S-Creek Bank	27.1902	2.153
M-12c	M-12-Creek Bank	4.0988	-5.441
M-0c	M-0-Creek Bank	3.0238	-2.2854
Pe	P-Edge of Vegetation	-5.109	-7.8938
Se	S-Edge of Vegetation	-11.3267	-7.7726
M-12e	M-12-Edge of Vegetation	-2.8666	2.2896
M-0e	M-0-Edge of Vegetation	-5.0807	7.6387
Eigenvalue		150.325440	50.533225
Percentage of Variation		74.76	25.13
Cumulative Percentage		74.76	99.9

Pearson Correlation Coefficients between sediment characteristics and the principal components.

Sediment Characteristics	PC1	PC2
Silt	0.44059	0.89771*
Sand	0.85726*	-0.51488
Clay	-0.98967*	-0.14339
Organic Matter	0.53790	0.03455

*Indicates significant correlation coefficient ($P<0.05$)

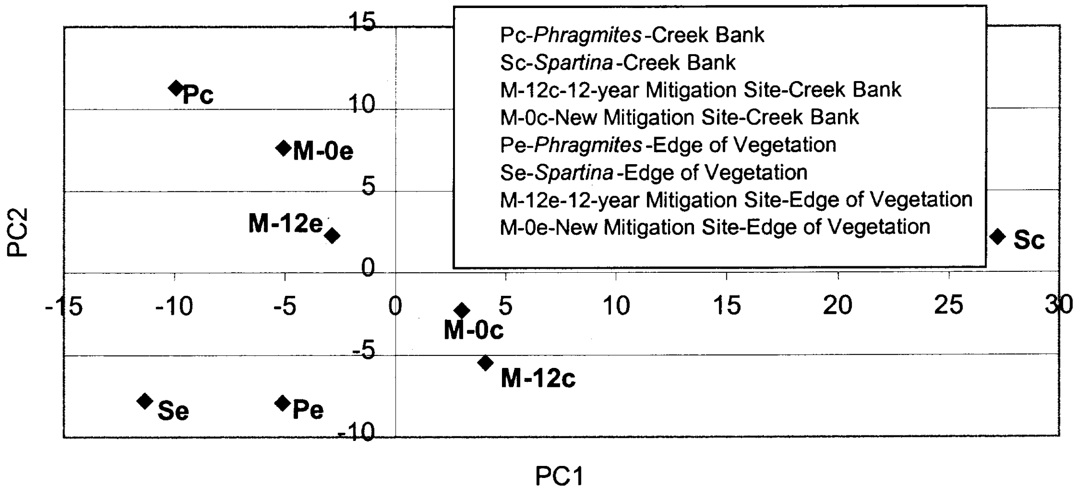

Figure 3.4 Principal Component Analysis for sediment characteristics at the creek bank and edge of the vegetation habitats.
Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) during September 1999.

The sediment from the Tuckerton salt marsh taken from the creek bank that was used in the recolonization experiment was also analyzed for the percentage of silt, sand, and clay (Figure 3.2). The sediment at Tuckerton was similar to the Hackensack Meadowlands sites (P, S, M-12, and M-0) at the creek bank habitat in regards to the percentage of silt. The Tuckerton sediment was significantly coarser than P, M-12, and M-0, but similar to S. It was also similar to S for the percentage of clay therefore P, M-12, and M-0 were significantly finer than the Tuckerton sediment.

3.1.3 Salinity

Brackish estuarine systems are classified as having three types of salinity modifiers, which are polyhaline (18.0-30.0-ppt), mesohaline (5.0-18.0-ppt), and oligohaline (0.5-5.0-ppt) (Mitsch Gosselink 1993). Freshwater systems have a salinity level of <0.5-ppt. Marine systems have a salinity range of ≥30-ppt

The salinity was different between the two creeks. At SMC, the salinity range from June 1999 to September 1999 was 8-ppt in June; 15-ppt in July; 20-ppt in August; and 4-ppt in September (Figure 3.5). At MC (M-12), the salinity over the same period and sampled on the same days was 3-ppt in June; 10-ppt in July; 10-ppt in August; and 2-ppt in September (Figure 3.5). The increase in salinity from June to August was due to the drought that occurred in New Jersey during the 1999 summer. The salinity decreased drastically in September due to the rainfall associated with Hurricane Floyd that went through New Jersey on September 16, 1999, which was the week before my last sampling event.

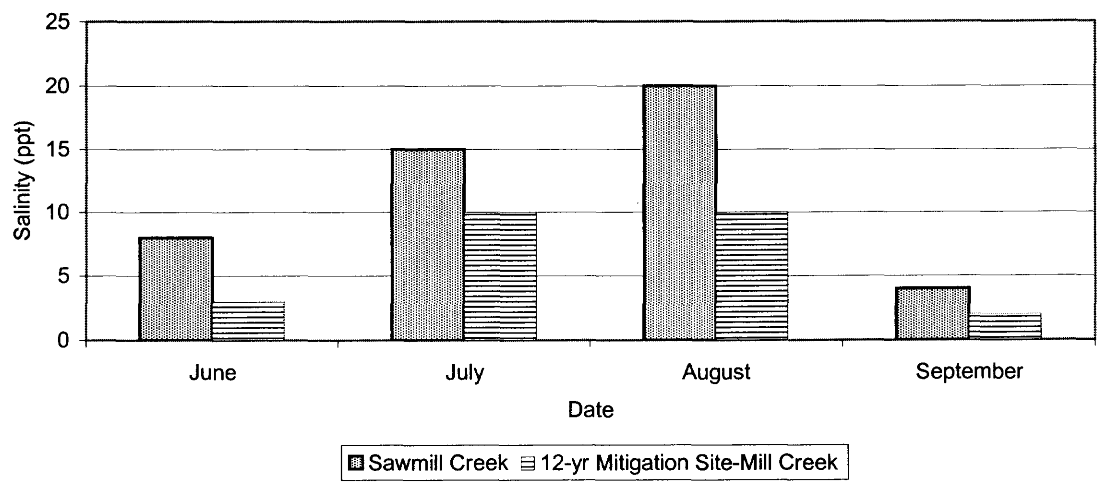

Figure 3.5 Salinity levels at Sawmill Creek and Mill Creek (M-12) monitored from June to September 1999.

The SMC and MC sites are considered brackish estuarine systems since their salinity levels can be classified as oligohaline to polyhaline. MC can be classified as oligohaline and mesohaline. SMC can be classified as oligohaline to polyhaline. Although, the low numbers in September are not indicative of the typical salinity levels for the Summer of 1999 at MC and SMC since the low salinity levels are due to the rainfall after the Hurricane.

3.2 Benthic Samples

The overall mean abundance, mean taxa richness, mean Shannon-Wiener and Simpson's diversity indices for each site has been compiled (Table 3.2). All data has been pooled together for both habitats and all sampling dates for each site (Table 3.2). The following sections on abundance, taxa richness, and diversity indices look at the data more specifically with regard to habitat and sampling dates.

Table 3.2 Mean abundance (#/m^2), mean taxa richness, and mean Shannon-Wiener and Simpson's Taxa Diversity Index at Sawmill Ceek-P. australis (P), Sawmill Creek-S. alterniflora (S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) sampled from June to September 1999. N=3.

Site	Mean Abundance +/- Standard Error (#/m^2)	Mean Taxa Richness +/- Standard Error	Mean Shannon-Wiener Diversity Index +/- Standard Error	Mean Simpson's Diversity Index +/- Standard Error
June				
P-Creek Bank	155030+/-15.52	9.67+/-0.88	1.66+/-0.06	0.76+/-0.01
S-Creek Bank	98297+/-26.87	6.67+/-0.88	1.33+/-0.08	0.64+/-0.04
M-12-Creek Bank	1262279+/-511.38	9.00+/-0.58	1.05+/-0.18	0.52+/-0.09
M-0-Creek Bank	528778+/-57.36	8.00+/-0.58	1.12+/-0.13	0.58+/-0.06
July				
P-Creek Bank	35447+/-15.34	7.68+/-1.45	1.64+/-0.14	0.75+/-0.04
P-Edge of Vegetation	100811+/-30.61	6.00+/-0.58	1.06+/-0.22	0.52+/-0.12
S-Creek Bank	46928+/-28.83	7+/-1.53	1.50+/-0.18	0.73+/-0.04
S-Edge of Vegetation	122097+/-31.54	4.67+/-0.67	0.71+/-0.12	0.37+/-0.08
M-12-Creek Bank	1660916+/-622.00	10.00+/-0.58	0.94+/-0.17	0.45+/-0.09
M-12-Edge of Vegetation	554756+/-49.11	8.33+/-0.33	1.13+/-0.02	0.65+/-0.01
M-0-Creek Bank	578220+/-106.29	7.33+/-0.88	0.79+/-0.09	0.39+/-0.04
M-0-Edge of Vegetation	611991+/-158.60	5.67+/-0.33	0.94+/-0.09	0.53+/-0.05
August				
P-Creek Bank	100309+/-30.59	10.33+/-0.88	1.77+/-0.03	0.78+/-0.01
P-Edge of Vegetation	125113+/-36.43	9.67+/-0.88	1.37+/-0.08	0.67+/-0.04
S-Creek Bank	84889+/-41.38	8.67+/-0.33	1.65+/-0.14	0.738+/-0.05
S-Edge of Vegetation	240506+/-80.48	5.33+/-0.88	1.06+/-0.07	0.62+/-0.04
M-12-Creek Bank	2139414+/-720.27	10.67+/-0.33	0.66+/-0.08	0.33+/-0.05
M-12-Edge of Vegetation	1670721+/-608.09	7.67+/-0.67	0.80+/-0.10	0.44+/-0.07
M-0-Creek Bank	2508134+/-574.03	7.00+/-0.00	0.37+/-0.06	0.15+/-0.03
M-0-Edge of Vegetation	343329+/-71.01	7.00+/-1.00	1.33+/-0.10	0.66+/-0.05
September				
P-Creek Bank	38799+/-20.38	6.67+/-1.45	1.49+/-0.19	0.72+/-0.05
P-Edge of Vegetation	85225+/-6.57	6.33+/-0.33	1.33+/-0.14	0.67+/-0.05
S-Creek Bank	91342+/-14.47	7.67+/-0.88	1.30+/-0.06	0.61+/-0.01
S-Edge of Vegetation	109778+/-3.46	6.00+/-0.58	1.11+/-0.14	0.62+/-0.05
M-12-Creek Bank	1791393+/-446.71	9.67+/-0.67	0.63+/-0.13	0.29+/-0.07
M-12-Edge of Vegetation	3470745+/-506.24	6.67+/-0.67	0.33+/-0.08	0.15+/-0.04
M-0-Creek Bank	1652536+/-119.98	5.67+/-0.33	0.38+/-0.02	0.17+/-0.01
M-0-Edge of Vegetation	128801+/-4.91	6.67+/-0.33	1.31+/-0.10	0.66+/-0.05

3.2.1 Abundance

The abundances of organisms were analyzed on the creek bank at all sites for all months sampled (Figure 3.6). There was a significant difference ($P<0.05$) in the total number of organisms for all months sampled on the creek bank. During the months of June, July, August, and September 1999, the SMC sites had a significantly less abundance than the MC sites. The SMC sites were also significantly different from each other during September, when S had more abundance than P.

The mean abundance at the edge of the vegetation was also significantly different for all three months sampled (Figure 3.7). During July, P and S had significantly less abundance than M-12 and M-0. During August, M-12 had significantly more abundance than the other three sites. M-12 once again had significantly more abundance than the other sites in September, but the other three sites were also significantly different from each other, in that M-0 and S had significantly more abundance than P. There was a significant difference between the two habitats at M-12 for July and M-0 for August and September. The creek bank had more abundance than the edge of the vegetation at M-12. During August and September, at M-0, the creek bank had more abundance than the edge of the vegetation. In comparing the SMC sites, S had a greater abundance of benthic invertebrates then P at both habitats, and at MC, M-12 had greater abundance than M-0 at both habitats as well. Overall, both MC sites had a more abundant benthic community than both SMC sites at both habitats.

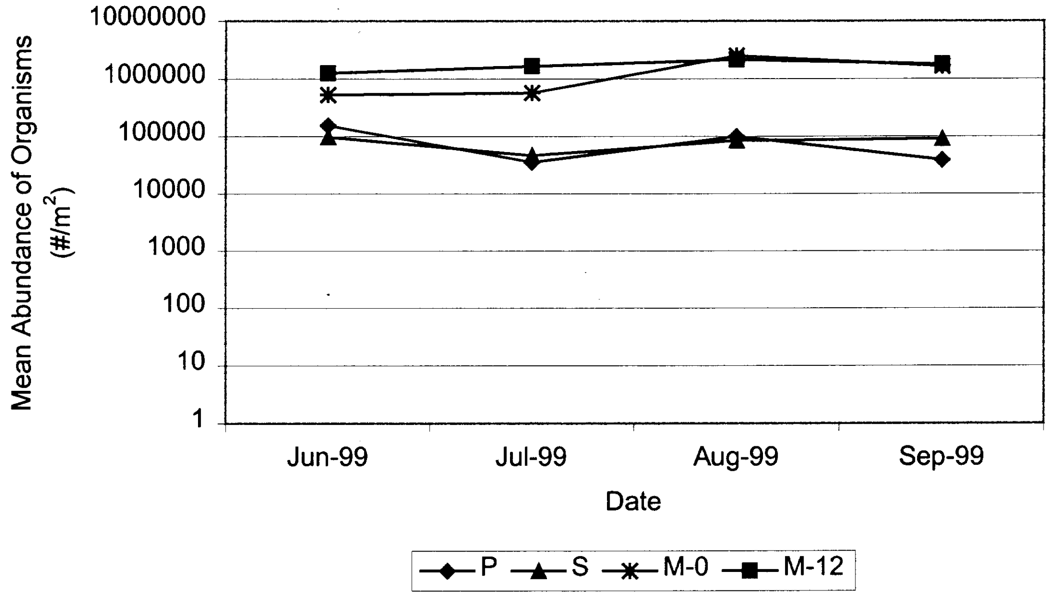

Figure 3.6 Mean abundance (#/m^2) of benthic invertebrates at the creek bank habitat Sampled at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) from June to September 1999.

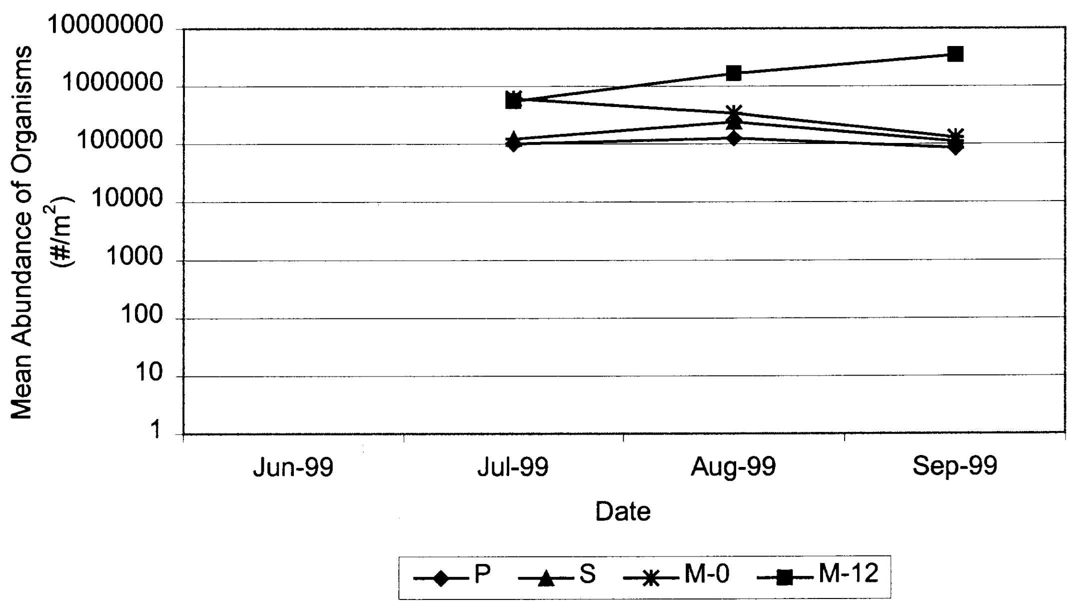

Figure 3.7 Mean abundance (#/m^2) of benthic invertebrates at the edge of the vegetation habitat. Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) from July to September 1999.

3.2.2 Taxa Richness

There was a significant difference ($P<0.05$) for the month of August 1999 on the creek bank in which M-0 was lower than the other three sites (Figure 3.8). The taxa richness of the other three months sampled did not reveal any significant difference between sites. The mean taxa richness at the edge of the vegetation did have a significant difference ($P<0.05$) for the months of July and August (Figure 3.9). During July, M-12 had more taxa present than the other sites. During the month of August, P had a significantly ($P<0.05$) higher number of taxa than S. The mean taxa richness between the two habitats at each site was significantly different at M-12 for August and September when the creek bank had more taxa present than the edge of the vegetation. There was a significant difference as well at S for the month of August when the creek bank had more taxa present than the edge of the vegetation. The taxa richness was significantly different between the habitats at some of the sites in which there were more taxa present at the creek bank than the edge of the vegetation. As for the overall taxa richness, no one site had more taxa present throughout the study period.

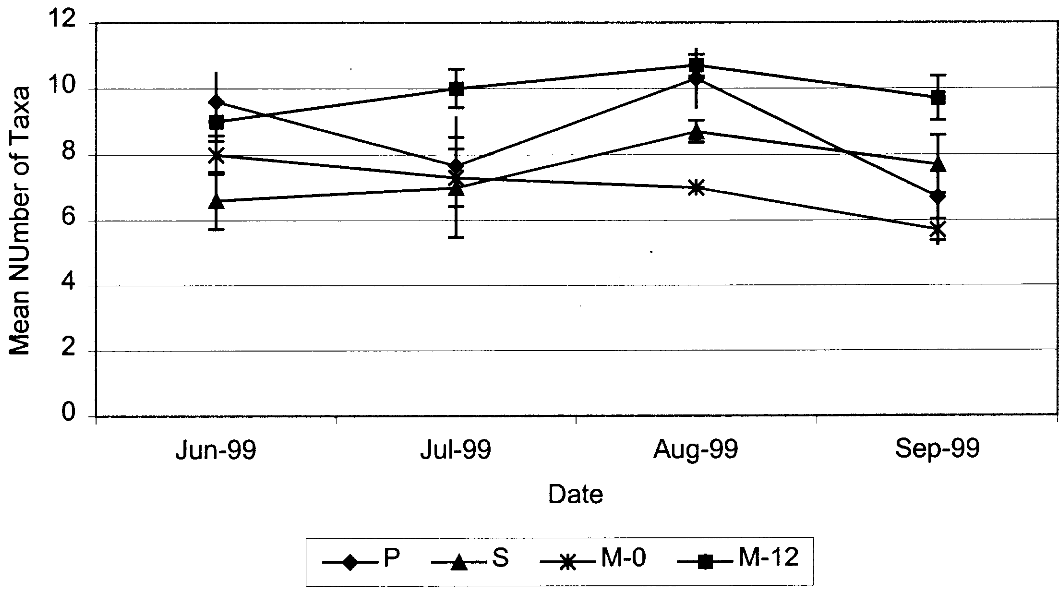

Figure 3.8 Mean taxa richness at the creek bank habitat.
Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) from June to September 1999.

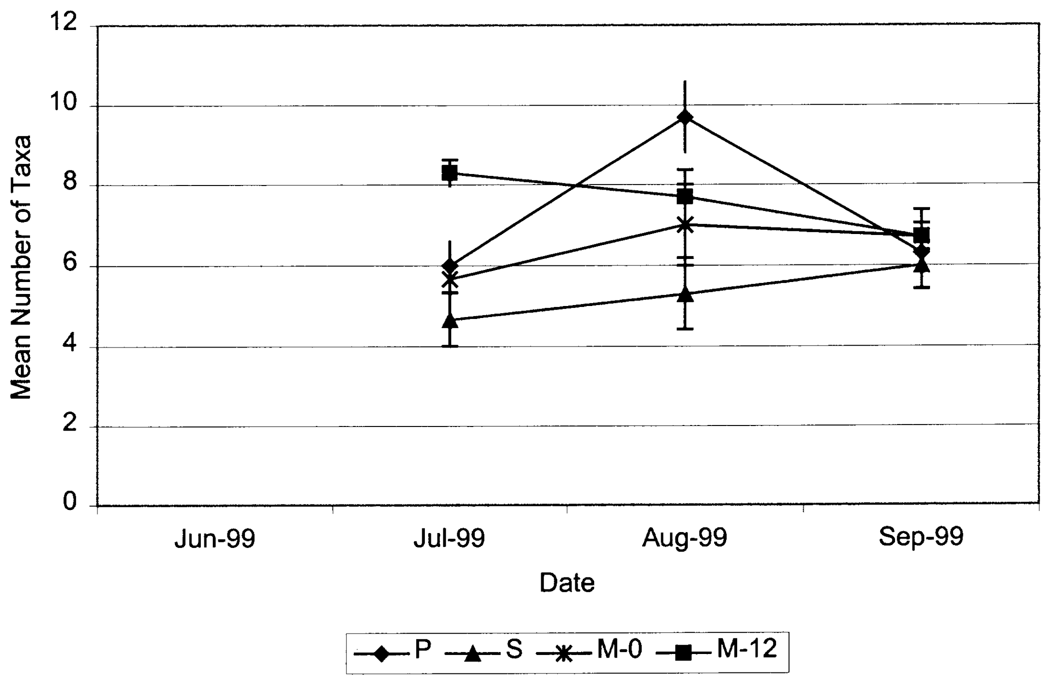

Figure 3.9 Mean taxa richness at the edge of the vegetation habitat.
Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) from July to September 1999.

3.2.3 Taxa Diversity Index

3.2.3.1 Shannon-Wiener Taxa Diversity Index P on the creek bank had the highest species diversity for June, July, August, and September 1999 (Table 3.3). For the month of June, the lowest species diversity index was found at the M-12 on the creek bank. S at the edge of the vegetation had the lowest species diversity index for July. M-0 at the creek bank had the lowest species diversity index in August. M-12 at the edge of the vegetation was the lowest in September.

There is a significant difference between the species diversity found on the creek bank between the sites. For the month of June, P was significantly more diverse than both M-12 and M-0 sites. During July and September, P and S had significantly higher diversity than the M-12 and M-0 sites. For the month of August, both SMC sites were significantly more diverse than the MC sites, but the Mill Creek sites were significantly different from each other as well, with M-12 having more diversity than M-0. At the edge of the vegetation, M-12 had significantly less diversity than P and S for the month of August. During September, M-12 was significantly less diverse than the other three sites.

3.2.3.2 Simpson's Taxa Diversity Index P on the creek bank had the highest species diversity for June, July, August, and September 1999 (Table 3.4). For the month of June, the lowest species diversity index was found at the M-12 on the creek bank. S at the edge of the vegetation had the lowest species diversity index for July. M-0 at the creek bank had the lowest species diversity index in August. M-12 at the edge of the vegetation was the lowest in September.

There was a significant difference between the species diversity found on the creek bank between the sites for all months sampled except for June. During July and September, P and S had significantly higher diversity than the M-12 and M-0 sites. For the month of August, both SMC sites were significantly more diverse than the MC sites, but the Mill Creek sites were significantly different from each other as well, with M-12 having more diversity than M-0. At the edge of the vegetation, there was no significant difference between the sites for the months of July and August. During September, M-12 was significantly less diverse than the other three sites.

Overall, the species diversity was the greatest at the SMC sites compared to the MC sites, and P was the most diverse of all of the sites at both habitats. Shannon-Wiener and Simpson's diversity indices both found the same general pattern in which P at the creek bank was the most diverse and S at the creek bank was the second most diverse site.

Table 3.3 Shannon-Wiener Taxa Diversity Index at the creek bank and edge of the vegetation habitats. Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) from June to September 1999.

Site	June	July	August	September
P-Creek Bank	**1.66**	**1.64**	**1.77**	**1.49**
P-Edge of Vegetation	-	1.06	1.37	1.33
S-Creek Bank	1.33	1.50	1.65	1.30
S-Edge of Vegetation	-	0.71	1.06	1.11
M-12-Creek Bank	1.05	0.94	0.66	0.63
M-12-Edge of Vegetation	-	1.23	0.80	0.33
M-0-Creek Bank	1.12	0.79	0.37	0.38
M-0-Edge of Vegetation	-	0.94	1.33	1.31

Bolded numbers represent the highest taxa diversity index for each month sampled.

Table 3.4 Simpson's Taxa Diversity Index at the creek bank and edge of the vegetation habitats. Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) from June to September 1999.

Site	June	July	August	September
P-Creek Bank	**0.76**	**0.75**	**0.78**	**0.72**
P-Edge of Vegetation	-	0.52	0.67	**0.67**
S-Creek Bank	0.64	**0.73**	**0.74**	0.61
S-Edge of Vegetation	-	0.37	0.62	0.62
M-12-Creek Bank	0.52	0.46	0.33	0.29
M-12-Edge of Vegetation	-	0.65	0.44	0.15
M-0-Creek Bank	0.59	0.39	0.15	0.17
M-0-Edge of Vegetation	-	0.53	**0.66**	**0.66**

Bolded numbers represent the highest taxa diversity index for each month sampled.

3.2.5 Composition

The presence (+) and absence (-) of the 25 taxa found over the four months at the sites have been compiled, and the data has been pooled together for both habitats and all sampling dates for each site (Table 3.5). There were three species of meiofauna (Nematoda, Copedoda, and Ostracoda) for each site. The sites were composed primarily of macrofauna taxa. Although the abundances of macrofauna compared to meiofauna were greater at M-12 and M-0 while the meiofauna compared to macrofauna was greater at P and S (Table 3.5). This section on composition looks specifically at the taxa that were classified as common (≥1.0% of the total abundance) and uncommon (>0.1% but <1.0%) (LaSalle and Rozas 1991).

Table 3.5 Taxa present (+) and absent (-) found at Sawmill Creek-*P. australis* (P), Sawmill Creek-*S. alterniflora* (S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) sampled from June to September 1999.

Taxa	P	S	M-12	M-0
Meiofauna				
P: Nematoda	+	+	+	+
P: Arthropoda				
C: Crustacea				
SC: Copepoda	+	+	+	+
SC: Ostracoda	+	+	+	+
Total Number of Meiofauna Taxa Present	3	3	3	3
Total Individuals of Meiofauna	995	1000	37475	19258
Macrofauna				
P: Arthropoda				
C: Crustacea				
SC: Cirripedia				
O: Amphipoda				
F: Gammaridae	+	+	-	-
O: Isopoda				
F: Anthuridae				
G: *Cyanthura polita*	+	+	-	-
F: Idoteidae				
G: *Edotea triloba*	+	-	-	-

Table 3.5 Taxa present (+) and absent (-) found at Sawmill Creek-*P. australis* (P), Sawmill Creek-*S. alterniflora* (S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) sampled from June to September 1999. Continued.

Taxa	P	S	M-12	M-0
C: Insecta				
Unknown Insecta Larvae	+	-	-	-
O: Diptera				
F: Ceratopogonidae	+	+	+	+
F: Chironomidae	+	-	+	+
F: Culicidae (Mosquito Pupae)	+	-	-	-
O: Collembola				
G: *Anurida martima*	+	-	+	-
C: Arachnida				
O: Acarina	+	+	+	+
P: Mollusca				
C: Bivalvia (Pelecypoda)				
F: Tellinidae				
G: Macoma balthica	+	-	-	-
C: Gastropoda	+	+	+	+
P: Annelida				
C: Oligochaeta	+	+	+	+
C: Polychaeta				
F: Sabellidae				
G: *Manayunkia aesturina*	+	+	+	+
F: Nereidae	+	+	+	+
F: Ampharetidae				
Unknown Ampharetidae	-	-	+	-
G: *Hobsonia florida*	+	+	+	+
F: Spionidae				
Unknown Spionidae	+	+	+	+
G: *Streblospio benedicti*	+	+	+	-
P: Cnidaria				
C: Anthozoa	+	-	-	-
P: Nemertinea	+	+	-	-
P: Platyhelminthes				
C: Turbellaria	+	+	+	+
P: Foraminifera	+	+	+	+
Total Number of MacrofaunaTaxa Present	21	14	14	11
Total Individuals of Macrofauna	1299	1846	7463	3482

Overall Total Number of Taxa Present	24	17	17	14
Overall Total Individuals	2294	2846	44938	22740

The ten common taxa (Figure 3.10) found at P were Oligochaeta (24.4%), Nematoda (22.9%), the sabellid polychaete *Manayunkia aestuarina* (16.8%), Copepoda (12.9%), the spionid polychaete *Streblospio benedicti* (6.0%), the ampharetid polychaete *Hobsonia florida* (5.3%), other unknown spionidae polychaetes (3.3%), nereidae polychaetes (2.5%), the dipteran larvae Ceratopogonidae (1.1%), and the anthurid isopod *Cyanthura polita* (1.0%). The uncommon taxa found at this site were unidentified insect larvae (0.96%), Turbellaria (0.83%), Foraminifera (0.61%), the dipteran larvae Chironomidae (0.26%), the anthurid isopod *Edotea triloba* (0.22%), Acarina (0.22%), the collembola *Anurida martima* (0.17%), and the marine bivalve *Macoma balthica* (0.13%).

Eight common taxa were found at S (Figure 3.10), which are the sabellid polychaete *Manayunkia aestuarina* (33.0%), Nematoda (30.7%), Oligochaeta (19.0%), the spionid polychaete *Streblospio benedicti* (6.5%), Copepoda (4.4%), nereidae polychates (2.2%), the ampharetid polychaete *Hobsonia florida* (1.4%), and other unknown spionidae polychaetes (1.1%). The uncommon taxa found at this site were Turbellaria (0.39%), the anthurid isopod *Cyanthura polita* (0.35%), the dipteran larvae Ceratopogonidae (0.32%), Foraminifera (0.25%), and gammaridae amphipods (0.11%).

Four common taxa were found at M-12 (Figure 3.11): Nematoda (79.8%), Oligochaeta (13.7%), Copepoda (2.8%), and the sabellid polychaete *Manayunkia aestuarina* (1.8%). The uncommon taxa at this site were Ostracoda (0.81%) and Gastropoda (0.61%).

Four common taxa are found at M-0 (Figure 3.11): Nematoda (77.3%), Oligochaeta (12.6%), Ostracoda (6.8%), and Gastropoda (2.0%). The uncommon taxa at this site were Copepoda (0.65%), the dipteran larvae Chironomidae (0.46%), and the ampharetid polychaete *Hobsonia florida* (0.14%).

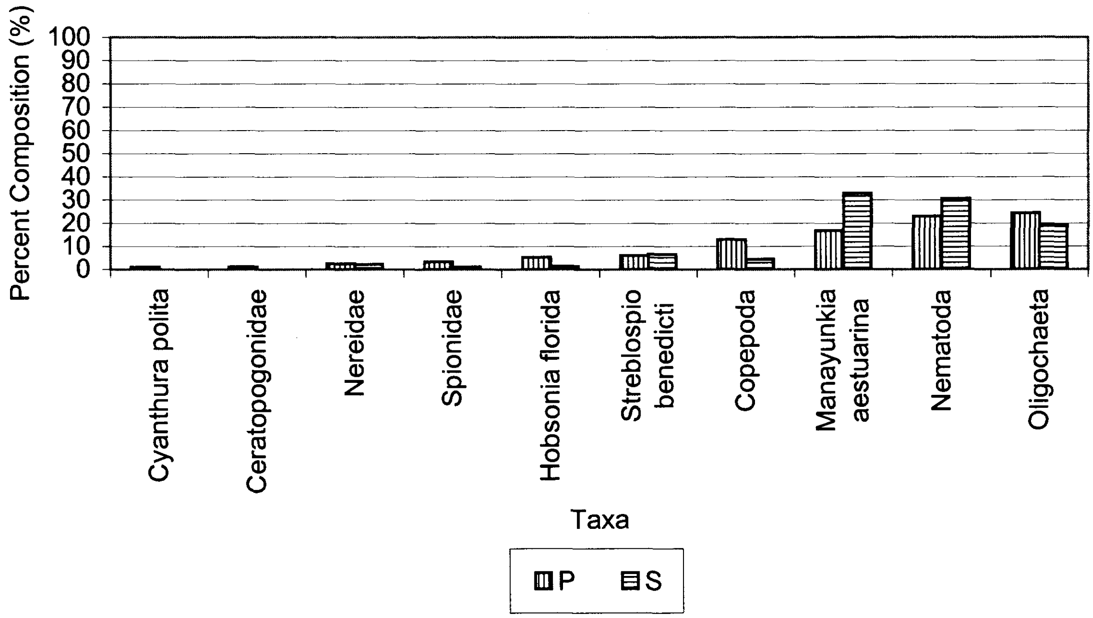

Figure 3.10 Percent composition of the common taxa.
Samples were taken at Sawmill Creek-S. alterniflora (S) and Sawmill Creek-P. australis (P) from June to September 1999.

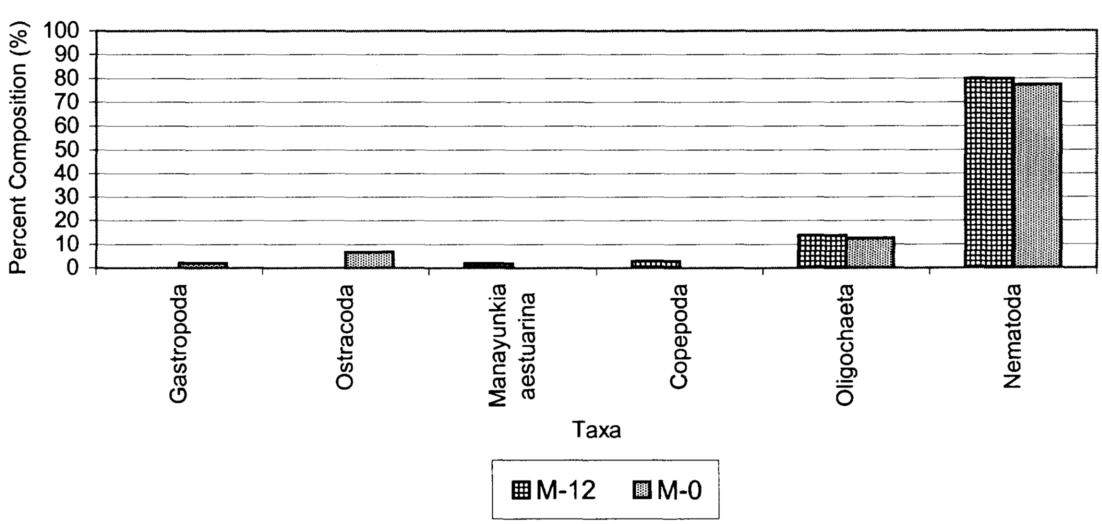

Figure 3.11 Percent composition of the common taxa.
Samples were taken at 12-year Mitigation Site-Mill Creek (M-12) and New Mitigation Site Mill Creek (M-0) from June to September 1999.

PCA was conducted on the 10 most abundant taxa, which includes 6 common (≥1.0% of the total abundance) and 4 uncommon (>0.1% but <1.0% of the total abundance) taxa from the total abundance (LaSalle and Rozas 1991). The total abundance was analyzed by taking the mean abundance for each month sampled for each sampling site. The first principal component (PC1) consisted of 98.1% of the total variation, and the second principal component (PC2) had only 1.4% of the total variation (Table 3.6). The results of the PCA revealed that there was no separation between P and S, and there was also no separation between the two habitats for these two sites therefore there was no variability (Figure 3.12). However, there was more variability for the results of the mitigation sites. There was some separation between the creek bank and the edge of the vegetation habitats for these two sites, but there was still no clear separation (Figure 3.12).

Table 3.6 Principal component analysis results of ten most abundant taxa of benthic sampling. Principal component loadings, eigenvalues, and percentage of variation of each principal component, and the cumulative percentage of variation found at Sawmill Creek-*P. australis*(P), Sawmill Creek-*S. alterniflora*(S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) sampled from June to September 1999.

Site ID	Site	PC1	PC2
	June		
1	P-Creek Bank	-608.42	-56.386
2	S-Creek Bank	-591.88	-63.959
3	M-0-Creek Bank	-314.98	68.908
4	M-12-Creek Bank	401.98	88.94
	July		
5	P-Edge of Vegetation	-646.32	2.97
6	P-Creek Bank	-657.23	-62.266
7	S-Edge of Vegetation	-610.97	-68.877
8	S-Creek Bank	-649.3	-58.933
9	M-0-Edge of Vegetation	-399.38	348.721
10	M-0-Creek Bank	-137.36	-88.835
11	M-12-Edge of Vegetation	-354.36	123.887
12	M-12-Creek Bank	820.75	150.4

Table 3.6 Principal component analysis results of ten most abundant taxa of benthic sampling. Principal component loadings, eigenvalues, and percentage of variation of each principal component, and the cumulative percentage of variation found at Sawmill Creek-*P. australis*(P), Sawmill Creek-*S. alterniflora*(S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) sampled from June to September 1999. Continued.

Site ID	Site	PC1	PC2
	August		
13	P-Edge of Vegetation	-625.49	-31.091
14	P-Creek Bank	-623.63	-64.353
15	S-Edge of Vegetation	-604.42	12.393
16	S-Creek Bank	-650.17	-53.52
17	M-0-Edge of Vegetation	-442.35	-21.329
18	M-0-Creek Bank	2094.71	-197.807
19	M-12-Edge of Vegetation	780.24	309.035
20	M-12-Creek Bank	1456.92	133.718
	September		
21	P-Edge of Vegetation	-623.16	-58.823
22	P-Creek Bank	-647.36	-65.901
23	S-Edge of Vegetation	-614.89	-54.814
24	S-Creek Bank	-594.99	-62.465
25	M-12-Edge of Vegetation	-638.24	-5.934
26	M-12-Creek Bank	1135.42	-128.541
27	M-0-Edge of Vegetation	3173.05	-86.189
28	M-0-Creek Bank	1171.84	-8.948
Eigenvalue		1006072.66	14791.31
Percentage of Variation		98.14	1.44
Cumulative Percentage		98.14	99.58

Pearson Correlation Coefficients between taxa and the principal components.

Taxa	PC1	PC2
Chironomid	0.21445	-0.09864
Copepod	0.04264	0.42633*
Gastroopod	0.25942	0.10875
Hobsonia florida	-0.16867	-0.08709
Manayunkia aestuarina	0.02609	0.12923
Nematodes	0.99996*	-0.00868
Nereidae	-0.0675	-0.13647
Oligochaetes	0.51975*	0.85330*
Ostracods	0.37763*	-0.1868
Streblospio benedicti	-0.24381	-0.17788

*Indicates Significant Correlation Coefficient (P<0.05)

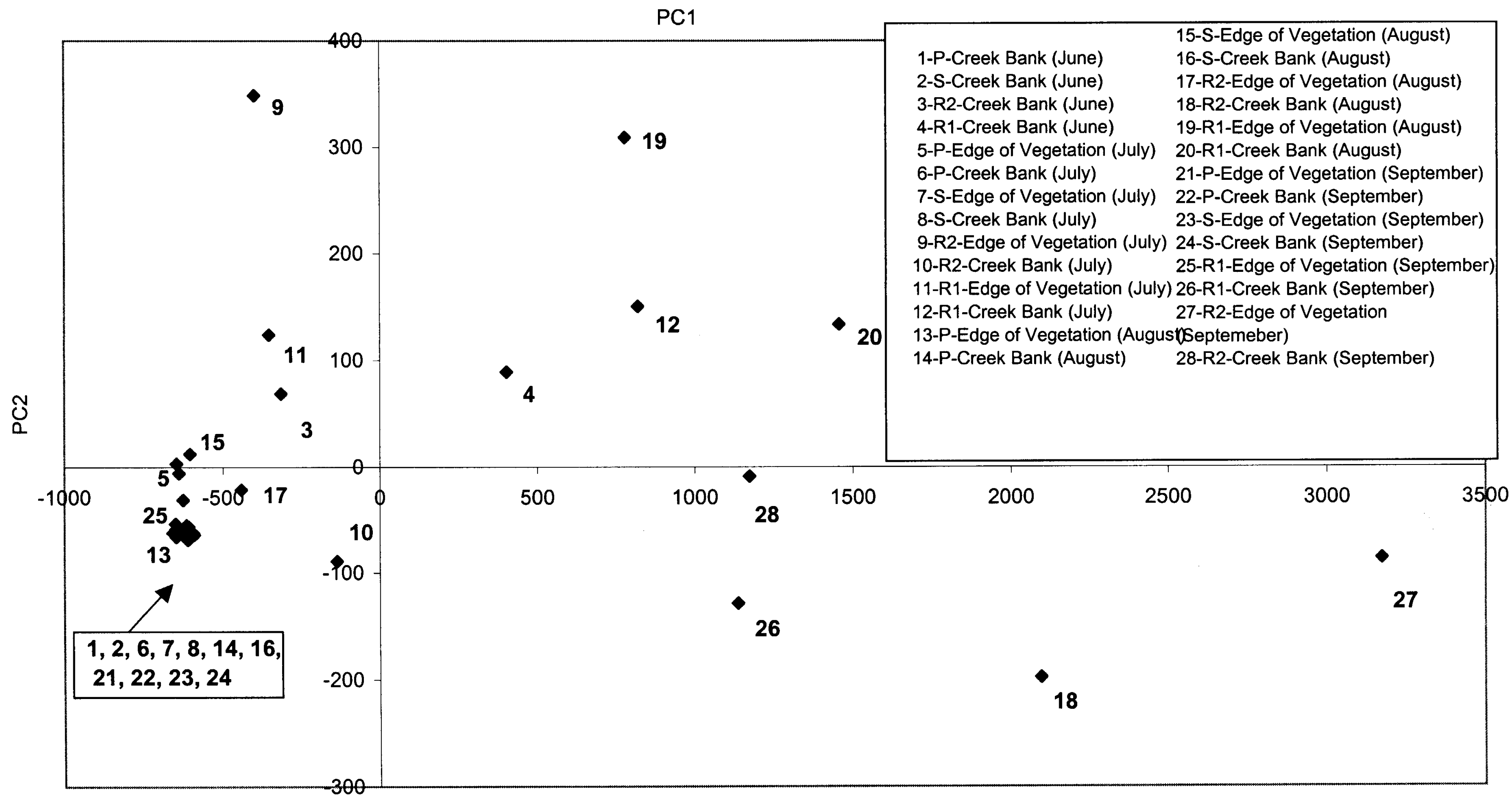

Figure 3.12 Principal Component Analysis results of the ten most abundant taxa.

Samples were taken at Sawmill Creek-*P. australis* (P), Sawmill Creek-*S. alterniflora* (S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) from June to September 1999.

P had more common taxa present (10) throughout the study period compared to the other three sites, S (8), M-12 (4), and M-0 (4). The common taxa that were present in all four sites are Oligochaeta and Nematoda. Oligochaeta had the highest percent composition at P, while *Manayunkia aestuarina* had the highest percent composition at S (Figure 3.10). Nematoda had the highest percent composition at M-12 and M-0 (Figure 3.11). The MC sites were heavily dominated by nematodes at approximately 77-80% of all taxa. The PCA revealed that the SMC sites had no separation between them, but the MC sites were slightly separated (Figure 3.12). The results of the mitigation sites may be due to the high abundances of organisms found there.

3.3 Recolonization Experiment

The containers that contained the Tuckerton sediment, which were inserted in the sediment in the Hackensack Meadowlands salt marshes were affected by the tidal conditions. During the July sampling event, there was approximately a ½ to 1-cm of sediment accumulation on the containers at both MC sites, but at the SMC sites the containers appeared to have lost some sediment, possibly due to the incoming tide that was covering the sampling station. During the August sampling event, the containers at the MC sites had additional accumulation of sediment on them.

3.3.1 Abundance

The recolonization samples were analyzed by comparing the ambient and the recolonized samples at each site for July and August (Figure 3.13). There was a significant difference between the ambient and the recolonized samples for the months of July and August at both SMC sites, in which the recolonized samples had higher abundance than the ambient samples. The recolonized samples and the ambient samples from the Sawmill Creek and Mill Creek Sites followed the same trends from July to August. Both the recolonized and the ambient samples from the Mill Creek Sites were an order of magnitude higher in mean abundance than both Sawmill Creek Sites

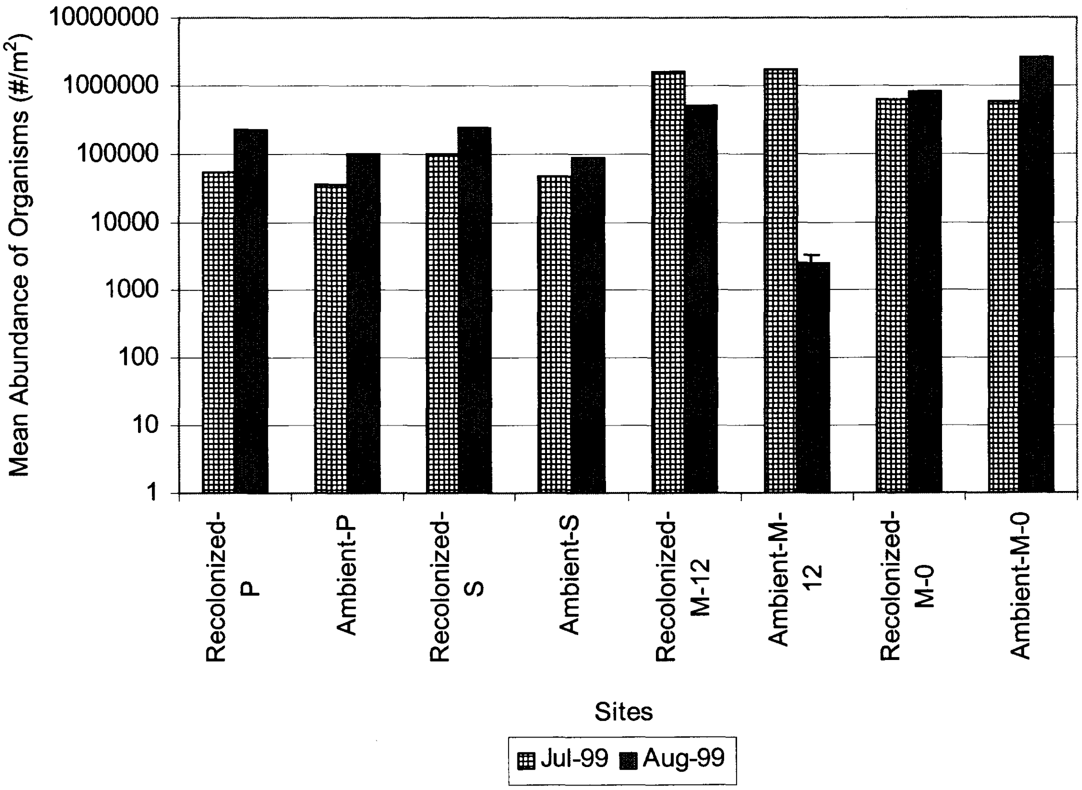

Figure 3.13 Mean abundance (#/m^2) of benthic invertebrates found in the recolonized and ambient samples. Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek, (M-12) and New Mitigation Site-Mill Creek (M-0) in July and August 1999.

3.3.2 Taxa Richness

The mean taxa richness was analyzed in the same manner as the mean abundance samples (Figure 3.14). For the mean taxa richness, there was a significant difference between the ambient and the recolonized samples at S and M-12 for August. The recolonized samples had significantly more taxa than the ambient samples at S, while at M-12; the ambient sample had more taxa present than the recolonized sample.

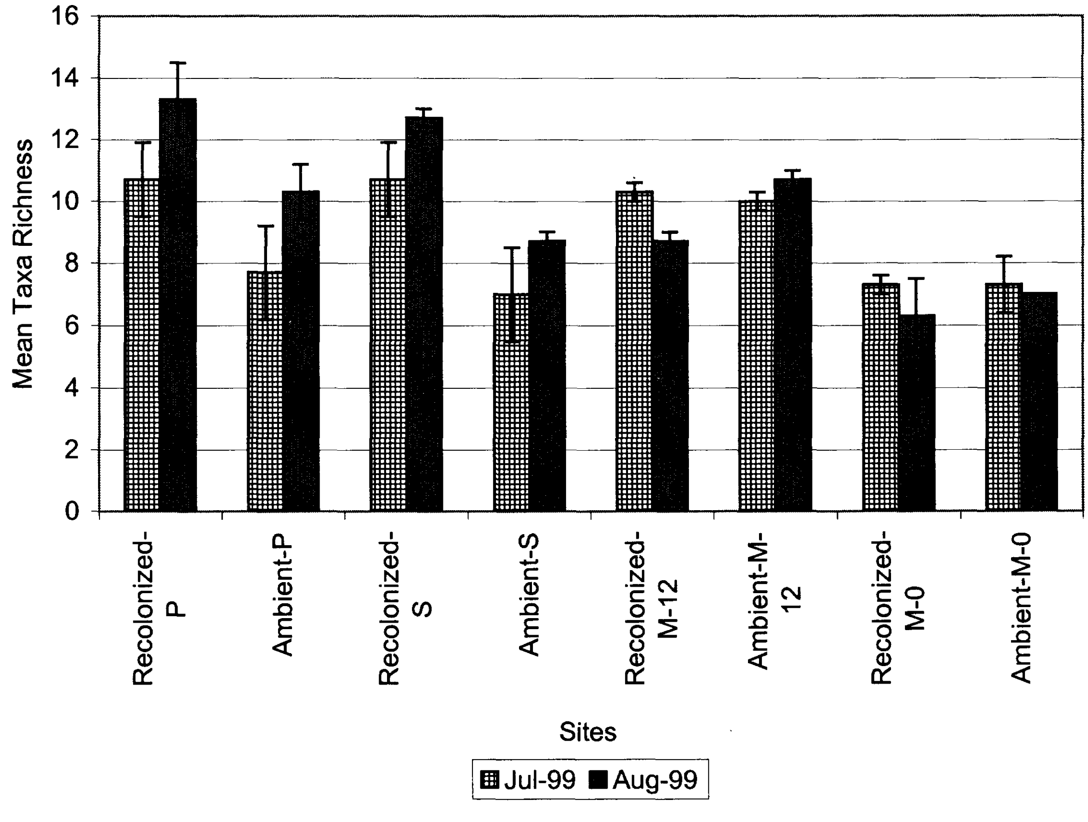

Figure 3.14 Taxa richness of benthic invertebrates found in the recolonized and ambient samples. Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) in July and August 1999.

3.3.3 Taxa Diversity Index

3.3.3.1 Shannon-Wiener Taxa Diversity Index The SMC sites had a higher diversity for July and August in recolonized samples compared to the MC Sites (Figure 3.15). The diversity index for P was 1.71 for both July and August. At S, the diversity index for July and August was 1.76 and 1.64, respectively. At M-12, the diversity index for July and August was 0.96 and 0.85, respectively and at M-0, it was 0.89 and 0.84 for July and August, respectively. There was no significant difference during the month of July between the sites, but there was a significant difference during the month of August between the SMC sites and the MC sites, in that the SMC sites had higher diversity index than the MC sites (Figure 3.15).

3.3.3.2 Simpson's Taxa Diversity Index The SMC sites had a higher diversity for July and August in recolonized samples compared to the MC Sites (Figure 3.16). S was the most diverse (0.77) while M-0 was the least diverse (0.41) for the month of July. P was the most diverse (0.75) while M-0 was the least diverse (0.43) for the month of August. However, there was no significant difference between the sites for both month sampled.

The Shannon-Wiener and Simpson's diversity indices both had the same general pattern. P and S were both more diverse than M-12 and M-0. The only difference was that there was no significant difference between the sites with regards to the Simpson's diversity index, but there was a significant difference with the Shannon-Wiener diversity index.

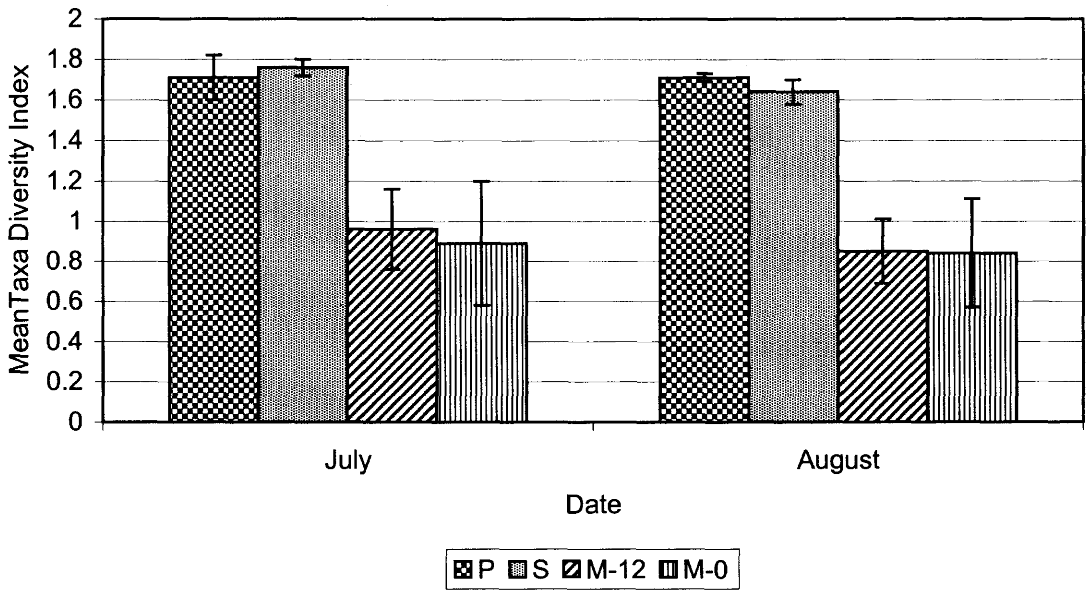

Figure 3.15 Mean Shannon-Wiener Taxa Diversity Index for Recolonization Experiment.

Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) in July and August 1999.

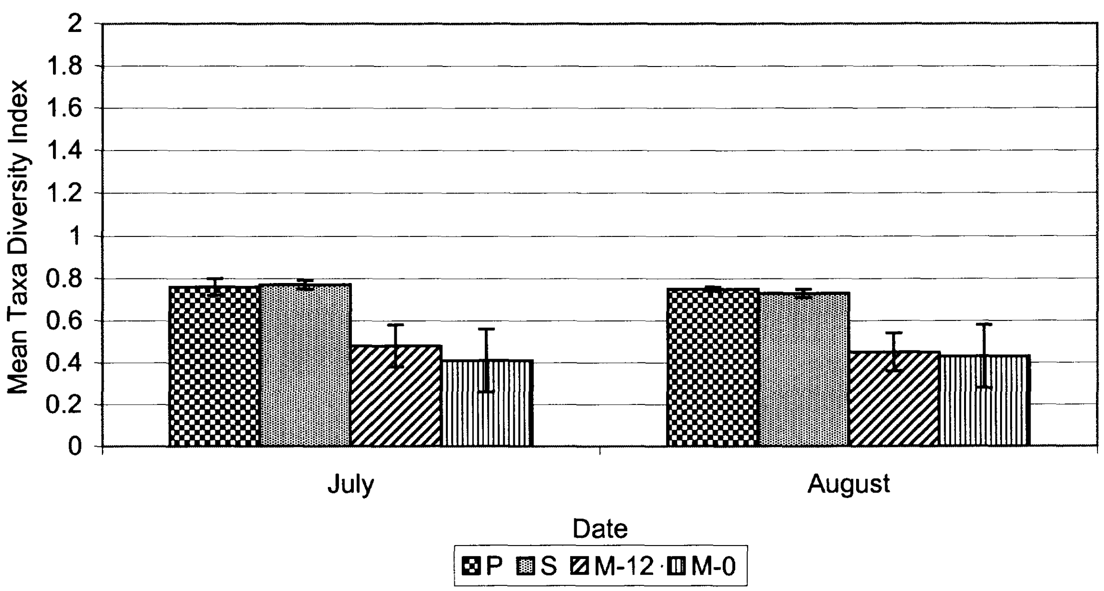

Figure 3.16 Mean Simpson's Taxa Diversity Index for Recolonization Experiment.

Samples were taken at Sawmill Creek-*S. alterniflora* (S), Sawmill Creek-*P. australis* (P), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site Mill Creek (M-0) in July and August 1999.

3.3.4 Composition

The taxa found within the recolonization samples were also classified as common and uncommon. There were eight common taxa found at P (Figure 3.17): unknown Spionidae (34.6%), Copepoda (21.6%), Nematoda (17.1%), Nereidae (7.4%), Oligochaeta (7.0%), the ampharetid polychaete *Hobsonia florida* (5.8%), the sabellid polychaete *Manayunkia aestuarina* (2.9%), and Turbellaria (1.6%). The uncommon taxa found at this site were the anthurid isopod *Cyanthura polita* (0.76%), the spionid polychaete *Streblospio benedicti* (0.19%), Nemertinea (0.19%), phyllodocidae polychaete *Eteone sp.* (0.19%), and Ostracoda (0.14%).

Eight common taxa were found at S (Figure 3.17): Spionidae (30.1%), Nematoda (27.1%), Copepoda (15.5%), Oligochaeta (9.0%), Nereidae (7.3%), the sabellid polychaete *Manayunkia aestuarina* (5.7%), the spionid polychaete *Hobsonia florida* (2.1%), and Turbellaria (2.0%). The uncommon taxa at this site were the anthurid isopod *Cyanthura polita* (0.39%), Ostracoda (0.19%), Gammaridae (0.19%), Nemertinea (0.12%), and other unknown Ampharetidae (0.12%).

Six common taxa were found at M-12 (Figure 3.18): Nematoda (74.3%), Oligochaeta (19.1%), the sabellid polychaete *Manayunkia aestuarina* (2.3%), Copepoda (1.4%), Ostracoda (1.2%), and Gastropoda (1.1%). The uncommon taxa at this site were the dipteran larvae Chironomidae (0.25%).

Five taxa were found at M-0 (Figure 3.18), which were Nematoda (79.5%), Ostracoda (10.4%), Oligochaeta (5.4%), Gastropoda (2.3%), and the dipteran larvae Chironomidae (1.3%). The uncommon taxa at this site are Turbellaria (0.79%) and Copepoda (0.26%).

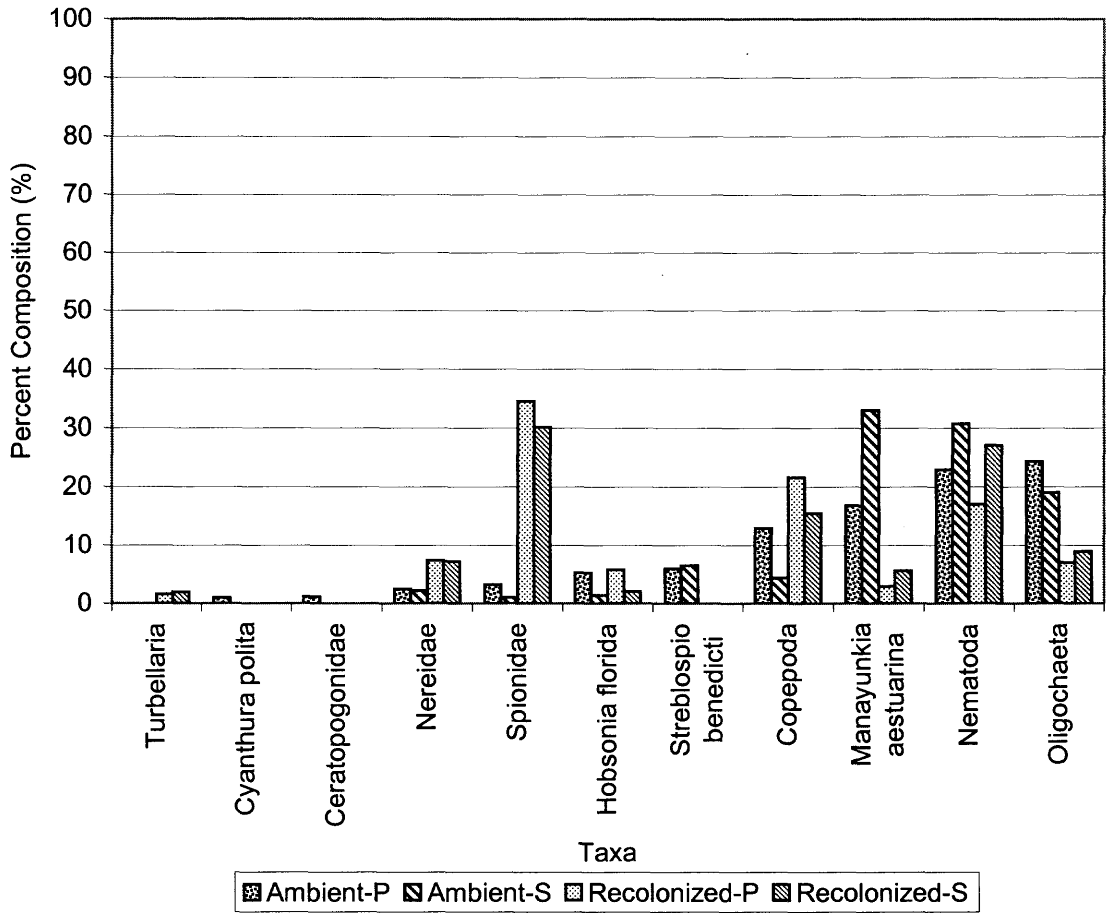

Figure 3.17 Percent composition of the common taxa.

Samples were taken in the recolonized and ambient samples at Sawmill Creek-*S. alterniflora* (S) and Sawmill Creek-*P. australis* (P) from July to August 1999.

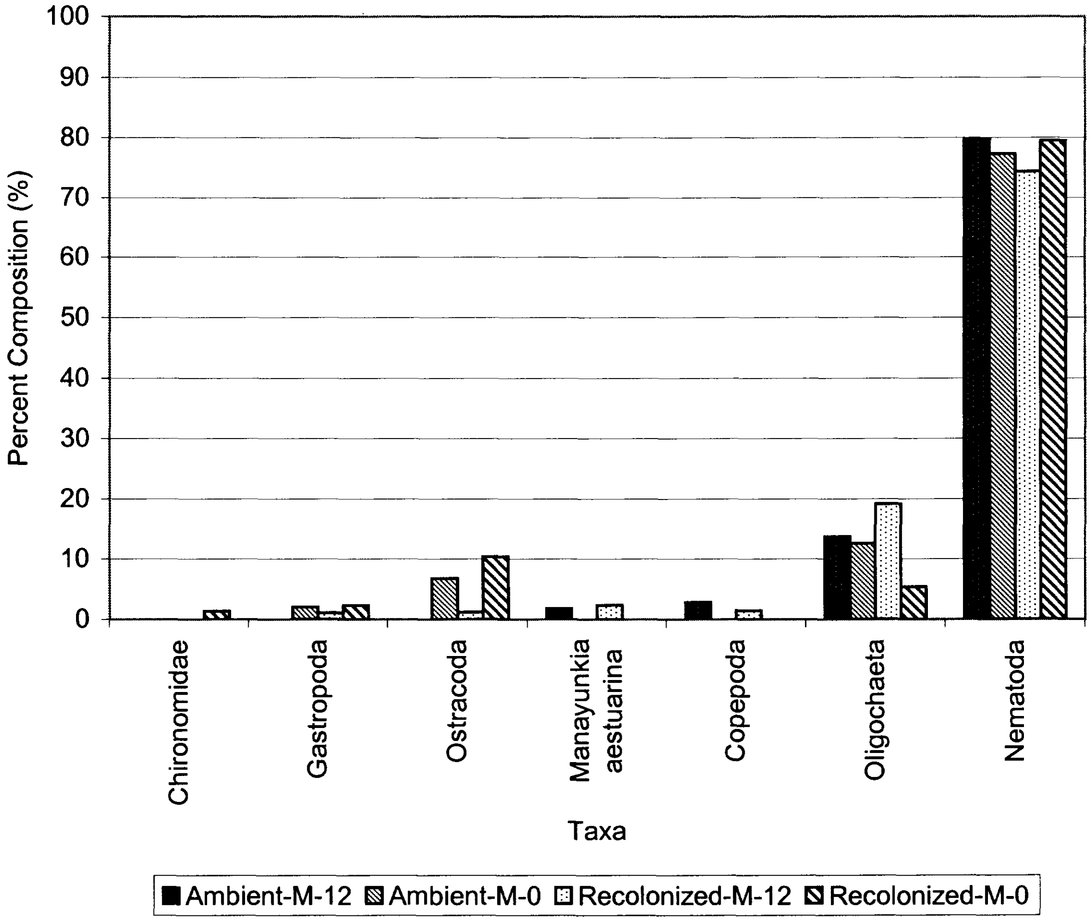

Figure 3.18 Percent composition of the common taxa found in the recolonized and ambient samples. Samples were taken at 12-year Mitigation Site-Mill Creek (M-12) and New Mitigation Site Mill Creek (M-0) from July to August 1999.

PCA was conducted on the 11 most abundant taxa, which are from the recolonized and ambient samples. The total abundance was analyzed by taking the mean abundance for each month sampled for each sampling site. The first principal component (PC1) consisted of 97.3% of the total variation, and the second principal component (PC2) had only 1.8% of the total variation (Table 3.7).

M-12 and M-0 were separated by PC1, and S and P were also separated by PC1 (Figure 3.19). The SMC and the MC sites were separated by PC2. The results revealed that oligochaetes, nematodes, *Manayunkia aestuarina*, and ostracods were significant variables (Table 3.7). The ambient M-0 site increased in abundance of nematodes from July to August (Figure 3.19). The recolonized M-0 site increased in abundance of ostracods from July to August. The recolonized M-12 site decreased in abundance of nematodes while increasing in oligochaetes and *M. aestuarina* from July to August. The ambient M-12 increased in abundance of nematodes from July to August (Figure 3.19). The results for M-12 and M-0 (the mitigation sites) may be due to the dynamics of it being such a young system.

Table 3.7 Principal component analysis results of the 11 most abundant taxa in recolonized and ambient samples. Principal component loadings, eigenvalues, and percentage of variation of each principal component, and the cumulative percentage of variation for the recolonized and ambient samples at Sawmill Creek-*P. australis* (P), Sawmill Creek-*S. alterniflora* (S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) sampled during July and August 1999.

Site ID	Site	PC1	PC2
	Recolonized Samples		
1	P-July	-874.83	8.084
2	S-July	-839.37	36.357
3	M-0-July	453.94	-76.463
4	M-12-July	2223.59	303.524
5	P-August	-793.07	32.976
6	S-August	-712.63	18.721
7	M-0-August	647.96	-255.734
8	M-12-August	95.35	122.525
	Ambient Samples		
9	P-July	-889.54	2.533
10	S-July	-881.41	5.236
11	M-0-July	-370.73	-86.838
12	M-12-July	600.77	128.127
13	P-August	-856.3	-0.224
14	S-August	-881.69	11.926
15	M-0-August	1845.46	-313.273
16	M-12-August	1232.51	62.524
Eigenvalue		1112532.40	20492.89
Percentage of Variation		97.27	1.79
Cumulative Percentage		97.27	99.07

Pearson Correlation Coefficients between sediment charactersitics and the principal components.

Taxa	PC1	PC2
Chironomid	0.41792	-0.23395
Copepod	-0.22928	0.41992
Gastroopod	0.79838*	-0.15993
Hobsonia florida	-0.43569	0.17418
Manayunkia aestuarina	0.49688	0.65647*
Nematodes	0.99985*	-0.01658
Nereidae	-0.42800	0.11955
Oligochaetes	0.76093*	0.63910*
Ostracods	0.40623	-0.65187*
Spionidae	-0.30865	0.07905
Streblospio benedicti	-0.32439	0.04533

*Indicates Significant Correlation Coefficient (P<0.05)

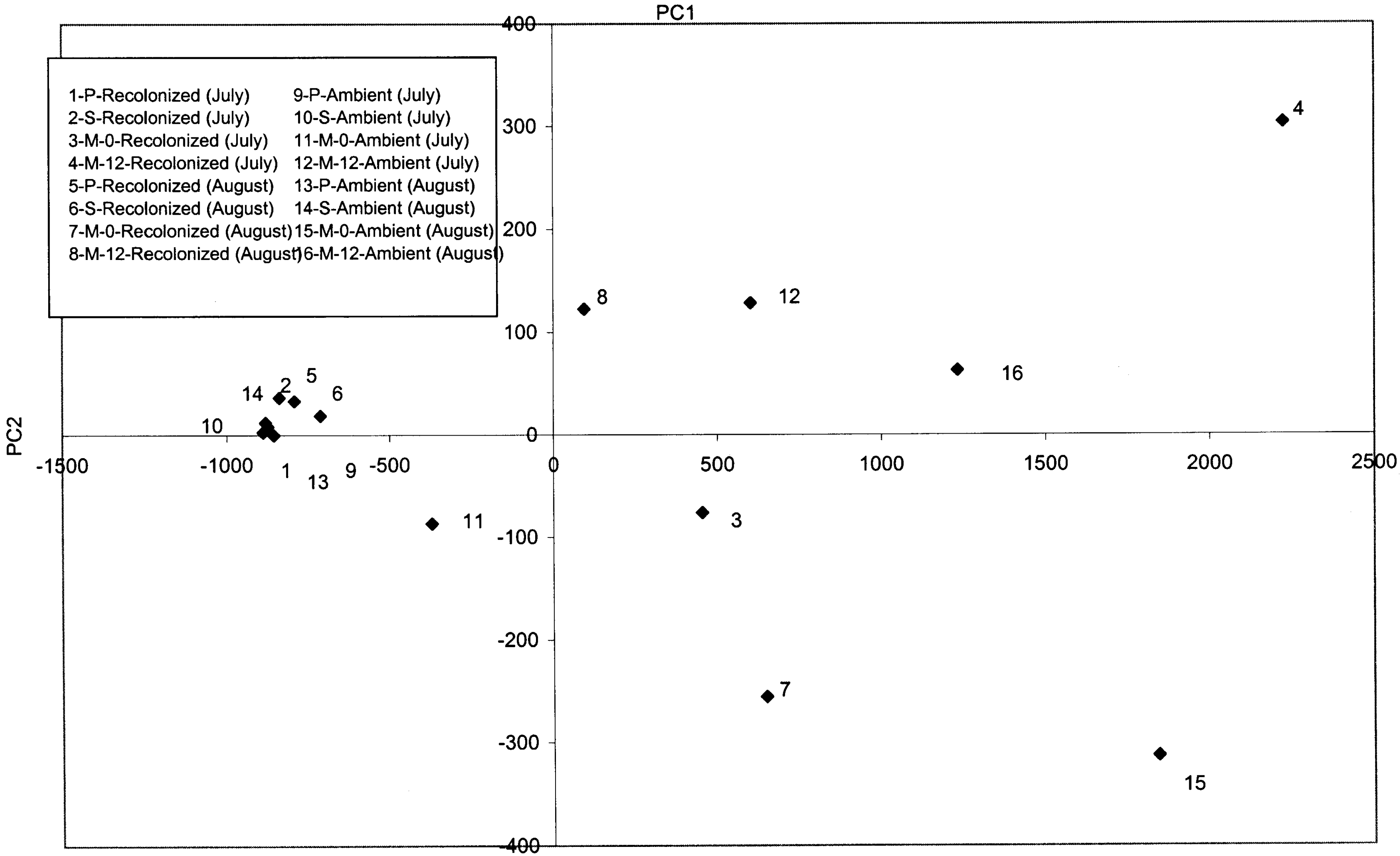

Figure 3.19 Principal Component Analysis of the 11 most abundant taxa in recolonized and ambient samples.

Samples were taken at Sawmill Creek-*P. australis* (P), Sawmill Creek-*S. alterniflora* (S), 12-year Mitigation Site-Mill Creek (M-12), and New Mitigation Site-Mill Creek (M-0) from July to August 1999.

Overall, the ambient and recolonized samples from the SMC sites and the MC sites followed the same trends for abundance and taxa richness (Figure 3.13 and 3.14). Regarding the abundance of benthic invertebrates, the ambient and the recolonized MC sites were an order of magnitude higher than the ambient and recolonized SMC sites (Figure 3.13). The recolonized samples had significantly greater abundance than the ambient samples at SMC. There was no significant difference in regarding the abundance of benthic invertebrates between the recolonized and the ambient samples at MC. There was no significant difference in regards to the abundance of benthic invertebrates between the recolonized and the ambient samples at MC. The taxa richness results revealed that there was a difference between the ambient samples and the recolonized samples at S and M-12 for August (Figure 3.14).

The Shannon-Wiener taxa diversity index found significant differences between the sites while Simpson's taxa diversity index did not. P and S had eight common taxa present, while M-12 had six and M-0 had five. The common taxa present at all four sites were Nematoda and Oligochaeta. Spionidae had the highest percent composition (30-34%) at the SMC sites (Figure 3.17), and Nematoda had the highest percent composition (74-79%) at the MC sites (Figure 3.18). The recolonization experiment results reveal that the recolonized samples follow the same trends as the its respective ambient samples.

CHAPTER 4

DISCUSSION

4.1 Benthic Invertebrates

The natural *S. alterniflora* and *P. australis* marshes as well as the mitigated *S. alterniflora* and bare marshes support a benthic community in the low marsh in the intertidal zone. However, different levels of abundance, taxa richness, diversity, and composition were found. This study suggests that differences in the benthic communities were attributed to the different types of vegetation and salinity levels, and whether or not the marsh was a mitigated or natural system. This study also suggests that substrate was not a factor. Contamination levels may have been a factor according to heavy metals data collected from other sources (Appendix A). Since this study did not investigate contaminate levels, the affect of contamination and pollution cannot be speculated.

This study reveals that a mitigated marsh after 12-years in the low marsh zone does not have a diverse benthic community as a natural marsh. The mitigated marshes at MC, specifically M-12, do not resemble the natural SMC sites, including the *P. australis* marsh, which is the type of marsh that managers are removing. These data suggest that significant progress in the benthic community has not occurred at the 12-year Mitigation Site at the low marsh since the TAMS (1990) and Kraus and Kraus (1988) studies. TAMS (1990) evaluated the benthic community at the Western Brackish Marsh (M-12), and found a benthic community with a low diversity that contained pollution tolerant species. Kraus and Kraus (1988) compared the benthic communities at SMC and MC in 1986 when the Hartz Mountain Mitigation at Mill Creek (M-12) was underway, and found that SMC had a more diverse benthic community than MC.

It has been over ten years since the TAMS (1990) study was conducted at M-12 in the open water channels, and this study shows that a low diversity still persists. TAMS (1990) found the benthic community at M-12 to consist of pollution tolerant organisms, such as oligochaetes and hydrobiidae gastropods that comprised over 80% percent of the population. In 1999, oligochaetes and gastropods consisted of 14.3% of the composition, but oligochaetes had the second highest percent composition at M-12. This study also found that M-12 was dominated by few taxa with large abundances.

The results of this study were also similar to Kraus and Kraus' (1988) study conducted at MC and SMC. Sawmill Creek and the mitigated (*S. alterniflora*) and non-mitigated (*P. australis*) sections of Mill Creek were sampled (Kraus and Kraus 1988). However, the sections (*S. alterniflora* versus *P. australis* sections) sampled at Sawmill Creek were not specified, and there was also no specification about where the Mill Creek control sites were located or which type of vegetation was present there. Kraus and Kraus (1988) found that SMC had a greater abundance of benthic invertebrates, higher taxa richness, and higher diversity than the MC sites, which were dominated by gastropods and nematodes. The current study found that SMC had more common (≥1.0% of the total abundance) taxa and higher taxa diversity than MC, and the MC sites were dominated by nematodes. Polychaetes were found only at Sawmill Creek and not Mill Creek (Kraus and Kraus 1988) while this study found polychaetes at the SMC and MC, although the numbers were low at MC.

This study suggests that *P. australis* supports a healthy benthic community. Angradi et al. (2001) compared the benthic communities in *P. australis* and *Spartina* marshes. The *Spartina* marsh had a greater abundance than the *P. australis* marsh

(Angradi et al. 2001) while this study found that the restored marshes (*S. alterniflora* and the bare vegetation) had greater abundance than the natural marshes (*S. alterniflora* and *P. australis*). Angradi et al. (2001) found more taxa present at the *Spartina* marsh while this study found more common taxa ($\geq$1.0% of the total abundance) present at the *P. australis* marsh. The dominant taxa were oligochaetes, nematodes, and *Manayunkia aestuarina* in Angradi et al.'s (2001) study, which is similar to this study at both the *P. australis* and *S. alterniflora* marshes. Fell et al. (1998) also looked at benthic communities in non-*P. australis* and *P. australis* marshes, and found them to be equivalent.

The benthic communities at SMC and MC could be the result of the different salinity levels at these two creeks. Salinity is known to affect the composition and diversity of benthic communities (Levin and Talley 2000). The surface water salinity level was consistently higher at SMC (mesohaline to polyhaline) than MC (oligohaline to mesohaline) from June to August. Both MC and SMC had salinity levels that were classified as oligohaline in September, and this may be due to the rainfall associated with Hurricane Floyd. The MC sites had a low diversity along with low salinity levels compared to the SMC sites, and other studies have shown that low salinity has been found to decrease abundance, taxa richness, and diversity of benthic communities (Levin and Talley 2000; Boesch 1972; West and Ambrose 1992). Insects and oligochaetes usually dominate benthic communities in a low salinity system, while a high salinity system is known to favor polychaetes (Levin and Talley 2000), and this can be seen at MC and SMC in which SMC had considerably more polychaetes than MC, but both sites had an abundance of oligochaetes. Although, oligochaetes were one of the few common taxa found at Mill Creek while at Sawmill Creek oligochaetes were one of the many common taxa.

Structural and textural sediment characteristics did not seem to have an affect on the benthic communities of this study. There was only a significant difference between the sites at the creek bank habitat, but not at the edge of the vegetation. Therefore, any differences found between the sites in abundance, taxa richness, taxa diversity, and composition cannot be attributed to the structural and textural sediment characteristics at the edge of the vegetation. However, according to Levin and Talley (2000) soil organic matter and soil grain size do have an effect on benthic communities found in salt marshes.

Studies have shown that contaminants and pollutants affect benthic communities (Gray et al. 1990; Pocklington and Wells 1992; Gaston and Young 1992; Whaley et al. 1989; Maltby 1999). This study cannot speculate about the affects of contamination on benthic communities in the Hackensack Meadowlands, but there is heavy metals data from this area that can be referred to (Appendix). Kraus and Kraus (1988) reported that the water quality was better at SMC than MC, but the Center for Information Management, Integration and Connectivity (CIMIC) through the Meadowlands Environmental Research Institute (MERI) that conducts water quality testing in the Hackensack Meadowlands District determined that water quality was better at MC than SMC during the dates of this study (CIMIC 1999). However, water quality on a single date is far less important than long-term sediment concentration, which would impact benthos. Greater contamination persists in MC sediments (Appendix).

Another source of contamination that may be affecting the benthic community at MC is the sewage treatment plant along Mill Creek. According to Levin and Talley (2000), a marsh that was exposed to sewage had more of an abundance of the oligochaete *Monopylephorus rubroniveus* and the amphipod *Talorchestia longicornis*. Mill Creek is exposed to a sewage treatment plant, which is near the newly mitigated section, and the MC sites had an abundance of oligochaetes. Mill Creek's exposure to the sewage treatment plant could be a factor in the types of organisms being found there, but this should be further researched.

This study suggests that significant progress has not occurred at M-12 at the low marsh since the commencement of the mitigation in 1988. Significant progress may not have occurred due to the area being a stressed polluted environment where low salinity persists. Changing the vegetation alone from *P. australis* to *S. alterniflora* may not be able to alter these environmental conditions (Packard and Stiverson 1976; Allen et al. 1994; Sacco et al. 1994) at Mill Creek that affect the benthic community. This study also suggests that the *P. australis* marsh does support a healthy benthic community that was comparable to the *S. alterniflora* marsh.

4.2 Recolonization Experiment

Opportunism plays a major role in recolonizing sediments. The basis of opportunism is that ambient species will settle in disturbed sediments to increase their population above their level of abundance in the surrounding sediment (Zajac and Whitlatch 1982). The benthic invertebrates in the recolonized samples at SMC increased in abundance, and had more abundance of benthic invertebrates than the ambient samples throughout the study period. This increase in population size may be due to the opportunistic species that will settle in disturbed sediments at a greater abundance than that is found in the surrounding ambient sediments (Zajac and Whitlatch 1982; Thrush et al. 1996). After 4-6 weeks of initial colonization, this population reaches a maximum and then decreases. This decrease can be seen in the recolonized samples at the MC sites since the abundance of benthic invertebrates decreases after two months into the experiment.

The recolonized samples from the SMC sites always had more abundance and number of taxa than the ambient samples, and the recolonized samples at MC did not always follow this trend. The more taxa and abundance of benthic invertebrates may have been found in the recolonized SMC samples than MC because the presence of the Tuckerton sediment may have been more of a disturbance at SMC than at MC. SMC is more of an established salt marsh than MC; especially M-0, therefore more opportunistic species may be colonizing the Tuckerton sediment at SMC.

Some opportunistic species are the spionidae polychaetes *Polydora ligni* and *Streblospio benedicti*, the amphipod *Ampelisca abdita,* and the *Capitella spp.* (Zajac and Whitlatch 1982; McCall 1977). These species are considered opportunistic species because they have a fast rate of development, high rate of reproduction per year, high recruitment, and high death rate (McCall 1977). The taxa found in the recolonization experiment at SMC were dominated by opportunistic species, such as polychaetes, especially Spionidae.

Benthic recolonization is variable and ambient population fluctuations can affect it (Zajac and Whitlatch 1982). This dependency on the ambient population is apparent since the recolonized samples and the ambient samples have similar to identical organisms (Zajac and Whitlatch 1982; Thrush et al. 1996). According to McCall (1977), the species composition of the recolonized and the ambient samples were not similar until a year after the commencement of the experiment, but the number of species were similar prior to this. Thrush et al. (1996) conducted a study in New Zealand, and found that the defaunated sediments were similar to the ambient sediment in reduction oxidation characteristics after two tidal cycles.

Substrate and contamination may not be responsible for the differences found in the recolonized samples between SMC and MC. The Hackensack Meadowlands sediment is contaminated while the Tuckerton sediment is uncontaminated. The Tuckerton sediment was texturally and structurally different from the Hackensack Meadowlands sediment. Regardless of the differences in sediment structure and texture and the contamination levels, the composition of benthic invertebrates was similar between the recolonized and ambient sediments. McCall (1978) found that sediment was not a factor in the distribution of benthic invertebrates, but depth and disturbance were more important factors in species distribution. This study found that the recolonization of benthic invertebrates did not indicate an effect from substrate and contamination.

This recolonization experiment followed the concept of opportunism that was proposed by Zajac and Whitlatch (1982) since the recolonized samples had always more abundance of benthic invertebrates than the ambient samples at SMC, and sometimes at MC. The abundance of benthic invertebrates in the recolonized samples at MC may have decreased below the abundance found in the ambient samples because a benthic invertebrate population that reaches a maximum after 4-6 weeks of initial recolonization will decrease (Zajac and Whitlatch 1982; McCall 1977). Some of the known opportunistic species, such as Spionidae polychaetes and *Streblospio benedicti* were found in the recolonized samples. This study corroborates Zajac and Whitlatch (1982) and Thrush et al.'s (1996) findings that the recolonized population of benthic invertebrates is dependent upon the ambient population.

CHAPTER 5

CONCLUSIONS

The functional value of *P. australis* should be further evaluated to determine its affects on salt marshes. According to this study, *P. australis* supports a healthy benthic community that is comparable to *S. alterniflora*, and the benthic community of the mitigated *S. alterniflora* marsh had not improved since its establishment. However, more research needs to be conducted on *P. australis* and its affects on benthic communities. Since the knowledge on *P. australis* is incomplete, the management of *P. australis* should be conducted by trying to control its invasive nature.

P. australis should be managed because it is beneficial to the environment in such ways as being a buffer against storm wave damage, keeping estuarine banks stabilized, serving as a refuge for wildlife, colonizing a disturbed area, and detoxifying sediments (Hellings and Gallagher 1992; Rice et al. 2000). *P. australis* can be managed through increasing tidal flooding and salinity levels onto the marsh, which supports flood-tolerant and salt-tolerant shortgrass species (Windham and Lathrop 1999), instead of mowing, burning, or spraying it with herbicides (Hellings and Gallagher 1992). Rice et al. (2000) suggest that controlling recently established stands of *P. australis* in a marsh with diverse vegetation instead of trying to control large well established monotypic stands might be easier and more efficient. Meyer et al. (2001) state that conserving, restoring, and creating marshes is the better option than transforming existing marshes, and knowing the functional value of *P. australis* is necessary before trying to manage it.

Suggestions for future research in the Hackensack Meadowlands District are that benthic communities in *P. australis* and *S. alterniflora* dominated salt marshes should be studied along a salinity gradient. The sites of this study have very different salinity levels, and this could be a factor in the types and abundances of benthic invertebrates being found at the sites. Salinity levels may affect the diversity of the benthic communities. Conducting a study along a salinity gradient, could determine whether or not salinity is a factor. Benthic communities should be investigated on the marsh surface on the high marsh since this study concentrated on the creek bank and the edge of the vegetation in the intertidal zone on the low marsh. The effects of heavy metal contamination on the benthic communities in the Hackensack Meadowlands should also be further researched since the area is highly contaminated.

APPENDIX

SUMMARY OF HEAVY METALS CONTAMINATION

	Cadmium (ppm)	Chromium (ppm)	Copper (ppm)	Iron (ppm)	Lead (ppm)	Mercury (ppm)	Nickel (ppm)	Zinc (ppm)
Sawmill Creek								
Sediments								
HMDC-1982*	16.0	-	-	-	347.0	8.0	-	918.0
Kraus and Kraus-1986*	2.4	253.5	164.5	-	151.5		63.3	296.6
Water Quality								
CIMIC-1999	0.0571	0.0272	0.0387	0.595	0.37	-	0.244	0.0503
Mill Creek								
Sediments								
TAMS-1985*	5.0	-	-	-	850.0	6.0	-	1800.0
Kraus and Kraus-1986*								
non-mitigated section	1.0	1092.7	717.83	-	503.5	-	64.8	-
mitigated section	1.7	174.7	79.5	-	247	-	53.9	419
Water Quality								
CIMIC-1999	0.0165	0.0085	0.0132	0.403	0.155	-	0.0816	0.0304
Soil Cleanup Criteria (NJDEP 1999)								
Residential Direct Contact Soil Cleanup Criteria (RDCSCC)	39	240/270 (CrVI), 120000 (CrIII)	600	-	400	14	250	1500
Non-Residential Direct Contact Soil Cleanup Criteria (NRDCSCC)	100	6100/20(CrVI)	600	-	600	270	2400	1500

REFERENCES

Able, K.W. and S.M. Hagan. 2000. Effects of common reed (*Phragmites australis*) invasion on marsh surface macrofauna: response of fishes and decapod crustaceans. Estuaries. 23: 633-646.

Allen, E.A., P.E. Fell, M.A. Peck, J.A. Gieg, C.R. Guthke. and M.D. Newkirk. 1994. Gut Contents of common mummichogs, *Fundulus heteroclitus* L., in a restored impounded marsh and in natural reference marshes. Estuaries. 17: 462-471.

Angradi, T.R., S.M. Hagan, and K.W. Able. 2001. Vegetation type and the intertidal macroinvertebrate fauna of a brackish marsh: *Phragmites* vs. *Spartina*. Wetlands. 21: 75-92.

Bertness, M.D. 1991. Zonation of *Spartina patens* and *Spartina alterniflora* in New England salt marsh. Ecology. 72: 138-148.

Boesch, D.F. 1972. Species diversity of marine macrobenthos in the Virginia area. Chesapeake Science. 13: 206-211.

Britton, N.L. 1889. Catalogue of plants found in New Jersey. From the Final Report of the State Geologist, Vol. II. Trenton, New Jersey: The John L. Murphy Publishing Company, 293.

Center for Information Management, Integration and Connectivity (CIMIC). 1999. Meadowlands Environmental Research Institute (MERI). Digital Meadowlands. Hackensack Meadowlands Development Commission Monitoring. Water Quality Program. http://cimic.rutgers.edu/hmdc_public/monitoring/

Chapman, V.J. 1940. Studies in salt-marsh ecology: Sections VI and VII. Comparisons with marshes on the East Coast of North America. Journal of Ecology. 28: 118-152.

Chapman, V.J. 1960. Salt marshes and salt deserts of the world. New York: Intersciences Publishers, Inc., 1-11, 277-289.

Chapman, V.J. (ed.) 1977. Ecosystems of the world 1: Wet coastal ecosystems. Amsterdam: Elsevier Scientific Publishing Company, 2-13, 23-27.

Craft, C.B., S.W. Broome, and J.A. Sacco. 1998. Twenty five years of ecosystem succession on newly created salt-marshes. Abstract in International Conference. Concepts and controversies in tidal marsh ecology. April 5-9, 1998. Cumberland County College, Vineland, New Jersey.

Fell, P.E., S.P. Weissbach, D.A. Jones, M.A. Fallon, J.A. Zeppieri, E.K. Faison, K.A. Lennon, K.J. Newberry, L.K. Reddington. 1998. Does invasion of oligohaline tidal marshes by reed grass, *Phragmites australis* (Cav.) Trin. ex Steud., affect the availability of prey resources for the mummichogs, *Fundulus heteroclitus* L.? Journal of Experimental Marine Biology and Ecology. 222: 59-77.

Flint, R.W. and R.D. Kalke. 1983. Environmental disturbance and estuarine benthos functioning. Bulletin of Environmental Contamination and Toxicology. 31: 501-511.

Flynn, M.N., Y. Wakabara, and A.S. Tararam. 1998. Macrobenthic associations of the lower and upper marshes of a tidal flat colonized by *Spartina alterniflora* in Cananeia Lagoon Estuarine region (Southeastern Brazil). Bulletin of Marine Science. 63: 427-442.

Gaston, G.R., D.L. Lee, and J.C. Nasci. 1988. Estuarine macrobenthos in Calcasieu Lake, Louisiana: Community and Trophic Structure. Estuaries. 11: 192-200.

Gaston, G. R. and J.C. Young. 1992. Effects of contaminants of macrobenthic communities in the upper Calcasieu Estuary, Louisiana. Bulletin of Environmental Contamination and Toxicology. 49: 922-928.

Gleason, H.A. and A. Cronquist. 1991. Manual of vascular plants of Northeastern United States and Adjacent Canada. 2nd ed. New York: The New York Botanical Garden. 781.

Graca, M.A., S.Y. Newell, and R.T. Kneib. 2000. Grazing rates of organic matter and living fungal biomass of decaying *Spartina alterniflora* by three species of salt-marsh invertebrates. Marine Biology. 136:281-289.

Gray, J.S., K.R. Clarke, R.M. Warwick, and G. Hobbs. 1990. Detection of initial effects of pollution on marine benthos: an example from the Ekofisk and Eldfisk oilfields, North Sea. Marine Ecology Progress Series. 66: 285-299.

Hackensack Meadowlands Development Commission (HMDC). February 2000. Internal report for t he National Academy of Sciences Meadowlands Mitigation Project Summary.

Harshberger, J.W. and V.G. Burns. 1919. The Vegetation of the Hackensack Marsh: a typical American fen. Transactions of the Wagner Free Institute of Science of Philadelphia. 9: 1-35.

Havens, K.J., Varnell, L.M., and J.G. Bradshaw. 1995. An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia. Ecological Engineering. 4: 117-141.

Healy, B. and K. Walters. 1994. Oligochaeta in *Spartina* stems: the microdistribution of Enchytraeidae and Tubificidae in a salt marsh, Sapelo Island, USA. Hydrobiologia. 278: 111-123.

Hellings, S.E. and J.L. Gallagher. 1992. The effects of salinity and flooding on *Phragmites australis*. Journal of Applied Ecology. 29: 41-49.

Hettler, W.J. 1989. Nekton use of regularly-flooded salt marsh cordgrass habitat in North Carolina, USA. Marine Ecology Progress Series. 56:111-118.

Heusser, C.J. 1949. History of an estuarine bog at Secaucus, NJ. Bulletin of the Torrey Botanical Club. 76: 385-406.

Hopkinson, C.S., A.E. Giblin, J. Tucker, and R.H. Garritt. 1999. Benthic metabolism and nutrient cycling along an estuarine salinity gradient. Estuaries. 22: 863-881.

Ishikawa, K. 1989. Relationship between bottom characteristics and benthic organisms in the shallow water of Oppa Bay, Miyagi. Marine Biology. 102: 265-273.

Kneib, R.T. 1984. Patterns of invertebrate distribution and abundance in the intertidal salt marsh: causes and questions. Estuaries. 7: 392-412.

Kneib, R.T. 1988. Testing for indirect effects of predation in an intertidal soft-bottom community. Ecology. 69: 1795-1805.

Kneib, R.T. 1993. Growth and mortality in successive cohorts of fish larvae within an estuarine nursery. Marine Ecology Progress Series. 94:115-127.

Kneib, R.T. and M.K. Knowlton. 1995. Stage-structured interactions between seasonal and permanent residents of an estuarine nekton community. Oecologia. 103: 425-434.

Kneib, R.T., S.Y. Newell, and E.T. Hermeno. 1997. Survival, growth and reproduction of the salt-marsh amphipod *Uhlorchestia spartinophlia* reared on natural diets of senescent and dead *Spartina alterniflora* leaves. Marine Biology. 128: 423-431.

Kneib, R.T. and A.E. Stiven. 1978.Growth, reproduction, and feeding of the *Fundulus heteroclitus* (L.) on a North Carolina salt marsh. Journal of Experimental Marine Biology and Ecology. 31: 121-140.

Kneib, R.T. and A.E. Stiven. 1982. Benthic invertebrate responses to size and density manipulations of the common mummichog, *Fundulus heteroclitus*, in an intertidal salt marsh. Ecology. 63: 1518-1532.

Kneib, R.T. and S.L. Wagner. 1994. Nekton use of vegetated marsh habitats at different stages of tidal inundation. Marine Ecology Progress Series. 106: 227-238.

Kneib, R.T. and C.A. Weeks. 1990. Intertidal distribution and feeding habits of the mud crab, *Eurytium limosum*. Estuaries. 13:462-468.

Kraus, D.B. and M.L. Kraus. 1988. The establishment of a fiddler crab (*Uca minax*) colony on a manmade *Spartina* mitigation marsh, and its effect on invertebrate colonization. Proceedings of the National Wetland Symposium: Mitigation of Impacts and Losses. New Orleans, Louisiana. October 8-10, 1986. Association of State Wetland Managers (ASWM) Technical Report 3. 343-348.

Krebs, C.J. 1999. Ecological methodology. 2nd ed. New York: Addison Wesley Longman, Inc. 440-452.

LaMotte Company. 1999. Soil Texture Unit. Code 1067. Chestertown, Maryland.

LaSalle, M.W., M.C. Landin, and J.G. Sims. 1991. Evaluation of the flora and fauna of a *Spartina alterniflora* marsh established on dredged material in Winyah Bay, South Carolina. Wetlands. 11: 191-208.

LaSalle, M.W. and L.P. Rozas. 1991. Comparing benthic macrofaunal assemblages of creek bank beds of the spikerush *Eleocharis parvula* (R&S) link and adjacent unvegetated areas in a Mississippi brackish marsh. Wetlands. 11: 229-244.

Levin, L.A. 1982. Interference interactions among tube-dwelling polychaetes in a dense infaunal assemblage. Journal of Experimental Marine Biology and Ecology. 65: 107-119.

Levin, L.A., D. Talley, and G. Thayer. 1996. Succession of macrobenthos in a created salt marsh. Marine Ecology Progress Series. 141: 67-82.

Levin, L.A. and T. S. Talley. 2000. Influences of vegetation and abiotic environmental factors on salt marsh invertebrates. In: M.P. Weinstein and D.A. Kreeger (eds.), Concepts and controversies in tidal marsh ecology. Dordrecht, Netherlands: Kluwer Academic Publishers, 661-707.

Maltby, L. 1999. Studying stress: the importance of organism-level responses. Ecological Applications. 9: 431-440.

McCall, P.L. 1977. Community patterns and adaptive strategies of the infaunal benthos of Long Island Sound. Journal of Marine Research. 35: 221-266.

McCall, P.L. 1978. Spatial-temporal distributions of Long Island Sound infauna: the role of bottom disturbance in a nearshore marine habitat. In: M.L. Wiley (ed.), Estuarine Interactions. New York: Academic Press, 191-219.

Meyer, D.L., J.M. Johnson, and J.W. Gill. 2001 A Comparison of nekton use of *Phragmites australis* and *Spartina alterniflora* marshes in the Chesapeake Bay, USA. Marine Ecology Progress Series. 209: 71-83.

Moy, L.D. and L.A. Levin. 1991. Are *Spartina* marshes a replaceable resource? A functional approach to evaluation of marsh creation efforts. Estuaries. 14: 1-16.

Minello, T.J. and J.W. Webb, Jr. 1997. Use of natural and created *Spartina alterniflora* salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series. 151: 165-179.

Minton, M.D. 1999. Coastal wetland restoration and its potential impact on fishery resources in the Northeastern United States. American Fisheries Society Symposium. 22: 405-420.

Mitsch, W.J. and J.G. Gosselink. 1993. Wetlands. 2nd ed. New York: Van Nostrand Reinhold, 5-17, 219-265.

Multivariate Statistical Package (MVSP). 1985-2000. Version 3.12c. [Computer Program]. Kovach Computing Services.

Newcomb, L. 1977. Wildflower guide. New York: Little, Brown and Company. 418-419.

Niering, W.A. R.S. Warren. 1977. Salt Marshes. In J. Clark. Coastal ecosystem management: a technical manual for the conservation of coastal zone resources. New York: John Wiley and Sons, 697-702.

Niering, W.A. and R.S. Warren. 1980. Vegetation patterns and processes in New England salt marshes. BioScience. 30: 301-307.

Osenga, G.A. and B.C. Coull. 1983. *Spartina alterniflora* Loisel root structure and meiofaunal abundance. Journal of Experimental Marine Biology and Ecology. 67: 221-225.

Packard, G.C. and R.K. Stiverson. 1976. Abundance and production of macroinvertebrates from natural and artificially established salt marshes in North Carolina. The American Midland Naturalist. 96: 487-493.

Pocklington, P. and P.G. Wells. 1992. Polychaetes key taxa for marine environmental quality monitoring. Marine Pollution Bulletin. 24: 593-598.

Posey, M.H., T.D. Alphin, and C.M. Powell. 1997. Plant and infaunal communities associated with a created marsh. Estuaries. 20: 42-47.

Rader, D.N. 1984. Salt-marsh benthic invertebrates: small-scale patterns of distribution and abundance. Estuaries. 7: 413-420.

Raichel, D.L. 2001. The influence of *Phragmites* dominance on marsh resident fish in the Hackensack Meadowlands, New Jersey. M.S. Thesis. Rutgers University, NJ, USA.

Rice, D., J. Rooth, and J.C. Stevenson. 2000. Colonization and expansion of *Phragmites australis* in upper Chesapeake Bay tidal marshes. Wetlands. 20: 280-299.

Rilling, G.C., P.E. Fell, and R.S. Warren. March 26, 1998. Fish use of brackish high marsh areas in the lower Connecticut River: a comparison of a *Phragmites australis*-dominated versus a restored marsh. Connecticut Department of Environmental Protection. Office of Long Island Sound Programs. Final Report.

Ringold, P. 1979. Burrowing, root mat density, and the distribution of fiddler crabs in the Eastern United States. Journal of Experimental Marine Biology and Ecology. 36: 11-21.

Sacco, J.N., E.D. Seneca, and T.R. Wentworth. 1994. Infaunal community development of artificially established salt marshes in North Carolina. Estuaries. 17: 489-500.

Sarda, R., K. Foreman, and I. Valiela. 1992. Controls of benthic invertebrate populations and production of salt marsh tidal creeks: experimental enrichment and short- and long-term effects. In: G. Colombo, I. Ferrari, V. Ceccherelli, R. Rossi (eds.), Marine eutrophication and population dynamics. Fredensburg, Denmark: Olsen and Olsen, 85-91.

Sarda, R., K. Foreman., and I. Valiela. 1995. Macroinfauna of a southern New England salt marsh: seasonal dynamics and production. Marine Biology. 121: 431-445.

Smith, C.R. and S.J. Brumsickle. 1989. The effects of patch size and substrate isolation on colonization modes and rates in an intertidal sediment. Limonology and Oceanography. 34: 1263-1277.

Statistical Analysis System (SAS). 1999-2000. Version 8. [Computer Program.] Cary, North Carolina: SAS Institute Inc.

TAMS. 1990. Comprehensive baseline studies: IR-2 site and off-site mitigation areas. Evaluation of Harmon Meadow Western Brackish Marsh Mitigation Area. Prepared for Hartz Mountain Industries, Inc. TAMS, New York, New York.

Teal, J.M. 1962. Energy flow in the salt marsh ecosystem of Georgia. Ecology. 43: 614-624.

Teal, J.M. and J.W. Kanwisher. 1966. Gas transport in the marsh grass, *Spartina alterniflora*. Journal of Experimental Botany. 17: 355-361.

Teal, J.M. and W. Wieser. 1966. The distribution and ecology of nematodes in a Georgia salt marsh. Limnology and Oceanography. 11: 217-222.

Thrush, S.F., R.B. Whitlatch, R.D. Pridmore, J.E. Hewitt, V.J. Cummings, and M.R. Wilkinson. 1996. Scale-dependent recolonization: the role of sediment stability in a dynamic sandflat habitat. Ecology. 77: 2472-247.

United States Department of Agriculture-Natural Resources Conservation Service, Department of Plant and Soil Sciences of the University of Massachusetts, and Massachusetts Agricultural Experiment Station. Describing and documenting soil conditions. Amherst, Massachusetts: Soil Characterization Laboratory, University of Massachusetts.

Van Dolah, R.F. 1978. Factors regulating the distribution and population dynamics of the amphipod *Gammarus palustris* in an intertidal salt marsh community. Ecological Monographs. 48: 191-217.

Waksman, S.A., H. Schulhoff, C.A. Hickman, T.C. Cordon, and S.C. Stevens. 1943. The peats of New Jersey and their utilization. Part B. New Jersey: Department of Conservation and Development.

Wainright, S.C., M.P. Weinstein, K.W. Able, and C.A. Currin. 2000. Relative importance of benthic microalgae, phytoplankton and the detritus of smooth cordgrass *Spartina alterniflora* and the common reed *Phragmites australis* to brackish-marsh food webs. Marne Ecology Progress Series. 200: 77-91.

Weinstein, M.P., J.H. Balletto, J.M. Teal, and D.F. Ludwig. 1997. Success criteria and\ adaptive management for a large-scale wetland restoration project. Wetlands Ecology and Management. 4: 111-127.

Weinstein, M.P. and J.H. Balletto. 1999. Does the common reed, *Phragmites australis*, affect essential fish habitat?. Estuaries. 22:793-802.

Weinstein, M.P., S.Y. Litvin, K.L. Bosley, C.M. Fuller, and S.C. Wainright. 2000. The role of tidal salt marsh as an energy source for marine transient and resident finfishes: A stable isotope approach. Transactions of t he American Fisheries Society. 129: 797-810.

Weiss, H.M. 1995. Marine animals of southern New England and New York. State Geological and Natural History Survey of Connecticut. Hartford, Connecticut: Connecticut Department of Environmental Protection.

West, T.L. and W.G. Ambrose, Jr. 1992. Abiotic and biotic effects on population dynamics of oligohaline benthic invertebrates. In: G. Colombo, I. Ferrari, V. Ceccherelli, R. Rossi (eds.), Marine eutrophication and population dynamics. Fredensburg, Denmark: Olsen and Olsen, 189-194.

Willis, O.R. 1877. Catalogue of plants growing without cultivation in the state of New Jersey, with a specific description of all the species of a violet found therein. New York: A.S. Barnes and Company, 75.

Whaley, M., R. Garcia, and J. Sy. 1989. Acute bioassays with benthic macroinvertebrates conducted in situ. Bulletin of Environmental Contamination and Toxicology. 43: 570-575.

Windham, L. and R.G. Lathrop, Jr. 1999. Effects of the (common reed) invasion on aboveground biomass and soil properties in brackish tidal marsh of the Mullica River, New Jersey. Estuaries. 22: 927-935.

Zajac, R.N. and R.B. Whitlatch. 1982. Responses of estuarine infauna to disturbance. I. Spatial and temporal variation of initial recolonization. Marine Ecology Progress Series. 10:1-14.

Zedler, J.B. 1993. Canopy architecture of natural and planted cordgrass marshes: selecting habitat evaluation criteria. Ecological Applications. 3: 123-138.

Zedler, J.B. 1996. Ecological issues in wetland mitigation: an introduction to the forum. Ecological Applications. 6: 33-37.