EFFECTS OF ALTERED SHORELINES ON MACROFAUNA AND DIEL-CYCLING HYPOXIA IN TIDAL TRIBUTARIES OF DELAWARE BAY AND DELAWARE COASTAL BAYS

by

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ABSTRACT

The physical structure of estuarine habitats has been, and continues to be modified by human activity. Understanding how the biophysical structure of shoreline affects the functional value and habitat quality of estuaries for shore-zone biota is important in determining impacts of shoreline modification on estuarine systems. The Mid-Atlantic region has seen a decrease in the area of *Spartina alterniflora* (smooth cordgrass) within estuaries during the past few decades, while invasive *Phragmites australis* (common reed) and several types of shoreline hardening including bulkhead, riprap and riprap-sill structures have become more common. These hardened shorelines and invasive grass species have caused a change in the ecological character of the intertidal zone, particularly in urbanized watersheds. This research seeks to understand how changes in shoreline structure affect the ecological character of tidal marsh creeks and basins along Delaware Bay and Delaware Coastal Bays through estuarine fish and crab assemblages, tributary water quality, and fish spawning activity.

In the first part of this study, I used quantitative sampling to examine the spatiotemporal use of estuarine systems by fish and blue crabs in association with environmental characteristics of the habitat, in particular shoreline type and water quality. Fish and blue crabs were collected from the shore-zone of five shoreline types, *S. alterniflora* marsh, *P. australis* marsh, sandy beach, riprap, and bulkhead from

August through September 2009 and June through September 2010. Over the two-year study, in total 102,343 individuals in 28 species of fish and 3,607 blue crabs were collected in the shore-zone of Indian River and Pepper Creek. The greatest abundances of fish were collected from the shore-zone of native *S. alterniflora* marsh, while the greatest abundances of blue crabs were caught in the shore-zone of *P. australis* marsh. Individual species of abundant estuarine fish had unique relationships with hardened and unhardened shorelines. Total fish abundance was greatest when dissolved oxygen (DO) was above the EPA dissolved oxygen criterion of 2.3 mg O_2/l for the survival of aquatic organisms, though this relationship differed among fish species and shoreline type. Minimal associations were found between fish and blue crab abundance and other environmental variables including water temperature, watershed land use, and watershed impervious surface area. Overall fish community assemblages did not differ in any perceptible pattern among shoreline types. Fish assemblages at all sampling stations were more idiosyncratic than coupled based on shoreline type.

In the second part of this study, I used quantitative sampling to examine the spatiotemporal use of estuarine systems by fish and blue crabs in association with riprap-sill. Wetland managers generally consider riprap-sill structures (a type of ‘living shoreline’ consisting of a rock sill placed low in the intertidal zone with native vegetation planted between the sill and shore) to be more ecologically sound than riprap for shoreline stabilization in estuaries. However, little research has been conducted comparing the macrofauna associated with these structures with that inhabiting the more traditionally applied riprap. Fish and blue crab abundance and diversity were compared at riprap-sill shoreline, riprap shoreline, and shoreline fringed with smooth cordgrass (*Spartina alterniflora*) marsh in the Delaware Coastal Bays, USA, from June through September 2010. Seining was conducted to quantitatively sample the shore-zone and shallow subtidal regions, and minnow traps were used to determine presence/absence of fishes in the upper intertidal zone of each shoreline type. In total, 9777 fishes and 548 blue crabs were collected from the shore-zone of all three shoreline types during seining. The greatest abundance and density of fishes were caught along the *S. alterniflora* shoreline. Species densities and composition were generally more similar between the *S. alterniflora* and the riprap-sill shoreline than were the riprap-sill and traditional riprap shoreline. This study demonstrates that although *S. alterniflora* shoreline is inhabited by the highest abundance and density of fishes, riprap-sill structure provides a better alternative than does traditional riprap in terms of abundance and diversity of shore-zone estuarine fish and blue crabs.

In the third part of this study, multi-parameter water quality meters were positioned in the subtidal zone immediately adjacent to each of five shoreline types, *S. alterniflora* marsh, *P. australis* marsh, sandy beach, riprap, and bulkhead, within four tributary regions of Indian River and Pepper Creek. Multi-parameter sondes were alternated among each tributary region once at 2 week intervals to record differences in water temperature and DO concentrations during the summer of 2010. Measurements were recorded when tributaries were experiencing severe diel-cycling hypoxia (July 15th-September 9th, 2010). Negligible differences were found in water temperature among shoreline types. The subtidal area directly adjacent to *S. alterniflora* shorelines was found to experience less severe diel-cycling hypoxia than other shoreline types in Indian River and Pepper Creek. This was particularly evident between the hours of 0200 and 1000, when DO concentrations are typically at their lowest, wherein water adjacent to *S. alterniflora* shorelines had consistently greater DO concentrations than did water adjacent to other shoreline types.

In the fourth part of this study, I examined how Atlantic silverside (*Menidia menidia*) utilized six different shoreline types (*S. alterniflora* marsh, *P. australis* marsh, sandy beach, riprap-sill, riprap, bulkhead) for egg deposition. Atlantic silverside is among the most abundant forage fish species in Mid-Atlantic estuaries, and is an important prey for piscivores such as striped bass, Atlantic mackerel, bluefish, and others. Atlantic silverside eggs are demersal, adhesive, and are laid in estuarine intertidal zones. From April 14th to June 10th, 2010, egg deposition of Atlantic silverside, was measured daily at several sites near Roosevelt Inlet, Delaware, close to the mouth of Delaware Bay. Substrates utilized for egg attachment were noted at each shoreline type. Air/water temperature was recorded every 15 minutes at high, mid, and low intertidal elevations using iButton thermochrons. Over 3,000,000 eggs were collected during 50 sampling days. Eggs were deposited at all six shoreline types, with >93% of eggs collected from *S. alterniflora* shorelines. Choice of substrate for egg attachment was similar across shoreline types with >91% of eggs collected from filaments of the green alga *Enteromorpha* spp., a disproportionately high utilization rate in comparison with *Enteromorpha* spp.'s relative coverage. This study demonstrates that *S. alterniflora* shoreline, in association with *Enteromorpha* spp., is preferred spawning habitat for *M. menidia*, and that hardened shorelines and shorelines inhabited by *P. australis* support substantially reduced egg densities.

Note: All figures and tables contained within document created by author.

Chapter 1

EXAMINING RELATIONSHIPS AMONG SHORELINE TYPE, DIEL-CYCLING HYPOXIA AND LOCALIZED LAND USE ON SHORE-ZONE FISH ASSEMBLAGE AND BLUE CRAB ABUNDANCE IN TWO ESTUARINE TRIBUTARIES OF THE DELAWARE COASTAL BAYS

Introduction

Estuaries, with their varied aquatic habitats, serve as important nursery grounds for the early life stages of many species of fish and crustaceans (Weinstein 1979, Ryer et al. 1990, Lubbers et al. 1990, Peterson and Ross 1991, Rakocinski et al. 1992, Peterson and Turner 1994, Baltz et al. 1998, Peterson et al. 2000). Loss of estuarine aquatic habitat is among the most significant threats to fisheries (Burns 1991, Thomas 1995), as over 70% of commercial fishery species (Brouha 1993, Waste 1996) and over 66% of recreational fishery species utilize coastal and estuarine habitats during at least one stage of their life history (Thayer et al. 1996). Likewise, estuary-resident species such as the mummichog (*Fundulus heteroclitus*), which spend their entire life cycle within an estuary, are negatively impacted by habitat alteration, including the spread of invasive estuarine grasses (Teo and Able 2003a, Teo and Able 2003b). Anthropogenic modifications of coastal regions are significant contributors to the alteration of estuarine habitats. Modifications of subtidal habitats may alter biodiversity, trophic interactions, and faunal assemblages of the shore-zone ecosystem (Toft et al. 2007). Until recently there was little information on the impact of shoreline development and subsequent habitat alteration on fish assemblages and crab abundances directly associated with the shore-zone in estuarine environments

(Peterson and Lowe 2009). Data on fishes from freshwater ecosystems (e.g. review in Paul and Meyer 2001), as well as studies on estuarine benthic infauna (Seitz et al. 2006, Lawless 2008), have consistently shown negative impacts on ecological integrity associated with shoreline and watershed development. Ecosystem management within developed watersheds strives to keep ecosystem production high by preserving optimal habitat characteristics for both flora and fauna. Reaching and assessing this goal has been made difficult by a lack of data regarding the specific impacts of shoreline development on shore-zone estuarine fish assemblages and blue crab abundance (Peterson and Lipcius 2003).

Despite the well documented benefits of healthy coastal wetlands, they are increasingly impacted by watershed urbanization (Lee et al. 2006). This is apparent in Delaware's Coastal Bays and their tributaries. The Delaware Coastal Bays are comprised of three major embayments and their respective tributaries and canals. All three bays are characterized by fringing *Spartina alterniflora* (smooth cordgrass) marsh and increasing human development (Daiber 1969, Tyler 2005). Indian River Bay and Rehoboth Bay receive high levels of nutrient enrichment and have extensive microalgal drift communities during summer months (Timmons and Price 1996, Price 1998).

The tidal tributaries of the Delaware Coastal Bays have been the subject of frequent macrofaunal sampling and water quality monitoring by University of Delaware researchers as well as state natural resources departments (Tyler 2004, Tyler 2005, Tyler and Targett 2007, Stierhoff et al. 2009a). The present study focuses on two tributaries of Indian River Bay: Indian River and Pepper Creek. Tributary creeks are of particular interest in this study because they have a much greater watershed area to water volume ratio than do coastal bays, allowing for clearer investigation involving land use effects. Both tributaries also exhibit diel-cycling hypoxia and are lined by a variety of shoreline hardening features and natural marsh plant assemblages. Shoreline hardening is a frequent form of shore-zone human modification in the Delaware Coastal Bays and their tributaries.

Riprap, riprap-sills (riprap structures positioned low in the intertidal zone with native grasses planted on the landward side), and bulkheads are often installed in areas of human development in an attempt to reduce property erosion and provide protection from storm surges. Shoreline development can negatively impact estuarine water quality and aquatic assemblages through alteration of intertidal habitat, changes in hydrology, loss of allochthanous materials, increased recreational use, and the loss of natural erosion control (Bilkovic and Roggero 2008). Shoreline hardening can result in increased water depth at the land-water interface, leading to a decrease in intertidal faunal community integrity through reductions in nekton diversity, reduced integrity of fish assemblage structure, and reduced macrobenthic assemblage biotic indexes (Jennings et al. 1999, Peterson et al. 2000, Bilkovic et al. 2006). Bulkhead structures alter the natural curve of the shoreline, remove undercut crevice habitat, change shore-zone wave dynamics, and reduce shallow-water habitat. Bulkhead structures often eliminate plant communities in the subtidal regions. Destruction of these plant communities and their subsequent detrital inputs limits the number of detrital-algal consumers such as amphipods and mysids, which form the base of many fishes' food webs in estuarine systems (Boesch and Turner 1984, Peterson and Turner 1994, Jennings et al. 1999). Such impacts on the land-water interface and subsequent biotic responses have been found in both fresh and saltwater wetland regions around the world (Goforth and Carman 2005, Lee et al. 2006).

Shoreline hardening most prominently affects the zone of transition, including the intertidal and shallow subtidal zones, the most ecologically productive regions in estuaries (Odum 1970, Toft et al. 2007, Bilkovic and Roggero 2008). Riprap and bulkhead structures have been shown to negatively impact a range of populations of fauna (primarily fishes, crustaceans, and benthic invertebrates) living adjacent to them over a large spatial and temporal scale (Tourtellotte and Dauer 1983, Herke and Rogers 1989, Able et al. 1998, Duffy-Anderson and Able 1999, Seitz et al. 2006, Bilkovic and Roggero 2008). Shoreline development on a watershed scale has also been shown to alter growth rates, reproduction success, and tissue health of fishes (Burns 1991). The work of Bilkovic and Roggero (2008) demonstrated that bulkheads negatively affect the shore-zone environment and related nekton assemblages in large tributaries of Chesapeake Bay, regardless of the level of watershed development. They found the lowest Fish Community Index (FCI, an index to quantify the quality of an estuary as fish habitat) values are associated with bulkhead shorelines, while riprap and natural shoreline sites shared similar, higher values.

The blue crab (*Callinectes sapidus*) is an important member of estuarine macrofaunal assemblages and is impacted by shoreline hardening. Blue crabs utilize estuaries as habitat during much of its benthic life (Newcombe 1945, Ryer et al. 1990, Wilson et al. 1990, Fitz and Wiegert 1991). Blue crabs play an important role in the trophic structure of estuarine marshes as both predator and prey (Millikin and Williams 1980). Blue crabs use marsh habitats as feeding areas and are preyed upon by various macrofauna including fishes and shorebirds (Fontenot and Rogillio 1970, Adkins 1972, Heck and Thoman 1984, Orth et al. 1991), allowing potential differences in food availability and predator protection to drive habitat preference. Blue crab abundance, though highly variable, was lower along hardened shorelines and unaltered shorelines in Chesapeake Bay (Seitz et al. 2006). Jivoff and Able (2003) found blue crabs were also more abundant in reference marshes than in disturbed salt marsh hay marshes of New Jersey. How blue crab abundance differs among a wide range of shoreline types has not yet been studied at a fine spatiotemporal scale.

Disturbance of natural shorelines by the installation of shoreline hardening structures is one of the primary contributors to the expansion of a non-native genotype of *Phragmites australis* (common reed) in the Mid-Atlantic region (King et al. 2007). *Phragmites australis* invasion has led to a displacement of native macrophyte communities, degradation of wildlife habitat, and alteration of ecosystem processes (Weinstein and Balletto 1999, Windham and Meyerson 2003, Minchinton et al. 2006). While other factors such as eutrophication due to changes in watershed land use also contribute to the spread of *P. australis*, findings from Chesapeake Bay tributaries suggest development directly along shorelines alone allows for the establishment of *P. australis* stands (King et al. 2007).

Phragmites australis-dominated marshes are utilized as habitat by estuary-resident and estuary-dependent fish species. However, within *P. australis*-dominated areas, fish movement into the marsh interior is deterred due to the thick stands of *P. australis* at the marsh edge, as well as increased elevation of established *P. australis* marshes (Weinstein and Balletto 1999, Able and Hagan 2003, Able et al. 2003). With reduced access to the marsh surface, estuarine fish are limited in their feeding capabilities at high tide, potentially reducing their long-term fitness (Able et al. 2003).

Kneib and Wagner (1994) demonstrated that nekton (among fish, primarily *Fundulus* spp.) abundance and species richness in an undisturbed Georgia salt marsh were greatest at slack high tide on the marsh surface suggesting the importance of high marsh habitat. In Louisiana, many species of macrofauna utilized the marsh surface (Rozas 1992, Rozas and Reed 1993). Even abundance of prey species of resident marsh fish species was lower in *P. australis* marshes than in *S. alterniflora* marshes in marsh creeks in southern New Jersey (Raichel et al. 2003).

Both Indian River and Pepper Creek experience low dissolved oxygen (DO), or hypoxia, during the summer months. Hypoxia is characterized as conditions in which DO is below saturation. When DO is < 2.3 mg O_2/l it is characterized as severe hypoxia, a condition which is harmful to aquatic fauna (USEPA 2000, Diaz 2001, Tyler et al. 2009). Hypoxia resulting from eutrophication of coastal estuarine ecosystems is increasing in both temporal and spatial scale worldwide, as the past 60 years have seen a rapid acceleration in the number of occurrences of severe hypoxic and anoxic conditions in estuarine environments (Diaz and Rosenberg 1995, Diaz et al. 2004, Diaz and Rosenberg 2008). Increases in severe hypoxia have paralleled human population growth in estuarine watersheds. As human populations in coastal regions expand, nutrient and organic matter input into aquatic ecosystems increases. Increased nutrient input, particularly nitrogen, from developed watersheds results in excessive algal growth and is commonly associated with severe hypoxia (Nixon 1995, Valiela et al. 1997, Bricker et al. 1999). Decomposition of dead algae and other organic matter by bacteria as well as respiration by living algae can reduce DO concentrations low enough to negatively impact fisheries (Breitburg 2002, Diaz et al.

2004). Beyond fisheries, hypoxic conditions may disrupt or entirely eliminate aquatic faunal communities (Diaz and Rosenberg 1995).

Diel-cycling hypoxia occurs most commonly in the photic zone of the water column. The photic zone may encompass the entirety of the water column in shallow coastal waters such as small marsh creeks, or the water column above the pycnocline in deeper waters. Duration and magnitude of diel-cycling hypoxia is determined by the daily, light-driven cycle of DO production and respiration by algae and bacteria (Kemp and Boynton 1980, D'Avanzo and Kremer 1994, Tyler et al. 2009). The daily cycle of DO fluctuation during summer months is predictable, with the lowest concentrations of DO occurring near dawn, following nighttime community respiration and the highest concentrations of DO occurring in the late afternoon, following daytime algal photosynthesis (D'Avanzo and Kremer 1994, Beck and Bruland 2000, Tyler et al. 2009). Within the daily cycle of diel-cycling hypoxia, DO can range from anoxia (0 mg O_2/l) to highly supersaturated (> 15 mg O_2/l), with severe hypoxia most frequently occurring between the hours of 0200 and 1000 (D'Avanzo and Kremer 1994, Tyler et al. 2009). For coastal water bodies such as estuaries, the proximity to oceanic exchange also affects the magnitude of DO fluctuation (Boynton et al. 1996). Estuarine tributaries, particularly those which feed coastal bays, often have inherent characteristics that favor the growth of high algal density, the biological driver of diel-cycling hypoxia. These characteristics include proximity to high nutrient runoff areas, low tidal current, low flushing rates, high water temperatures, and low rates of wind driven mixing. The two tributaries studied in this research are particularly susceptible to high algal growth due to very high nitrogen concentrations, an order of magnitude greater than in similar tributaries of

Chesapeake Bay (Jordan unpublished data). Recent studies in tributaries of Delaware's Coastal Bays have demonstrated that the duration and spatial extent of diel-cycling hypoxia can be explained in hierarchical order by water temperature, previous day's insolation, hours since morning ebb tide, and daily stream flow conditions (Tyler et al. 2009). Specifically, the longest duration and most severe hypoxic events are associated with high water temperatures, low insolation, and high stream flow.

Field studies frequently demonstrate that abundance and diversity of demersal fishes decline as DO decreases (Breitburg et al. 2001, Tyler and Targett 2007). In the Neuse River Estuary of North Carolina, all ten species of fish studied (including pelagic and demersal estuary-dependent species) avoided areas with DO < 2.0 mg O_2/l (Eby and Crowder 2002). Behavioral avoidance of hypoxic conditions can allow mortality rates in the field to be low. In their natural habitat, external conditions such as water temperature, prey availability, and predation risk influence fish and blue crab behavior. In this respect, behavioral avoidance strategies in hypoxic waters exhibit situational dependence (Kramer 1987, Eby and Crowder 2002). Several estuary-dependent fish species alter swimming speeds in hypoxic conditions (Shoji et al. 2005, Brady et al. 2009). Fish have been found to migrate away from hypoxic conditions and return immediately following the onset of slightly higher DO (Tyler and Targett 2007). However, many species of estuary-resident fish, including mummichog, have very limited home ranges (Lotrich 1975) and are incapable of avoiding wide spread hypoxia. In laboratory studies, fishes (larval, juvenile, and adult) responded to oxygen gradients by moving upwards or laterally away from waters with physiologically stressful or potentially lethal DO concentrations and toward areas of higher DO (Breitburg 2002, Stierhoff et al. 2009b).

Hypoxic conditions may lead to a reduction in fish growth rates. Survival to maturity of larval and juvenile fishes is dependent on their growth rates; higher growth rates in early life stages decrease stage duration and predation mortality, thus increasing survival (Houde 1987, Kneib 1993, Billerbeck et al. 2001, Lankford et al. 2001). Many species of fish utilize estuarine habitats as nurseries during larval and juvenile stages because estuaries provide habitat conducive to fast growth due to warm water temperatures and high prey concentrations (Able and Fahay 1998). The effects of hypoxia on growth rates may be deleterious to the growth benefits provided by estuaries. Currently, the United States Environmental Protection Agency has established a concentration of 4.8 mg O_2/l as the growth protective criteria for the most sensitive species in the saltwater faunal community in the Mid-Atlantic Bight (USEPA 2000). DO concentrations below this level are frequently encountered in the estuarine tributaries of Delaware’s Coastal Bays.

Hypoxic conditions in coastal waters result in the alteration of valuable aquatic habitat. Bearing in mind the mobility of most fishes and adult blue crabs, one must consider the concept of a mosaic of high and low oxygen areas within an estuarine system at any given time. Certain habitats may provide better refuge from hypoxia while others may exacerbate physiological strain. How each shoreline type within an estuary is utilized in different DO conditions is important in our understanding of fish and crab habitat.

As human population in coastal regions grows, forested and marsh areas become cleared and developed resulting in greater proportions of urban and agricultural land use and increased impervious surface area in watersheds. As watershed development occurs, many terrestrial-aquatic linkages are compromised.

Urbanization most visibly affects habitat loss in estuaries through the alteration of shorelines. However, the major drivers of estuary dynamics: hydrological regimes, sedimentation regimes, and nutrient dynamics are impacted in both structure and function by urbanization in a watershed (Lee et al. 2006).

How an individual estuary responds to changes in watershed land use can vary significantly depending on the chemical makeup of runoff, gradients in geomorphology of the watershed and the innate biogeochemical cycling regime of the particular estuary (Cloern 2001, Aikman and Lanerolle 2005). The potential responses of biota to changes in watershed land use are equally unpredictable on small scales (i.e. single stretches of shoreline). Urbanization of a watershed causes adverse outcomes in freshwater stream ecosystems, including degraded water quality and a loss of invertebrate and fish communities (e.g. see review in Paul and Meyer 2001).

Estuaries with different watershed land use characteristics and impervious surface areas experience differences in content and magnitude of freshwater input off of their watershed (e.g. see review in Allan 2004). These differences lead to quantifiable variations in estuarine water quality. Changes in the composition of freshwater input affect the physical and biogeochemical balance of estuarine ecosystems by altering the input of inorganic nutrients, particulate organic matter, dissolved organic matter, and toxic metals (Sanger et al. 1999, Dame et al. 2000, Lerberg et al. 2000, Bowen and Valiela 2004). Increases in impervious surface area modifies the timing, magnitude, and composition of freshwater delivery to the coastal zone, which subsequently alters inputs of associated nutrients and pollutants from urbanized watersheds, altering the biogeochemical attributes of estuarine ecosystems (Dame et al. 2000, Holland et al. 2004). Increasing impervious surface area in watersheds by 10-30% reduced flushing time patterns of estuaries (Buzzelli et al. 2007). With decreased water retention times, various water quality characteristics, including salinity and temperature, change more rapidly, potentially making it physiologically more difficult for fauna to react. Fauna have been noted to respond to sublethal changes in water conditions by moving to maintain preferred physiological conditions (Brett 1971, Beitinger et al. 1975, Reynolds and Casterlin 1976, Major 1978, Stierhoff et al. 2006, Stierhoff et al. 2009b).

Research in South Carolina estuaries found maximum chlorophyll a concentrations, an indicator of eutrophication, to be higher in urbanized estuaries than undeveloped estuaries (White et al. 2004). King et al. (2005) found severe hypoxia present in over 20% of developed watersheds studied in Chesapeake Bay while no hypoxic conditions were observed in estuaries with predominantly forested watersheds. Nutrient loading in Indian River Bay has increased in recent years due to growing urbanization within the watershed as well as continued agricultural runoff (CIB 2007). One of the largest agricultural practices in the region, chicken farming, is a significant source of nutrients to the estuaries. Seventy million chickens, producing roughly 95 million tons of manure yearly, are located within the Delaware Coastal Bays watershed (CIB 2007). This large input of nutrients contributes to the eutrophication of the tributaries and, in turn, the Coastal Bays. The effects of degraded water quality on estuarine fauna often lead to differences in biotic communities between estuaries in urbanized watersheds and undeveloped watersheds.

In an extensive study conducted in Chesapeake Bay macrobenthic community biotic indices were highest in estuaries with primarily forested watersheds (Bilkovic et al. 2006). A similar study in Chesapeake Bay identified ecological thresholds for macrobenthic species at > 23% developed land use within a 1000 m buffer of the shoreline (Bilkovic and Roggero 2008). Watershed land use was significantly correlated with blue crab abundance in Chesapeake Bay (King et al. 2005). Relatively fewer crabs were found in subestuaries with developed and agricultural watersheds than in subestuaries with forested watersheds. Because blue crabs are sensitive to low DO concentrations, their distribution and abundance may be directly affected by eutrophication, commonly associated with high levels of developed and agricultural land use within a watershed (Taylor and Eggleston 2000).

Prior research has found some correlations between watershed land use and fish communities. A study in the large Hudson River estuary found that a 10% increase in watershed development was linked to degradation of the fish community (Limburg and Schmidt 1990). The Maryland Department of Natural Resources suggests that watersheds with 10% or greater impervious surface area will have poor fish habitat (Uphoff et al. 2006). More extensive studies on this subject have been completed in freshwater stream environments. A synthesis of these and other studies led Paul and Meyer (2001) to conclude that impervious surface area comprising between 10 and 20% of a watershed is directly tied to degradation of fish communities in freshwater streams. Likewise, watersheds with greater than 10-25% development display compromised ecological integrity in both fresh and estuarine systems (Limburg and Schmidt 1990, Wang et al. 1997, Paul and Meyer 2001, Bilkovic et al. 2006, Brooks et al. 2006).

Rountree and Able (2007) define habitat quality as “the ability of the habitat to provide conditions appropriate for individual, population or species persistence”. It seems an innate fact that habitat quality will differ in the shore-zone among different shoreline types and that fish and crabs should select the highest quality habitat available (Rountree and Able 2007). The present study was the first to use quantitative sampling on a relatively short temporal scale to examine the ecological impacts of shoreline hardening, including spatial dynamics of the shore-zone fish and the blue crab assemblage and magnitude of diel-cycling hypoxia along five different shoreline types in marsh creek habitats of the Delaware Coastal Bays. Prior studies have often only concentrated on hardened shoreline vs. “natural” shorelines or native marsh plant marshes vs. invasive marsh plant marshes, never all concurrently. Only recently has any research been conducted on how macrofauna utilize a wide range of shoreline structures (Currin et al. 2008, Davis et al. 2008, Stayer et al. 2012). This present study utilized frequent sampling of five unique shoreline habitat types in estuaries during periods before, during and after diel-cycling hypoxia. By utilizing a standardized sampling technique along differing shoreline types, habitat quality was assessed for shore-zone fish assemblages and blue crab abundance. Assessing the abundance and diversity of nekton assemblages along various shore-zone environments has helped to elucidate the habitat quality provided by each shoreline type. The specific objectives of this study were to a) characterize the relationship of five shoreline types (*S. alterniflora*, *P. australis*, sandy beach, riprap, and bulkhead) with shore-zone fish assemblages and blue crab abundance in Pepper Creek and Indian River and b) determine how the magnitude of diel-cycling hypoxia relates to shore-zone fish assemblage and blue crab abundance in Delaware Coastal Bay tributaries.

Materials and Methods

Study Area

The Delaware Coastal Bays are comprised of three major embayments and their respective tributaries and canals (Figure 1.1). All three bays are classified as coastal lagoons: inland water bodies, oriented parallel to the coast, separated from the ocean by a barrier, connected to the ocean through a restricted inlet, and comprised primarily of shallow-waters (Phleger 1969). The bays have an average depth of ~ 1.5 m and are polyhaline to mesohaline systems (Daiber 1969). They include Rehoboth Bay, Indian River Bay and Little Assawoman Bay. All three bays are characterized by fringing *S. alterniflora* marsh and increasing human development (Daiber 1969, USEPA 1996, Tyler 2005). Indian River Bay, Rehoboth Bay, and their respective tributaries receive high levels of nutrient enrichment and have extensive microalgal drift communities during summer months (Timmons and Price 1996, Price 1998).

Macrofaunal sampling and water quality monitoring was conducted in two mesohaline tributaries of Indian River Bay, Indian River and Pepper Creek (Figure 1.2). Both tributaries are lined by a variety of shoreline hardening features and natural marsh assemblages. Indian River and Pepper Creek have a watershed size of approximately 263 km^2 and 108 km^2, respectively. The Indian River watershed comprises ~ 54% of the Indian River Bay watershed, and Pepper Creek comprises ~ 22% of the Indian River Bay watershed. Sediments in these tributaries are primarily soft mud, and patches of macroalgae may be present in summer months, particularly *Ulva* spp., *Gracilaria* spp., and *Aghardiella* spp. (Daiber 1969). Indian River ranges from ~900 m wide at its mouth to ~200 m wide at the upper bounds of the sampling area. Pepper Creek ranges from ~750 m wide at its mouth to ~175 m wide at the upper bounds of the sampling area. Both Indian River and Pepper Creek contain excessively high levels of nutrients, particularly nitrogen (T. Jordan unpublished data), most likely due to agricultural and residential runoff. Both tributaries exhibit severe diel-cycling hypoxia during summer months (Tyler and Targett 2007, Tyler et al. 2009). Macrofaunal sampling for fish and crabs was conducted in two tributary regions, an upper and lower, within each creek in 2009 and 2010. Within Indian River, the upper (IRU) and lower (IRL) tributary regions were sampled at five shoreline types (sampling stations). Within Pepper Creek the upper (PCU) tributary region was sampled at five shoreline types and the lower (PCL) tributary region was sampled at four shoreline types. A third tributary region was sampled near the mouth of Indian River (IRLL) at four shoreline types during the summer of 2009. Water quality monitoring to assess the prevalence and magnitude of diel-cycling hypoxia was conducted at six tributary regions, three in each creek. Water quality monitoring tributary regions correspond with macrofaunal tributary regions.

Water Quality Monitoring

YSI model 600XLM multi-parameter sondes were deployed in each study tributary for continuous water quality monitoring from May through September during both 2009 and 2010. Sondes were positioned just outside the navigation channel, centrally located among sampling stations within each tributary region (Figure 1.2). Each sonde was suspended approximately 15 cm above the substrate using a steel pyramid stanchion. Salinity (psu), dissolved oxygen (DO saturation % and mg/l), pH, and water temperature (°C) were recorded at fifteen-minute intervals. Sondes were cleaned weekly to remove biofouling accumulation. Water quality data were retrieved from each multi-parameter sonde biweekly by removing the sonde from its stanchion and downloading data. When recovered from the field, each sonde was cleaned and calibrated in the lab. Only newly calibrated sondes were positioned in the field to record measurements.

Macrofaunal Sampling

Seining was conducted to study fish and crab abundance and diversity inhabiting the shore-zone and shallow subtidal region at four tributary regions during the summers of 2009 and 2010. Fish and blue crabs occupying the shore-zone were sampled weekly using a 10 m seine with center bag (2 m high; 5 mm mesh) in 2009 and a 15 m seine with center bag (2 m high; 5 mm mesh) in 2010. At each tributary region, five types of shoreline were sampled, *S. alterniflora*, *P. australis*, sandy beach, riprap, and bulkhead (each shoreline type sampled at least 30 meters in length) (only four shoreline types sampled in PCL and IRLL) (Figure 1.3). Each individual section of shoreline where seining occurred is referred to as a sampling station; 23 sampling stations were sampled during the summers of 2009 and 2010. In 2009, seine hauls sampled an area of ~ 21 m^2, and in 2010, seine hauls sampled an area of ~ 33 m^2. Both years were fished using the following method: One end of the net was placed on shore using a shallow draft boat. The net was unfurled in an arc, 3 m from shore at its apogee, along the shoreline. The far end of the net was then brought to shore. A PVC pole was used to startle fish and blue crabs from the area between the net and shoreline towards the bag of the net. The two ends of the net were walked together along the shoreline. The net was then gathered and fish and blue crabs further directed into the bag, which was cinched tight in the water and subsequently placed in the boat where the sample was removed and placed in a bucket containing tributary water. The sampling station was then be immediately sampled again, fishing the net with the same method and in the same direction as the first haul. Fish from the second seine haul were placed into a second bucket. To keep seine efficiency similar among sampling stations, as well as the fact that sediment type and structural habitats (aside from the shoreline structure itself) can influence species composition and assemblage structure, all sampling stations were positioned along areas where substrate was sand to dense mud, with little to no offshore structure (i.e. woody debris or rock) (Szedlmayer and Able 1996).

During sampling in 2009, catch rate was much greater in the first of the two successive seine hauls, with only 7% of the total catch coming from the second haul. Efficiency decreased in 2010 with 83% of the catch coming in the first seine haul. The catch from the two seine hauls was combined into one sample. Bag seine sampling in tributaries of Chesapeake Bay found slightly lower catch efficiencies of fish and crabs along natural marsh and riprap shorelines in comparison with beach and bulkhead shorelines (D. Breitburg personal communication).The first two rapidly repeated seine hauls collected >90% of individual fish and >95% of blue crabs at all five shoreline types. Due to these similarities, no catchability coefficient was applied for each respective shoreline type. In this present study, the catch from the two seine hauls were combined to form a single collection sample. All fish and crabs caught were identified to species and measurements (total length for fishes and carapace width for blue crabs) were collected on a maximum of 20 individuals per collection. All individuals were measured for species represented by ≤ 20 individuals and 20 haphazardly selected individuals were measured for species with > 20 individuals. All macrofauna were released adjacent to the shoreline where they were collected. Sandy beach, riprap, and bulkhead sites were sampled within two hours of high tide. Natural marsh sites, *S. alterniflora* and *P. australis*, were sampled within two hours of high tide (2009) as well as on ebbing tides when the water level had just reached the marsh edge (2010). Every effort was made to account for water level in the present study (Rozas and Minello 1997). In 2010 sampling did not occur when marsh surface was available to species, thus low water level concentration factors were equivalent for all shoreline types. Additionally water level was used as a correction in calculating species density along each shoreline type. These sampling methods were used to standardize catch efficiencies among shoreline types. Water depths were taken at the shoreline and at the apogee of the seine haul in 2010 to calculate volume of water seined, and subsequently fish density (individuals/m^3). Sampling was conducted between August and September of 2009, and June and September of 2010. Eighty-four seines (combined two seine hauls) were conducted in 2009 and 299 seines were conducted in 2010. Temperature, salinity, and DO were measured directly at each sampling station immediately prior to seining using a handheld YSI 85 Dissolved Oxygen Meter.

Land Use

Watershed attributes for both Indian River and Pepper Creek were calculated using ESRI ArcGis 9.2. Hydrography was obtained from National Hydrography Dataset (NHD) through the USGS (http://nhd.usgs.gov/). Hydrologic Unit Code 14 (HUC 14) watershed delineations were obtained through DNREC (http://www.dnrec.state.de.us/DNRECeis/). Pepper Creek and Indian River are subwatersheds of the Indian River Bay HUC 14 watershed. The tributaries respective watersheds were further refined using 2’ LIDAR contour data available through Delaware State Datamil website (http://datamil.delaware.gov/tiles/) along with drainage ditch and various other watershed specific water flow information provided through NHD. GIS layers containing shoreline characteristics including shoreline type and near shore land use were provided by the Center for the Inland Bays (CIB unpublished data). Shoreline type, near shore land use, watershed land use, and impervious surface area were calculated for each tributary, tributary region, and sampling station at buffer distances of 50 m, 100 m, 200 m, and 1000 m from tributary center lines.

Data Analysis

Water quality data was analyzed using three different approaches. Data was plotted over time and distinct differences in temperature, salinity, and DO concentrations were visually noted among sampling stations. Direct statistical comparisons among sampling stations and shoreline types were only made where data was available from all locations to remove variability in daily and hourly temperature, salinity, and DO concentrations. The time each sampling station spent in “extreme” conditions of water temperature (> 30°C, > 31°C, etc.) and DO (< 4.8 mg O_2/l, < 2.3 mg O_2/l, < 0.4 mg O_2/l) were calculated. Thirdly, one-way repeated measures analysis of variance was used to compare temperature and DO among sampling stations. If significant differences were found the Holm-Sidak method of pairwise multiple comparison post-hoc test was performed. If parametric assumptions of normality or variance were not passed after data transformation, Friedman’s related samples two-way analysis of variance by rank was used to compare differences. If significant differences were found in this test, stepwise step-down multiple comparison post-hoc test were performed.

Fish and blue crab abundance and density data from 2009 and 2010 were summarized descriptively. Because different gear (10 m seine vs. 15 m seine) was utilized in 2009 and 2010, statistical analyses are kept separate for the two years. Species richness, mean fish abundance, and density (individuals/m^3), and mean blue crab abundance and density were calculated for each creek, tributary region, shoreline type, and sampling station. Three biotic indexes were calculated for each creek, tributary region, shoreline type, and sampling station: Margalef's Richness Index which relates the number of species to the total abundance at a site, Shannon Diversity Index which relates species evenness and richness within a site, and Pielou's Evenness Index which relates the Shannon Diversity Index to overall site-species evenness.

One-way repeated measures analysis of variance was used to compare weekly fish and blue crab abundance and density data among creek, tributary region, shoreline type, and sampling station for 2010 data. Limited sampling dates from 2009 proved data to be inadequately robust for ANOVA analysis. If significant differences were found, the Holm-Sidak method of pairwise multiple comparison post-hoc test was performed. If parametric assumptions of normality or variance were not passed after data transformation, Friedman's related samples two-way analysis of variance by rank was used to compare differences. If significant differences were found in this test, a stepwise step-down multiple comparison post-hoc test was performed.

Correlations were conducted on fish and crab abundance and density data with various abiotic characteristics of water quality and land use. If parametric assumptions of normality were not met after data was transformed, Spearman's rank correlation was performed. All statistical analyses presented above were conducted using SPSS (version 11.2.0.5) and Sigmaplot (version 18.0.2).

Tributary region, shoreline type, and sampling station differences in fish and blue crab assemblage structure were analyzed with multivariate approaches using PRIMER-E (version 6.0). Station by species similarity matrices based on the Bray-Curtis similarity measure were constructed from fourth-root transformed mean abundance and density of each species at each station within each year. Non-metric multidimensional scaling (MDS) based on the similarity matrices was then used to illustrate the overall spatial groupings of stations. Along with the MDS plots, group-average hierarchical cluster analysis of the similarity matrices was used to define biologically significant levels of similarity among observed station groupings.

Constrained correspondence analysis (CCA) was used to determine which abiotic variables (water quality, watershed land use, shoreline type, sampling location) most closely ordinate with fish assemblage data for 2010 data. Limited sampling dates from 2009 proved data to be inadequately robust for CCA analysis. CCA is similar to multiple regression analysis, except that more than one dependent variable is predicted based on chi-squared distances (Tabachnick and Fidell 1996, Oksanen 2011). Canonical variates, similar to those produced in principal component analysis, are produced for each dataset, with the additional qualification that the resulting variates are strongly correlated with each other (Manly 2005). In the present research, each abiotic variable was time and/or site matched to each seine haul conducted in 2010 to determine which temporal and spatial scales more closely correspond with assemblage data. Correspondence analysis is a weighted averaging method, all CCAs run in this research were scaled such that sites were weighted averages of species assemblages (scaling=1). CCAs were run with species density data from the 13 most abundant fish species (all species with >50 individuals caught) and blue crabs; in correspondence analysis rare species have an unduly high influence on final results. The package "vegan" in R was used to automatically select constraints into the final CCA model using the function "step". This function uses Akaike's information criterion (AIC) as the selection criterion. Using the "step" function to build a model, a CCA was performed with all available predictor variables (65 variables including water quality, land use, time, location, and tide), non-significant variables were removed and the model was refit and run again. This was iterated until only significant variables remain. Two other sets of CCAs were run with no factor variables (i.e. shoreline type and location); one for water quality data and two for land use data (developed land use at 100 m buffer and developed land use at watershed/1000 m buffer) All CCAs were performed in R (version 2.10.1).

Results

Water Quality

Temperature, salinity, and DO conditions were monitored at PCU and PCL through the duration of the summers (May-September) of 2009 and 2010. Temperature, salinity, and DO conditions were monitored at IRU and IRL in early (May 28 – June 25) and late (August 12 – September 22) summer of 2009 and through the duration of the summer (May-September) of 2010. Temperature, salinity and DO conditions were monitored at IRLL and PCLL in early (May 28 – June 25) and late

(August 12 – September 22) summer of 2009 and in early (May 26 – July 15) summer of 2010. Temperature was generally similar between Indian River and Pepper Creek during the sampling timeframe. Temperature was generally higher at the upper tributary region of each creek and decreased downstream towards the mouth. Mean temperatures and temperature ranges are summarized in Table 1.1 and Figures 1.5-1.6.

Salinity across all sampling regions ranged from 2.5-29.1 psu (Table 1.1). Salinity increased along the axis of each tributary, with IRU and PCU having their tributaries respective lowest mean salinities and IRLL and PCLL having their tributaries respective greatest mean salinities, as expected.

DO concentrations showed much greater daily and hourly variability than temperature and salinity. Between the two tributaries, DO ranged from 0 mg O_2/l (anoxia) to 23.5 mg O_2/l (super-saturated conditions) (Table 1.1). Severe diel-cycling hypoxia began to occur frequently in late May in both 2009 and 2010 (Figures 1.7 and 1.8). DO dropped to anoxic (or very nearly anoxic) conditions during diel-cycling events at all three tributary regions in both tributaries in both years. In 2009 IRU and PCU stations had DO concentrations below the EPA DO criterion of 2.3 mg O_2/l for the survival of aquatic organisms for the greatest amount of time (16.8% and 25.0%) (Table 1.2). In 2010, IRU and PCU stations had DO concentrations below the EPA DO criterion of 4.8 mg O_2/l for the safe growth of aquatic organisms for the greatest amount of time (45.9% and 47.2%) (Table 1.3). IRU and PCU also had DO concentrations below the EPA criterion of 2.3 mg O_2/l for the survival of aquatic organisms for the greatest amount of time (20.9% and 16.7%). For data from 2010, between the hours of 0200 and 1000, DO concentrations were below the EPA DO criterion of 4.8 mg O_2/l for safe growth of aquatic organisms and below the EPA criterion of 2.3 mg O_2/l for the survival of aquatic organisms most commonly at IRU (82.4%, 50.0% respectively) and PCU (83.9%, 35.6% respectively) (Table 1.3). Similar findings were present in 2009 (Table 1.2).

In non-parametric repeated measures analysis of DO concentrations, the significantly lowest DO concentrations were at PCL and the significantly greatest DO concentrations were at IRL and PCU in 2009 (Friedman's; $p<0.01$). In the summer of 2010, IRLL had the significantly greatest DO concentrations while IRU had the significantly lowest DO concentrations (Friedman's; $p<0.01$). When analyzing DO concentrations between 0200 and 1000, the significantly lowest DO concentrations were found at PCU and IRU in both 2009 and 2010 (Friedman's; $p<0.01$).

Land Use

The watershed of Indian River has a higher percentage of its total area in developed land use (residential, commercial, industrial) (20.8%) than does Pepper Creek (13.2%) (Table 1.4). Near-shore (100 m buffer) developed landuse along Indian River (36.0%) is greater than that along Pepper Creek (28.5%). The watershed of Pepper Creek (45.2%) has a slightly higher percentage of its total area in agricultural land use than Indian River (40.3%). Indian River (6.6%) and Pepper Creek (6.8%) have near-shore (100 m buffer) agricultural land use in very similar percentages.

The watershed of Indian River (4.1%) has a higher percentage of impervious surface area within its watershed than Pepper Creek (3.3%) (Table 1.5). Impervious surface area is greater at all analyzed buffer widths (100 m, 200 m, 1000 m) in the watershed of Indian River than in the watershed of Pepper Creek. Among tributary regions, impervious surface area percentage is greatest within the PCU watershed

(5.5%), while the lowest percentage impervious surface area within a watershed is found at PCL (4.3%). At the near shore buffer width of 100 m impervious surface area percentage is greatest along IRU (5.7%) and lowest along PCL (3.9%).

The length of bulkhead shoreline is greater along Indian River (10.5%) than Pepper Creek (6.5%) (Table 1.6), while the lengths of riprap shoreline are nearly equivalent between Indian River (2.6%) and Pepper Creek (2.7%). Among tributary regions in Indian River, bulkhead and riprap shorelines are by far most prevalent at IRLL (27.7% and 4.6% respectively). Between tributary regions in Pepper Creek, PCL has a greater length of bulkhead shorelines (8.4%), while PCU has greater lengths of shoreline in riprap (3.3%). Areas around unhardened shorelines tended to remain undeveloped, even at a 1000 m buffer (Table 1.7). Areas around riprap and bulkhead shoreline were dominated by hardened shoreline structures at a 100 m buffer, but had similar ratios to unhardened sampling stations of hardened to unhardened shorelines at a larger 1000 m buffer.

Fish Assemblages

In total, 102,343 individuals in 28 species of fish were collected at 23 sampling stations over the two-year study. Collections were dominated by a limited number of fish species: Atlantic silverside (*Menidia menidia*) (70.2% of total catch) in 2009 (Table 1.8-1.10) and mummichog (*Fundulus heteroclitus*) (65.3% of total catch) in 2010 (Table 1.11-1.13). Other dominant species in 2009 including bay anchovy (*Anchoa mitchilli*) (7.6%), silver perch (*Bairdiella chrysoura*) (7.3%) and mummichog (7.3%), and together with Atlantic silverside accounted for 92.4% of the total catch. Other dominant species in 2010 including menhaden (*Brevoortia tyrannus*) (12.3%)

and Atlantic silverside (12.3%), and together with mummichog accounted for 89.9% of the total catch. Species caught during this study included juvenile and adult estuarine resident and dependent species as well as marine transient fish species.

Species richness and fish abundance/density varied among creek, tributary region, shoreline type, and sampling station during both sampling years. In 2009, species richness was greater in Indian River (21 species) than in Pepper Creek (15 species) (Table 1.14). The tributary region with the greatest species richness in 2009 was IRL (16 species), while the lowest species richness was found in PCU (11 species). Among shoreline types in 2009 species richness ranged from 16 species along beach shorelines to 13 species along *S. alterniflora* shorelines (Table 1.15). In 2010 species richness was greater in Pepper Creek (25 species) than in Indian River (22 species). The tributary region with the greatest species richness in 2010 was PCL (23 species), while IRL (20 species) had the lowest species richness. Among shoreline types in 2010 species richness ranged from 22 species along beach shorelines to 17 species along *S. alterniflora* shorelines.

Mean fish abundance was greater, though not significantly (Repeated Measures ANOVA; p=0.74), in Pepper Creek than in Indian River during 2009. The greatest mean fish abundances among tributary regions in 2009 were PCU and IRL, though no significant differences existed in mean abundance among tributary region (Repeated Measures ANOVA; p=0.68). Beach shorelines had the greatest mean fish abundance (Repeated Measures ANOVA; p<0.01) while all other shorelines had lower, similar abundances. Much of the increased abundance along beach shorelines in 2009 was due to high abundances of Atlantic silverside.

Mean fish abundance (Repeated Measures ANOVA; p=0.03) and density (Repeated Measures ANOVA; p=0.03) were significantly greater in Indian River (IR) than in Pepper Creek (PC) in 2010. Significantly greater densities of summer flounder (Friedman's; p=0.033), Atlantic silverside (Friedman's; p=0.046), and sheepshead minnows (Friedman's; p<0.01) were caught in Indian River than in Pepper Creek. Significantly greater densities of weakfish were caught in Pepper Creek than in Indian River (Friedman's; p<0.01). Mean fish abundance was statistically similar among tributary regions in 2010 (Friedman's; p=0.219). However, mean fish density was significantly greater (Friedman's; p=0.035) in IRL and PCU than in PCL. Significantly lower densities of mummichog were caught at PCL than any other tributary region (Friedman's; p<0.01). Significantly lower densities of menhaden were caught at IRL than any other tributary region (Friedman's; p<0.01). Mean fish abundance was significantly greater (Friedman's; p=0.01) along *S. alterniflora* shorelines than along *P. australis* and bulkhead shorelines. Mean fish density was found to be significantly greater (Friedman's; p<0.01) along *S. alterniflora* shorelines than any other shoreline type. Beach shorelines had significantly greater fish densities than bulkhead shorelines while beach, *P. australis*, and riprap shorelines shared statistically similar mean fish densities. Significantly greater densities of silver perch (Friedman's; p<0.01) and mummichog (Friedman's; p<0.01) were caught along unhardened shorelines than along hardened shorelines.

Habitat utilization of shore-zone areas adjacent to different shoreline types differed among abundant species in 2010 (limited sampling dates in 2009 make direct comparison weak) (Figure 1.9). Sandy beach shorelines were dominated by high relative densities of mummichog. Striped mullet (*Mugil cephalus*) and striped killifish

(*Fundulus majalis*) utilized beach shore-zones in greater densities than along any other shoreline type. Among abundant species, only silver perch were found in relatively low densities along sandy beach habitats. The shore-zone assemblage along bulkhead shorelines was dominated by menhaden. Bay anchovy and menhaden utilized bulkhead shore-zones in greater densities than along any other shoreline type. *Phragmites australis* shorelines were dominated by high relative densities of mummichog. Silver perch utilized *P. australis* shore-zones frequently, and only *S. alterniflora* shorelines had greater relative densities of silver perch than did *P. australis* shorelines. No other abundant species was found in relatively high densities along *P. australis* shorelines. Riprap shorelines were dominated by Atlantic silverside and mummichog. Atlantic silversides and bay anchovy appear to prefer riprap shoreline habitat, while all other abundant species are found in relatively low densities along riprap structures. *Spartina alterniflora* shorelines were dominated by mummichog. Mummichog, silver perch, striped killifish, and menhaden all were found with relative high densities along *S. alterniflora* shorelines. Among abundant species only striped mullet and bay anchovy were found to have low relative densities along *S. alterniflora* shorelines.

Overall mean fish abundance and density was greatest during periods when DO was ≥ 4.8 mg O_2/l (Figure 1.10, Table 1.16). This relationship between species abundance/density and DO does differ among abundant species and shoreline type (Table 1.16). Among the most abundant species, mummichog and Atlantic silverside had the greatest mean abundance and density during periods when DO was ≥ 4.8 mg O_2/l. Menhaden and silver perch had the greatest mean abundance and density during periods when DO was < 4.8 mg O_2/l but > 2.3 mg O_2/l. Striped killifish had the greatest mean abundance and density during periods when DO was < 2.3 mg O_2/l. Along beach and *S. alterniflora* shorelines overall mean fish abundance and density was greatest during periods when DO was ≥ 4.8 mg O_2/l. Along bulkhead shorelines mean fish abundance and density was greatest during periods when DO was < 4.8 mg O_2/l but > 2.3 mg O_2/l. Along *P. australis* and riprap shorelines mean fish abundance and density was greatest during periods when DO was < 2.3 mg O_2/l.

Mean fish standard lengths differed between tributaries and among tributary regions and shoreline types in 2009 (Tables 1.17-1.19). Only mummichog had large differences in mean lengths among shoreline types with the largest individuals found along bulkhead and *S. alterniflora* shorelines and the smallest individuals found along riprap shorelines. Small samples sizes of all species, including mummichog, may have contributed to the large differences in mean length found.

Larger sample sizes of fish length were collected in 2010 (Tables 1.20-1.22). Length assemblage structures from length-frequency histograms (Figure 1.11-1.16) suggest shore-zone assemblages of mummichog, Atlantic silverside, striped mullet, striped killifish, silver perch, and bay anchovy are similar in Pepper Creek and Indian River. Among the most abundant species, only menhaden displayed differences in length-frequency structure between tributaries, with Pepper Creek having higher relative abundance of larger individuals than Indian River. Length assemblage structure from length-frequency histograms suggest shore-zone assemblages of menhaden, striped mullet, striped killifish, silver perch, and bay anchovy are similar among tributary regions. At IRU sites, a greater relative abundance of smaller individual mummichog was found than at other tributary regions. A greater relative abundance of larger individual Atlantic silversides was found at IRU sites than at other tributary regions.

Length assemblage structure from length-frequency histograms suggest shore-zone communities of bay anchovy, silver perch, striped killifish, striped mullet, and Atlantic silversides to be relatively similar across shoreline types. A greater relative abundance of larger individual mummichog was found at hardened shorelines than at unhardened shorelines. A greater relative abundance of larger individual menhaden was found along bulkhead shorelines than along any other shoreline type.

Biotic index scores varied between Indian River and Pepper Creek in 2009 (Table 1.23). Margalef's Richness Index was higher in Indian River while Shannon Diversity Index and Pielou's Evenness Index scores were slightly higher in Pepper Creek. IRU had the greatest index values associated with it while IRL had the lowest index scores among tributary regions in 2009. *Spartina alterniflora* and *P. australis* shorelines had slightly higher index values than beach and hardened shorelines in 2009 (Table 1.24). In 2010 Pepper Creek had higher biotic index values than Indian River. IRL had the lowest index scores associated with it, while all other tributary regions shared similar, greater values. *Spartina alterniflora* shorelines had much lower biotic index values than all other shorelines, which shared similar greater values in 2010.

Results of Spearman rank correlations show that temperature, salinity, DO, and tidal height had significant effects on the occurrence of some fish species, although most values were comparatively low (< 0.50) (Table 1.25). Mummichog, striped mullet, and weakfish were associated with higher water temperatures, while silver perch were generally associated with cooler water temperatures in the early summer and fall. Minimal associations were found among abundant species and salinity (< 0.20). Menhaden, spot (*Leiostomus xanthurus*), and summer flounder were found to be associated with higher DO concentrations while silver perch were associated with lower DO concentrations. Mummichog, striped killifish, sheepshead minnow (*Cyprinodon varigatus*), and naked goby (*Gobiosoma bosc*) were all associated with low tidal heights while bay anchovy and weakfish were associated with higher tidal heights.

MDS ordination of shore-zone fish assemblages by tributary region, shoreline type, and sampling station indicated some spatial structuring of biotic similarity patterns among stations. In general, among tributary regions, the two Pepper Creek tributary regions (PCU, PCL) tended to group together, while the Indian River regions tended to be more dispersed (Figure 1.18 and 1.18). In 2009 MDS ordination indicated the mouth section of Indian River (IRLL) to have the least related fish assemblage of all other tributary regions. All other tributary regions (IRU, IRL, PCU, PCL) were found to be at least 60% similar and PCU, PCL, and IRL to be 75% similar. In 2010 all tributary regions were found to be over 70 % similar and PCL, PCU, and IRU to be over 80% related.

Among shoreline types, high similarity existed among fish assemblages along all five shorelines in both 2009 (68% similarity) (Figure 1.20) and in 2010 (75% similarity) (Figure 1.21). In 2009 *S. alterniflora* and *P. australis* shorelines (82%) ordinate most closely together, while in 2010 *S. alterniflora* had the least similar fish assemblages of any shoreline. In both 2009 and 2010 individual sampling stations do not clearly ordinate closely to one another based on creek, tributary region, or shoreline type (Figures 1.21 and 1.22). This lack of clear grouping in further emphasized graphically by cluster analysis from dendrograms of fish assemblage data (Figures 1.23 and 1.24).

CCAs were used to describe the association among water quality, watershed land use, sampling location, and fish/blue crab abundance and density data. From 2010 macrofaunal sampling, using the thirteen most abundant fish species, blue crab abundance/density, and all available predictor variables, CCA model building determined three variables to most accurately correspond with species assemblage data: the factors sampling station (i.e. IRL-BE) and sampling week and the vector tidal height (Table 1.26). This CCA indicated that 59.7% of the variance was explained by the first two canonical correlations and 100% of variance was explained by the constrained axes. The first canonical variate calculated for the environmental data was positively correlated with tidal height and explained 31.7% of the variance in fish assemblage density (graphical ordination not presented as factor results plot directly atop species, biplot scored used for analysis). The first canonical variate calculated for the species data was most positively correlated (biplot scores >|.5|) with the presence of menhaden and spot and most negatively correlated with sheepshead minnow and mummichog. Taken as a pair, these variates indicated that the presence of menhaden and spot positively correspond with tide height while mummichog and sheepshead minnow negatively correspond with tide height, which suggests that menhaden and spot were caught in greater densities at higher tides while mummichog and sheepshead minnow were caught in greater densities at lower tides.

Results from a CCA performed with water quality as the only environmental predictor variables indicated that water quality (temperature, salinity, DO) explained much less variation in fish and crab assemblages than did sampling location, sampling week, and tidal height. Permutation tests indicated that constrained axes in the water quality CCA explained only 7.8% of total variability in fish assemblage density (Table 1.27). None of the constrained axes was statistically significant in the water quality CCA. Axis 1, explaining only 3.8% of the variance in fish assemblage density was primarily explained by a positive association with water temperature (Figure 1.26). White mullet were associated with higher water temperatures while summer flounder and silver perch were associated with cooler water temperatures. Axis 2 was negatively associated with salinity, and explained a further 3.4% of the variance in fish assemblage density. Bay anchovy and Atlantic silverside were associated with higher salinities while menhaden and sheepshead minnows were associated with lower salinities.

Results from a CCA performed with developed upland land use attributes within 100 m (DEV100) of the sampling station as the only environmental predictor variables indicated that near shore land use explained much less variation in fish and crab assemblages than did sampling location, sampling week, and tidal height. Permutation tests indicated that constrained axes in the DEV100 CCA explained only 12% of total variability in fish assemblage density (Table 1.28), with two significant axes explaining 96% of that variation. Axis 1, explaining 8.4% of the variance in fish assemblage density was primarily explained by a negative association with all three model selected variables; near shore land use development proportion within 100 m of the sampling station, impervious surface area proportion within 100 m of the sampling station, and riprap length within 100 m of the sampling station (Figure 1.27). Mummichog and sheepshead minnows were positively associated with axis 1, suggesting they are associated with lower development, while bay anchovy and menhaden were negatively associated with axis 1, suggesting a more positive association with development in the near shore region. Axis 2 was negatively associated with impervious surface area within 100 of the sampling station, and explained a further 3.2% of the variance in fish assemblage density. Striped killifish and menhaden were associated with higher levels of impervious surface area while white mullet and sheepshead minnows were associated with lower proportions of impervious surface area. These are contradictory results and further support the poor ordination values found in the DEV100 CCA.

Results from a CCA performed with developed land use within the watershed (DEVWS) of the sampling station as the only environmental predictor variables indicated that watershed land use explained much less variation in fish and crab assemblages than did sampling location, sampling week, and tidal height, though explained a higher amount of variation than the DEV100 CCA. Predictor vectors include bulkhead length within 1000 m of sampling station, riprap length within 1000 m of sampling station, proportion developed land use within sampling station watershed, proportion impervious surface area within watershed, and proportion of near shore land use developed within 1000 m buffer of sampling station (Figure 1.28). Permutation tests indicated that constrained axes in the DEVWS CCA explained only 13.7% of total variability in fish assemblage density (Table 1.29), with two significant axes explaining 94% of that variation. Axis 1, explaining 8.2% of the variance in fish assemblage density was not strongly explained by an association with any of the land use variables. Axis 2 was negatively associated with bulkhead length within 1000 m of the sampling station and positively associated with developed land use proportion and impervious surface area proportion with the sampling station watershed, and explained a further 3.2% of the variance in fish assemblage density. These are contradictory results and further support the poor ordination values found in the DEVWS CCA.

Blue Crabs

In total, of 3,607 blue crabs were collected in the shore-zone during the two-year study (Tables 1.30-1.38). Blue crabs were present in 58.3% of seines in 2009 and 97.0% of seines in 2010 (only mature blue crabs, over 40 mm carapace width, were enumerated in 2009). Mean crab abundance was greater in Indian River than in Pepper Cree during both 2009 and 2010, though no statistical differences in crab abundance (Repeated Measures ANOVA; p=0.531) or crab density (Repeated Measures ANOVA; p=0.541) were found between Indian River and Pepper Creek.. IRL had the greatest mean abundances of crabs in 2009, however this tributary region had the lowest mean abundance in 2010. PCL had low mean abundance of crabs in both years of the study. Among shoreline types, mean crab abundance was greatest at beach shorelines, followed by *P. australis* shorelines in 2009. Mean crab abundance was greatest at *P. australis* shorelines, followed by *S. alterniflora* shorelines in 2010. Repeated measures analysis of variance was not used on crab data from 2009 due to limited data, and all following results are from 2010. IRU had significantly greater blue crab abundance (Friedman's; p<0.01) and density (Friedman's; p<0.01) than IRL and PCL. *Phragmites australis* shorelines had greater crab abundances than bulkhead shorelines (Repeated Measures ANOVA; p=0.011). *Phragmites australis, S. alterniflora* and beach shorelines had significantly greater densities (Repeated Measures ANOVA; p<0.01) of crabs than hardened shorelines.

Mean crab carapace length was slightly greater in Indian River (50.3 mm) than in Pepper Creek (46.5 mm) during 2010. IRU had the greatest mean carapace width (50.8 mm) of blue crabs in 2010. The greatest mean carapace widths were collected from riprap shorelines (55.5 mm). Bulkhead shorelines (43.3 mm) had the lowest mean carapace widths in 2010. Length assemblage structure from length-frequency histograms (Figure 1.29) suggests shore-zone assemblages of blue crabs to be similar between Indian River and Pepper Creek. Greater relative abundances of smaller individual crabs are found in the upper section of each tributary than in the lower tributary regions. Relative abundances of blue crab size classes were generally similar across all five shoreline types.

Mean blue crab abundance and density was greatest when DO was < 2.3 mg O_2/l (Table 1.16). Mean blue crab abundance and density was similar when DO was < 4.8 mg O_2/l but > 2.3 mg O_2/l and when DO was ≥ 4.8 mg O_2/l. Mean blue crab abundance and density was greatest when DO was < 2.3 mg O_2/l at all five shoreline types.

The results of the Spearman rank correlations revealed that salinity and tidal height had significant effects on the occurrence of blue crabs, although values were comparatively low (<0.50) (Table 1.25). Temperature and DO were not significantly correlated with blue crab density.

Discussion

Fish Assemblages

In this present study 3 species, mummichog, Atlantic silverside, and menhaden accounted for 89.2% of all fish caught. Nekton assemblages in estuarine environments are often dominated by few species with high relative abundances (Hettler 1989, Kneib and Wagner 1994, Thomas and Connolly 2001, Boutin 2009). The fish and blue crab assemblage structure found in the present study were similar to those reported from other estuarine creeks along the east coast of the United States (Derickson and Price 1973, Weinstein 1979, Rountree and Able 1992, Wagner 1999, Able et al. 2001, Boutin 2009). Mummichog and Atlantic silverside have previously been reported to be dominant components of the shore-zone assemblage in Delaware Bay (de Sylva et al. 1962) and the Delaware Coastal Bays (Derickson and Price 1973, Price 1998, Boutin 2009). Boutin (2009) reported that mummichog and Atlantic silverside account for 77.2% of total catch in the shore-zone of tributaries in the Delaware Coastal Bays. Likewise, mummichog and Atlantic silverside were among the top five most abundant species along the shore-zone in the Delaware coastal bays and surrounding tidal creeks several decades earlier (Derickson and Price 1973). Across the Delaware Bay in New Jersey, mummichog and Atlantic silverside were, year-round, the two most abundant fish species captured in the shore-zone (Rountree and Able 1992). These species use marsh creeks for foraging, refuge and spawning (Richards and Castagna 1970, Shenker and Dean 1979, Conover and Ross 1982, Able et al. 2003, Raichel et al. 2003).

However, menhaden (the third most abundant species in the present study) had been previously reported to be scarce in the Delaware Coastal Bays (Derickson and

Price 1963, Boutin 2009), though common in the shore-zone of lower Delaware Bay (de Sylva et al. 1962). High variability in menhaden abundance was found in the present study wherein only two menhaden were caught in the shore-zone in the summer of 2009, but was among the top two most abundant species in 2010. This is further emphasized by the fact that despite high overall mean density, menhaden were only caught in only 29.5% of seines in 2010, in stark contrast with the much more frequently encountered mummichog (78.5% of seines) and Atlantic silverside (87.2% of seines). These differences may be due to the schooling nature of menhaden, changes in water quality, as well as changes in abundance by year class.

Singular relationships among shoreline type and abundant species were found in this study. Mummichog's generally high relative density among all shoreline types (greatest mean density among all fish species at unhardened shorelines, second greatest mean density among all fish species at hardened shorelines) was to be expected given the large populations of this species known to inhabit estuarine creeks, as well as their high physiological tolerances to changing oxygen and temperature conditions, such as those found in Pepper Creek and Indian River (USFWS 1989). Mummichog may be the most important species to consider in this research due to their overall pervasiveness as well as their role in marsh food webs, where they are instrumental in the movement of organic material within and out of salt marsh ecosystems, as well as their inextricable ties to shoreline habitats (Kneib and Wagner 1994). Mummichog are the most abundant estuary-resident species in the Mid-Atlantic and have a very limited home range (18 m), suggesting very discrete relationships to the shoreline they are caught adjacent to (Lotrich 1975). Additionally, mummichog utilize intertidal marsh zones for reproduction, further asserting their strong ties to shoreline development (Able 1984). How mummichog density responds to shoreline development may serve as a bellwether for other estuarine fauna. Greater relative densities of mummichog were found along *S. alterniflora* shorelines than any other shoreline type. Significantly greater densities of mummichog were caught along unhardened shorelines than along hardened shorelines. This disparity among densities along shoreline types may be due to the species preferential use of marsh pools, ditches, and depressions found on the marsh surface at high tide. *Spartina alterniflora* marshes experience greater levels of marsh surface inundation than other shorelines (Able and Hagan 2003). This marsh surface habitat, which a certain proportion of the mummichog community will leave for subtidal waters as the tide recedes, is reduced along *P. australis* marsh (due to increased marsh elevation) and non-existent along hardened shorelines. Weisberg and Lotrich (1982) demonstrated that mummichog require utilization of the marsh surface to supplement energy consumption. So though mummichog were ubiquitous across all shoreline types and tributary regions, they did display greater densities along unhardened shorelines, particularly *S. alterniflora* marsh.

Densities of three small estuarine-resident species, striped mullet, striped killifish, and silver perch were greater along unhardened shorelines than along hardened shorelines. Overall shoreline preference for striped mullet is difficult to generalize about (no significant differences exist in density by shoreline type); they were present in only 28% of seine tows and were generally caught in large schools. Because of this, mean densities among shoreline type within tributary region are idiosyncratic with large standard errors. Striped killifish, another abundant estuarine resident species, displayed a more discernible pattern of shoreline preference in this study. Prior studies have found striped killifish to be abundant over sandy substrates (Peterson and Peterson 1979). Though this present study attempted to keep substrate composition among sampling stations as homogenous as possible, sandy beach shorelines tended to have generally sandier subtidal substrates than other shoreline types. The findings from this study match well with this prior research, as the significantly greatest relative densities of striped killifish were found along sandy beach shorelines, while the significantly second greatest relative densities were found along *S. alterniflora* shorelines, the habitat which provides the greatest marsh surface area for feeding and predator avoidance. Like mummichog, striped killifish, another member of the genus *Fundulus*, are known to actively utilize marsh surface habitats (Kneib and Wagner 1994) Silver perch were an important component of the faunal assemblage of Pepper Creek and Indian River as the sixth most abundant fish species and present in over half of all seine hauls. Though silver perch have been reported from marsh creeks and estuarine shorelines, little information is available on habitat utilization (Rountree and Able 1992). In this present study, the vast majority of silver perch caught were young-of-the-year. These individuals displayed a strong preference for the grass marsh habitats, with the significantly greatest relative densities found along *S. alterniflora* and *P. australis* shorelines. Silver perch densities were greatest and grass marsh habitats across all tributary regions further supporting the concept of silver perch showing strong preference for those habitat types.

Menhaden, Atlantic silverside, and bay anchovy all displayed idiosyncratic relative densities with shoreline types among tributary regions. This may be due in part to the schooling nature of these species, allowing single seine hauls to account for significant proportions of total fish catch over the course of the summer. This is particularly true for menhaden and bay anchovy, which were present in only 30% and 35% of tows respectively. Though the sum total of menhaden density at all tributary regions show high relative densities at bulkhead, beach, and *S. alterniflora* shorelines, very high variability in menhaden density among tributary regions was found for each of these shoreline types. For instance, the greatest mean density of menhaden at any sampling site was found at the *S. alterniflora* marsh of IRU, however, menhaden were almost entirely absent from *S. alterniflora* marsh sampling stations in IRL and PCL. Likewise, bulkhead shorelines in IRU and PCL had very high mean densities of menhaden, while menhaden were found in very low densities along bulkhead shorelines in IRL and PCU. This suggests menhaden utilization of shoreline type is potentially highly variable temporally and spatially, thus drawing conclusions about habitat preference is difficult and perhaps unwarranted. Length analysis of menhaden by shoreline type does however suggest some differences in preference among shoreline type (see explanation pg. 44). Atlantic silverside also displayed high variability in density within shoreline type among tributary region. This is most clearly seen at riprap sites, which had the greatest relative density of Atlantic silverside, though this was almost entirely driven by only one riprap site, IRL-RR, while the three riprap sites in the other tributary regions had relatively low densities of Atlantic silversides. Were this IRL-RR station taken out of analysis Atlantic silverside density would be fairly even across all shoreline types, a more likely reality for the schooling, pelagic species. Likewise, bay anchovy, another schooling pelagic species showed high variability within shoreline type among tributary regions suggesting limited direct relationships with shoreline type.

Overall strong similarity in fish assemblage length-frequency structure between Pepper Creek and Indian River is a testament to the overall similarities in watershed land use, water quality, and geographic location between these two tributaries. Given the generally similar size and salinity regimes of the two study tributaries, it is somewhat predictable that fish length-frequency assemblages within species should be quite similar. Among the most abundant species caught in 2010, only menhaden displayed any difference in length-frequency structure between the two creeks with Pepper Creek having a higher relative abundance of larger individuals. This may be attributable to the sampling sites in Pepper Creek being physically closer to the open water habitats of Indian River Bay utilized by late juvenile and adult menhaden. Much like the lack of differences in length-frequency structure between creeks, minimal differences existed in length-frequency structure of the most abundant fish species among tributary regions.

Among shoreline types, greater relative abundances of larger mummichog were found along bulkhead shorelines, than unhardened shorelines. Beach and *S. alterniflora* shorelines in particular had length-frequency structures shifted smaller then bulkhead shorelines. One could argue that smaller individuals preferred unhardened shorelines due to potentially increased cover and therefore reduced predation rates, however, it is unlikely that sandy beach shorelines provide increased cover in comparison with bulkhead shoreline. Bulkhead shorelines do however provide deeper water at the land-water interface than any other shoreline type, therefore serving as more open water than shore-zone habitat. As such, larger mummichog individuals may have a greater predilection to venture into the deeper water habitats along bulkhead shorelines. This situation is magnified for menhaden.

Length-frequency histograms for unhardened and riprap shorelines display the greatest counts of menhaden in the 20-40 mm range, while along bulkhead shorelines the greatest counts are found in the 60-80 mm range. Clearly schools of larger menhaden prefer the open water habitats along bulkhead shorelines to the shallower waters found in the shore-zone of the other shoreline types.

Several species of economic importance to recreational and commercial fisheries were frequently found to be part of the faunal assemblage in Pepper Creek and Indian River including menhaden, spot, weakfish (*Cynoscion regalis*), striped bass (*Morone saxatilis*), and summer flounder. These species were most frequently represented by young-of-year individuals and have previously been noted to utilize estuarine environments as nursery habitats, but are not necessarily frequently reported from these habitats (Able and Fahay 1998). Other, large economically important species captured less frequently in this study included bluefish (*Pomatomus salatrix*), Atlantic croaker (*Micropogonias undulatus*), American eel (*Anguilla rostrata*), and black drum (*Pogonias cromis*). This current study generally supports the widely held view that estuaries and marsh creeks provide nursery habitat for many important fish species (Able and Fahay 1998). Though low abundances of many of these species were caught, a few observations may be made about their relationship to shoreline type. Spot, weakfish, and striped bass were most prevalent along *P. australis* and beach shorelines. Spot and weakfish were also caught in high abundance along riprap shorelines, while striped bass were abundant along the open water habitat provided by bulkhead shorelines. Summer flounder were most prevalent along *S. alterniflora* and beach shorelines.

Prior research has found fish and macrobenthic community integrity to be similar among natural and riprap shorelines, while bulkhead shorelines displayed reduced community integrity (Jennings et al.1999, Seitz et al. 2006, Bilkovic et al. 2006, Bilkovic and Roggero 2008). This present research, however, found all shoreline types to have very similar fish biotic index scores among hardened and unhardened shorelines with the exception of *S. alterniflora* shorelines. Low overall species richness and evenness caused biotic indexes from macrofaunal sampling to be relatively low across all shoreline types. This was particularly apparent at sites where mummichog were found in extremely high relative densities (SP, IRL). A matching effect between biotic index scores and ordination plots from non-metric multidimensional scaling can be seen wherein sampling locations where biotic index scores are markedly lower than other locations tend to be the least similar in overall assemblage in ordination. Overall, results from biotic indexes are not very illustrative, most sampling locations had very similar scores, and those with lower scores are known to be directly tied to the prevalence of one species, mummichog.

No clear association exists among DO concentrations in each tributary region and overall fish abundance and density at that tributary region. The diel-cycling hypoxia regimes of Pepper Creek and Indian River were extremely similar during the summers of 2009 and 2010. Both creeks experienced severely hypoxic to anoxic conditions throughout both summers. Consistently high densities of fish were caught at sampling stations at PCU in 2010, however this tributary region experienced the most severe magnitude diel-cycling hypoxia. In contrast, PCL had lower densities of fish in 2010, but less severe diel-cycling hypoxia.

Varying results were found when examining fish density at different DO concentrations across all tributary regions. The overall results of this research match with earlier studies whereby fish abundance and density is reduced when DO drops below 4.8 mg O_2/l and is severely reduced when DO drops below 2.3 mg O_2/l (Eby and Crowder 2002, Tyler 2004, Tyler and Targett 2007). However, this research did not find this to be true across all fish species. In fact, rather unexpectedly, species with known susceptibility to low DO (weakfish, striped bass) were more prevalent during low-DO events while species known to be tolerant of severely hypoxic waters (mummichog) had greatly reduced densities during low-DO events. These findings coincide with the high variability found in weekly catch rates. Given the limited sampling area fished by the seine net, tows where large schools of fish were caught very strongly affected mean fish data at that site and for that species leading to high standard errors. It then seems plausible to suggest a few individual tows may weight mean fish densities greatly, allowing one catch during a low DO event to drive final results. This is supported by the fact that the majority of the species which were found to have greater densities at low DO conditions are schooling species which were infrequently caught, but when caught were almost always found in large schools (striped mullet, menhaden). This suggests a longer-term study, in more tributaries which experience markedly different magnitudes of diel-cycling hypoxia, with a larger net may be more capable of defining the effects of DO on individual species in the Delaware Coastal Bays. Results from this research suggest that estuarine systems with widely pervasive diel-cycling hypoxia will not see changes in shore-zone fish assemblage with hourly changes in DO. This indicates the possibility that shoreline stressors throughout an estuary may have just as great an effect on fish assemblages as does high magnitude diel-cycling hypoxia.

Generally low associations were found among environmental variables and fish densities in nonparametric correlations. Water temperature and salinity range vastly in shallow estuarine waters over the course of a summer, and estuarine resident and transient species are capable of adapting to these changing conditions. Fishes' ability to adjust to these changing conditions allows their presence to not be entirely dependent on these environmental variables, thus low correlation coefficients among temperature, salinity, and fish density were found. Given the wide-spread nature of diel-cycling hypoxia found in Indian River and Pepper Creek (severe hypoxia was encountered in every tributary region of both years), in connection with the fish and crab sampling method used in this project, it was unlikely that strong correlations among DO and fish density would be found. By the very nature of using a small seine net in very close proximity to the land-water interface, only relatively small fauna (usually estuarine-resident species or juveniles/young-of-the-year of estuarine transient species) are captured. These individuals are less capable of physically avoiding low DO prevalent throughout an entire tributary than are larger fish species that may be caught in deeper waters. Thus they are generally present in both low- and high-DO conditions, reducing strong linear correlations among fish density and DO. Results from correlations involving tidal height support the fact that estuarine resident species that utilize the marsh surface during high tide for feeding and predation protection (mummichog, striped killifish, sheepshead minnow) only return to the subtidal waters of the main stem of a creek when the ebbing tide leaves the marsh surface (Kneib and Wagner 1994). Higher tide levels also produce more open-water habitat along hardened shorelines in close connection to the land-water interface, which some abundant species, including bay anchovy, actively utilize.

MDS ordination and cluster analysis of shore-zone fish density data did not indicate strong groupings of fish assemblages by creek, tributary region, or shoreline type. In both 2009 and 2010 ordination results indicated that fish assemblages at individual sampling sites were all highly similar to one another, though idiosyncratic in grouping hierarchy. In 2009 fish assemblages at individual sampling stations were less similar to one another than in 2010, most likely due to limited sampling dates in 2009, allowing for greater variability in abundances from week to week. In 2010, 17 of the 19 sampling stations were at least 75% similar to one another, suggesting fairly equivalent utilization by fish of shore-zone habitats between Indian River and Pepper Creek, among each tributary region, and among all five shoreline types. This high overall similarity of assemblage structures, as well as the lack of discrete clustering by any single organization level implies that there is limited difference in the functional value and habitat quality for resident and transient nekton directly at the shore-zone between Indian River and Pepper Creek, among each tributary region, or among all five shoreline types. Estuarine fish assemblages, as a whole, seem to be utilizing shore-zones quite similarly laterally and longitudinally through these tributaries. However, I have also earlier documented how single species do in fact utilize tributary regions and shoreline types uniquely (through density and length-frequency structure), suggesting that at least some difference exists in the functional value and habitat quality of different shoreline types. So, within highly developed tributaries of Delaware’s Coastal Bays that experience widely pervasive diel-cycling hypoxia, shore-zone fish assemblages are rather homogenous among differing shoreline types.

But at the microhabitat scale of single stretches of shoreline, individual species do exhibit distinct preferences for certain shoreline types.

Results from CCA have provided a synoptic view of fish assemblages within Pepper Creek and Indian River and which environmental variables most affect that assemblage. Though salinity (Able et al. 2001), temperature (Desmond et al. 2002), and DO (Eby and Crowder 2003) have previously been identified as abiotic variables which influence fish assemblage structure, not one of these, or combination of these environmental variables corresponded well with fish assemblage structure in Indian River and Pepper Creek. The variables that most affect fish assemblage were found to be the location of a specific sampling site, the week sampling occurred, and tidal height. For all intents and purposes, these results summarize as: the most predictive variables of fish assemblage are the exact place (i.e. PCL-SP or IRU-BU) and time at which samples were collected. This concept further supports conclusions drawn from MDS ordination suggesting sampling sites to be idiosyncratic in terms of overall fish assemblage structure. Limited associations were found among water quality variables, land use characteristics, and fish assemblage density using CCA. Thus differences in overall seine catches can generally only be identified in retrospection, with little predictive capabilities among shoreline type. This strongly suggests that these estuarine waters are serving as habitat for shore-zone fishes in an almost homogenous sense among shoreline types and that at similar levels of watershed and shoreline development, fish assemblage structure will not differ among shoreline types across a creek.

Bilkovic and Roggero (2008) found an ecological threshold of 23% developed land use within 1000 m of shorelines in Chesapeake Bay where above this development percentage biotic integrity of the estuary was significantly compromised. Results from CCAs performed for this present study confirm that analyzing land use at a 1000 m to a whole watershed scale is more successful than examining only land use directly along shorelines (100 m buffer) when attempting to research land use effects on macrofaunal assemblages. In this present study, the percent of developed land use was very similar between Pepper Creek and Indian River, as well as among tributary regions, all of which had developed land use greater than 23% in the watershed. Given these similarities in land use among tributary regions, it is unsurprising that this study did not find significant correlations or relationships of fish abundance/density and land use composition. Potentially, because both tributaries have such high development, both within their watersheds as well as directly along their shorelines, fish assemblages have been shifted on the scale of the entire tributary, so only minimal differences can be seen at the microhabitat (single stretch of shoreline) scale. Though the differences in biotic integrity between developed and undeveloped estuarine creeks are well documented, future research should concentrate on how fish utilization of single stretches of hardened and unhardened shoreline differ between tributaries with large differences in development.

While extensive spatiotemporal sampling was conducted in this research, a few limitations of this analysis should be considered. As the sampling season only extended from August through September in 2009 and June through September in 2010, results do not represent an entire year. To address this concern, monthly macrofaunal samples were collected from October through December in 2010 to determine if frequent sampling during late fall/early winter would provide insight into macrofaunal assemblages during these times. Very limited differences were found in fish assemblage structure during October and November sampling; primarily reduced fish abundance/density and reduced species richness of estuarine transient fish species. Sampling in early December found total species richness in Indian River and Pepper Creek to be one (Atlantic silverside) with abundances well over an order of magnitude reduced from summer and fall collections. These results suggest that a lack of year-round sampling may not be of great concern for fish studies in the Mid-Atlantic Bight estuaries, as most resident and transient fish species are found in high abundances in the spring through fall when perceived nursery functions such as warm waters for fast growth and plentiful forage occur (Able and Fahay 1998).

A secondary limitation in this study is the restriction of sampling to daylight hours. A prior study in similar marsh creek habitats found significant diel differences in marsh creek assemblages (Rountree and Able 1993). Diel sampling should be a consideration for future shore-zone sampling, particularly when considering the different cover and land-water interface depth differences among shoreline types. It should be reiterated that this study is only from a summer and a half of sampling, and that annual variation in occurrence and abundance of estuarine resident and transient fish species can be quite variable (Able and Fahay 1998).

Blue Crabs

Blue crabs comprised a large proportion of macrofauna captured in this study. Blue crabs use the shore-zone and adjacent subtidal areas of estuarine marshes as juvenile nursery areas and adult habitat and feeding areas (Weinstein 1979, Orth and van Montfrans 1987, Wilson et al. 1990, Rountree and Able 1992). Blue crab abundance and density differed among shoreline types during both years of the study.

Generally, abundances were lower along hardened shorelines than unhardened shorelines. Low variability within shoreline types among tributary regions was found, suggesting a strong relationship of blue crab abundance/density and shoreline type. Also, statistically significant differences in crab abundance and density by shoreline type were found. This result is similar to recent findings from the Chesapeake Bay where blue crab abundance was found to be greater (though not significantly) along natural shorelines than along riprap and bulkhead shorelines (Seitz et al. 2006). Carrol (2003) reported abundance of juvenile crabs were lower in riprap structures compared to natural shorelines. Likewise, crabs were least abundant along bulkhead and riprap shorelines and most abundant along marsh habitats in the Gulf of Mexico shore-zone (Peterson et al. 2000). This present study worked to further synthesize this prior research by examining five different shoreline types (as opposed to three each) over a long sampling period.

Over the course of a summer, juvenile and young crabs migrate into shallower, less-saline waters in upper estuaries where they then grow and mature, while mature crabs are generally found in more saline waters (Fischler and Walburg 1962). Greater relative abundances of smaller (20 mm-60 mm), younger crabs were found in the upper section of each tributary. The relative abundance of blue crabs followed a generally similar pattern across all five shoreline types. Riprap shorelines had slightly greater relative abundance of crabs between 70 mm and 100 mm than other shoreline types while beach and *P. australis* shorelines had slightly greater relative abundances of blue crabs between 10 and 30 mm than other shoreline types. Neither of these differences is striking enough in their magnitude or consistency to suggest true differences in habitat utilization among shoreline types by blue crabs of different size/age classes.

Mean blue crab density was over two-fold greater when DO was < 2.3 mg O_2/l than when it was ≥ 2.3 mg O_2/l. Conversely, results from a Spearman rank correlation found no (0.00) association between blue crab density and DO. Blue crab are negatively affected by low DO, however, this present research suggests that even severe diel-cycling hypoxia will not act to reduce blue crab density on a short time scale (Taylor and Eggleston 2000). One potential reason for increased crab density during low-DO events would be a general increase in blue crab density over the course of the summer as more juveniles migrate up-creek in direct association with a general increase in the magnitude of diel-cycling hypoxia over the summer. This overall result suggests that estuarine systems with widely pervasive diel-cycling hypoxia will not see large swings in blue crab abundance coincide with hourly changes in DO.

Generally low associations (<0.40) were found among environmental variables and blue crab density in nonparametric correlations. Negative associations with salinity further support the theory presented above wherein greater densities of blue crab were found in the less saline waters of the upper regions of each tributary. Blue crab density was also negatively associated with tidal height suggesting that blue crabs actively utilize the marsh surface during high tide, a finding similar to that of Rozas and Reed (1993). This further helps to explain increased blue crab abundance/density along *S. alterniflora* and *P. australis* shorelines as no marsh surface exists along hardened shorelines. Ordination of blue crab density in 2010 found sampling stations to be less idiosyncratic than in fish data. Similarities were more definable by shoreline type suggesting strong associations of blue crab density with shoreline type.

Significant differences exist in the habitat quality provided by differing shoreline types. These differences are exhibited through varying fish densities along each shoreline type. Differences in relative densities of estuary-resident species are particularly informative of the magnitude of the difference in habitat quality among these five shoreline types. Additionally we find small differences in shoreline habitat utilization at differing levels of DO, suggesting a potentially non-linear relationship between shifting DO conditions and shoreline habitat quality.

Table 1.1 Water quality characteristics by tributary region in Indian River and Pepper Creek for summers of 2009 and 2010. See Figure 1.2 for sampling station locations and Materials and Methods section for timing of water quality readings.

			Temperature (°C)		Salinity (psu)		DO (mg O_2/l)	
Creek	Station	Year	Mean	Range	Mean	Range	Mean	Range
Indian River	IRU	2009	25.7	20.1-32.8	12.8	2.5-21.7	5.5	0-18.3
		2010	28.2	20.4-35.6	17.8	8.2-24.8	5.8	0-23.0
	IRL	2009	25.8	19.9-32.2	16.2	7.7-25.3	4.7	0.2-14.5
		2010	28.1	21.0-33.8	22.6	14.7-29.1	6.1	0.1-17.6
	IRLL	2009	25.6	19.9-31.4	21.8	13.7-27.8	5.1	0.1-13.7
		2010	27.6	21.2-32.4	23.8	19.0-28.2	5.6	0.9-10.9
Pepper Creek	PCU	2009	26.1	19.1-32.3	19.0	9.7-25.3	5.0	0-23.5
		2010	28.4	22.3-33.7	22.5	14.2-26.6	5.1	0-18.4
	PCL	2009	25.9	19.3-32.0	20.6	11.9-25.9	5.9	0.3-21.0
		2010	27.8	22.8-33.9	24.0	17.6-27.0	5.6	0.1-17.9
	PCLL	2009	25.6	19.7-31.6	21.9	14.7-27.4	4.9	0-14.2
		2010	27.5	22.5-32.3	23.3	17.7-26.4	5.4	0.2-10.4

Table 1.2 Percentage of water quality measurements in “extreme” ranges from 2009 by tributary region in Indian River and Pepper Creek. See Figure 1.2 for sampling station locations and Materials and Methods section for timing of water quality readings. Data presented is from when data is available simultaneously for each tributary region, thus more data is used for “matched sampling station” section.

	All Matched Data						Matched Sampling Station			
	Indian River			Pepper Creek			Indian River		Pepper Creek	
	Upper	Lower	Mouth	Upper	Lower	Mouth	Upper	Lower	Upper	Lower
Temperature										
% measurements >30°C	13.2	14.0	6.8	11.9	7.2	2.0	11.0	11.7	9.9	6.0
% measurements >31°C	4.6	3.1	1.6	3.0	1.7	0.4	3.8	2.6	2.5	1.4
% measurements >32°C	0.8	0.3	0.0	0.1	0.0	0.0	0.7	0.3	0.1	0.0
% measurements >33°C	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
% measurements >34°C	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dissolved Oxygen										
% measurements <4.8 mg O_2/l	43.6	51.9	49.4	51.7	40.5	53.1	46.0	50.9	51.7	37.1
% measurements <2.3 mg O_2/l	15.8	14.0	9.2	25.4	8.2	10.0	16.8	12.7	25.0	7.3
% measurements <0.4 mg O_2/l	1.0	0.1	0.0	7.9	0.1	1.5	1.9	0.1	6.7	0.1
Dissolved Oxygen (0200-1000)										
% measurements <4.8 mg O_2/l	43.9	51.6	49.7	51.3	40.7	52.8	46.1	50.8	51.8	37.1
% measurements <2.3 mg O_2/l	15.9	14.1	9.2	24.9	8.3	10.0	16.9	12.7	25.0	7.3
% measurements <0.4 mg O_2/l	1.0	0.1	0.0	7.8	0.1	1.5	1.9	0.1	6.7	0.1

Table 1.3 Percentage of water quality measurements in “extreme” ranges from 2010 by tributary region in Indian River and Pepper Creek. See Figure 1.2 for sampling station locations and Materials and Methods section for timing of water quality readings. Data presented is from when data is available simultaneously for each tributary region, thus more data is used for “matched sampling station” section.

	All Matched Data						Matched Sampling Station			
	Indian River			Pepper Creek			Indian River		Pepper Creek	
	Upper	Lower	Mouth	Upper	Lower	Mouth	Upper	Lower	Upper	Lower
Temperature										
% measurements >30°C	30.1	28.8	16.1	31.7	17.9	10.8	27.7	26.1	25.4	16.2
% measurements >31°C	18.1	11.9	5.0	17.7	5.0	1.1	14.5	11.3	12.4	5.3
% measurements >32°C	6.4	2.3	0.5	4.2	1.0	0.2	5.2	2.4	4.0	1.3
% measurements >33°C	1.3	0.0	0.0	0.0	0.0	0.0	1.2	0.3	0.5	0.2
% measurements >34°C	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0
Dissolved Oxygen										
% measurements <4.8 mg O_2/l	61.7	48.5	23.5	58.4	43.9	42.1	45.9	31.1	47.2	37.0
% measurements <2.3 mg O_2/l	28.3	1.6	0.4	22.3	13.5	6.2	20.9	2.1	16.7	7.3
% measurements <0.4 mg O_2/l	3.3	0.2	0.0	2.1	0.0	0.1	2.3	0.1	3.4	0.9
Dissolved Oxygen (0200-1000)										
% measurements <4.8 mg O_2/l	94.6	83.3	55.9	90.4	75.8	81.0	82.4	58.4	83.9	72.6
% measurements <2.3 mg O_2/l	61.6	3.3	1.0	44.7	27.3	15.0	50.0	4.5	35.6	15.9
% measurements <0.4 mg O_2/l	6.1	0.4	0.0	4.6	0.0	0.2	5.2	0.2	7.0	2.1

Table 1.4 Percent landuse type at 100 m, 1000 m, and entire watershed (WS) buffer width of each tributary and tributary region.

Tributary	Tributary Region	Buffer Width	Residential	Commercial	Industrial	Agricultural	Natural
Indian River	IR	100	15.0	3.9	17.1	6.6	57.4
		1000	14.8	6.4	11.7	30.3	36.8
		WS	11.4	3.0	6.4	40.3	38.9
	IRU	100	13.2	16.0	6.0	5.8	59.0
		1000	14.7	17.8	9.9	34.8	22.8
		WS	10.7	3.6	6.7	40.9	38.2
	IRL	100	14.0	9.3	8.3	8.8	59.5
		1000	12.2	11.3	11.9	36.5	28.3
		WS	10.4	3.2	6.3	41.2	38.8
	IRLL	100	13.7	4.1	17.8	6.8	57.6
		1000	13.8	6.6	12.1	29.9	37.6
		WS	10.8	3.0	6.5	40.6	39.1
Pepper Creek	PC	100	27.4	0.6	0.5	6.8	64.7
		1000	20.7	1.7	2.2	35.2	40.3
		WS	8.7	3.3	1.2	45.2	41.6
	PCU	100	37.3	0.0	0.0	10.5	52.2
		1000	19.3	4.0	3.5	46.9	26.3
		WS	8.1	5.8	2.2	50.4	33.6
	PCL	100	31.1	0.0	0.0	7.8	61.1
		1000	21.4	3.2	2.8	41.4	31.3
		WS	8.2	3.5	1.2	45.6	41.5

Table 1.5 Percent impervious surface area within 100 m, 1000 m and entire watershed (WS) buffer of each tributary and tributary region.

Tributary	Tributary Region	Buffer Width	% Impervious Surface Area
Indian River	IR	100	4.1
		1000	8.3
		WS	5.2
	IRU	100	5.7
		1000	13.0
		WS	5.2
	IRL	100	4.6
		1000	11.0
		WS	5.0
	IRLL	100	4.2
		1000	8.4
		WS	5.1
Pepper Creek	PC	100	3.3
		1000	5.6
		WS	4.4
	PCU	100	5.0
		1000	8.6
		WS	5.5
	PCL	100	3.9
		1000	7.6
		WS	4.3

Table 1.6 Natural marsh and shoreline structure percentage within 100 m and 1000 m buffer of each tributary and tributary region.

Tributary	Tributary Region	Buffer	Natural	Bulkhead	Riprap
Indian River	IR	100	71.3	22.3	6.1
		1000	85.5	10.5	2.6
	IRU	100	68.8	19.5	11.6
		1000	91.8	6.1	2.1
	IRL	100	84.3	13.5	2.2
		1000	97.2	1.2	1.6
	IRLL	100	58.0	36.7	4.2
		1000	63.0	27.7	4.6
Pepper Creek	PC	100	68.1	8.2	23.7
		1000	89.6	6.5	2.7
	PCU	100	66.9	7.8	25.4
		1000	89.6	8.4	2.0
	PCL	100	69.7	8.8	21.5
		1000	89.5	4.9	3.3

Table 1.7 Natural marsh and shoreline structure percentage at summarized sampling stations by shoreline type at 100 m and 1000 m buffers.

Shoreline Type	Buffer Width	Natural	Bulkhead	Riprap
S. alterniflora	100	86.4	3.3	10.3
	1000	88.2	8.4	2.4
P. australis	100	100.0	0.0	0.0
	1000	88.2	8.0	2.5
Beach	100	86.4	12.3	1.4
	1000	85.1	10.5	2.7
Riprap	100	45.5	24.7	29.9
	1000	91.9	4.7	2.8
Bulkhead	100	41.9	42.0	20.4
	1000	83.5	12.2	2.8

Table 1.8 Mean abundances (individuals/2 seine hauls) per tributary, total abundance and percent of total catch for all fishes collected in 2009. Species are listed in descending order of percent contribution to the total shore-zone catch in 2009.

Species	Indian River	Pepper Creek	Total Catch	% Catch
All Fish Species (Mean Abundance)	60.7	68.0	5383	100
Menidia menidia (Atlantic silverside)	44.5	45.5	3777	70.2
Anchoa mitchilli (Bay anchovy)	5.5	4.1	408	7.6
Bairdiella chrysoura (Silver perch)	2.1	7.6	394	7.3
Fundulus heteroclitus (Mummichog)	1.6	8.3	393	7.3
Fundulus majalis (Striped killifish)	4.5	0.4	218	4.0
Trinectes maculatus (Hogchoker)	0.8	1.1	78	1.4
Cynoscion regalis (Weakfish)	0.4	0.2	23	0.4
Paralichthys dentatus (Summer flounder)	0.3	0.2	20	0.4
Mugil curema (White mullet)	0.2	0.3	19	0.4
Gobiosoma bosc (Naked goby)	0.2	0.1	12	0.2
Mugil cephalus (Striped mullet)	0.1	0.0	5	0.1
Symphurus plagiusa (Blackcheek tonguefish)	0.1	0.0	5	0.1
Pomatomus salatrix (Bluefish)	0.0	0.1	5	0.1
Menticirrhus saxatilis (Northern kingfish)	0.1	0.0	5	0.1
Leiostomus xanthurus (Spot)	0.0	0.1	5	0.1
Anguilla rostrata (American eel)	0.0	0.1	4	0.1
Alosa sapidissima (American shad)	0.1	0.0	3	0.1
Menticirrhus americanus (Southern kingcroaker)	0.0	0.0	2	<0.1
Pogonias cromis (Black drum)	0.0	0.0	2	<0.1
Caranx hippos (Crevalle jack)	0.0	0.0	2	<0.1
Brevoortia tyrannus (Menhaden)	0.0	0.0	2	<0.1
Opsanus tau (Oyster toadfish)	0.0	0.0	1	<0.1

Table 1.9 Mean abundances (individuals/2 seine hauls) per tributary region, total abundance and percent of total catch for all fishes collected in 2009. Species are listed in descending order of percent contribution to the total shore-zone catch in 2009.

	Indian River			Pepper Creek		Total	
Species	Upper	Lower	Mouth	Upper	Lower	Catch	% Catch
All Fish Species (Mean Abundance)	42.5	62.5	82.4	82.8	44.4	5383	100
Menidia menidia (Atlantic silverside)	19.8	48.8	70.7	57.9	25.6	3777	70.2
Anchoa mitchilli (Bay anchovy)	1.2	7.3	8.2	6.1	1.0	408	7.6
Bairdiella chrysoura (Silver perch)	1.1	3.4	1.4	6.8	8.9	394	7.3
Fundulus heteroclitus (Mummichog)	3.7	0.7	0.0	10.8	4.2	393	7.3
Fundulus majalis (Striped killifish)	13.3	0.1	0.0	0.4	0.5	218	4.0
Trinectes maculatus (Hogchoker)	2.4	0.1	0.0	0.0	2.7	78	1.4
Cynoscion regalis (Weakfish)	0.1	0.4	0.6	0.1	0.3	23	0.4
Paralichthys dentatus (Summer flounder)	0.0	0.5	0.2	0.1	0.4	20	0.4
Mugil curema (White mullet)	0.1	0.3	0.0	0.4	0.2	19	0.4
Gobiosoma bosc (Naked goby)	0.2	0.3	0.2	0.0	0.1	12	0.2
Mugil cephalus (Striped mullet)	0.0	0.1	0.4	0.0	0.0	5	0.1
Symphurus plagiusa (Blackcheek tonguefish)	0.0	0.2	0.1	0.0	0.0	5	0.1
Pomatomus salatrix (Bluefish)	0.0	0.1	0.0	0.1	0.0	5	0.1
Menticirrhus saxatilis (Northern kingfish)	0.0	0.0	0.5	0.0	0.0	5	0.1
Leiostomus xanthurus (Spot)	0.0	0.1	0.0	0.0	0.2	5	0.1
Anguilla rostrata (American eel)	0.1	0.1	0.0	0.0	0.1	4	0.1
Alosa sapidissima (American shad)	0.2	0.0	0.0	0.0	0.0	3	0.1
Menticirrhus americanus (Southern kingcroaker)	0.0	0.0	0.2	0.0	0.0	2	<0.1
Pogonias cromis (Black drum)	0.1	0.1	0.0	0.0	0.0	2	<0.1
Caranx hippos (Crevalle jack)	0.1	0.0	0.0	0.0	0.0	2	<0.1
Brevoortia tyrannus (Menhaden)	0.1	0.0	0.0	0.0	0.0	2	<0.1
Opsanus tau (Oyster toadfish)	0.0	0.0	0.0	0.0	0.1	1	<0.1

Table 1.10 Mean abundances (individuals/m^2 seine hauls) per shoreline, total abundance and percent of total catch for all fishes collected in 2009. Species are listed in descending order of percent contribution to the total shore-zone catch in 2009.

Species	*S. alterniflora*	*P. australis*	Beach	Riprap	Bulkhead	Total Catch	% Catch
All Fish Species (Mean Abundance)	26.3	29.1	154.1	67.5	36.2	5383	100
Menidia menidia (Atlantic silverside)	10.7	19.1	124.9	43.0	21.6	3777	70.2
Anchoa mitchilli (Bay anchovy)	7.7	3.3	1.7	1.3	9.3	408	7.6
Bairdiella chrysoura (Silver perch)	6.5	2.9	3.7	8.8	1.4	394	7.3
Fundulus heteroclitus (Mummichog)	0.2	0.3	7.9	13.1	1.8	393	7.3
Fundulus majalis (Striped killifish)	0.1	0.3	11.6	0.3	0.0	218	4.0
Trinectes maculatus (Hogchoker)	0.1	1.4	2.8	0.4	0.1	78	1.4
Cynoscion regalis (Weakfish)	0.3	0.7	0.3	0.0	0.2	23	0.4
Paralichthys dentatus (Summer flounder)	0.2	0.2	0.1	0.1	0.6	20	0.4
Mugil curema (White mullet)	0.1	0.0	0.4	0.1	0.5	19	0.4
Gobiosoma bosc (Naked goby)	0.2	0.2	0.3	0.1	0.0	12	0.2
Mugil cephalus (Striped mullet)	0.0	0.3	0.0	0.0	0.1	5	0.1
Symphurus plagiusa (Blackcheek tonguefish)	0.0	0.2	0.1	0.1	0.0	5	0.1
Pomatomus salatrix (Bluefish)	0.1	0.1	0.1	0.1	0.1	5	0.1
Menticirrhus saxatilis (Northern kingfish)	0.0	0.0	0.2	0.0	0.1	5	0.1
Leiostomus xanthurus (Spot)	0.1	0.1	0.0	0.1	0.2	5	0.1
Anguilla rostrata (American eel)	0.1	0.1	0.0	0.0	0.1	4	0.1
Alosa sapidissima (American shad)	0.0	0.0	0.0	0.0	0.2	3	0.1
Menticirrhus americanus (Southern kingcroaker)	0.0	0.0	0.1	0.0	0.0	2	<0.1
Pogonias cromis (Black drum)	0.0	0.0	0.1	0.1	0.0	2	<0.1
Caranx hippos (Crevalle jack)	0.0	0.0	0.1	0.0	0.1	2	<0.1
Brevoortia tyrannus (Menhaden)	0.0	0.0	0.0	0.0	0.0	2	<0.1
Opsanus tau (Oyster toadfish)	0.0	0.0	0.0	0.1	0.0	1	<0.1

Table 1.11 Mean densities (individuals/m^3 water seined) per tributary, total abundance and percent of total catch for all fishes collected in 2010. Species are listed in descending order of percent contribution to the total shore-zone catch in 2010. Significant differences in mean density denoted by superscript letters.

Species	Indian River	Pepper Creek	Total Catch	% Catch
All Fish Species (Mean Density)	42.65^a	14.25^b	96960	100
Fundulus heteroclitus (Mummichog)	33.75	7.99	63277	65.3
Brevoortia tyrannus (Menhaden)	2.93	1.64	11969	12.3
Menidia menidia (Atlantic silverside)	3.58^a	1.97^b	11898	12.3
Mugil cephalus (Striped mullet)	0.24	1.39	3302	3.4
Fundulus majalis (Striped killifish)	0.87	0.56	2217	2.3
Bairdiella chrysoura (Silver perch)	0.38	0.27	1482	1.5
Anchoa mitchilli (Bay anchovy)	0..16	0.22	1160	1.2
Cyprinodontidae variegatus (Sheepshead minnow)	0.55^a	0.00^b	789	0.8
Leiostomus xanthurus (Spot)	0.09	0.10	437	0.5
Cynoscion regalis (Weakfish)	0.01^b	0.04^a	113	0.1
Morone saxatilis (Striped bass)	0.02	0.02	84	0.1
Gobiosoma bosc (Naked goby)	0.03	0.02	73	0.1
Paralichthys dentatus (Summer flounder)	0.01^a	0.01^b	59	0.1
Trinectes maculatus (Hogchoker)	0.01	0.00	20	<0.1
Pomatomus salatrix (Bluefish)	0.00	0.00	16	<0.1
Caranx hippos (Crevalle jack)	0.00	0.00	12	<0.1
Mugil curema (White mullet)	0.00	0.00	10	<0.1
Menticirrhus saxatilis (Northern kingfish)	0.00	0.00	9	<0.1
Chasmodes bosquianus (Striped blenny)	0.00	0.01	8	<0.1
Opsanus tau (Oyster toadfish)	0.00	0.00	8	<0.1
Micropogonias undulatus (Atlantic croaker)	0.00	0.00	7	<0.1
Anguilla rostrata (American eel)	0.00	0.00	6	<0.1
Microgobius thalassinus (Green goby)	0.00	0.00	2	<0.1
Menidia berrylina (Inland silverside)	0.00	0.00	1	<0.1
Pogonias cromis (Black drum)	0.00	0.00	1	<0.1

Table 1.12 Mean densities (individuals/m^3 water seined) per tributary region, total abundance and percent of total catch for all fishes collected in 2010. Species are listed in descending order of percent contribution to the total shore-zone catch in 2010. Significant differences in mean density denoted by superscript letters.

	Indian River		Pepper Creek		Total	
Species	Upper	Lower	Upper	Lower	Catch	% Catch
All Fish Species (Mean Density)	17.81^{ab}	68.13^{a}	17.80^{a}	9.80^{b}	96960	100
Fundulus heteroclitus (Mummichog)	8.30^{a}	59.86^{a}	11.76^{a}	3.26^{b}	63277	65.3
Brevoortia tyrannus (Menhaden)	5.67^{a}	0.11^{b}	1.59^{a}	1.69^{ab}	11969	12.3
Menidia menidia (Atlantic silverside)	0.85^{b}	6.39^{a}	2.49^{ab}	1.32^{ab}	11898	12.3
Mugil cephalus (Striped mullet)	0.42	0.06	0.39	2.65	3302	3.4
Fundulus majalis (Striped killifish)	1.46^{a}	0.26^{b}	0.76^{ab}	0.32^{ab}	2217	2.3
Bairdiella chrysoura (Silver perch)	0.59	0.16	0.24	0.30	1482	1.5
Anchoa mitchilli (Bay anchovy)	0.27	0.04	0.31	0.10	1160	1.2
Cyprinodontidae variegatus (Sheepshead minnow)	0.02^{ab}	1.10^{a}	0.00^{b}	0.00^{ab}	789	0.8
Leiostomus xanthurus (Spot)	0.12	0.06	0.13	0.05	437	0.5
Cynoscion regalis (Weakfish)	0.02^{ab}	0.00^{b}	0.05^{ab}	0.03^{a}	113	0.1
Morone saxatilis (Striped bass)	0.03	0.00	0.04	0.00	84	0.1
Gobiosoma bosc (Naked goby)	0.02	0.05	0.01	0.03	73	0.1
Paralichthys dentatus (Summer flounder)	0.01	0.02	0.01	0.01	59	0.1
Trinectes maculatus (Hogchoker)	0.01	0.00	0.00	0.00	20	<0.1
Pomatomus salatrix (Bluefish)	0.00	0.01	0.00	0.00	16	<0.1
Caranx hippos (Crevalle jack)	0.01	0.00	0.00	0.00	12	<0.1
Mugil curema (White mullet)	0.01	0.00	0.00	0.00	10	<0.1
Menticirrhus saxatilis (Northern kingfish)	0.00	0.00	0.00	0.01	9	<0.1
Chasmodes bosquianus (Striped blenny)	0.00	0.00	0.00	0.01	8	<0.1
Opsanus tau (Oyster toadfish)	0.00	0.00	0.00	0.01	8	<0.1
Micropogonias undulatus (Atlantic croaker)	0.00	0.00	0.00	0.00	7	<0.1
Anguilla rostrata (American eel)	0.00	0.00	0.00	0.00	6	<0.1
Microgobius thalassinus (Green goby)	0.00	0.00	0.00	0.00	2	<0.1
Menidia berrylina (Inland silverside)	0.00	0.00	0.00	0.00	1	<0.1
Pogonias cromis (Black drum)	0.00	0.00	0.00	0.00	1	<0.1

Table 1.13 Mean densities (individuals/m^{3} water seined) per shoreline, total abundance and percent of total catch for all fishes collected in 2010. Species are listed in descending order of percent contribution to the total shore-zone catch in 2010. Significant differences in mean density denoted by superscript letters.

Species	*S. alterniflora*	*P. australis*	Beach	Riprap	Bulkhead	Overall	% Catch
All Fish Species (Mean Density)	91.17^{a}	9.56^{bc}	22.21^{b}	9.82^{bc}	7.34^{c}	96960	100
Fundulus heteroclitus (Mummichog)	81.06^{a}	6.51^{b}	11.16^{a}	2.77^{c}	1.70^{c}	63277	65.3
Brevoortia tyrannus (Menhaden)	3.15^{ab}	0.71^{ab}	2.92^{ab}	0.95^{b}	3.36^{a}	11969	12.3
Menidia menidia (Atlantic silverside)	2.99^{ab}	1.24^{b}	3.32^{ab}	4.72^{a}	1.41^{ab}	11898	12.3
Mugil cephalus (Striped mullet)	0.09	0.09	2.71	0.52	0.32	3302	3.4
Fundulus majalis (Striped killifish)	1.34^{ab}	0.26^{b}	1.54^{a}	0.26^{c}	0.08^{c}	2217	2.3
Bairdiella chrysoura (Silver perch)	0.92^{a}	0.38^{a}	0.12^{b}	0.17^{b}	0.06^{b}	1482	1.5
Anchoa mitchilli (Bay anchovy)	0.05	0.14	0.20	0.25	0.29	1160	1.2
Cyprinodontidae variegatus (Sheepshead minnow)	1.36^{a}	0.00^{b}	0.01^{ab}	0.01^{ab}	0.00^{ab}	789	0.8
Leiostomus xanthurus (Spot)	0.01	0.09	0.10	0.09	0.04	437	0.5
Cynoscion regalis (Weakfish)	0.01	0.03	0.04	0.03	0.02	113	0.1
Morone saxatilis (Striped bass)	0.01	0.03	0.03	0.01	0.02	84	0.1
Gobiosoma bosc (Naked goby)	0.03^{ab}	0.06^{a}	0.02^{ab}	0.00^{b}	0.02^{ab}	73	0.1
Paralichthys dentatus (Summer flounder)	0.02	0.00	0.02	0.01	0.01	59	0.1
Trinectes maculatus (Hogchoker)	0.00	0.01	0.00	0.00	0.00	20	<0.1
Pomatomus salatrix (Bluefish)	0.00	0.00	0.01	0.00	0.00	16	<0.1
Caranx hippos (Crevalle jack)	0.01	0.00	0.01	0.00	0.00	12	<0.1
Mugil curema (White mullet)	0.00	0.00	0.00	0.00	0.00	10	<0.1
Menticirrhus saxatilis (Northern kingfish)	0.00	0.00	0.01	0.00	0.00	9	<0.1
Chasmodes bosquianus (Striped blenny)	0.00	0.00	0.00	0.01	0.00	8	<0.1
Opsanus tau (Oyster toadfish)	0.00	0.00	0.00	0.01	0.00	8	<0.1
Micropogonias undulatus (Atlantic croaker)	0.00	0.00	0.01	0.00	0.00	7	<0.1
Anguilla rostrata (American eel)	0.00	0.00	0.00	0.00	0.00	6	<0.1
Microgobius thalassinus (Green goby)	0.00	0.00	0.00	0.00	0.00	2	<0.1
Menidia berrylina (Inland silverside)	0.00	0.00	0.00	0.00	0.00	1	<0.1
Pogonias cromis (Black drum)	0.00	0.00	0.00	0.00	0.00	1	<0.1

Table 1.14 Fish species richness within each tributary and tributary region in 2009 and 2010.

Tributary	Tributary Region	Year	Species Richness
Indian River	IR	2009	21
		2010	22
	IRU	2009	14
		2010	21
	IRL	2009	16
		2010	20
	IRLL	2009	10
		2010	NA
Pepper Creek	PC	2009	15
		2010	25
	PCU	2009	11
		2010	21
	PCL	2009	13
		2010	23

Table 1.15 Fish species richness within each shoreline type in 2009 and 2010.

Shoreline Type	Year	Species Richness
S. alterniflora	2009	13
	2010	17
P. australis	2009	14
	2010	19
Beach	2009	16
	2010	23
Riprap	2009	14
	2010	21
Bulkhead	2009	15
	2010	19

Table 1.16 Mean fish density (individuals/m^3) of 13 most abundant species and blue crab during differing periods of DO (mg O_2/l) concentrations during the summer of 2010, overall and at each shoreline type individually.

	Overall Mean Density				*S. alterniflora* Mean Density			
	Overall	> 4.8	4.8-2.3	<2.3	Overall	> 4.8	4.8-2.3	<2.3
All Species	29.12	44.69	28.99	18.99	91.0	146.4	95.8	38.9
Fundulus heteroclitus (Mummichog)	21.47	35.23	19.32	12.59	81.1	134.6	78.8	29.9
Menidia menidia (Atlantic silverside)	2.82	3.01	2.59	0.97	1.3	0.6	4.9	0.7
Brevoortia tyrannus (Menhaden)	2.31	2.42	3.90	0.21	3.2	6.6	5.3	0.7
Mugil cephalus (Striped mullet)	0.79	1.44	0.46	0.94	0.1	0.1	0.1	0.1
Fundulus majalis (Striped killifish)	0.72	0.70	0.54	1.62	0.0	0.7	1.0	4.4
Bairdiella chrysoura (Silver perch)	0.32	0.19	0.59	0.16	0.9	0.3	2.4	0.2
Cyprinodon varigatus (Sheepshead minnow)	0.29	0.46	0.36	0.02	1.4	2.0	1.9	0.0
Anchoa mitchilli (Bay anchovy)	0.19	0.20	0.28	0.01	3.0	0.0	0.1	0.0
Leiostomus xanthurus (Spot)	0.09	0.15	0.08	0.04	0.0	0.2	0.1	0.2
Cynoscion regalis (Weakfish)	0.03	0.02	0.01	0.08	0.0	0.0	0.0	0.0
Gobiosoma bosc (Naked goby)	0.02	0.03	0.03	0.04	0.0	0.0	0.1	0.1
Morone saxatilis (Striped bass)	0.02	0.03	0.03	0.00	0.0	0.0	0.0	0.0
Paralichthys dentatus (Summer flounder)	0.01	0.02	0.01	0.01	0.0	0.0	0.0	0.0
Callinectes sapidus (Blue crab)	0.90	0.81	0.76	2.25	0.0	1.1	1.1	2.4

Table 1.16 (Continued)

	P. australis Mean Density				Beach Mean Density			
	Overall	> 4.8	4.8-2.3	<2.3	Overall	> 4.8	4.8-2.3	<2.3
All Species	9.6	11.3	10.9	13.5	22.2	31.7	20.4	16.5
Fundulus heteroclitus (Mummichog)	6.5	6.7	7.5	6.7	11.2	15.4	10.8	11.3
Menidia menidia (Atlantic silverside)	0.3	0.6	1.1	1.2	1.5	4.5	2.4	1.0
Brevoortia tyrannus (Menhaden)	0.7	1.6	0.3	0.2	2.9	1.8	3.3	0.0
Mugil cephalus (Striped mullet)	0.1	0.2	0.0	0.0	2.7	6.3	1.2	0.3
Fundulus majalis (Striped killifish)	0.0	0.1	0.2	0.8	0.0	1.8	1.4	1.4
Bairdiella chrysoura (Silver perch)	0.4	0.4	0.5	0.2	0.1	0.1	0.1	0.1
Cyprinodon varigatus (Sheepshead minnow)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Anchoa mitchilli (Bay anchovy)	1.2	0.3	0.1	0.0	3.3	0.4	0.1	0.0
Leiostomus xanthurus (Spot)	0.1	0.2	0.0	0.0	0.1	0.1	0.1	0.0
Cynoscion regalis (Weakfish)	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.3
Gobiosoma bosc (Naked goby)	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.2
Morone saxatilis (Striped bass)	0.1	0.0	0.0	0.0	0.2	0.1	0.0	0.0
Paralichthys dentatus (Summer flounder)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Callinectes sapidus (Blue crab)	0.0	1.0	1.0	4.2	0.0	1.0	0.9	1.7

Table 1.16 (Continued)

	Riprap Mean Density				Bulkhead Mean Density			
	Overall	> 4.8	4.8-2.3	<2.3	Overall	> 4.8	4.8-2.3	<2.3
All Species	9.8	10.9	10.4	16.7	7.3	5.0	12.8	5.0
Fundulus heteroclitus (Mummichog)	2.8	1.1	3.8	9.0	1.7	2.0	1.6	2.4
Menidia menidia (Atlantic silverside)	0.3	8.1	2.4	0.4	0.1	0.9	2.1	1.6
Brevoortia tyrannus (Menhaden)	0.9	0.7	2.2	0.0	3.4	0.8	7.4	0.0
Mugil cephalus (Striped mullet)	0.5	0.1	0.1	4.0	0.3	0.3	0.6	0.1
Fundulus majalis (Striped killifish)	0.0	0.2	0.2	0.8	0.0	0.2	0.0	0.1
Bairdiella chrysoura (Silver perch)	0.2	0.1	0.2	0.1	0.1	0.0	0.0	0.1
Cyprinodon varigatus (Sheepshead minnow)	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Anchoa mitchilli (Bay anchovy)	4.7	0.1	0.7	0.0	1.4	0.3	0.4	0.0
Leiostomus xanthurus (Spot)	0.1	0.1	0.1	0.0	0.0	0.1	0.1	0.0
Cynoscion regalis (Weakfish)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
Gobiosoma bosc (Naked goby)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Morone saxatilis (Striped bass)	0.2	0.0	0.1	0.0	0.3	0.0	0.0	0.0
Paralichthys dentatus (Summer flounder)	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Callinectes sapidus (Blue crab)	0.0	0.4	0.6	2.3	0.0	0.4	0.4	0.6

Table 1.17 Mean fish standard lengths (mm) by tributary in 2009.

Species	Indian River	Pepper Creek
Menidia menidia (Atlantic silverside)	56.5	56.8
Anchoa mitchilli (Bay anchovy)	48.0	41.8
Bairdiella chrysoura (Silver perch)	60.1	66.1
Fundulus heteroclitus (Mummichog)	53.7	34.5
Fundulus majalis (Striped killifish)	25.9	33.6
Trinectes maculatus (Hogchoker)	27.3	22.2
Cynoscion regalis (Weakfish)	72.8	43.4
Paralichthys dentatus (Summer flounder)	119.3	120.5
Mugil curema (White mullet)	147.7	132.4
Gobiosoma bosc (Naked goby)	24.6	28.0
Mugil cephalus (Striped mullet)	130.4	NA
Symphurus plagiusa (Blackcheek tonguefish)	35.4	NA
Pomatomus salatrix (Bluefish)	102.5	87.3
Menticirrhus saxatilis (Northern kingfish)	73.4	NA
Leiostomus xanthurus (Spot)	147.0	144.8
Anguilla rostrata (American eel)	375.0	545.0
Alosa sapidissima (American shad)	73.3	NA
Menticirrhus americanus (Southern kingcroaker)	49.5	NA
Pogonias cromis (Black drum)	134.5	NA
Caranx hippos (Crevalle jack)	33.0	68.0
Brevoortia tyrannus (Menhaden)	77.5	NA
Opsanus tau (Oyster toadfish)	NA	34.0

Table 1.18 Mean fish standard lengths (mm) by tributary region in 2009.

	Indian River			Pepper Creek	
Species	Upper	Lower	Mouth	Upper	Lower
Menidia menidia (Atlantic silverside)	44.2	57.1	60.5	55.1	57.5
Anchoa mitchilli (Bay anchovy)	37.2	48.9	48.8	42.2	41.5
Bairdiella chrysoura (Silver perch)	71.1	58.4	55.1	73.6	57.8
Fundulus heteroclitus (Mummichog)	50.2	67.7	NA	32.5	42.8
Fundulus majalis (Striped killifish)	25.8	31.0	NA	35.1	31.6
Trinectes maculatus (Hogchoker)	27.4	22.0	NA	NA	22.2
Cynoscion regalis (Weakfish)	79.0	72.8	72.0	35.0	46.8
Paralichthys dentatus (Summer flounder)	NA	123.6	98.0	142.5	113.2
Mugil curema (White mullet)	204.0	138.3	0.0	143.1	103.7
Gobiosoma bosc (Naked goby)	27.7	24.4	20.5	NA	28.0
Mugil cephalus (Striped mullet)	NA	170.0	120.5	NA	NA
Symphurus plagiusa (Blackcheek tonguefish)	NA	36.8	30.0	NA	NA
Pomatomus salatrix (Bluefish)	NA	102.5	NA	87.3	NA
Menticirrhus saxatilis (Northern kingfish)	NA	0.0	73.4	NA	NA
Leiostomus xanthurus (Spot)	NA	147.0	NA	148.0	143.7
Anguilla rostrata (American eel)	190.0	560.0	NA	NA	545.0
Alosa sapidissima (American shad)	73.3	NA	NA	NA	NA
Menticirrhus americanus (Southern kingcroaker)	NA	NA	49.5	NA	NA
Pogonias cromis (Black drum)	128.0	141.0	NA	NA	NA
Caranx hippos (Crevalle jack)	33.0	NA	NA	68.0	NA
Brevoortia tyrannus (Menhaden)	77.5	NA	NA	NA	NA
Opsanus tau (Oyster toadfish)	NA	NA	NA	NA	34.0

Table 1.19 Mean fish standard lengths (mm) by shoreline type in 2009.

Species	*S. alterniflora*	*P. australis*	Beach	Riprap	Bulkhead
Menidia menidia (Atlantic silverside)	59.3	57.0	55.9	58.2	51.9
Anchoa mitchilli (Bay anchovy)	47.1	44.2	51.0	39.3	47.9
Bairdiella chrysoura (Silver perch)	61.6	70.1	55.0	70.9	61.5
Fundulus heteroclitus (Mummichog)	62.8	33.3	42.2	30.4	63.1
Fundulus majalis (Striped killifish)	34.0	29.3	26.3	29.3	NA
Trinectes maculatus (Hogchoker)	38.0	26.9	23.2	27.3	17.5
Cynoscion regalis (Weakfish)	35.8	73.6	66.4	NA	86.7
Paralichthys dentatus (Summer flounder)	137.0	112.7	131.0	146.0	112.6
Mugil curema (White mullet)	135.0	NA	105.0	204.0	154.0
Gobiosoma bosc (Naked goby)	27.8	18.5	27.6	16.0	NA
Mugil cephalus (Striped mullet)	NA	129.3	NA	NA	135.0
Symphurus plagiusa (Blackcheek tonguefish)	NA	36.5	30.0	37.0	NA
Pomatomus salatrix (Bluefish)	79.0	116.0	89.0	94.0	89.0
Menticirrhus saxatilis (Northern kingfish)	NA	NA	82.3	NA	60.0
Leiostomus xanthurus (Spot)	148.0	147.0	NA	77.5	143.7
Anguilla rostrata (American eel)	560.0	190.0	NA	NA	545.0
Alosa sapidissima (American shad)	NA	NA	NA	NA	73.3
Menticirrhus americanus (Southern kingcroaker)	NA	NA	49.5	NA	NA
Pogonias cromis (Black drum)	NA	NA	141.0	128.0	NA
Caranx hippos (Crevalle jack)	NA	NA	33.0	NA	68.0
Brevoortia tyrannus (Menhaden)	NA	NA	NA	NA	NA
Opsanus tau (Oyster toadfish)	NA	NA	NA	34.0	NA

Table 1.20 Mean fish standard lengths (mm) by tributary in 2010.

Species	Indian River	Pepper Creek
Fundulus heteroclitus (Mummichog)	50.0	49.7
Brevoortia tyrannus (Menhaden)	38.2	54.1
Menidia menidia (Atlantic silverside)	50.3	47.4
Mugil cephalus (Striped mullet)	54.4	36.2
Fundulus majalis (Striped killifish)	36.9	45.3
Bairdiella chrysoura (Silver perch)	44.4	45.5
Anchoa mitchilli (Bay anchovy)	41.3	39.5
Cyprinodontidae variegatus (Sheepshead minnow)	41.3	29.5
Leiostomus xanthurus (Spot)	66.6	69.4
Cynoscion regalis (Weakfish)	43.8	43.5
Morone saxatilis (Striped bass)	79.5	73.2
Gobiosoma bosc (Naked goby)	28.1	30.3
Paralichthys dentatus (Summer flounder)	132.7	142.0
Trinectes maculatus (Hogchoker)	27.9	36.4
Pomatomus salatrix (Bluefish)	74.2	75.3
Caranx hippos (Crevalle jack)	45.1	38.7
Mugil curema (White mullet)	109.8	112.5
Menticirrhus saxatilis (Northern kingfish)	90.0	86.7
Chasmodes bosquianus (Striped blenny)	0.0	57.9
Opsanus tau (Oyster toadfish)	35.0	38.0
Micropogonias undulatus (Atlantic croaker)	21.2	14.0
Anguilla rostrata (American eel)	406.7	543.0
Microgobius thalassinus (Green goby)	27.0	29.0
Menidia berrylina (Inland silverside)	0.0	58.0
Pogonias cromis (Black drum)	0.0	143.0

Table 1.21 Mean fish standard lengths (mm) by tributary region in 2010.

	Indian River		Pepper Creek	
Species	Upper	Lower	Upper	Lower
Fundulus heteroclitus (Mummichog)	44.8	50.8	49.5	50.7
Brevoortia tyrannus (Menhaden)	38.0	47.9	41.5	64.9
Menidia menidia (Atlantic silverside)	49.9	50.3	46.9	48.3
Mugil cephalus (Striped mullet)	49.3	94.8	50.5	33.9
Fundulus majalis (Striped killifish)	33.6	58.1	48.9	35.7
Bairdiella chrysoura (Silver perch)	43.6	47.6	49.1	41.3
Anchoa mitchilli (Bay anchovy)	40.4	46.8	38.4	44.2
Cyprinodontidae variegatus (Sheepshead minnow)	30.5	41.5	34.0	25.0
Leiostomus xanthurus (Spot)	63.6	74.9	68.4	72.9
Cynoscion regalis (Weakfish)	45.9	29.3	42.2	47.0
Morone saxatilis (Striped bass)	72.3	157.0	64.6	149.0
Gobiosoma bosc (Naked goby)	26.2	28.9	28.7	30.9
Paralichthys dentatus (Summer flounder)	136.2	130.7	152.9	130.2
Trinectes maculatus (Hogchoker)	27.6	29.5	38.3	29.0
Pomatomus salatrix (Bluefish)	70.8	75.8	79.0	68.0
Caranx hippos (Crevalle jack)	41.9	56.5	NA	38.7
Mugil curema (White mullet)	109.8	NA	95.0	130.0
Menticirrhus saxatilis (Northern kingfish)	96.0	87.0	87.5	86.3
Chasmodes bosquianus (Striped blenny)	NA	NA	NA	57.9
Opsanus tau (Oyster toadfish)	NA	35.0	37.0	38.2
Micropogonias undulatus (Atlantic croaker)	21.5	23.0	14.0	NA
Anguilla rostrata (American eel)	322.5	575.0	524.5	580.0
Microgobius thalassinus (Green goby)	27.0	NA	NA	29.0
Menidia berrylina (Inland silverside)	NA	NA	58.0	NA
Pogonias cromis (Black drum)	NA	NA	NA	143.0

Table 1.22 Mean fish length by shoreline type in 2010.

Species	*S. alterniflora*	*P. australis*	Beach	Riprap	Bulkhead
Fundulus heteroclitus (Mummichog)	49.0	49.4	54.3	48.3	57.1
Brevoortia tyrannus (Menhaden)	38.0	74.0	37.9	33.7	49.4
Menidia menidia (Atlantic silverside)	48.4	49.2	47.1	51.2	47.9
Mugil cephalus (Striped mullet)	73.0	62.2	34.4	49.2	52.8
Fundulus majalis (Striped killifish)	39.8	36.3	40.5	39.1	54.0
Bairdiella chrysoura (Silver perch)	42.8	49.7	44.5	48.2	43.8
Anchoa mitchilli (Bay anchovy)	40.9	45.0	40.1	39.3	39.7
Cyprinodontidae variegatus (Sheepshead minnow)	41.3	NA	41.8	33.0	37.5
Leiostomus xanthurus (Spot)	63.8	54.9	72.0	74.8	77.1
Cynoscion regalis (Weakfish)	53.5	41.1	49.2	39.7	41.1
Morone saxatilis (Striped bass)	47.8	69.9	62.5	96.1	88.6
Gobiosoma bosc (Naked goby)	29.1	30.1	25.6	33.0	27.0
Paralichthys dentatus (Summer flounder)	146.2	145.0	124.2	148.7	128.1
Trinectes maculatus (Hogchoker)	20.0	30.1	19.3	24.3	47.0
Pomatomus salatrix (Bluefish)	NA	85.0	63.8	79.0	75.3
Caranx hippos (Crevalle jack)	42.0	41.0	45.8	NA	NA
Mugil curema (White mullet)	100.0	120.5	125.7	95.0	95.0
Menticirrhus saxatilis (Northern kingfish)	84.3	96.0	86.3	NA	96.0
Chasmodes bosquianus (Striped blenny)	NA	NA	65.0	56.9	0.0
Opsanus tau (Oyster toadfish)	NA	NA	NA	37.7	37.0
Micropogonias undulatus (Atlantic croaker)	NA	14.0	20.0	NA	NA
Anguilla rostrata (American eel)	NA	499.0	580.0	398.3	575.0
Microgobius thalassinus (Green goby)	NA	NA	27.0	29.0	NA
Menidia berrylina (Inland silverside)	NA	NA	NA	58.0	NA
Pogonias cromis (Black drum)	NA	NA	143.0	NA	NA

Table 1.23 Diversity Index Scores for tributaries and tributary regions in 2009 and 2010.

Tributary	Tributary Region	Year	Margalef's Richness Index	Shannon Diversity Index	Pielou's Evenness Index
Indian River	IR	2009	2.53	1.07	0.81
		2010	1.88	1.00	0.75
	IRU	2009	2.01	1.41	1.23
		2010	2.03	1.44	1.09
	IRL	2009	2.12	0.85	0.70
		2010	1.75	0.55	0.43
	IRLL	2009	1.32	0.57	0.57
		2010	NA	NA	NA
Pepper Creek	PC	2009	1.78	1.13	0.96
		2010	2.36	1.59	1.14
	PCU	2009	1.32	0.99	0.95
		2010	2.05	1.36	1.03
	PCL	2009	1.85	1.34	1.20
		2010	2.43	1.70	1.25

Table 1.24 Diversity Index Scores among shoreline types in 2009 and 2010.

Shoreline Type	Year	Margalef's Richness Index	Shannon Diversity Index	Pielou's Evenness Index
S. alterniflora	2009	1.93	1.34	1.2
	2010	1.46	0.63	0.51
P. australis	2009	2.19	1.27	1.11
	2010	2.11	1.33	1.04
Beach	2009	1.89	0.79	0.66
	2010	2.16	1.51	1.12
Riprap	2009	1.86	1.06	0.92
	2010	2.17	1.51	1.14
Bulkhead	2009	2.16	1.22	1.04
	2010	1.86	1.39	1.1

Table 1.25 Results of Spearman rank correlation of selected environmental variables on the density of abundant fish species and blue crabs. Correlations in bold are significant at $p<0.0033$ (Bonferroni correction α).

Species	Temperature	Salinity	Dissolved Oxygen	Tidal Height
Total Fish Density	**0.28**	**-0.18**	0.09	**-0.40**
Fundulus heteroclitus (Mummichog)	**0.32**	**-0.19**	0.06	**-0.53**
Brevoortia tyrannus (Menhaden)	0.11	-0.04	**0.22**	0.10
Menidia menidia (Atlantic silverside)	-0.03	0.10	-0.07	0.02
Mugil cephalus (Striped mullet)	**0.31**	0.03	0.13	-0.06
Fundulus majalis (Striped killifish)	0.14	-0.16	-0.05	**-0.21**
Bairdiella chrysoura (Silver perch)	**-0.28**	0.12	**-0.19**	0.13
Anchoa mitchilli (Bay anchovy)	-0.05	0.14	0.04	**0.18**
Cyprinodon variegatus (Sheepshead minnow)	0.13	-0.13	-0.02	-0.12
Leiostomus xanthurus (Spot)	-0.04	0.00	0.16	0.01
Cynoscion regalis (Weakfish)	0.16	0.09	0.07	0.15
Morone saxatilis (Striped bass)	0.11	0.13	0.03	0.45
Gobiosoma bosc (Naked goby)	0.14	0.07	0.09	**-0.24**
Paralichthys dentatus (Summer flounder)	-0.10	0.04	0.15	0.02
Callinectes sapidus (Blue crab)	0.08	**-0.19**	0.00	**-0.36**

Table 1.26 Canonical correspondence analysis (CCA) model summary for abundant fish and blue crab densities. Model fit by iterative model building, predictor factors include sampling station and sampling week, predictor vector is tidal height.

Canonical Variate	Eigenvalue	Proportion (variance explained)	Cumulative proportion (variance explained)	Species-Environment Correlation	Significance	Total inertia	Inertia explained by constrained axes
1	0.608	0.317	0.317	0.873	0.005	3.617	1.914
2	0.536	0.28	0.597	0.860	0.005		
3	0.444	0.232	0.829	0.802	0.005		
4	0.127	0.066	0.896	0.573	0.033		

Table 1.27 Canonical correspondence analysis (CCA) model summary for abundant fish and blue crab densities. Model fit by iterative model building using only water quality vectors, predictor vectors include water temperature, salinity and DO.

Canonical Variate	Eigenvalue	Proportion (variance explained)	Cumulative proportion (variance explained)	Species-Environment Correlation	Significance	Total inertia	Inertia explained by constrained axes
1	0.141	0.039	0.039	0.430	0.29	3.639	0.284
2	0.124	0.034	0.073	0.395	0.16		
3	0.019	0.005	0.078	0.203	0.83		

Table 1.28 Canonical correspondence analysis (CCA) model summary for abundant fish and blue crab densities. Model fit by iterative model building using only land use data at a 100 m buffer, predictor vectors include near shore landuse development proportion within 100 m, impervious surface area within 100 m and riprap length within 100 m.

Canonical Variate	**Eigenvalue**	**Proportion (variance explained)**	**Cumulative proportion (variance explained)**	**Species-Environment Correlation**	**Significance**	**Total inertia**	**Inertia explained by constrained axes**
1	0.304	0.084	0.084	0.594	0.005	3.617	0.4358
2	0.116	0.032	0.116	0.431	0.013		
3	0.016	0.004	0.120	0.232	0.670		

Table 1.29 Canonical correspondence analysis (CCA) model summary for abundant fish and blue crab densities. Model fit by iterative model building using only land use data at a 1000 m and entire watershed buffer, predictor vectors include Bulkhead length within 1000 m of sampling station, riprap length within 1000 m of sampling station percent developed land use within sampling station watershed, percent impervious surface area within watershed, percent of near shore landuse developed within 1000 m buffer of sampling station.

Canonical Variate	**Eigenvalue**	**Proportion (variance explained)**	**Cumulative proportion (variance explained)**	**Species-Environment Correlation**	**Significance**	**Total inertia**	**Inertia explained by constrained axes**
1	0.295	0.082	0.082	0.607	0.010	3.617	0.494
2	0.170	0.047	0.129	0.512	0.020		
3	0.017	0.005	0.133	0.131	0.990		
4	0.006	0.002	0.135	0.149	1.000		

Table 1.30 Mean blue crab abundances (individuals/2 seine hauls) collected during 2009 per tributary.

	Indian River	Pepper Creek
Callinectes sapidus (Blue Crab)	2.62	1.38

Table 1.31 Mean blue crab abundances (individuals/2 seine hauls) collected during 2009 per tributary region.

	Indian River			Pepper Creek	
	Upper	Lower	Mouth	Upper	Lower
Callinectes sapidus (Blue Crab)	1.93	3.05	2.82	1.83	0.67

Table 1.32 Mean blue crab abundances (individuals/2 seine hauls) collected during 2009 per shoreline type.

	S. alterniflora	*P. australis*	Beach	Riprap	Bulkhead
Callinectes sapidus (Blue Crab)	1.84	3.00	3.17	0.81	1.56

Table 1.33 Mean blue crab abundances (individuals/2 seine hauls) collected during 2010 per tributary.

	Indian River	Pepper Creek
Callinectes sapidus (Blue Crab)	12.23	10.75

Table 1.34 Mean blue crab abundances (individuals/2 seine hauls) collected during 2010 per tributary region. Significant differences in mean abundance denoted by superscript letters.

	Indian River		Pepper Creek	
	Upper	Lower	Upper	Lower
Callinectes sapidus (Blue Crab)	17.43^{a}	6.90^{b}	12.87^{ab}	8.10^{b}

Table 1.35 Mean blue crab abundances (individuals/2 seine hauls) collected during 2010 per shoreline type. Significant differences in mean abundance denoted by superscript letters.

	S. alterniflora	*P. australis*	Beach	Riprap	Bulkhead
Callinectes sapidus (Blue Crab)	13.57^{ab}	15.93^{a}	10.80^{ab}	10.02^{ab}	8.49^{b}

Table 1.36 Mean blue crab densities (individuals/l^{3}) collected during 2010 per tributary.

	Indian River	Pepper Creek
Callinectes sapidus (Blue Crab)	0.98	0.82

Table 1.37 Mean blue crab densities (individuals/l^3) collected during 2010 per tributary region. Significant differences in mean density denoted by superscript letters.

	Indian River		Pepper Creek	
	Upper	Lower	Upper	Lower
Callinectes sapidus (Blue Crab)	1.40^{a}	0.54^{b}	0.96^{ab}	0.65^{b}

Table 1.38 Mean blue crab densities (individuals/l^3) collected during 2010 per shoreline type. Significant differences in mean density denoted by superscript letters.

	S. alterniflora	*P. australis*	Beach	Riprap	Bulkhead
Callinectes sapidus (Blue Crab)	1.21^{a}	1.39^{a}	0.97^{a}	0.66^{b}	0.41^{b}

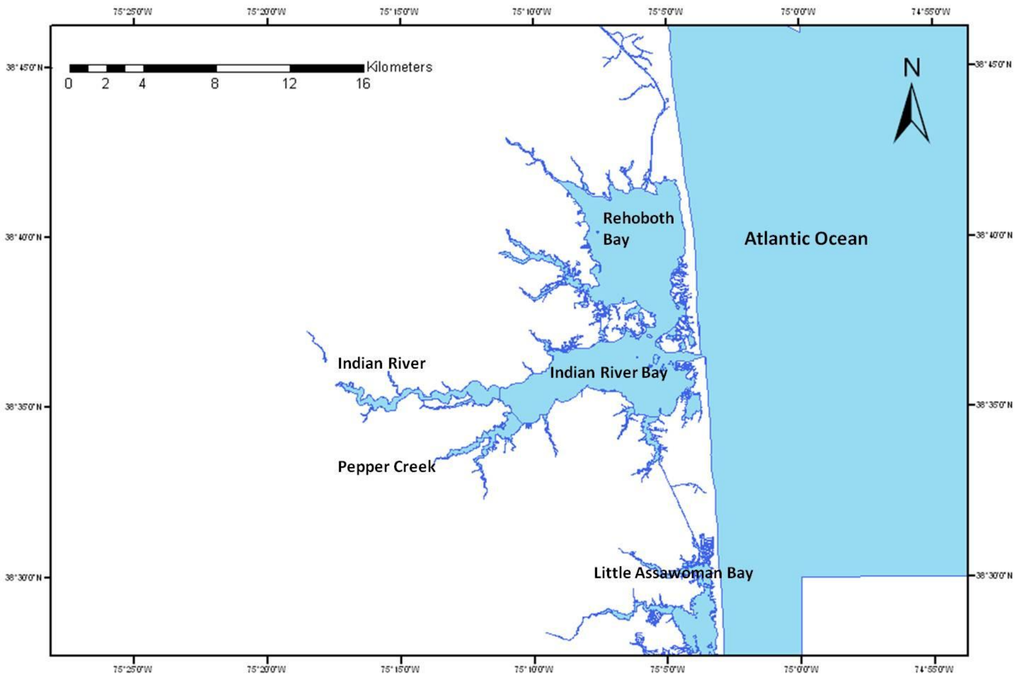

Figure 1.1 Delaware's Coastal Bays (USA) with location of Indian River and Pepper Creek.

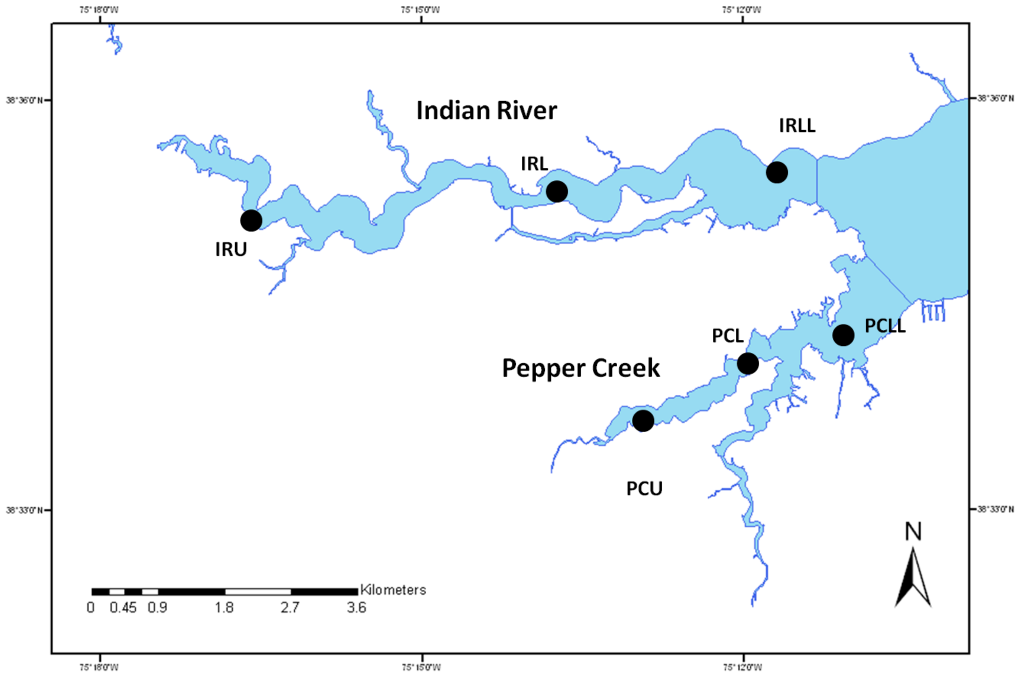

Figure 1.2 Indian River and Pepper Creek, tributaries off of Indian River Bay, Delaware Coastal Bays with general locations of general macrofaunal sampling stations at each tributary region, upper Indian River (IRU), lower Indian River (IRL), mouth of Indian River (IRLL) (2009), upper Pepper Creek (PCU), and lower Pepper Creek (PCL). These are also the location of multi-parameter sondes in 2009 and 2010 with the addition of the mouth of Pepper Creek (PCLL).

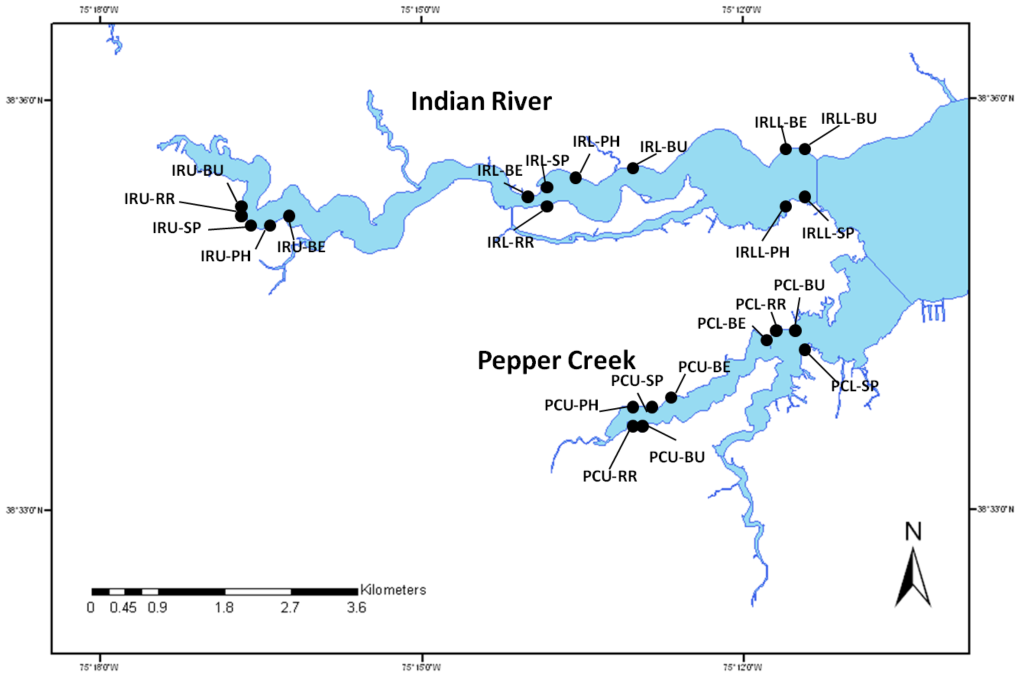

Figure 1.3 Indian River and Pepper Creek, tributaries off of Indian River Bay, Delaware Coastal Bays with specific locations of macrofaunal sampling stations. First part of codes describe tributary region, lower Pepper Creek (PCL), upper Pepper Creek (PCU), mouth of Indian River (IRLL), lower Indian River (IRL), and upper Indian River (IRU). Second part of codes describe shoreline type *S. alterniflora* marsh (SP), *P. australis* marsh (PH), sandy beach (BU), riprap (RR), and bulkhead (BU).

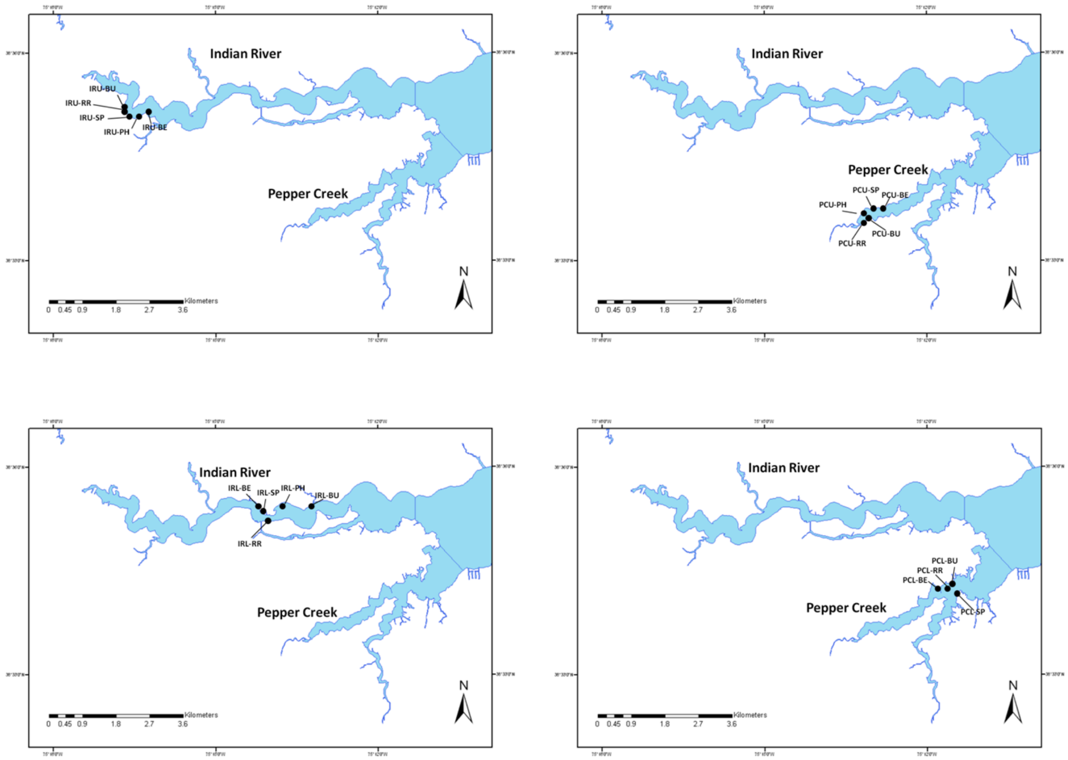

Figure 1.4 Locations of multi-parameter sondes for shoreline water quality monitoring during summer of 2010. Beginning in the top right frame and working clockwise, PCU locations in field from 7/15/2010 through 7/28/2010, PCL locations in field from 7/30/2010 through 8/11/2010, IRL locations in field from 8/12/2010 through 8/25/2010, IRU locations in field from 8/26/2010 through 9/9/2010.

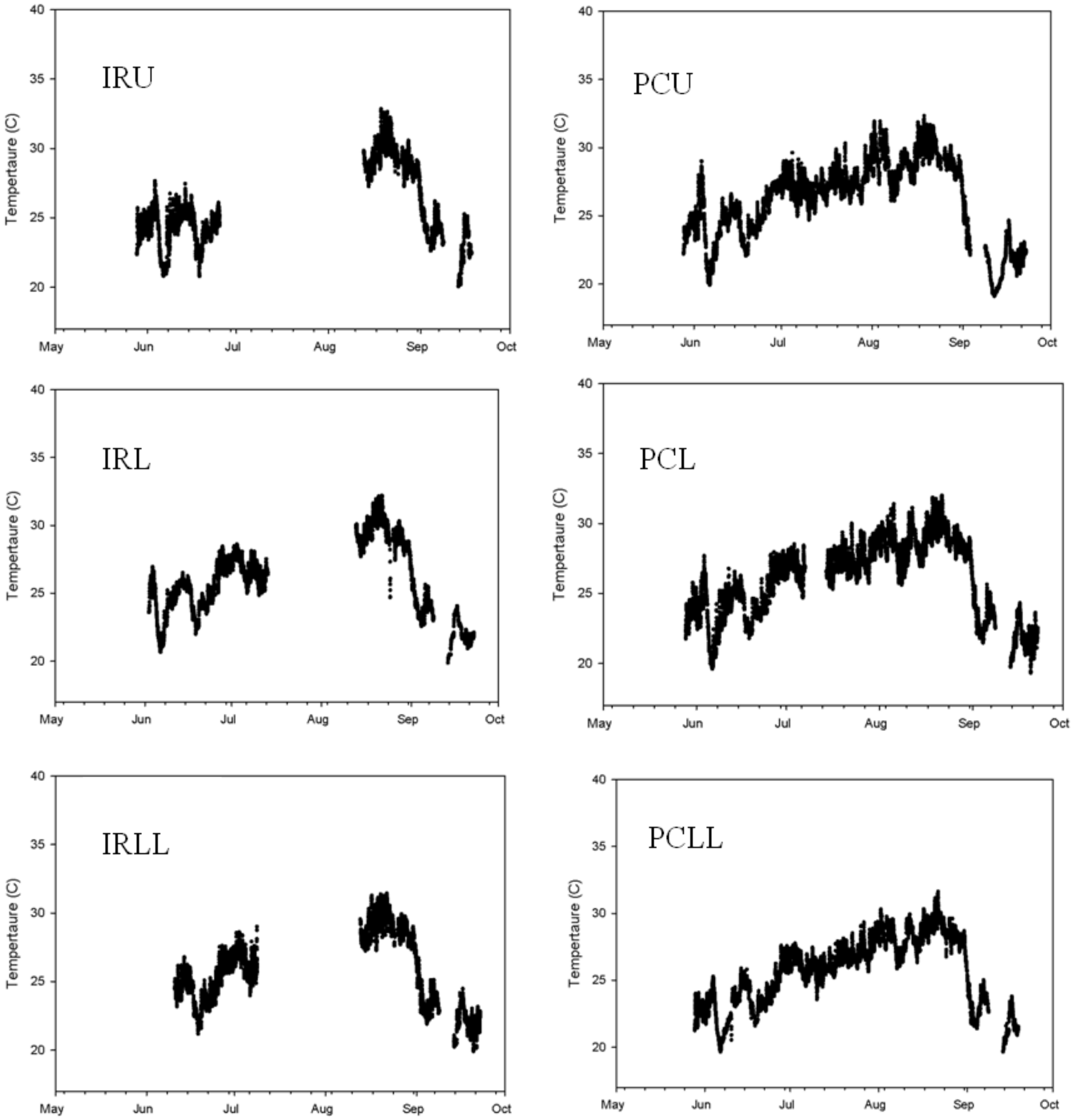

Figure 1.5 Water temperature (°C), data from Indian River and Pepper Creek in 2009.

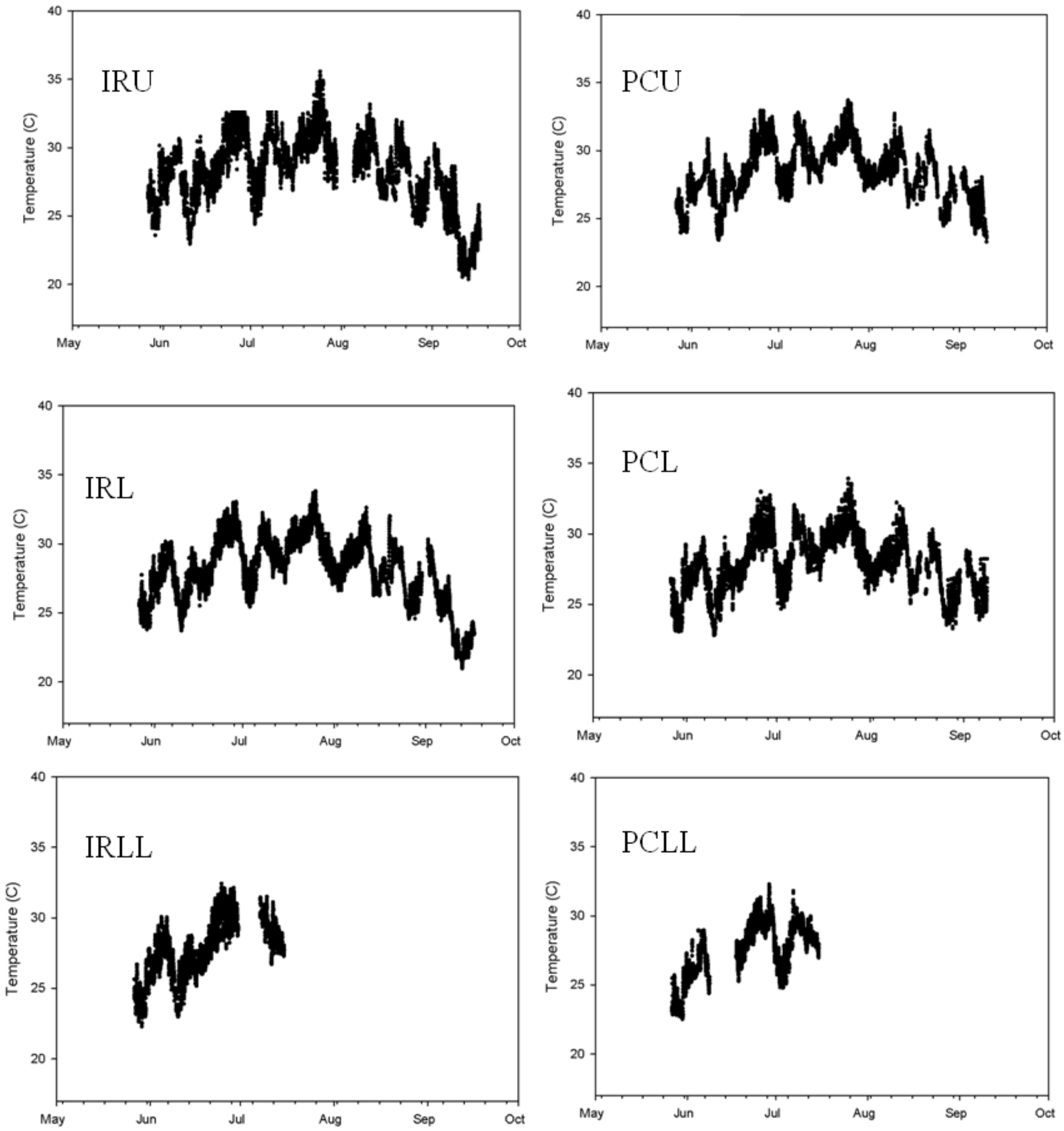

Figure 1.6 Water temperature (°C), data from Indian River and Pepper Creek in 2010.

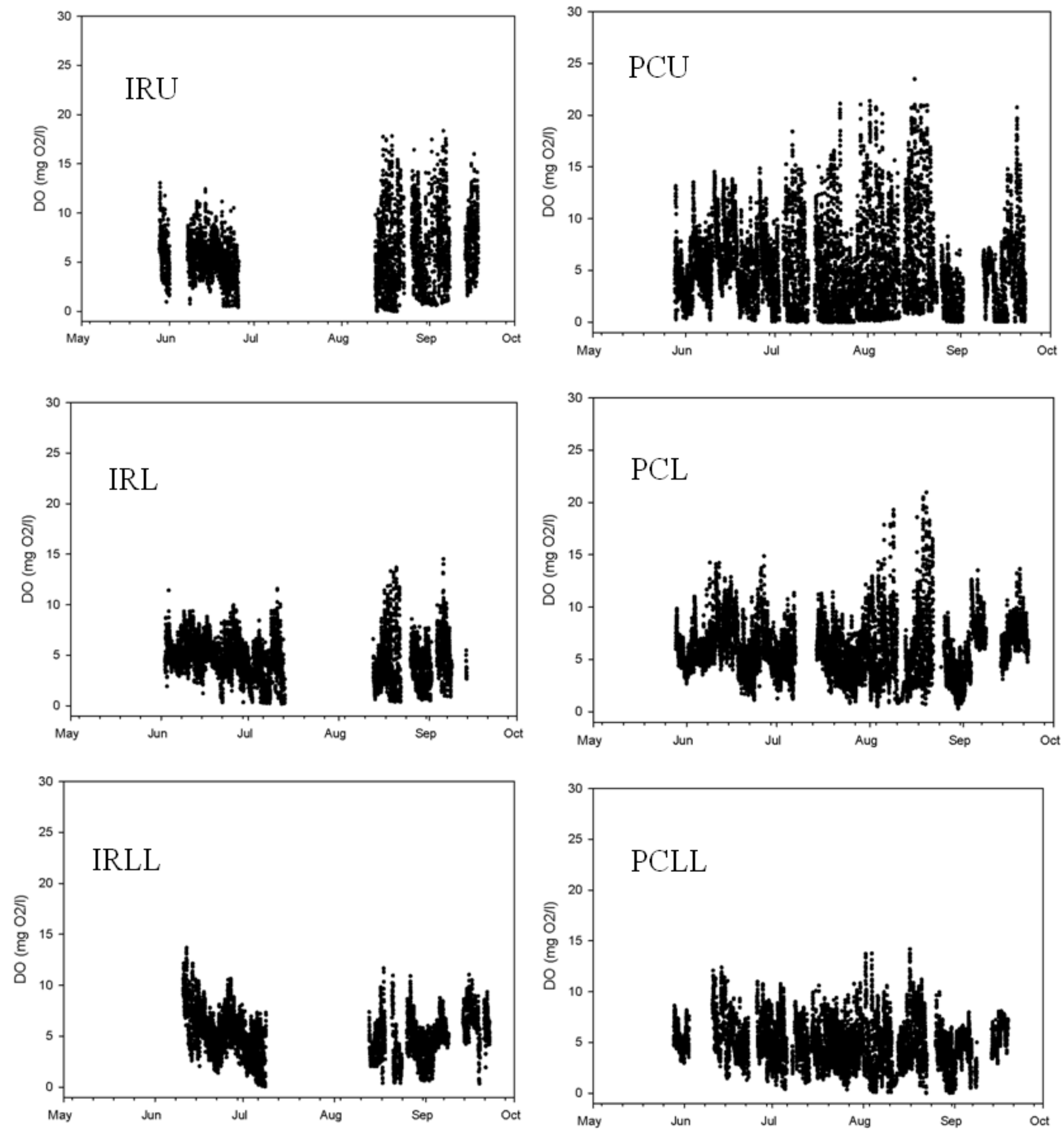

Figure 1.7 Dissolved Oxygen (mg O_2/l) data from Indian River and Pepper Creek in 2009.

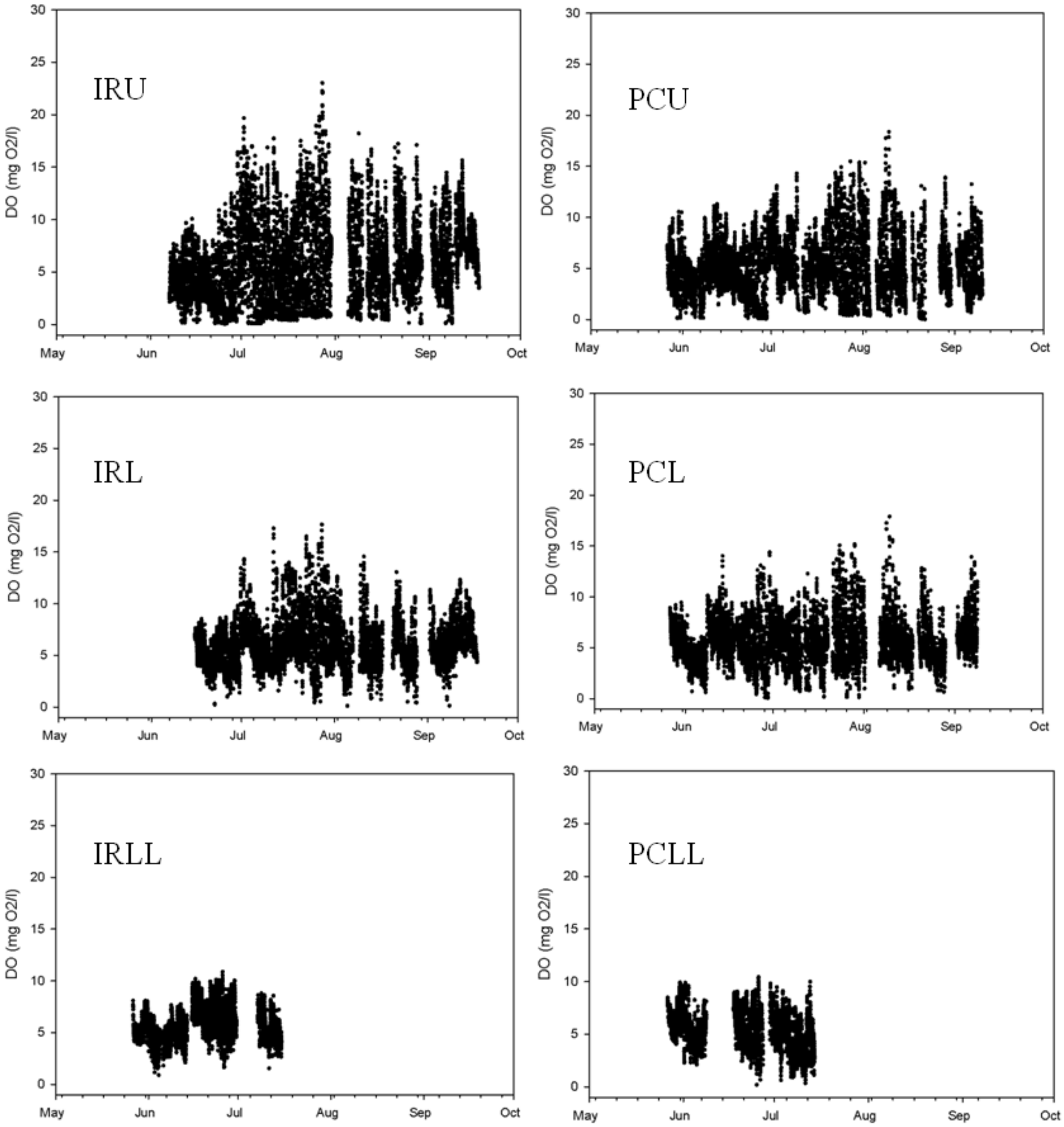

Figure 1.8 Dissolved Oxygen (mg O_2/l) data from Indian River and Pepper Creek in 2010.

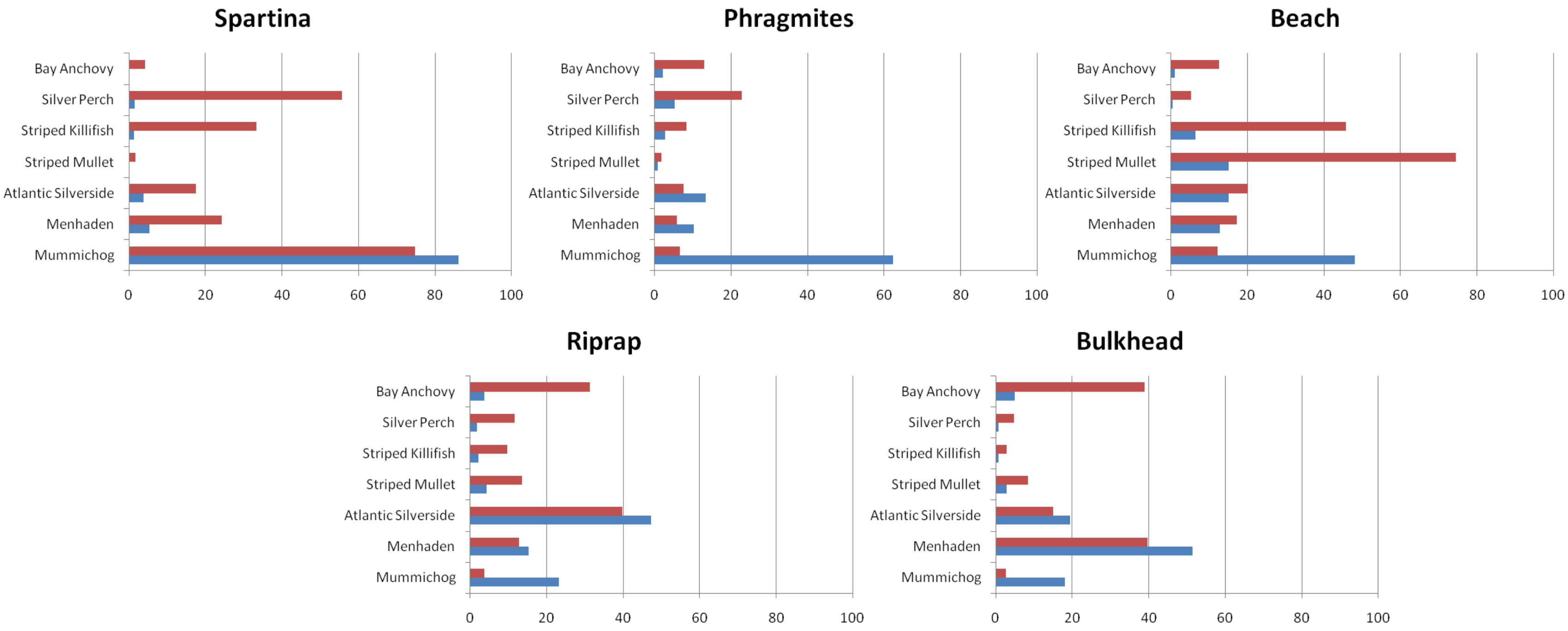

Figure 1.9 Relative densities of most abundant fish species by shoreline type. x axis represents relative density percent. Blue bars represent the percentage of the total catch at that shoreline type which that species composed. So, the combined blue bars within each shoreline type will sum to around 95%, as all other species are not included. Large blue bars represent a species which is particularly prevalent in comparison with other species at that shoreline type. Red bars represent the percentage of that species total catch (across all shoreline types) which was caught along that shoreline type. Large red bars represent habitat at which that species is particularly abundant. If species were equally abundant along all five shoreline types, all red bars would be 20%.

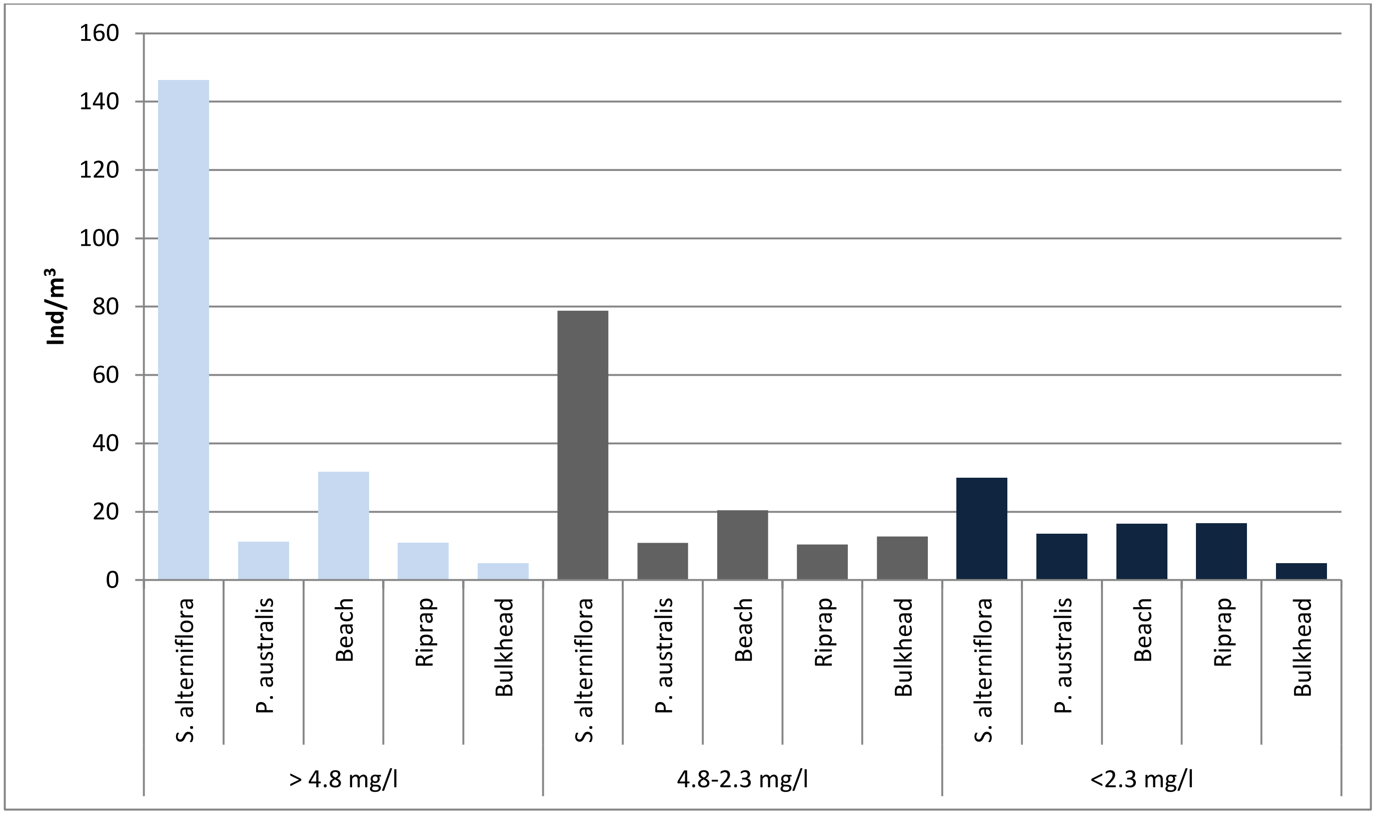

Figure 1.10 Mean fish densities along each shoreline during differing periods of DO (see table 1.16 for further detail).

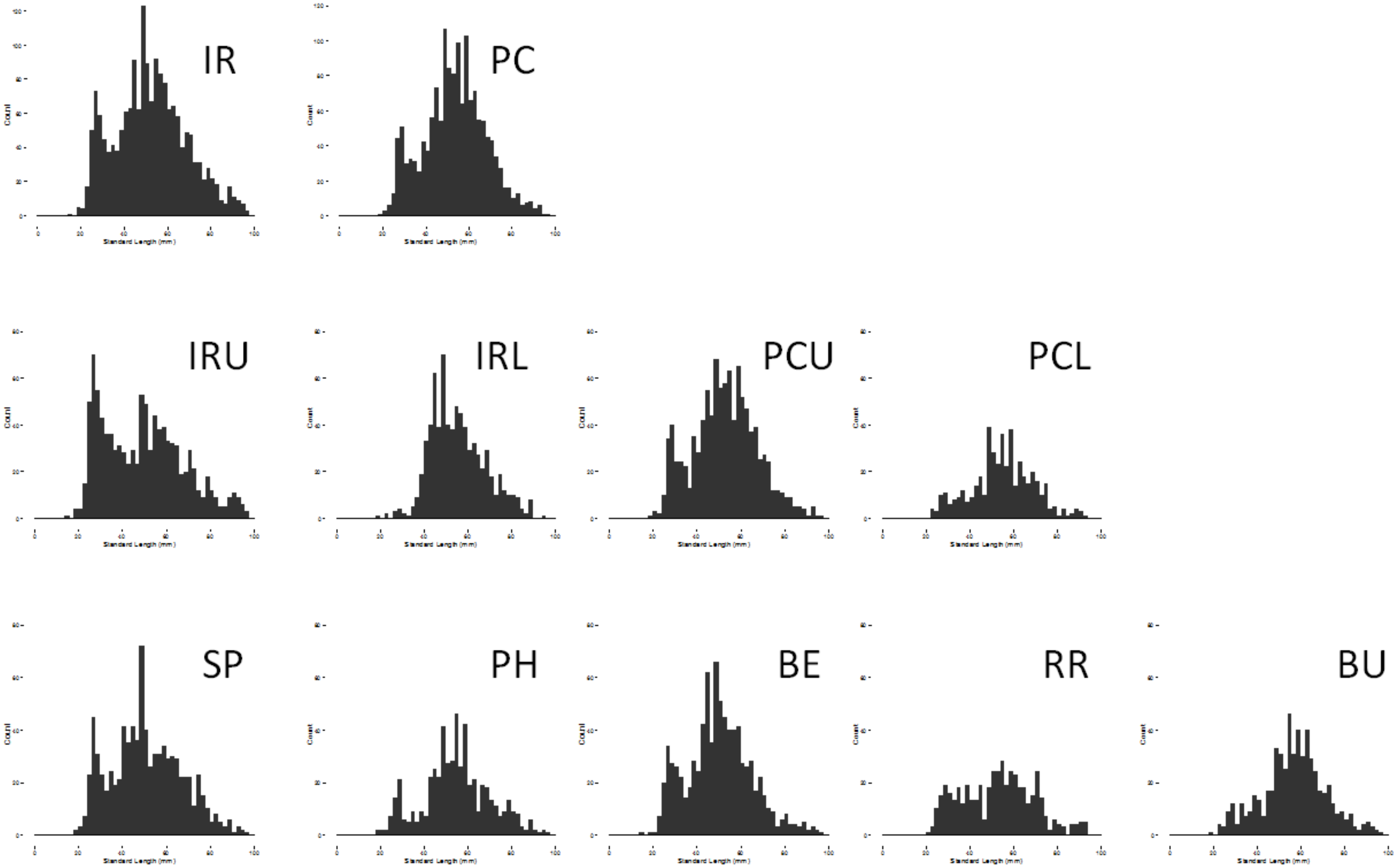

Figure 1.11   Mummichog length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

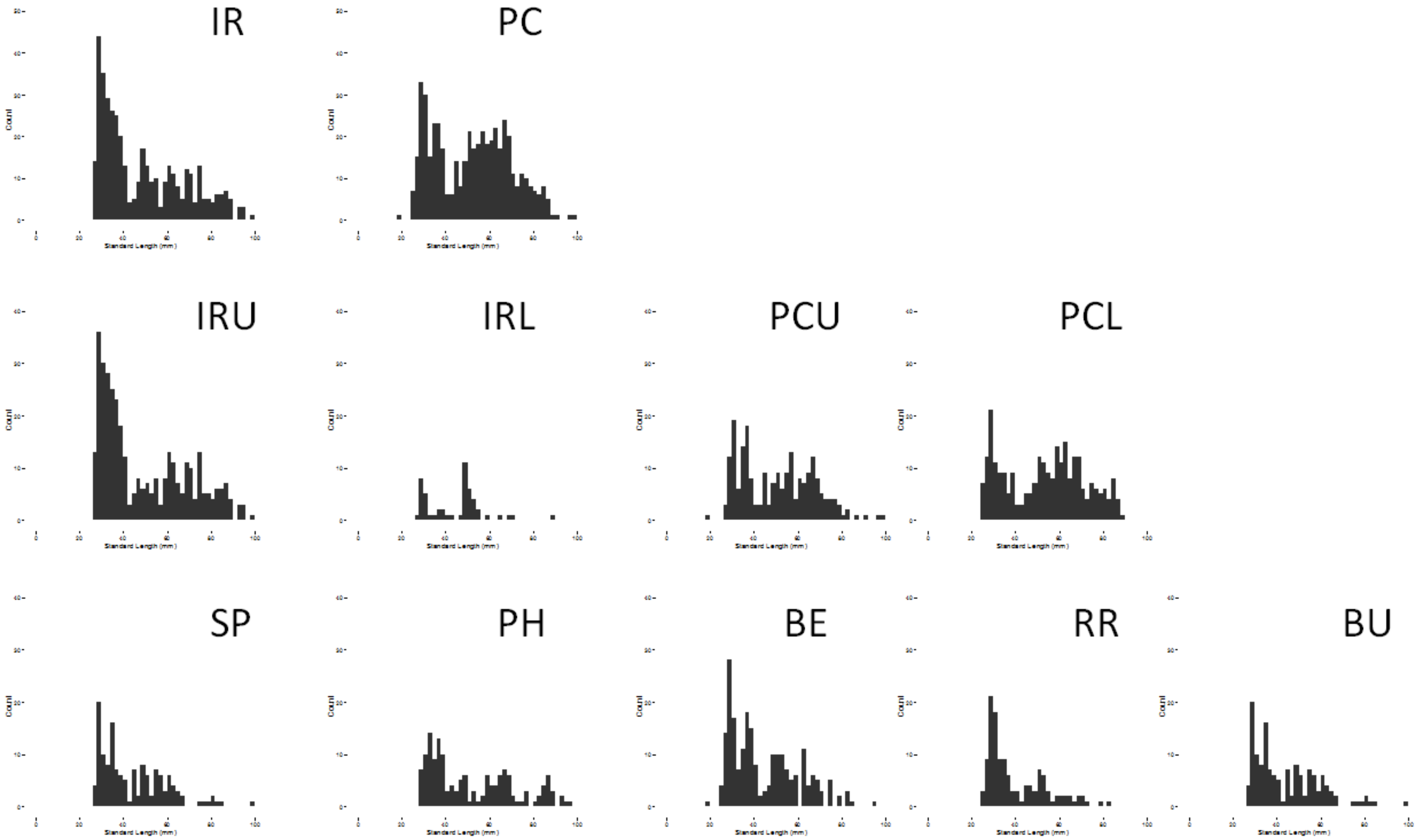

Figure 1.12 Menhaden length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

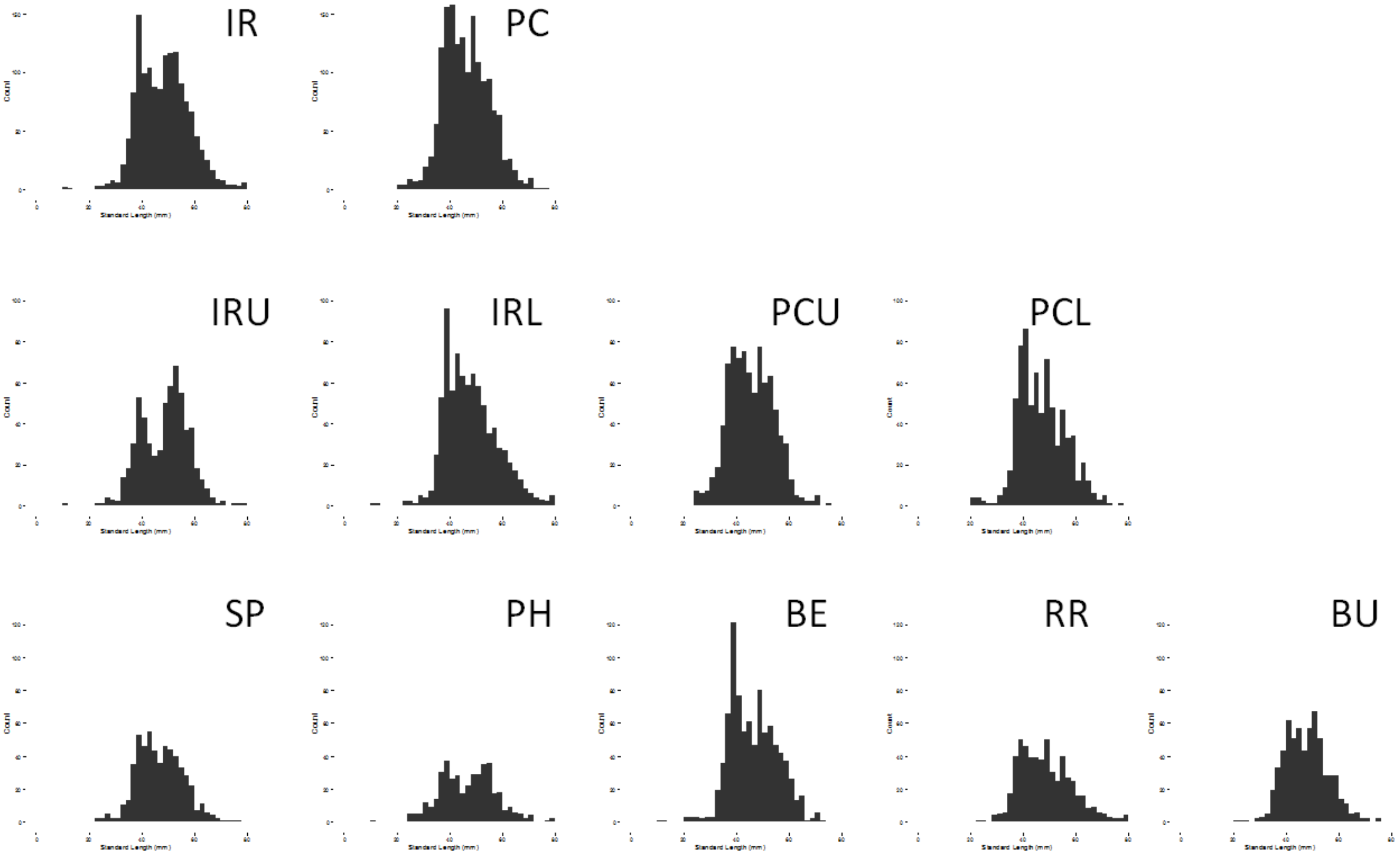

Figure 1.13 Atlantic silverside length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

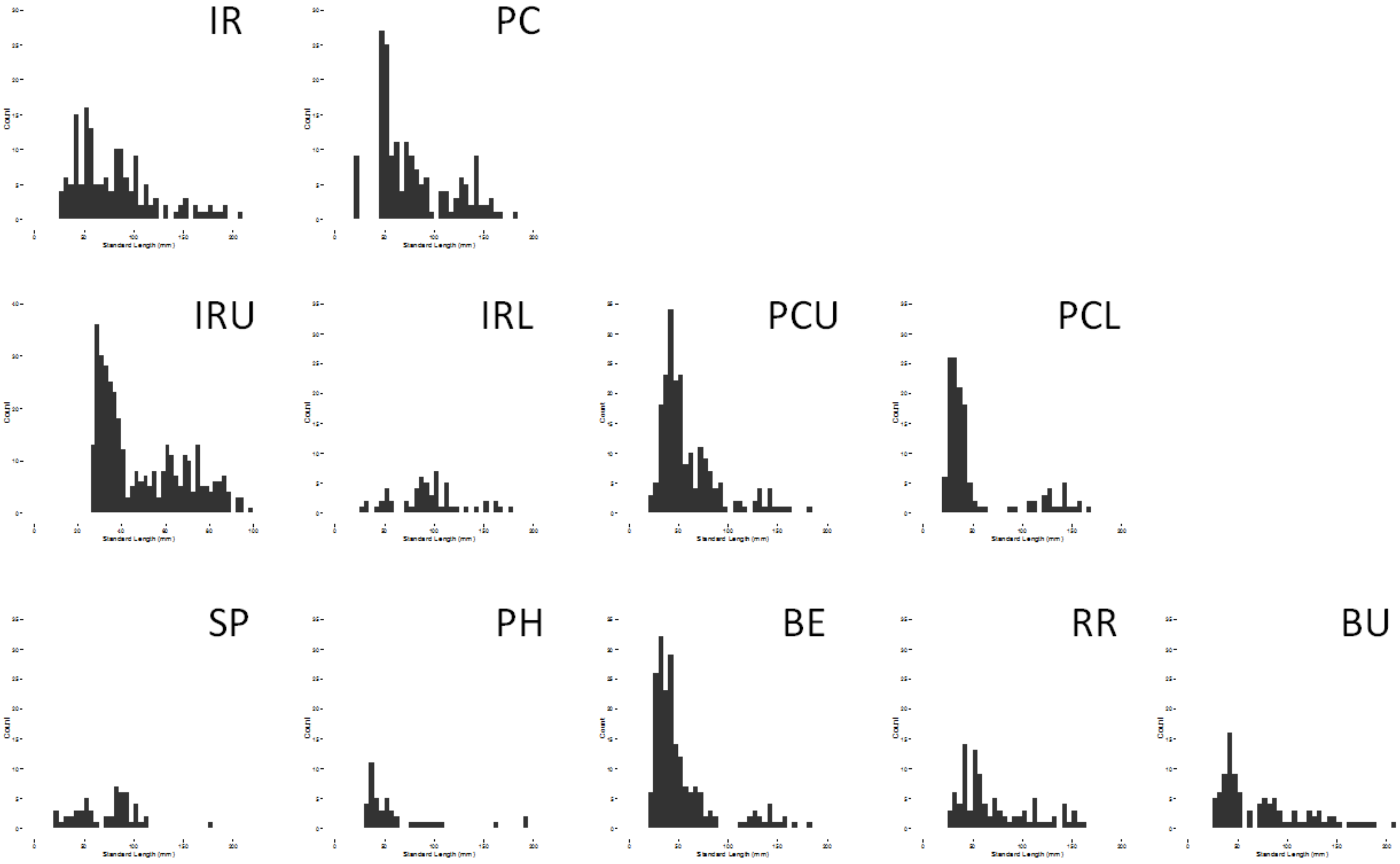

Figure 1.14 Striped mullet length-frequency histograms by tributary, tributary region, and shoreline type for 2010

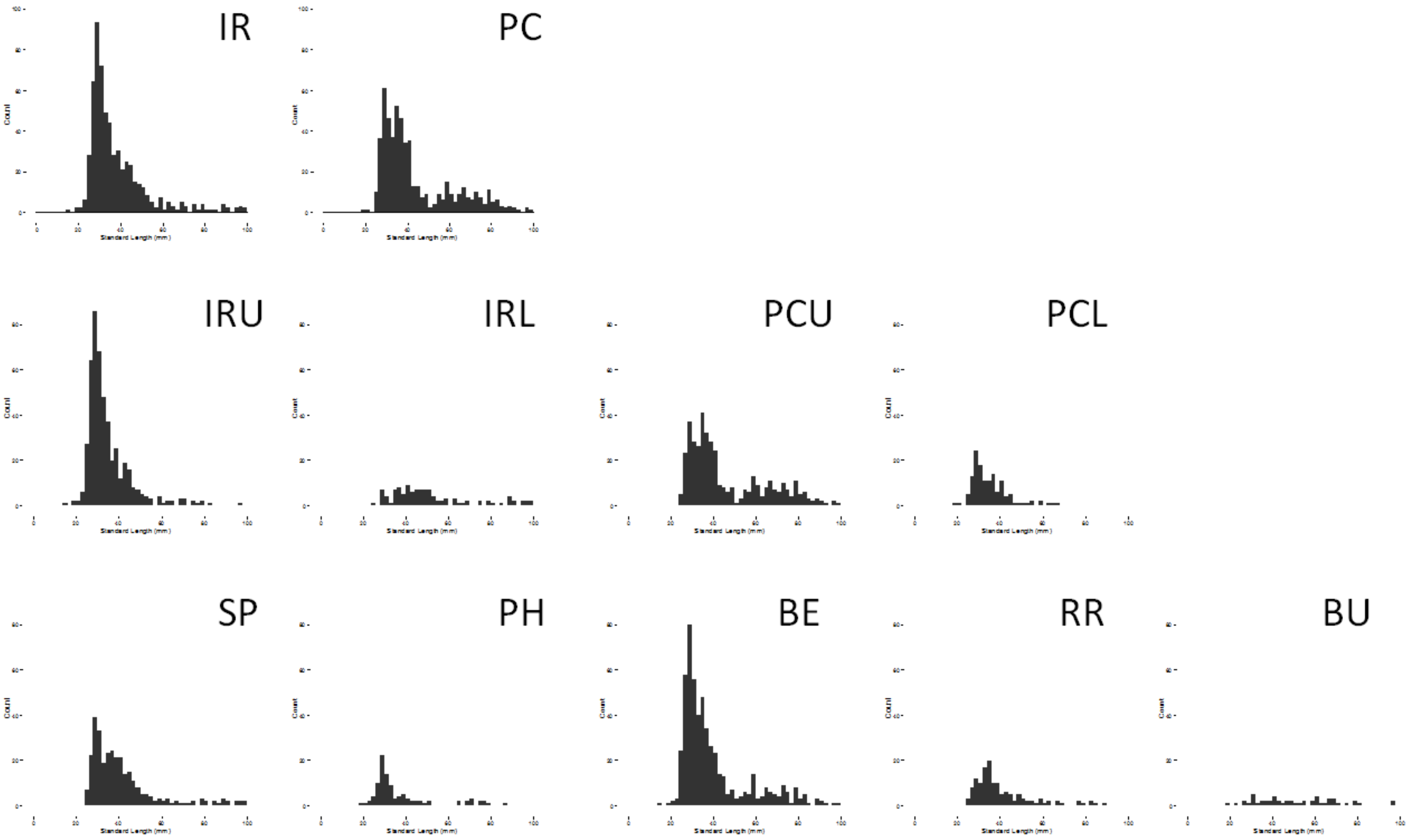

Figure 1.15 Striped killifish length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

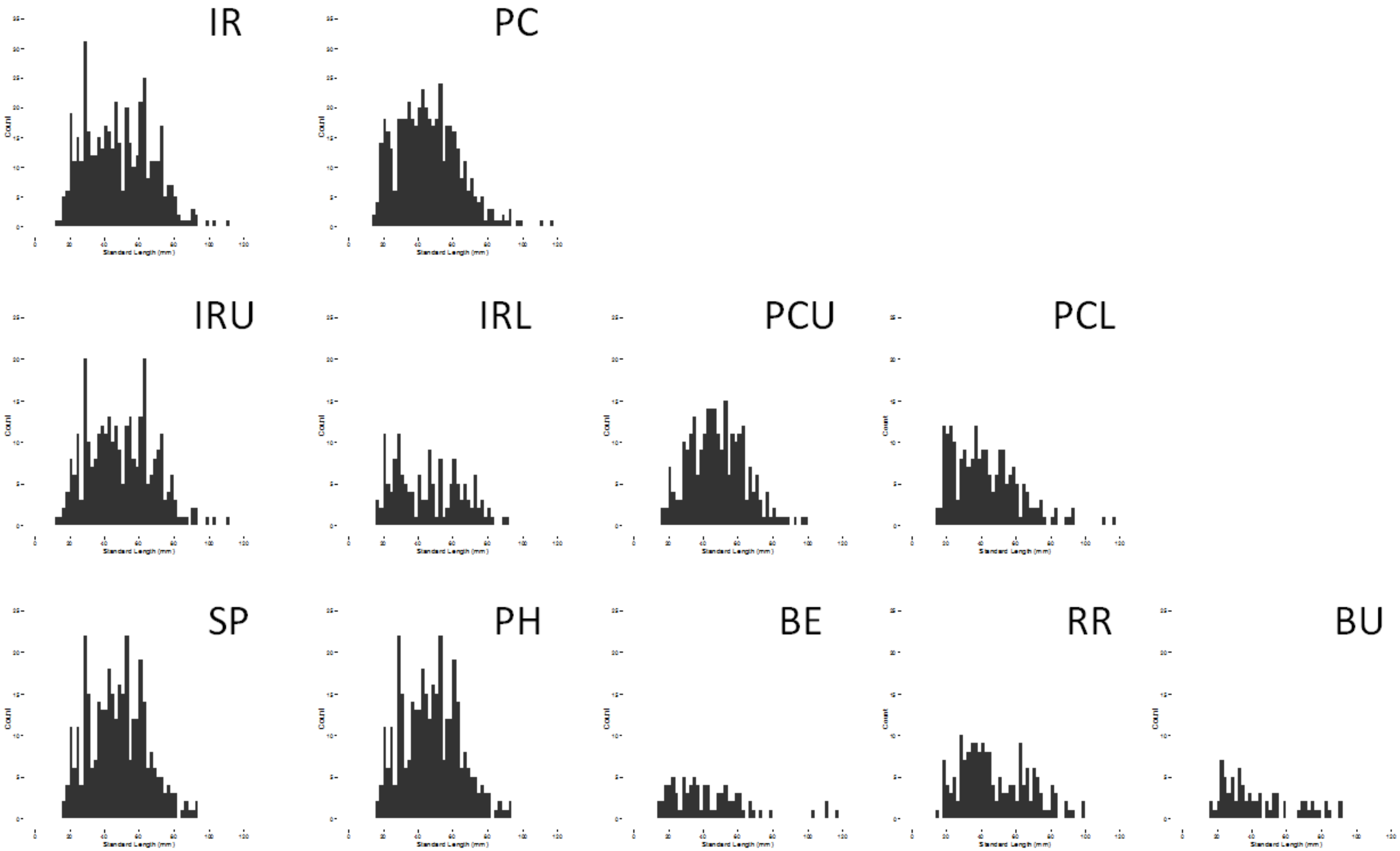

Figure 1.16 Silver perch length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

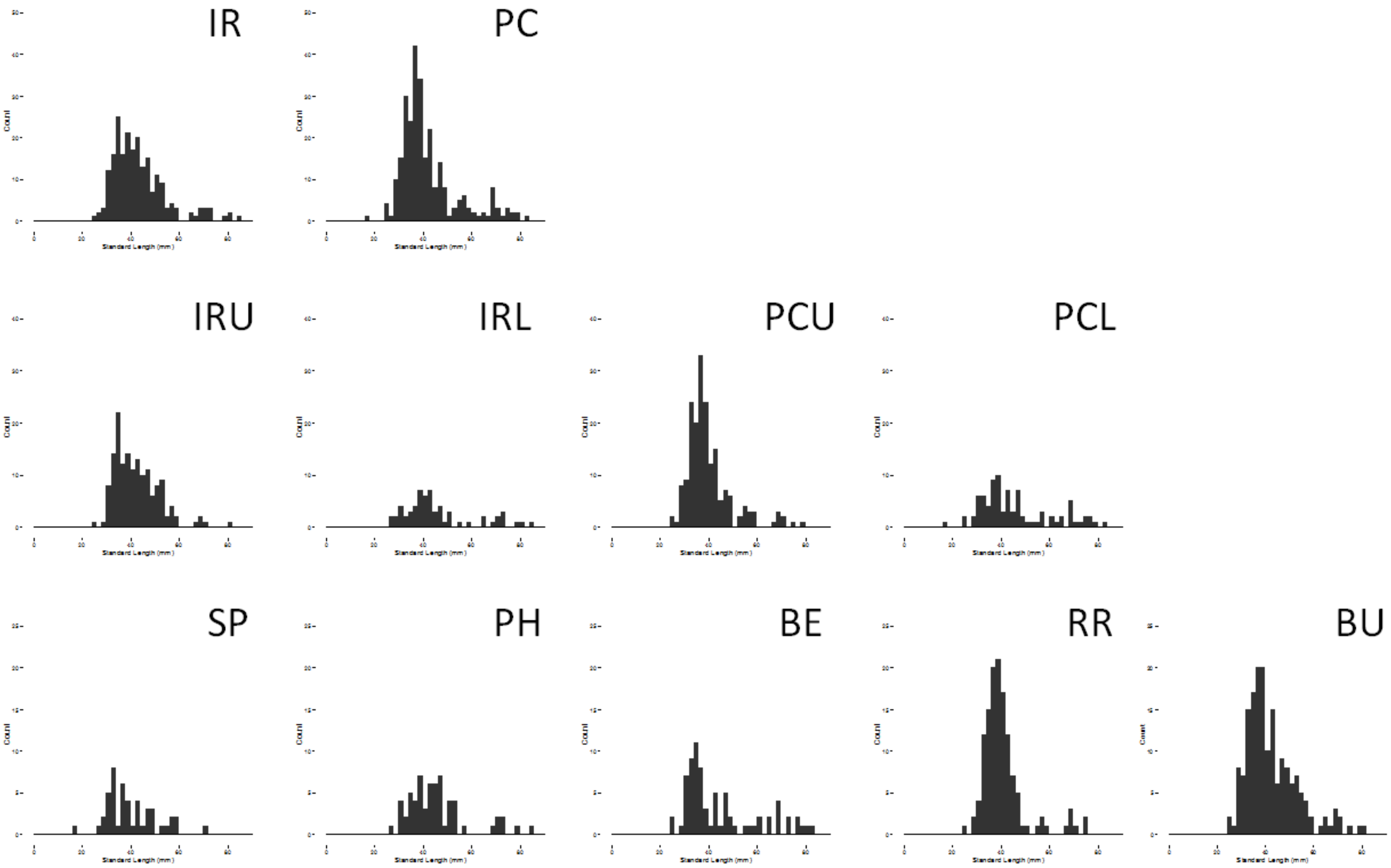

Figure 1.17 Bay anchovy length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

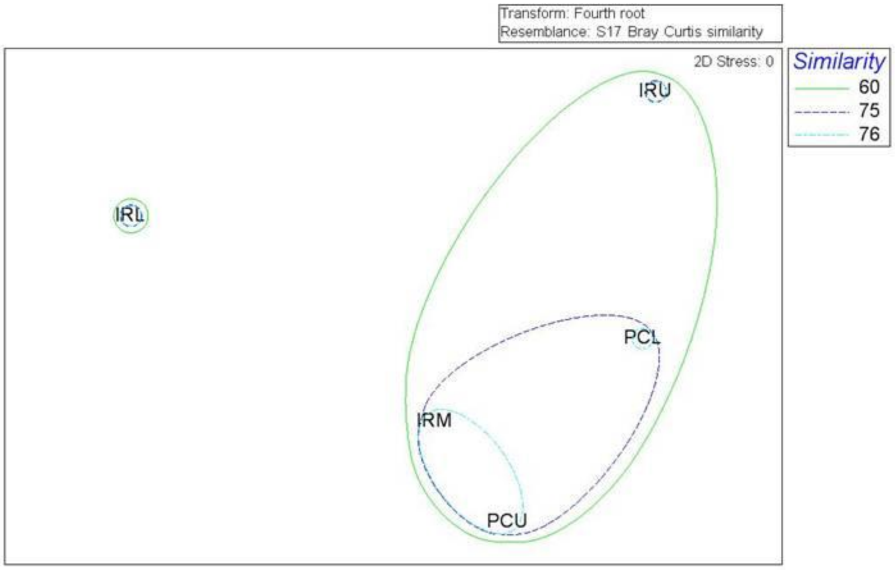

Figure 1.18 Non-metric multidimensional scaling plots based on fish assemblage at tributary regions in 2009.

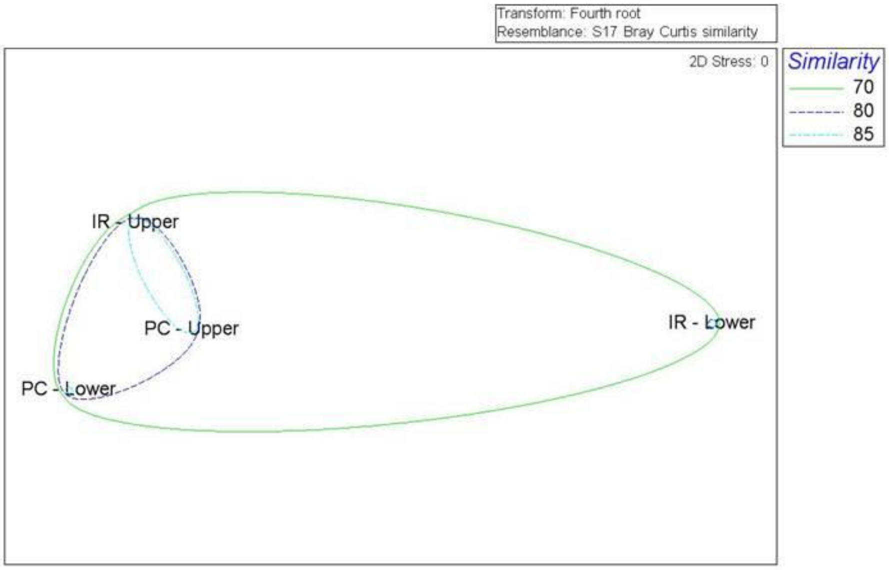

Figure 1.19 Non-metric multidimensional scaling plots based on fish assemblage at tributary regions in 2010.

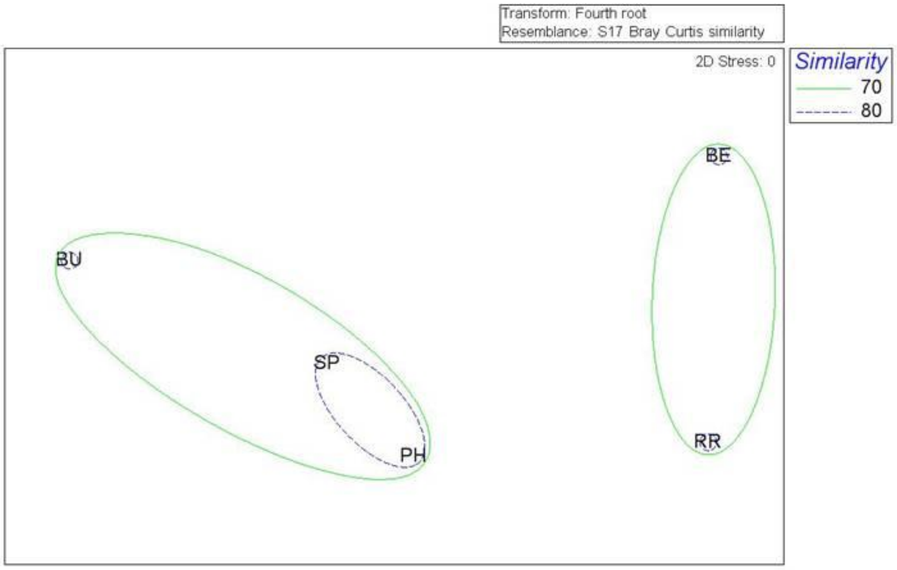

Figure 1.20 Non-metric multidimensional scaling plots based on fish assemblage at shoreline types in 2009.

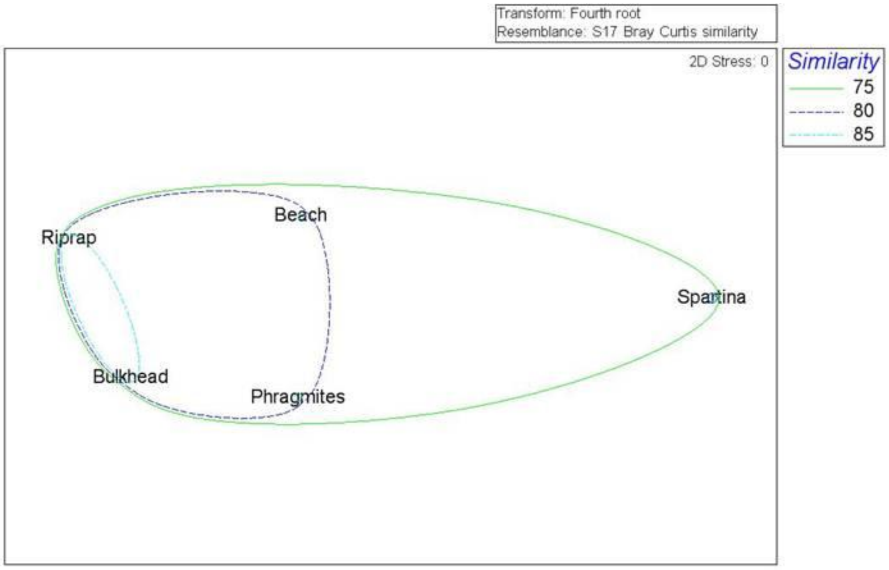

Figure 1.21 Non-metric multidimensional scaling plots based on fish assemblage at shoreline types in 2010.

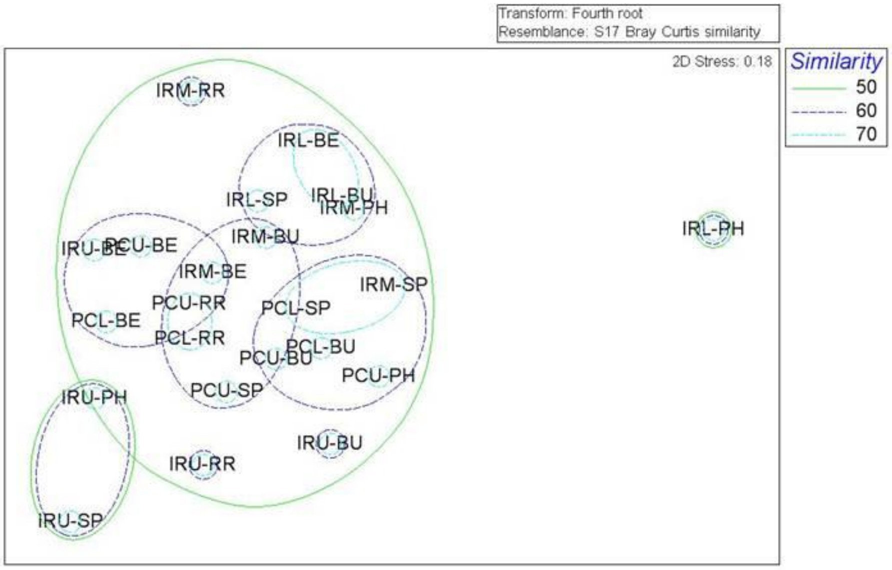

Figure 1.22 Non-metric multidimensional scaling plots based on fish assemblage at sampling stations in 2009.

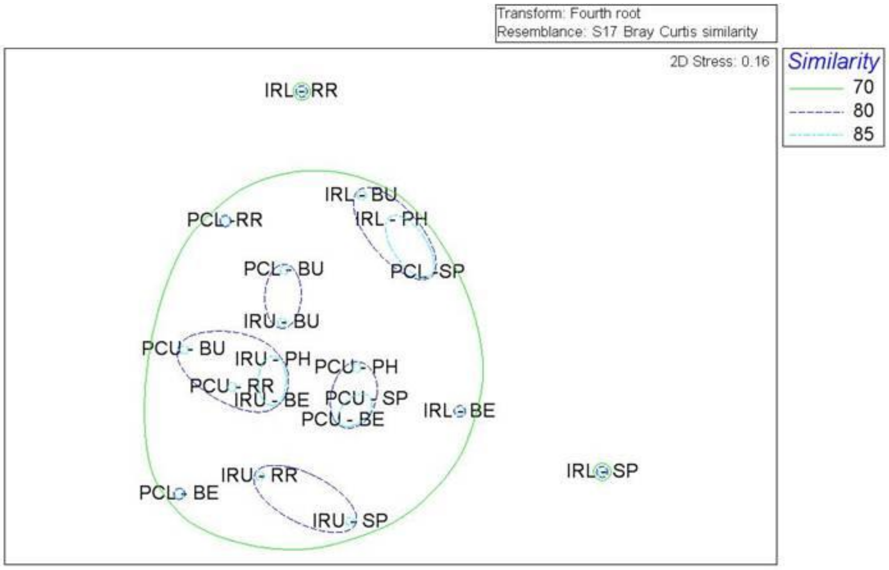

Figure 1.23 Non-metric multidimensional scaling plots based on fish assemblage at sampling stations in 2010.

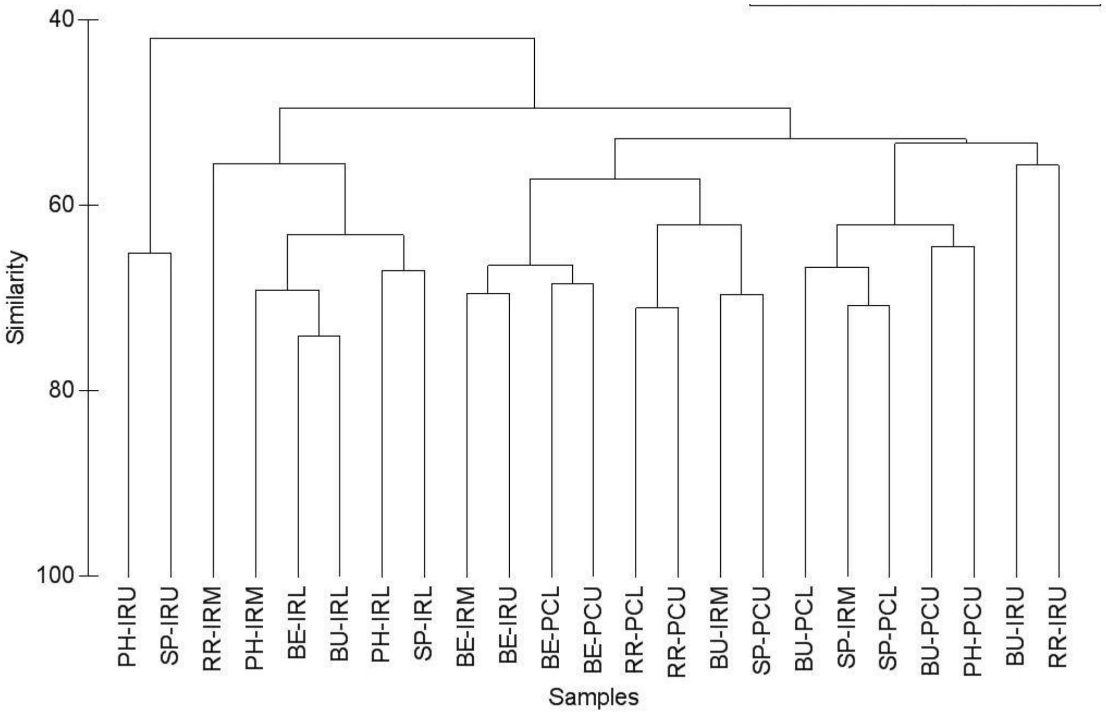

Figure 1.24 Cluster dendrogram for all sampling stations in 2009 (IRM represent lower Indian River and IRL represents mouth of Indian River).

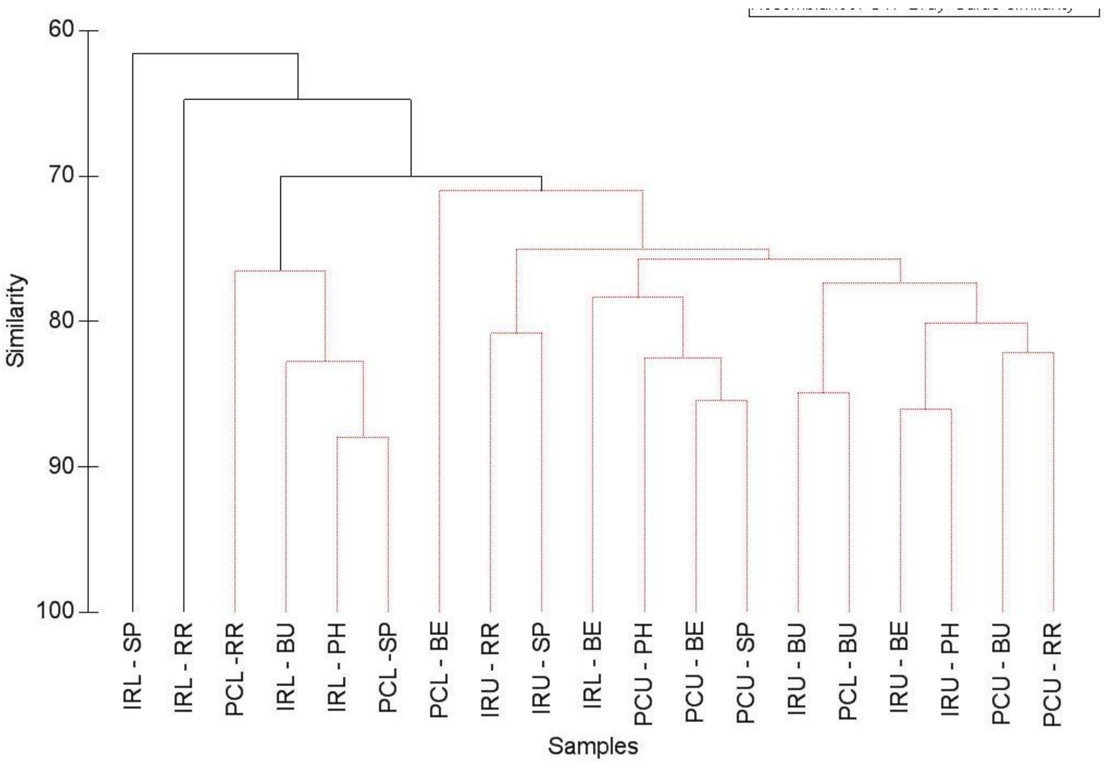

Figure 1.25 Cluster dendrogram for all sampling stations in 2010.

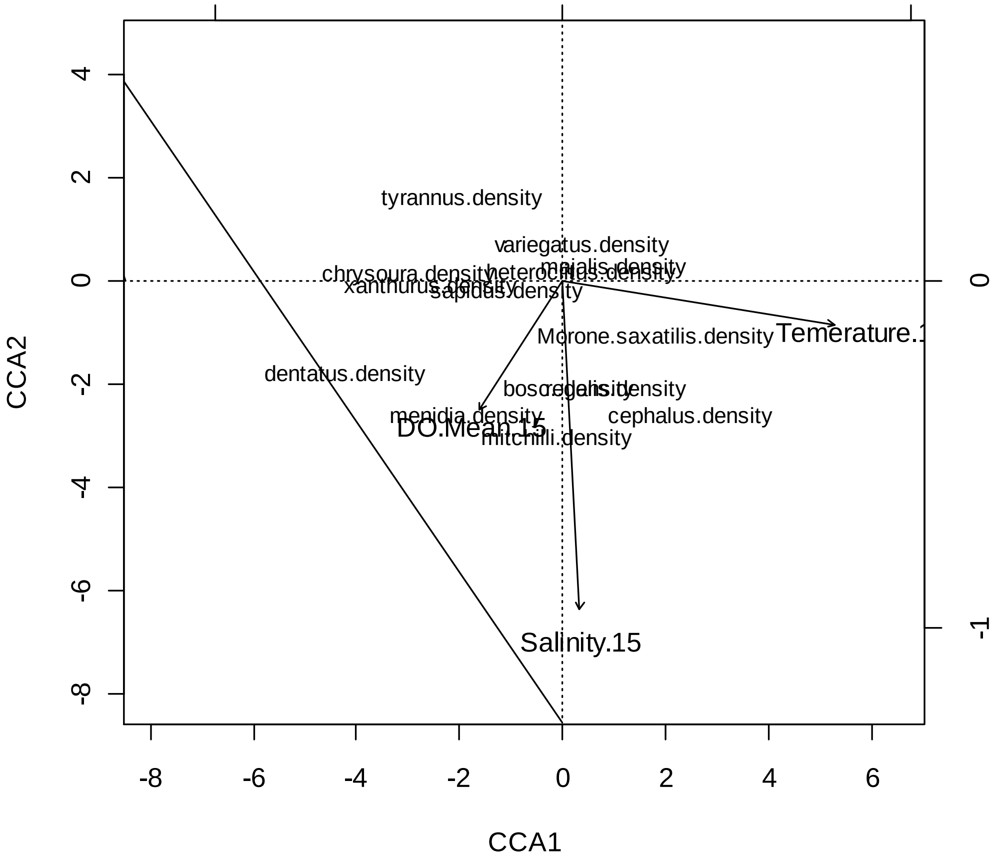

Figure 1.26 Ordination of species density of the thirteen most abundant fish species and blue crabs arranged according to response scores on significant CCA Axes (Table 1.25) from water quality predictors. Environmental variables are depicted as vectors and are positioned according to biplot scores on CCA axes.

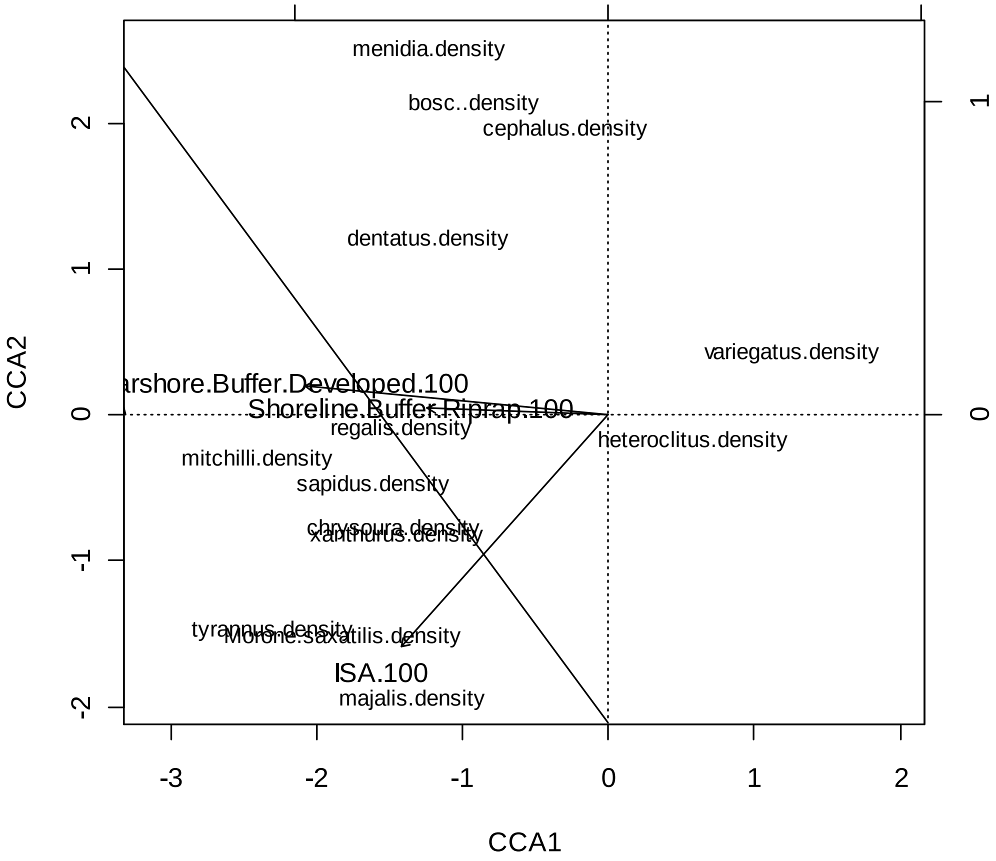

Figure 1.27 Ordination of species density of the thirteen most abundant fish species and blue crabs arranged according to response scores on significant CCA Axes (Table 1.26) from near shore land use predictors. Environmental variables are depicted as vectors and are positioned according to biplot scores on CCA axes.

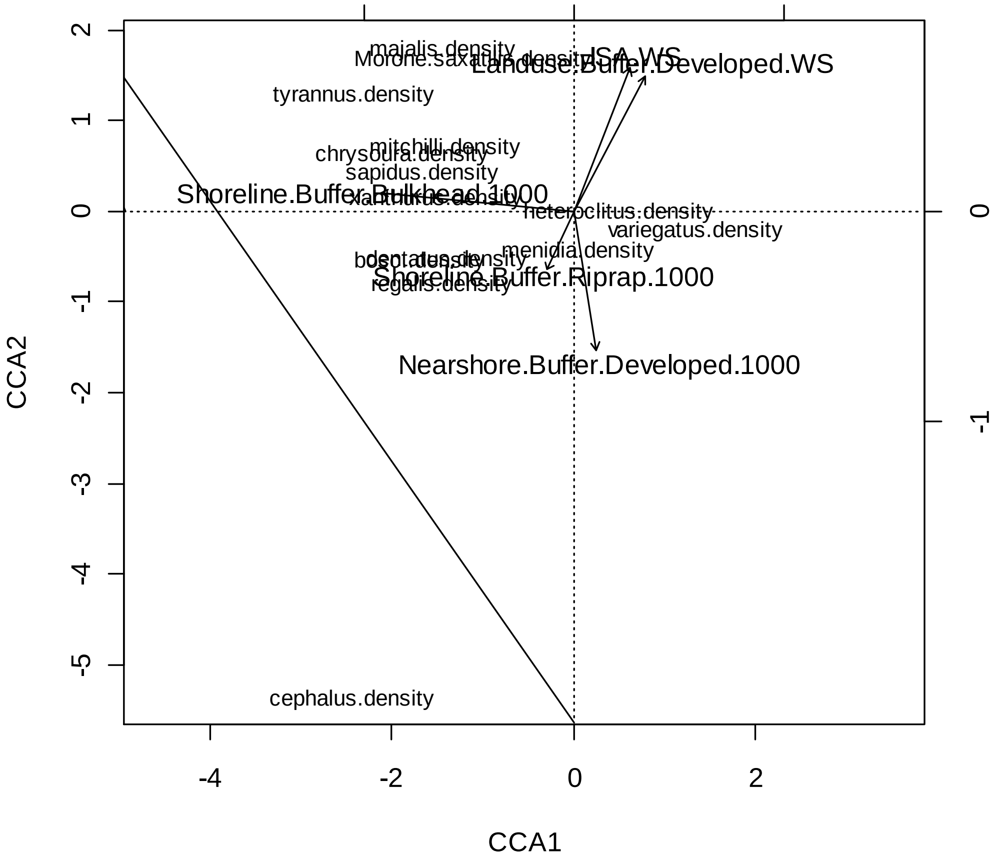

Figure 1.28 Ordination of species density of the thirteen most abundant fish species and blue crabs arranged according to response scores on significant CCA Axes (Table 1.27) from watershed land use predictors. Environmental variables are depicted as vectors and are positioned according to biplot scores on CCA axes.

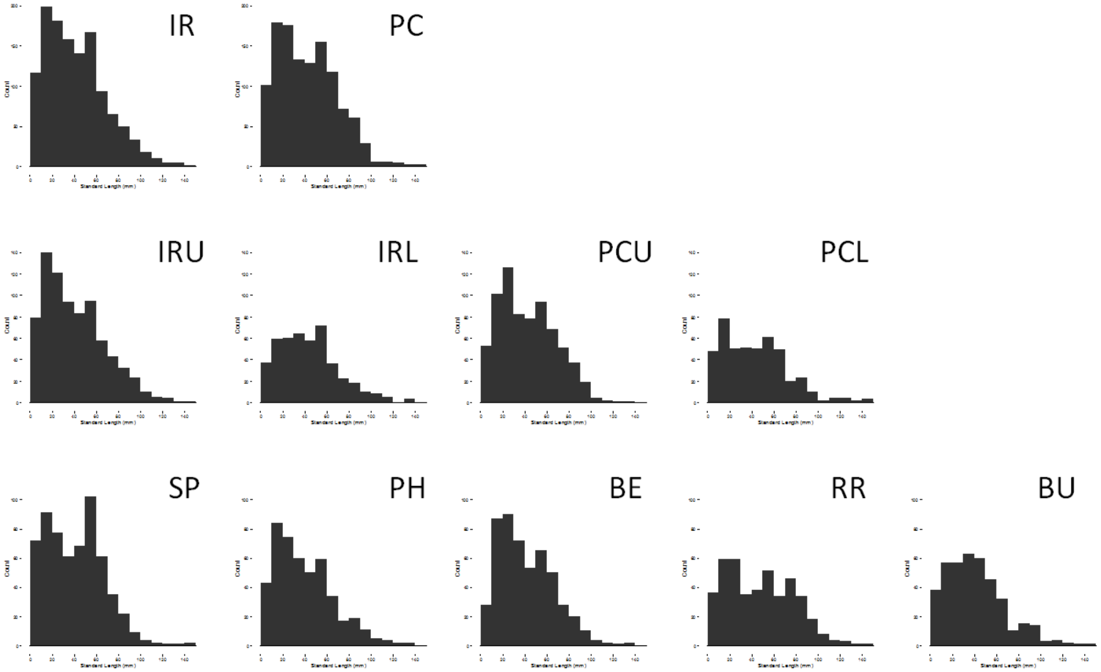

Figure 1.29 Blue length-frequency histograms by tributary, tributary region, and shoreline type for 2010.

Chapter 2

FISH AND BLUE CRAB ABUNDANCE ALONG A RIPRAP-SILL HARDENED SHORELINE: COMPARISONS WITH SPARTINA MARSH AND RIPRAP

Introduction

The physical structure of estuarine shorelines is impacted by modifications such as bulkhead (vertical walls) and riprap (rock and solid debris placed directly on the shoreline) which are installed for erosion control and to provide protection from storm surges. Several studies have found that these types of shoreline hardening negatively impact a range of estuarine macrofaunal communities through reduced abundance and diversity. Such negative effects have been reported for fishes, crustaceans, and benthic invertebrates (Herke and Rogers 1989, Able et al. 1998, Seitz et al. 2006, Bilkovic and Roggero 2008).

Riprap and bulkhead structures can negatively impact estuarine water quality and aquatic assemblages through alteration of intertidal habitat, changes in hydrology, loss of allochthanous materials, increased recreational use, and the loss of natural erosion control (Bilkovic and Roggero 2008). Shoreline hardening can also cause increased water depth at the land-water interface, leading to a decrease in intertidal faunal community integrity through reductions in nekton diversity, reduced integrity of fish assemblage structure, and reduced macrobenthic assemblage biotic indexes (Jennings et al. 1999, Peterson et al. 2000, Bilkovic et al. 2006). Estuaries along the

Mid-Atlantic, including Delaware's Coastal Bays and their tributaries, are increasingly impacted by shoreline hardening (USGS 2011).

Wetlands and subaqueous land managers frequently recommend that landowners install free-standing stone sills, with wetland plantings behind the sill (riprap-sill structures; a type of 'living shoreline'), rather than traditional riprap revetments for erosion control purposes. This recommendation is based on an assessment of the physical and erosional properties at the site, as well as the perceived ecological benefit of increased habitat diversity resulting from the wetlands and shallow-water habitat created behind the sill (L. Herr, personal communication). However, only recently has any research been conducted to compare how macrofauna utilize these shoreline structures (Currin et al. 2008, Davis et al. 2008, Strayer et al. 2012). Quantitative sampling is needed to establish whether there are differences in fish and blue crab densities between these two types of hardened shorelines. The objective of this research was to determine how riprap-sill shoreline is utilized by fish and blue crabs in comparison with traditional riprap and native *Spartina alterniflora* fringing marsh.

Materials and Methods

Seining was conducted to examine fish and blue crab abundance and diversity in the shore-zone and shallow subtidal region at riprap-sill, riprap revetment, and fringing *S. alterniflora* marsh shorelines in Indian River, a tributary of the Delaware Coastal Bays, USA (Figure 2.1). The riprap-sill structure was ~80 m long and located ~7 m offshore from mean high water level. Gaps approximately 1.5 m wide separated the sill structure every 20 m. The rocky sill is completely submerged at high tide and completely exposed at low tide. *Spartina alterniflora* (directly landward of the sill) and *Spartina patens* (higher in the intertidal) had been planted one growing season prior to sampling and was grown in to approximately 40% natural density. The riprap structure was ~120 m long, located directly on the shoreline, submerged to 2.5 m at high tide, and entirely exposed at low tide. The *S. alterniflora* sampling location was part of an unbroken stretch of *S. alterniflora* fringing marsh ~300 m long. Where sampled, the marsh surface extended ~9 m landward from the marsh edge. At all shoreline types, seining was conducted in areas where the near-shore substrate was sand to dense mud, with minimal structure (e.g., woody debris or rock). These locations were selected haphazardly along each shoreline type on each sampling day.

Sampling was conducted once every two weeks between June 18th and September 19th, 2010. Fishes and blue crabs occupying the shore-zone and shallow subtidal regions were sampled using a 15 m seine (2 m high; 5 mm mesh) with center bag. Sites were sampled within three hours of low tide, on both the flooding and ebbing tide. The riprap-sill was sampled on the seaward side of the rocky sill, alternately before the tide completely submerged the sill and just as the tide was receding from the sill (~ 1 m water height on structure). The riprap was sampled when the riprap structure was ~ 1 m inundated by the tide. The *S. alterniflora* site was sampled just as the water level reached the edge of the marsh surface. Every effort was made to account for water level in the present study. Sampling did not occur when marsh surface was available to species, thus low water level concentration factors were equivalent for all shoreline types (Rozas and Minello 1997). Additionally water level was used as a correction in calculating species density along each shoreline type.

Each seine haul was conducted using the following methods: One end of the net was placed on shore using a shallow-draft boat. The net was unfurled in an arc, 3 m from shore at its apogee, along the shoreline. The far end of the net was then brought to shore. A PVC pole was used to startle fish and blue crabs from the area between the net and shoreline toward the net bag. The two ends of the net were then walked together along the shoreline. The net was gathered and fish and blue crabs further directed into the bag, which was cinched tight in the water and placed in the boat where the sample was removed and placed in a container of water from the sampling site. A second seine haul was taken immediately using the same method, over the same area, and in the same direction as the first haul. Fish from the second seine haul were placed into a second container. Area swept by each seine haul was ~ 33 m^2. Water depths were taken at the shoreline and at the apogee of the seine haul to calculate volume of water seined, and subsequently fish and blue crab density (individuals/m^3).

Net efficiency was determined by enclosing a sampling station with a 33 m seine net and then using the 15 m sampling seine net to consecutively sample the station, removing the catch each time, until zero's were collected in two consecutive hauls. This was completed at each of the sampling stations prior to the sampling period. The first two rapidly repeated seine hauls collected >90% of fish and >95% of blue crabs at all three shoreline types. Due to these similarities, no catchability coefficient was applied for each shoreline type. Analysis of the two seine hauls indicated variability of the difference in catch between the first and second seine hauls was minimal, both among and within sampling stations. As a result, the catch from the two seine hauls was considered a single sample for analysis purposes.

All fishes and blue crabs were identified to species and measurements (standard length) for fishes and carapace width for blue crabs (±1.0 mm) were recorded on a maximum of 20 individuals per collection. All individuals were measured for species represented by ≤ 20 individuals and 20 haphazardly selected individuals were measured for species with > 20 individuals. All macrofauna was subsequently released adjacent to the shoreline where they were collected.

Species richness and mean abundance and density (individuals/m^3) for all fishes and blue crabs were calculated for each shoreline type. Three biotic indices were calculated for each shoreline type: Margalef's Richness Index which relates the number of species to the total abundance of individuals at a site, Shannon Diversity Index which relates species evenness and richness within a site, and Pielou's Evenness Index which relates the Shannon Diversity Index to overall site-species evenness.

One-way repeated measures analysis of variance (ANOVA) was used to compare weekly fish and blue crab abundance and density data among shoreline types. If significant differences were found, the Holm-Sidak method of pairwise multiple comparison post-hoc test was performed. Data was examined for normality using the Shapiro-Wilk test, Kolmogorov-Smirnov test and examination of Q-Q plots. When normality was not met, attempts at normalizing data using square root and log(x+1) transformations were made. When parametric assumptions of normality or homoscedasticity were not passed after data transformation, Friedman's related samples two-way analysis of variance by rank was used to compare differences. A stepwise step-down multiple comparison post-hoc test, adopted from a method from Campbell and Skillings (1985) was performed to determine homogeneous subsets in each non-parametric analysis. All statistics were conducted in SPSS (v. 18.0.2) and a significance level of $\alpha = 0.05$ was used throughout.

Seining was not possible in the upper intertidal zone at the shoreline types sampled, due to the complex structure (*S. alterniflora* or riprap) present. Thus, fish traps were used in these areas. Cylindrical traps (GEE Minnow Trap) composed of galvanized steel mesh (0.64 cm^2) with a funnel shaped throat on either end (2.5 cm mouth opening) and a volume of 10,330 cm^3 were fished without bait at each shoreline type on each of the same days as the seining described above. Four traps were placed 4 m apart in the mid to upper intertidal zone at each shoreline type. Traps were fished within three hours of low tide on both the flooding and ebbing tide for 15-120 minutes with the same soak time at all shoreline types on a given sampling day. Fish were identified to species and standard length measured (±1.0 mm). This was completed 2-4 times at each shoreline type on each sampling day (standard effort across all shorelines on each day). Quantitative estimation of fish abundance and density is not possible with this type of trapping, however qualitative estimates of species presence/absence are possible when fished for the same amount of time at each sampling location (Kneib and Craig 2001). These traps have been commonly used in fish studies, particularly for the collection of *Fundulus* species (Sweeny et al. 1998, Kneib and Craig 2001, Outerbridge et al. 2007). The percentage of traps sampled that contained each species on a given sampling day was calculated for each shoreline type (all traps counted individually) and a mean calculated across the sampling period. Also, the percentage of sampling days in which each species was collected was calculated for each shoreline type (all traps from a single sampling day at each shoreline type were combined).

Results

In total, 9777 fishes and 548 blue crabs were collected in seine sampling at the three shoreline types (Table 2.1). Greatest total abundance and density of fishes were in the shore-zone of fringing *S. alterniflora* marsh followed by riprap-sill and riprap (Tables 2.1 and 2.2). Significantly greater total fish abundance was along *S. alterniflora* than along riprap, whereas statistically similar total abundance was along *S. alterniflora* and riprap-sill (Friedman's; $p<0.01$). Significantly greater total fish density was along *S. alterniflora* than along riprap-sill and riprap (Friedman's; $p=0.015$).

Among the most abundant species (>150 individuals) caught (Table 2.1), no significant differences in abundance or density were found among shoreline types for mummichog (*Fundulus heteroclitus*) (abundance ANOVA $p=0.209$; density ANOVA $p=0.201$), Atlantic menhaden (*Brevoortia tyrannus*) (abundance Friedman's $p=0.670$; density Friedman's $p=0.417$), or blue crabs (*Callinectes sapidus*) (abundance Friedman's $p=0.177$; density ANOVA $p=0.294$). However, significantly greater abundance and density of Atlantic silverside (*Menidia menidia*) were caught along *S. alterniflora* and riprap-sill than along riprap (abundance Friedman's $p<0.01$; density Friedman's $p<0.01$). Significantly greater abundance and density of striped killifish (*Fundulus majalis*) were caught along *S. alterniflora* than along riprap-sill and riprap (abundance Friedman's $p<0.01$; Friedman's; $p<0.01$). Significantly greater abundance and density of bay anchovy (*Anchoa mitchilli*) were caught along riprap-sill than along riprap (abundance Friedman's $p<0.01$; density Friedman's $p<0.01$). Significantly greater abundance and density of silver perch (*Bardiella chrysoura*) were caught along

S. alterniflora and riprap-sill than along riprap (abundance ANOVA p<0.01; One-way ANOVA p<0.01). No species was found to be significantly more abundant or in greater densities along the riprap shoreline than along the *S. alterniflora* or riprap-sill shoreline.

Species richness was greatest along riprap-sill (17) followed by *S. alterniflora* (14) and riprap (12) shores. Biotic index scores were generally higher at riprap-sill than other shoreline types (Table 2.3). All three shoreline types were dominated by mummichog (Figure 2.2, crosshatched bars). Overall, a majority of all mummichog, Atlantic silverside, striped killifish, and silver perch caught were caught along *S. alterniflora* shoreline (Figure 2.2, black bars). A majority of bay anchovy was caught along riprap-sill and a majority of Atlantic menhaden was caught along riprap shorelines. Atlantic menhaden was the only abundant species that was caught in greater abundance at riprap than along either of the other shoreline types.

Structure of the length-frequency histograms suggests that somewhat larger mummichog were caught along riprap than along *S. alterniflora* or riprap-sill shorelines (Figure 2.3). Similar-sized individuals of Atlantic silverside, Atlantic menhaden, and striped killifish were caught at each shoreline type (Figure 2.3). Blue crabs < 30 mm carapace width were more commonly caught at *S. alterniflora* than along riprap-sill or riprap shorelines (Figure 2.3).

Fish and blue crab abundance (ANOVA p=0.452) and density (ANOVA p=0.473) were not statistically different between flood tide and ebb tide seine sampling. This was true both within and among shoreline types.

Mummichog, striped killifish, silver perch, and Atlantic silverside were the only species caught during minnow trap collections. Mummichog, Atlantic silverside, and striped killifish were most frequently present in traps along the *S. alterniflora* shoreline (Table 2.4). Silver perch were most frequently, and equally, caught in traps along *S. alterniflora* and riprap-sill shorelines. Mummichog were caught on 100% of sampling days along both the *S. alterniflora* and riprap-sill shorelines (Table 2.4). Striped killifish, silver perch, and Atlantic silverside were caught on more sampling days along *S. alterniflora* and riprap-sill than along riprap shorelines. Atlantic silverside were never captured in minnow traps along the riprap shoreline, and no species was caught on more sampling days at riprap shoreline than along the riprap-sill shoreline.

Discussion

Habitat quality for fishes in salt marsh creeks can be defined as "the ability of the habitat to provide conditions appropriate for individual, population or species persistence" (Rountree and Able 2007). Habitat quality differs in the intertidal and shallow subtidal zone along different shoreline types. Thus, fish and blue crab abundance/density and presence/absence should theoretically reflect overall habitat quality across species.

Very few prior studies have assessed macrofauna abundance along riprap-sill compared with riprap shoreline (Davis et al. 2008). The present research demonstrates that while native *S. alterniflora* fringing marsh provides the preferred shore-zone habitat for many small estuarine fishes, riprap-sill appears to provide more preferential habitat than does traditional riprap. In fact, a large majority of the most abundant species was found in statistically similar densities in seine hauls along *S. alterniflora* and riprap-sill shorelines. Seine sampling was conducted along the sill structure of the riprap-sill shoreline, an analogous structure to the riprap shoreline, although positioned in the lower intertidal zone as opposed to directly against the shore. The greater abundance of fishes along the riprap-sill than riprap may be due in part to the presence of *S. alterniflora*. This native marsh vegetation is accessible through the gaps and ends of the sill structure allowing access for small estuarine fish to access this habitat.

Traditional riprap structure completely removes any intertidal vegetated areas from the shore-zone. *Spartina alterniflora* marsh surface has been documented as important habitat for estuary-dependent and resident estuarine fish species, particularly *Fundulus* spp. (Rozas and Reed 1993, Kneib and Wagner 1994). Riprap-sill structures are designed as a type of 'living shoreline' by creating intertidal vegetated area, with native plantings in the area landward of the erosion control structure. Use of minnow traps in the present research demonstrated that several fish species, in particular the estuarine resident mummichog and the seasonal resident Atlantic silverside utilize this planted zone. There was a greater than two fold difference in *Fundulus* spp. abundance and density between *S. alterniflora* and riprap-sill structure, based on the seine data. As a result of the high relative abundance of mummichog along the *S. alterniflora* shoreline (ostensibly due to the more extensive marsh surface), biotic index scores were reduced, compared to riprap-sill structure. It is possible, however that a more mature *S. alterniflora* stand shoreward of the sill (the one sampled in the present research was only in its second growing season post-planting) might create even more similar habitat to the natural *S. alterniflora* fringing marsh sampled.

Atlantic menhaden, striped mullet and spot were caught in greater abundance and density (though not significantly so) in seine collections along the riprap structure than along the *S. alterniflora* and riprap-sill shorelines, the only abundant species to show this pattern. None of these three species are estuary-residents. The greater abundances along riprap may be due to the lack of structure the riprap shoreline creating more ‘open water’ habitat. Blue crabs <30 mm carapace width were more commonly caught at along *S. alterniflora* shoreline than along riprap-sill or riprap. These smaller individuals may have been using the structure provided by *S. alterniflora* as predator protection and also feeding on the well-developed marsh surface during high tide.

The present research has shown a riprap-sill shoreline to provide favorable habitat in comparison with a traditional riprap shoreline for small estuarine fish and blue crabs through increased species densities and greater biotic indices. Future research should focus on using passive trapping (fyke nets, bottomless lift nets) to better understand how the upper intertidal zone of riprap-sill is being utilized at high tide in comparison with *S. alterniflora* marsh surface. Additional riprap-sill structures in more estuarine environments should be sampled over a longer time period to get a more complete view of how macrofauna utilize this shoreline type, particularly in comparison with traditional riprap revetment. As coastal development continues worldwide, and as pressure for shoreline stabilization increases due to sea level rise, it is important for wetlands, shoreline and subaqueous land managers to have better information available to make informed planning decisions. More thorough investigation of shoreline hardening effects on macrofauna will help in this decision making.

Table 2.1 Mean fish and blue crab abundance in seine collections, by shoreline type, in a tributary of the Delaware Coastal Bays, USA, from June-September 2010. Sixteen seine hauls were conducted at each shoreline type. Significant differences ($p < 0.05$) among shoreline types are noted by superscript letters.

	S. alterniflora	Riprap-Sill	Riprap	Total Abundance	% of Total Abundance
All Fish Species (Total Abundance)	**5348^{a}**	**3065^{ab}**	**1912^{b}**	9777	100
Fundulus heteroclitus (Mummichog)	3546	1643	1027	6216	63.58
Menidia menidia (Atlantic silverside)	755^{a}	653^{a}	67^{b}	1475	15.09
Brevoortia tyrannus (Menhaden)	297	106	476	879	8.99
Fundulus majalis (Striped killifish)	380^{a}	70^{b}	53^{b}	503	5.14
Anchoa mitchilli (Bay anchovy)	40^{ab}	254^{a}	3^{b}	297	3.04
Bairdiella chrysoura (Silver perch)	87^{a}	67^{a}	17^{b}	171	1.75
Mugil cephalus (Striped mullet)	6	4	95	105	1.07
Leiostomus xanthurus (Spot)	16	29	35	80	0.82
Cynoscion regalis (Weakfish)	5	6	3	14	0.14
Paralichthys dentatus (Summer flounder)	3	8	0	11	0.11
Trinectes maculatus (Hogchoker)	5	3	0	8	0.08
Morone saxatilis (Striped bass)	1	4	2	7	0.07
Gobiosoma bosc (Naked goby)	2	5	0	7	0.07
Cyprinodontidae variegatus (Sheepshead minnow)	0	0	2	2	0.02
Pomatomus salatrix (Bluefish)	0	1	0	1	0.01
Caranx hippos (Crevalle jack)	0	1	0	1	0.01
Callinectes sapidus (Blue Crab)	205	211	132		

Table 2.2 Mean fish and blue crab density (individuals/m^{3}) in seine collections, by shoreline type, in a tributary of the Delaware Coastal Bays, USA, from June-September 2010. Significant differences ($p < 0.05$) among shoreline types are noted by superscript letters.

	S. alterniflora	Riprap-Sill	Riprap	Total Density	% Total Density
All Fish Species (Total Density)	**563.84^{a}**	**255.53^{b}**	**176.17^{b}**	947.37	100.00
Fundulus heteroclitus (Mummichog)	410.73	172.50	107.43	690.66	72.90
Menidia menidia (Atlantic silverside)	57.35^{a}	33.79^{a}	6.68^{b}	97.81	10.32
Fundulus majalis (Striped killifish)	48.67^{a}	10.06^{b}	5.80^{b}	64.53	6.81
Brevoortia tyrannus (Menhaden)	15.78	4.72	31.13	51.63	5.45
Bairdiella chrysoura (Silver perch)	6.64^{a}	4.99^{a}	1.87^{b}	13.51	1.43
Anchoa mitchilli (Bay anchovy)	1.68^{ab}	9.32^{a}	0.24^{b}	11.25	1.19
Mugil cephalus (Striped mullet)	0.67	0.40	8.29	9.36	0.99
Leiostomus xanthurus (Spot)	0.88	1.65	2.07	4.60	0.49
Gobiosoma bosc (Naked goby)	0.24	0.74	0.00	0.98	0.10
Cynoscion regalis (Weakfish)	0.25	0.31	0.24	0.80	0.08
Paralichthys dentatus (Summer flounder)	0.16	0.57	0.00	0.73	0.08
Trinectes maculatus (Hogchoker)	0.46	0.18	0.00	0.65	0.07
Morone saxatilis (Striped bass)	0.07	0.29	0.14	0.50	0.05
Cyprinodontidae variegatus (Sheepshead minnow)	0.00	0.00	0.27	0.27	0.03
Caranx hippos (Crevalle jack)	0.00	0.07	0.00	0.07	0.01
Pomatomus salatrix (Bluefish)	0.00	0.05	0.00	0.05	0.01
Callinectes sapidus (Blue Crab)	20.26	15.89	12.03		

Table 2.3 Biotic index scores by shoreline type

	S. alterniflora	Riprap-sill	Riprap
Margalef's Richness Index	1.52	2.01	1.47
Shannon Diversity Index	1.05	1.29	1.21
Pielou's Evenness Index	0.92	1.05	1.12

Table 2.4 Fishes collected in minnow traps in the upper intertidal zone of each shoreline type in a tributary of the Delaware Coastal Bays, USA, from June-September 2010. “Mean % of Traps Present” is the mean percentage traps that contained each species on a given sampling day, across the sampling period. “% of Sampling Days Present” is the percentage of sampling days on which each species was collected at each shoreline type.

		S. alterniflora	Riprap-Sill	Riprap
Mean % of Trap Presence	Mummichog	69.2	66.3	64.8
	Striped killifish	9.8	7.6	1.1
	Silver perch	8.7	8.7	2.2
	Atlantic silverside	23.5	1.1	0
Total % Presence	Mummichog	100	100	87
	Striped killifish	34.8	21.7	4.3
	Silver perch	34.8	21.7	8.7
	Atlantic silverside	34.8	4.3	0

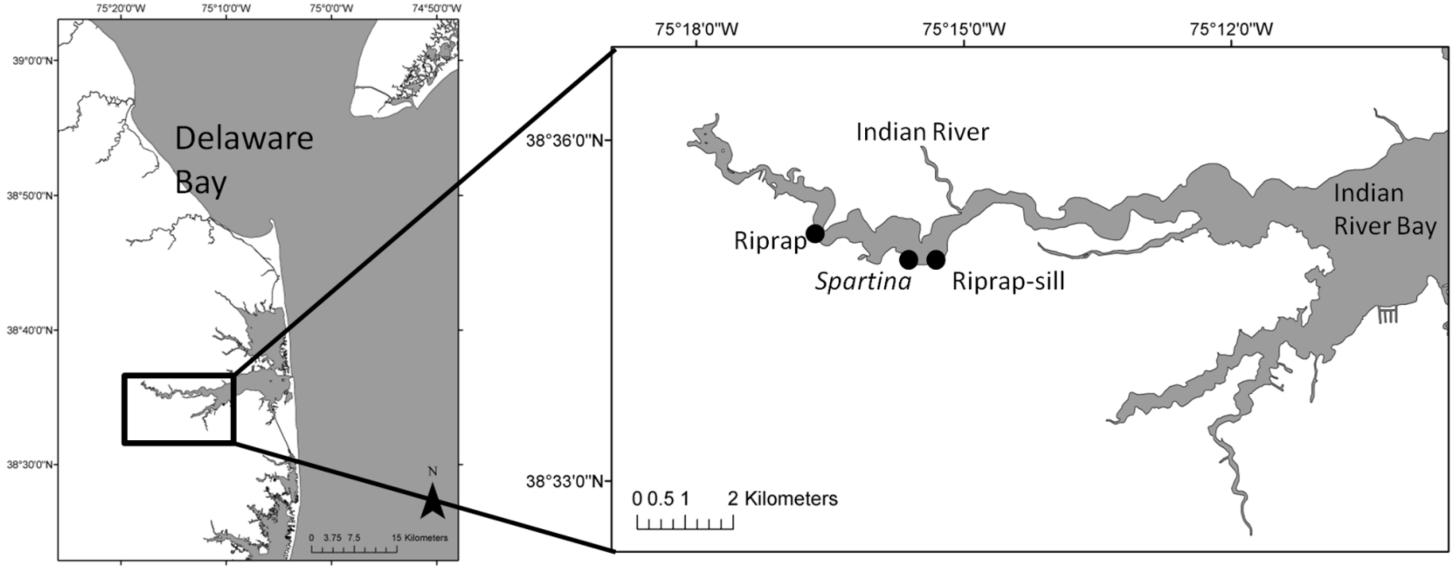

Figure 2.1 Sampling locations in Indian River, a tributary of the Delaware Coastal Bays, USA.

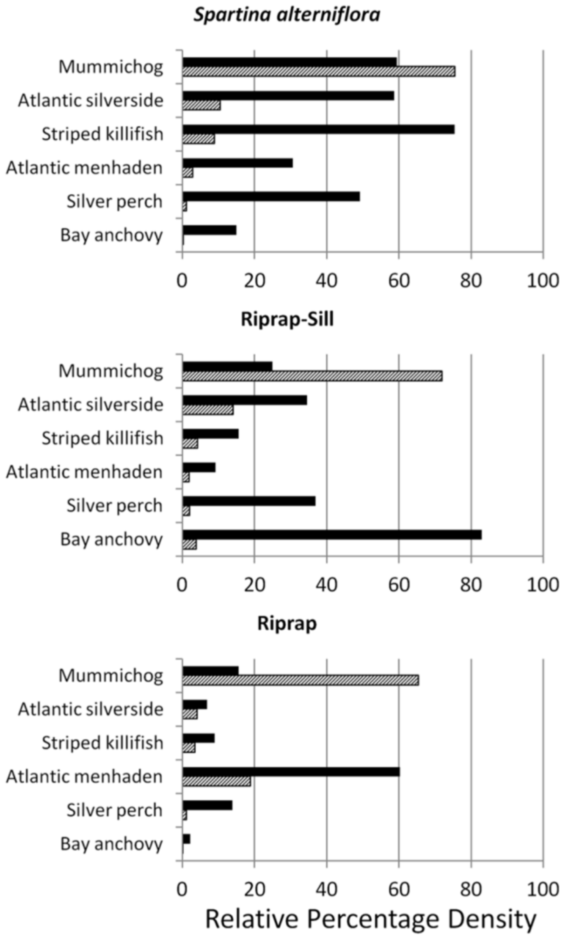

Figure 2.2 Relative density (individuals/m^3) of the most abundant fish species, by shoreline type. Crosshatched bars are the percentage of the total catch at that shoreline type which that species composed. The combined crosshatched bars within each shoreline type sum to ~ 95%, as all species collected are not included here. Therefore, long crosshatched bars indicate a species which is particularly prevalent in comparison with other species at that shoreline type. Black bars are the percentage of that species total catch (across all shoreline types) that was caught at that shoreline type. Therefore, long black bars a shoreline at which that species is particularly abundant. Therefore, if species were equally abundant along all five shoreline types, all black bars would be 33%.

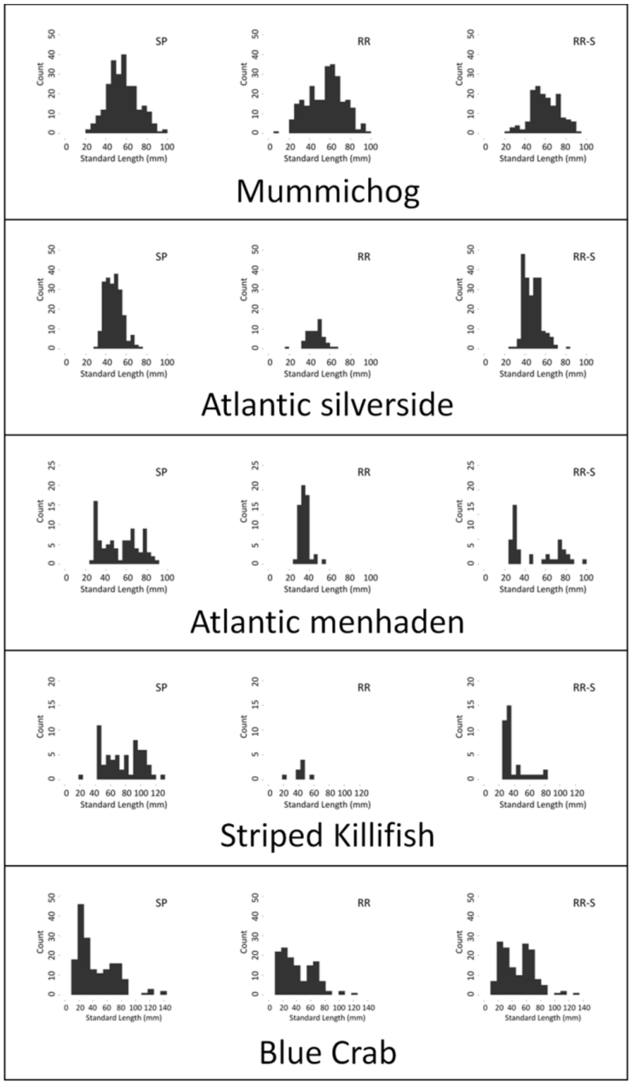

Figure 2.3 Length frequency histograms of mummichog, Atlantic silverside, Atlantic menhaden, striped killifish and blue crab by shoreline type (SP: S. alterniflora; RR-S: Riprap-Sill; RR: Riprap).

Chapter 3

THE EFFECT OF SHORELINE TYPE ON WATER TEMPERATURE AND DIEL-CYCLING HYPOXIA IN THE SHALLOW SUBTIDAL ZONE OF TIDAL TRIBUTARIES OF DELAWARE COASTAL BAYS

Introduction

Estuaries by their very nature have relatively quickly varying water conditions, including both water temperature and dissolved oxygen concentrations. This is attributable to many abiotic factors that act upon estuaries including tides, freshwater inputs, and wind mixing among others. Estuaries also have significant gradients in water temperature and dissolved oxygen concentrations spatially along gradients running their length, due in part to distance from inputs from the ocean. Despite the wide range of spatiotemporal water quality conditions, estuaries function as habitat for many types nekton. Due to the spatiotemporal range of water conditions found with a single estuary, microhabitats found within each estuary serve in particular as refugia for nekton. Certain microhabitats may enhance physiological processes such as providing optimum environmental conditions for growth (water temperature, dissolved oxygen concentrations) relative to other habitats. Alteration of an estuary’s shoreline through the installation of erosion-control structures and the spread of invasive grasses work to create further microhabitats.

For further details of how DO affects macrofauna please refer to the introduction of Chapter 1.

How nekton utilize each of these shoreline types (microhabitats) was explored in the prior two chapters. In this chapter, the water quality (specifically temperature and DO) along each of five shoreline types (bulkhead, riprap, sandy beach, *Phragmites australis, Spartina alterniflora*) is explored to see if differences in the quality of these microhabitats are driven by quantifiable differences in water quality, or if differences in water quality are negligible, and the physical structure of each shoreline type may be a more significant driver in determining nekton utilization.

Materials and Methods

In the summer, during periods of severe hypoxia, July 15th-September 9th, 2010, multi-parameter meters (sondes) were positioned in the subtidal zone immediately adjacent (< 3m) to each sampling station (and thus each shoreline type) at each tributary region, alternating at 2-week intervals (Figure 3.1). The majority of these sondes were within 500 m of one another. The farthest distance between two sondes within the same tributary region was 1.5 km. Each sonde was suspended approximately 15 cm above the substrate using a steel pyramid stanchion. Salinity (psu), dissolved oxygen (DO saturation % and mg/l), pH, and water temperature (°C) were recorded at 15-minute intervals. Sondes were cleaned weekly to remove biofouling accumulation. Water quality data were retrieved from each multi-parameter sonde biweekly by removing the sonde from its stanchion and downloading data. When recovered from the field, each sonde was cleaned and calibrated in the lab. Only newly calibrated sondes were positioned in the field to record measurements.

Water quality data (from both tributary regions as well as shoreline type comparison study) was analyzed using three different approaches. Data was plotted over time and distinct differences in temperature, salinity, and DO concentrations were visually noted among sampling stations. Direct statistical comparisons among sampling stations and shoreline types were only made where data was available from all locations to remove variability in daily and hourly temperature, salinity, and DO concentrations. The time each sampling station spent in “extreme” conditions of water temperature (> 30°C, > 31°C, etc.) and DO (< 4.8 mg O_2/l, < 2.3 mg O_2/l, < 0.4 mg O_2/l) were calculated. Thirdly, one-way repeated measures analysis of variance was used to compare temperature and DO among sampling stations. If significant differences were found the Holm-Sidak method of pairwise multiple comparison post-hoc test was performed. If parametric assumptions of normality or variance were not passed after data transformation, Friedman’s related samples two-way analysis of variance by rank was used to compare differences. If significant differences were found in this test, stepwise step-down multiple comparison post-hoc test were performed.

Results

As multi-parameter sondes were not positioned among tributary regions simultaneously, comparisons in water quality can only be made for shoreline type within tributary regions, not among tributary regions. Among shoreline types in IRL mean temperatures varied by 0.1°C among shoreline types, with riprap having the highest mean water temperature (Table 3.1). The maximum water temperature recorded was along the bulkhead shoreline. The lowest maximum temperature value was found along the beach shoreline. Temperatures over 32°C were only recorded along hardened and beach shorelines (Table 3.2). Riprap shorelines experienced the most time with water temperatures over 32°C. *Phragmites australis* and riprap shorelines had the greatest associated water temperatures (Friedman's; $p<0.01$) while bulkhead and beach shorelines had the significantly lowest water temperatures.

Among shoreline types in IRU, mean temperatures varied by 0.23°C, with *S. alterniflora* shorelines having the greatest mean temperature and bulkhead having the lowest mean temperature. The maximum water temperature recorded was along the *P. australis* shoreline at 32.6°C. The lowest maximum temperature value was along the bulkhead shoreline. Water temperatures greater than 32°C were only recorded at beach and unhardened shorelines, with beach shorelines experiencing them for the greatest percentage of time. Despite, the occasional high temperatures at the beach shorelines, repeated measures analysis showed riprap shoreline in IRU had significantly greater water temperatures (Friedman's; $p<0.01$) than other shoreline types. Beach and bulkhead shorelines had the significantly lowest water temperatures associated with them.

Among shoreline types in PCL, mean temperatures varied by 0.25°C, with beach shorelines having the greatest mean and maximum temperature and *S. alterniflora* shorelines having the lowest mean and maximum temperature. The beach shoreline spent the highest percentage of time with temperatures greater than 30°C of all the shorelines. Beach was also the only shoreline where temperatures exceeded 33°C. Beach shorelines had significantly greater water temperatures (Friedman's; $p<0.01$) than other shorelines. *Spartina alterniflora* shorelines had the significantly lowest water temperatures associated with them (Friedman's; $p<0.01$).

Among shoreline types in PCU, mean temperatures varied by 0.22°C among shoreline types with bulkhead having the greatest mean water temperatures and *S. alterniflora* having the lowest mean water temperatures. The maximum water temperature recorded was along the *P. australis* shoreline. The lowest maximum water temperature was along the bulkhead shoreline. Extreme temperatures varied greatly in PCU. Bulkhead shorelines had water temperatures greater than 30°C the greatest percentage of the time, though never getting above 33°C. Unhardened shorelines spent the greatest amount of time in extreme temperatures greater than 32°C. Bulkhead and riprap shorelines had the significantly greatest water temperatures (Friedman's; $p<0.01$) associated with them. Unhardened shorelines had the significantly lowest water temperatures associated with them (Friedman's; $p<0.01$).

Among shoreline types in IRL mean DO varied by 0.63 mg O_2/l among shoreline types, with the lowest mean DO located along the bulkhead shoreline and the greatest mean DO located along the *P. australis* shoreline (Figure 3.3, Table 3.2). The lowest minimum DO was found at the riprap shoreline (0.33 mg O_2/l). The greatest minimum DO was found at the beach shoreline (1.68 mg O_2/l). Water along the bulkhead shoreline spent the greatest amount of time at DO concentrations detrimental to fish growth (< 4.8 mg O_2/l). Water along the riprap shoreline spent the greatest amount of time at DO concentrations detrimental to fish survival (< 2.3 mg O_2/l). *Phragmites australis* and *S. alterniflora* shorelines had significantly greater DO concentrations (Friedman's; $p<0.01$) than any other shoreline, while the bulkhead shoreline had the significantly lowest DO concentrations.

Among shoreline types in IRL, between the hours of 0200 and 1000, mean DO varied by 0.5 mg O_2/l among shoreline types. The lowest mean DO was at the bulkhead shoreline. The greatest mean DO during this time period was along the *S. alterniflora* shoreline. *Spartina alterniflora* and riprap shorelines had the significantly greatest (Friedman's; $p<0.01$) DO concentrations during this time period while the bulkhead shoreline had the lowest DO concentrations.

Among shoreline types in IRU, mean DO varied by 2.96 mg O_2/l with the lowest mean DO occurring along the bulkhead shoreline and the greatest mean DO along the *S. alterniflora* shoreline (Figure 3.4). The lowest minimum DO was at the beach shoreline. The greatest minimum DO was at the riprap site. Water along the bulkhead shoreline spent the greatest amount of time at DO concentrations detrimental to fish growth. Water along the beach shoreline spent the greatest amount of time at DO concentrations detrimental to fish survival. The *S. alterniflora* shoreline had significantly greater DO concentrations (Friedman's; $p<0.01$) than any other shoreline while the bulkhead shoreline had the significantly lowest DO concentrations.

Among shoreline types in IRU, between the hours of 0200 and 1000, mean DO varied by 2.96 mg O_2/l among shoreline types. The lowest mean DO during this time period was at the bulkhead shoreline. The greatest mean DO during this time period was along the *S. alterniflora* shoreline. The bulkhead shoreline had the significantly lowest DO concentrations (Friedman's; $p<0.01$) while the *S. alterniflora* shoreline had the greatest DO concentrations.

Among shoreline types in PCL, mean DO varied by 1.07 mg O_2/l with lowest DO occurring along the *S. alterniflora* shoreline and the greatest DO occurring along the beach shorelines (Figure 3.5). The lowest minimum DO was at the beach shoreline. The greatest minimum DO was at the *S. alterniflora* site. Water along the *S. alterniflora* shoreline spent the greatest amount of time at DO concentrations detrimental to fish health. Water along the bulkhead shoreline spent the greatest amount of time at DO concentrations detrimental to fish survival. The beach shoreline had significantly greater DO concentrations (Friedman's; $p<0.01$) than any other shoreline while the *S. alterniflora* shoreline had the significantly lowest DO concentrations.

Among shoreline types in PCL, between the hours of 0200 and 1000, mean DO varied by 0.28 mg O_2/l among shoreline types. The lowest mean DO during this time period was at the bulkhead shoreline. The greatest mean DO during this time period was along the *S. alterniflora* shoreline. The bulkhead shoreline had the significantly lowest DO concentrations (Friedman's; $p<0.01$) while the *S. alterniflora* had the greatest DO concentrations.

Though DO data was collected for two weeks at five shoreline types at PCU, inadequate amounts of congruent (time-matched) data for all shoreline types were available for proper statistical analysis.

Comparing the rank of mean DO among shoreline types, beach and *P. australis* had the highest mean DO ranks while bulkhead had the lowest mean DO ranks (Table 3.3). Between the hours of 0200-1000, *S. alterniflora* had the highest mean DO ranks at all three tributary regions, while bulkhead shorelines had the lowest mean DO ranks at all three tributary regions (Table 3.4). For minimum DO values, *S. alterniflora* had the greatest minimum DO ranks and beach and bulkhead shorelines had the lowest mean minimum DO rank (Table 3.5).

Discussion

Temperature stress affects fish survival health and growth (see review Jobling 1981). Fish actively avoid regions of temperature that are harmful to their life processes. During the summer months in Delaware, water temperatures in estuarine waters may reach the high end of many species of fish thermal tolerance. Thus, if one shoreline type were shown to have higher water temperature adjacent to it than another, fish may actively avoid that area. They may potentially avoid it in two spatial dimensions. An individual or school of fish may seek to remain in the area of intertidal, shallow shore-zone waters to reduce predation risk, and if another shoreline type provides more tolerable water temperatures with the same water depth, then fish may simply move adjacent to this other shoreline type. The other possibility is that fish swim to deeper waters, where water temperatures will be cooler, but predation risk may be increased. In either case, fish may actively avoid a shoreline type due to increased water temperatures. In the first escape response, potential density-dependent issues arise as many fish school to the same region to avoid others. In the second escape response, predation mortality may increase.

However, data from this present research suggests shoreline type has little effect on water temperature in the subtidal region just offshore of shoreline structure. Though statistical differences in water temperatures were determined among shoreline types, the variability of these results, as well as the actual ecological significance of the magnitude of these differences, suggest overall strong similarity in water temperature among shoreline types. The minimal differences in mean water temperatures among shoreline types (0.25°C at the most) suggest that water is evenly mixed enough along each stretch of shoreline and that long-period differences in water temperature do not exist within tributary regions. Each tributary region had a different shoreline type (bulkhead, riprap, sandy beach, *S. alterniflora*, respectively) represent the greatest mean water temperature within its tributary region. Likewise, three different shoreline types (bulkhead, beach, *P. australis*) were represented in the greatest maximum water temperature from a tributary region. With regards to extreme water temperature events, of the four tributary regions studied, three different shoreline types (riprap, beach, *P. australis*) represented the shoreline spending the greatest amount of time with the greatest water temperature. Results from Friedman's test indicate that temperature varied among shoreline types differently in different tributary regions. Clearly no single shoreline type stands out as having distinct warm or cool water temperatures. As a result, it appears the shoreline type is not an effective indicator of shore-zone water temperature. It is therefore unlikely that differences in shore-zone fish and blue crab densities among shoreline types are temperature driven.

Similar to temperature stress, low dissolved oxygen may negatively impact fish and blue crab health. If water adjacent to one shoreline type were to exhibit greater DO during periods of diel-cycling hypoxia than water adjacent to another shoreline type, fish and crabs may preferentially utilize the shore-zone with greater DO. Data from this study demonstrates that DO appears to have some relationship to shoreline type, particularly related to diel-cycling hypoxia. Overall mean DO conditions varied among shoreline types within the different tributary regions sampled. Bulkhead shoreline had the lowest mean DO at both Indian River tributary regions, while *S. alterniflora* shorelines in these regions had the highest and second highest mean DO respectively. However, in PCL, the *S. alterniflora* shoreline had the lowest mean DO. Mean DO is not a very clear indicator of overall hypoxia, though. Diel-cycling hypoxia varies in magnitude by the amplitude and the wavelength of DO concentrations over the course of a day. It is the differences in the magnitude of DO flux that are most important to the health and survival of estuarine fish and crabs.

During periods of diel-cycling hypoxia, DO tends to be the lowest between the hours of 0200 and 1000 (Tyler et al. 2009). It is during these periods of time that low DO is of the most concern to the health of fish. In all tributary regions studied, bulkhead shorelines had the lowest mean DO of all shorelines between 0200 and 1000. *Spartina alterniflora* shorelines had the greatest mean DO at each of the tributary regions between 0200 and 1000. Likewise, bulkhead shorelines had among the lowest minimum DO readings at all three tributary regions between 0200 and 1000. *Spartina alterniflora* shorelines had the greatest or second greatest minimum DO values between 0200 and 1000. *Spartina alterniflora* shorelines also spent the least or second least amount of time with water with DO values detrimental to fish survival (< 2.3 mg O_2/l). However, all other shorelines fluctuated greatly in the comparative time spent with severely hypoxic DO, suggesting a minimal relationship of riprap, *P. australis*, and beach shorelines with DO.

Both *S. alterniflora* and bulkhead shorelines display a strong, consistent relationship with DO wherein DO concentrations appear to be greater between 0200 and 1000 along *S. alterniflora* shorelines than along any other shoreline type and lowest along bulkhead shorelines. *Spartina alterniflora* releases very little oxygen during dark hours and what little O_2 it does release is released into the soil where it is quickly consumed by bacteria (Sand-Jensen et al. 1982, Howes and Teal 1994). During daylight hours, any oxygen released by *S. alterniflora* into estuarine waters would be minimal in comparison with photosynthesis performed by algae. Denser growth of algae along a certain shoreline type would suggest greater magnitude diel-cycling hypoxia, as greater densities algae photosynthesize more during daylight, driving DO up, and respire more at night, driving DO down. Though not quantified in the field, it is possible that algae populations in the subtidal region near *S. alterniflora* shorelines are lower than along other shoreline types, as seen in Chesapeake Bay (Bradley and Seitz, in review). Alternatively, some other mechanism may be at work if, *S. alterniflora* shorelines retain higher concentrations of DO between 0200 and 1000. Future research should focus on more tributary regions from more creeks to extend the DO database. Multi-parameter sonde deployment should occur for shorter periods of time to keep data quality as high as possible.

The magnitude of diel-cycling hypoxia may be greatest at altered shorelines, with lower DO levels being reached. Native *S. alterniflora* shorelines have been shown to consistently have greater levels of DO associated with them. DO, a critical component of water quality at a habitat, is critical to the physiological health of estuarine fishes and invertebrates. Many species of fish have been shown to have negative physiological reactions to even small changes in DO at severly hypoxic conditions (Breitburg 1994, Wannamaker and Rice 2000, Taylor and Miller 2001, Robb and Abrahams 2003, Shoji et al. 2005, Stierhoff et al. 2006, Brady et al. 2009). As such, reduced levels of DO at altered shorelines may further exacerbate other stresses caused by shoreline type habitat quality on resident fauna.

Table 3.1 Water quality characteristics from shoreline hypoxia study 2010. Time matched data (only data used when available simultaneously from all data sondes at a specific time).

			Temperature (°C)		DO (mg O_2/l)		DO (0200-1000) (mg O_2/l)	
Creek	Station	Shoreline	Mean	Range	Mean	Range	Mean	Range
Indian River	IRU	*S. alterniflora*	27.59	23.07-32.01	10.86	1.22-22.23	7.05	1.22-14.94
		P. australis	27.51	22.96-32.6	9.04	0.89-18.89	5.85	0.89-10.08
		Beach	27.49	22.99-32.54	9.36	0.12-22.17	5.62	0.12-11.73
		Riprap	27.50	23.32-30.98	8.59	1.97-16.66	6.04	1.97-12.98
		Bulkhead	27.36	23.73-30.65	7.90	0.34-16.14	5.35	0.34-12
	IRL	*S. alterniflora*	27.67	24.9-32	5.42	0.95-10.04	4.43	1.34-8.2
		P. australis	27.71	24.98-31.96	5.49	0.53-9.46	4.25	0.53-7.1
		Beach	27.63	24.84-32.1	5.38	1.68-10.21	4.29	1.68-8.37
		Riprap	27.73	25.24-32.02	5.04	0.33-9.51	4.16	0.33-5.89
		Bulkhead	27.64	25.2-32.16	4.86	0.81-9.55	3.93	0.81-5.85
Pepper Creek	PCU	*S. alterniflora*	30.12	26.57-33.41	NA	NA	NA	NA
		P. australis	30.15	26.34-34.01	NA	NA	NA	NA
		Beach	30.14	26.99-32.93	NA	NA	NA	NA
		Riprap	30.29	27.6-33.13	NA	NA	NA	NA
		Bulkhead	30.34	27.54-32.89	NA	NA	NA	NA
	PCL	*S. alterniflora*	28.29	25.05-32.58	6.86	1.47-16.25	5.25	1.47-9.95
		P. australis	NA	NA	NA	NA	NA	NA
		Beach	28.54	25.82-34.46	7.93	0.38-21.38	5.07	0.38-14.42
		Riprap	28.47	25.5-32.72	7.59	0.64-18.27	5.18	0.64-13.43
		Bulkhead	28.38	25.67-32.67	7.31	0.46-19.57	4.97	0.46-12.2

Table 3.2 Percentage of water quality measurements in “extreme” ranges from 2010 for shoreline water quality study – time matched data. See Figure 1.4 for sampling station locations and Materials and Methods section for timing of water quality readings.

	Indian River - Lower					**Indian River - Upper**				
	Beach	Bulkhead	*P. australis*	Riprap	*S. alterniflora*	Beach	Bulkhead	*P. australis*	Riprap	*S. alterniflora*
Temperature										
% measurements >30C	6.84	6.08	6.84	6.58	6.84	7.26	1.69	7.02	4.24	7.14
% measurements >31C	2.91	2.66	2.53	2.41	2.66	1.82	0.00	1.82	0.00	1.94
% measurements >32C	0.13	0.13	0.00	0.51	0.00	0.61	0.00	0.48	0.00	0.12
% measurements >33C	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
% measurements >34C	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dissolved Oxygen										
% measurements <4.8 mg/l	45.70	47.63	36.65	44.36	44.36	14.80	16.91	10.70	10.04	3.83
% measurements <2.3 mg/l	1.34	1.48	2.08	6.08	1.19	6.47	4.62	2.38	0.13	0.53
% measurements <0.4 mg/l	0.00	0.00	0.00	0.45	0.00	3.96	0.13	0.00	0.00	0.00
Dissolved Oxygen (0200-1000)										
% measurements <4.8 mg/l	69.91	79.17	72.69	70.83	67.59	33.71	36.33	26.59	26.97	10.86
% measurements <2.3 mg/l	3.24	3.70	4.17	9.26	2.31	15.73	10.86	6.37	0.37	1.50
% measurements <0.4 mg/l	0.00	0.00	0.00	1.39	0.00	10.49	0.37	0.00	0.00	0.00

Table 3.2 Continued

	Pepper Creek - Lower					Pepper Creek - Upper				
	Beach	Bulkhead	*P. australis*	Riprap	*S. alterniflora*	Beach	Bulkhead	*P. australis*	Riprap	*S. alterniflora*
Temperature										
% measurements >30C	11.28	10.25	NA	9.97	9.23	53.48	62.82	56.96	59.71	54.76
% measurements >31C	4.75	4.10	NA	3.82	3.54	25.64	25.64	22.89	26.01	21.06
% measurements >32C	1.86	1.68	NA	1.68	1.12	6.23	4.21	6.78	6.41	4.76
% measurements >33C	0.65	0.00	NA	0.00	0.00	0.00	0.00	0.37	0.37	0.55
% measurements >34C	0.19	0.00	NA	0.00	0.00	0.00	0.00	0.18	0.00	0.00
Dissolved Oxygen										
% measurements <4.8 mg/l	21.41	22.23	NA	19.77	23.87	NA	NA	NA	NA	NA
% measurements <2.3 mg/l	3.28	4.30	NA	3.69	1.33	NA	NA	NA	NA	NA
% measurements <0.4 mg/l	0.10	0.00	NA	0.00	0.00	NA	NA	NA	NA	NA
Dissolved Oxygen (0200-1000)										
% measurements <4.8 mg/l	45.76	40.91	NA	38.79	43.64	NA	NA	NA	NA	NA
% measurements <2.3 mg/l	9.39	10.91	NA	10.00	3.94	NA	NA	NA	NA	NA
% measurements <0.4 mg/l	0.30	0.00	NA	0.00	0.00	NA	NA	NA	NA	NA

Table 3.3 Mean DO ranks by shoreline type at each tributary region. Rank of 1 is the lowest mean DO value, rank of 5 is greatest mean DO value (PCL only to 4).

Shoreline	Station	Rank
S. alterniflora	IRL	4
	IRU	5
	PCL	1
P. australis	IRL	5
	IRU	3
	PCL	NA
Beach	IRL	3
	IRU	4
	PCL	4
Riprap	IRL	2
	IRU	2
	PCL	3
Bulkhead	IRL	1
	IRU	1
	PCL	2

Table 3.4 Mean DO ranks by shoreline type at each tributary region from between the hours of 0200 and 1000. Rank of 1 is the lowest mean DO value, rank of 5 is greatest mean DO value (PCL only to 4).

Shoreline	Station	Rank
S. alterniflora	IRL	5
	IRU	5
	PCL	4
P. australis	IRL	3
	IRU	3
	PCL	NA
Beach	IRL	4
	IRU	2
	PCL	2
Riprap	IRL	2
	IRU	4
	PCL	3
Bulkhead	IRL	1
	IRU	1
	PCL	1

Table 3.5 Minimum DO ranks by shoreline type at each tributary region. Rank of 1 is the lowest minimum DO value, rank of 5 is greatest minimum DO value (PCL only to 4).

Shoreline	Station	Rank
S. alterniflora	IRL	4
	IRU	4
	PCL	4
P. australis	IRL	2
	IRU	3
	PCL	NA
Beach	IRL	5
	IRU	1
	PCL	1
Riprap	IRL	1
	IRU	5
	PCL	3
Bulkhead	IRL	3
	IRU	2
	PCL	2

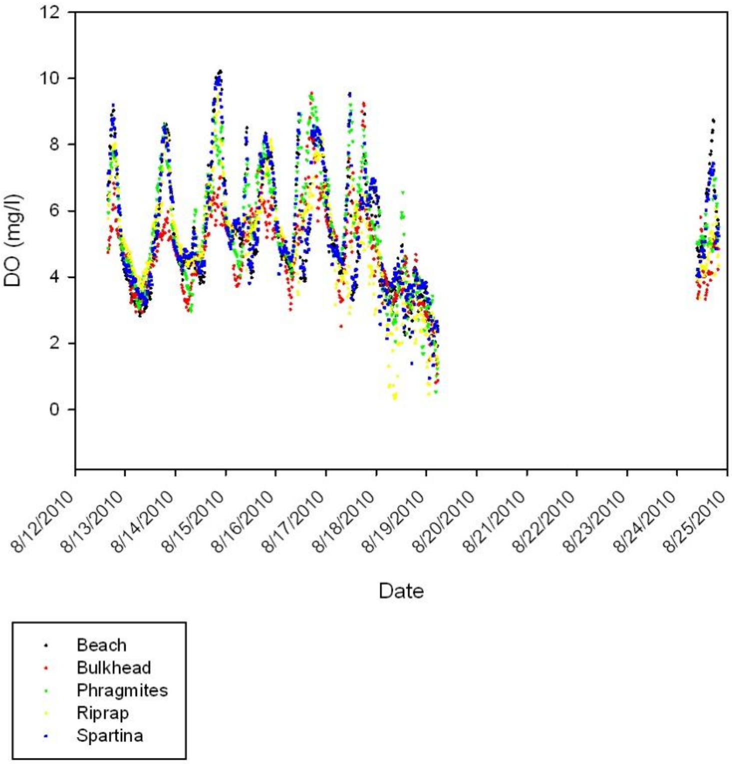

Figure 3.1 Dissolved oxygen in subtidal region along five shoreline types in lower Indian River, summer 2010.

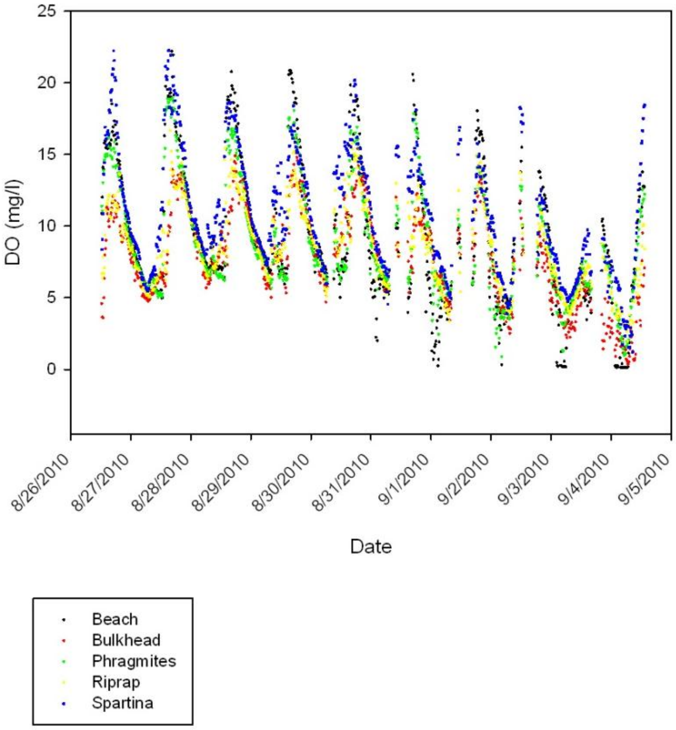

Figure 3.2 Dissolved oxygen in subtidal region along five shoreline types in upper Indian River, summer 2010.

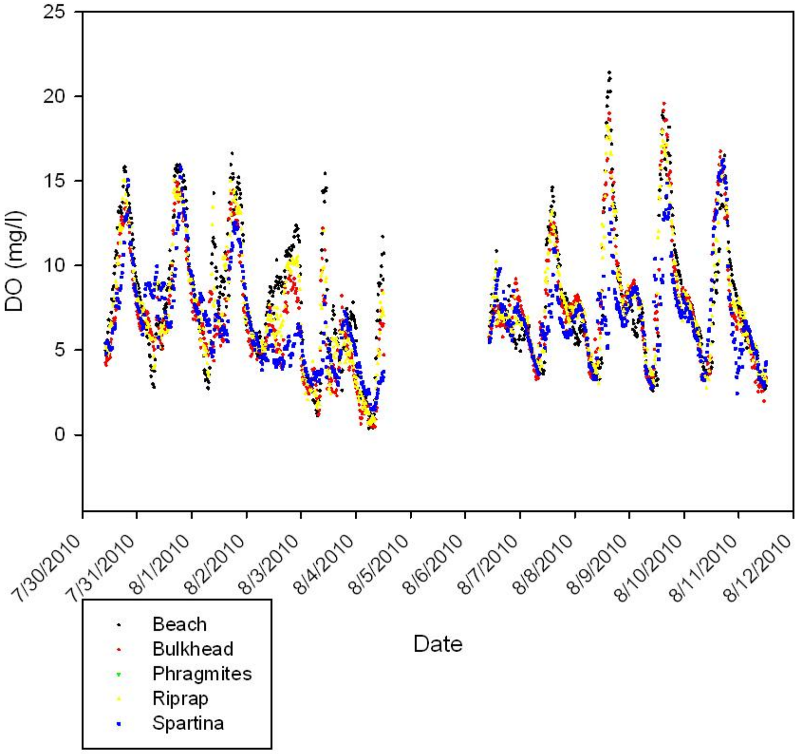

Figure 3.3 Dissolved oxygen in subtidal region along five shoreline types in lower Pepper Creek, summer 2010.

Chapter 4

EGG DEPOSITION BY ATLANTIC SILVERSIDE, MENIDIA MENIDIA: SUBSTRATE UTILIZATION AND COMPARISON OF SHORELINE TYPE

Introduction

The intertidal zone of estuaries serves as important spawning habitat for many species of fish (see review by DeMartini 1999). The physical structure of estuarine habitats, particularly the shore-zone continues to be modified by human activity. During the past few decades the area of intertidal fringing salt marsh comprised of *Spartina alterniflora* (smooth cordgrass) has decreased in US Mid-Atlantic estuaries, while invasive *Phragmites australis* (common reed) (King et al. 2007) and several types of shoreline hardening including bulkhead, riprap and riprap-sills have become more common (USGS 2011). Shoreline hardening has greatest effects on the zone of transition, including the intertidal and shallow subtidal zones, the most ecologically productive regions in estuaries (Toft et al. 2007; Bilkovic and Roggero 2008). Riprap and bulkhead structures have been shown to negatively impact, through reduced diversity, growth and abundance, a range of populations of fauna (primarily fishes and benthic invertebrates) living adjacent to them (Able et al. 1998; Seitz et al. 2006; Bilkovic and Roggero 2008; Pister 2009). Bulkhead and riprap structures alter the natural shape of the shoreline, remove undercut crevice habitat, change shore-zone wave dynamics, reduce shallow-water habitat and reduce or eliminate intertidal plant communities.

Disturbance of natural shorelines by shoreline hardening is one of the primary contributors to the expansion of a non-native genotype of *P. australis* along the Mid-Atlantic coast of the US (King et al. 2007). *Phragmites australis* has led to a displacement of native macrophyte communities (such as *S. alterniflora*), degradation of wildlife habitat and alteration of ecosystem processes (Weinstein and Balletto 1999; Minchinton et al. 2006). *Phragmites australis* dominated marshes are utilized by estuary-resident fishes for feeding and spawning (Able and Hagan 2003). However, due to increased elevation of established *P. australis* marshes, the length of time the intertidal marsh interior is submerged is reduced, resulting in a decrease in usage by estuarine fishes (Weinstein and Balletto 1999; Able and Hagan 2003; Able et al. 2003). Reduced access to intertidal area reduces viable locations for intertidal spawning.

Understanding how physical and biological structure of the shoreline and intertidal zone effect utilization by shore-zone biota for spawning habitat is important to further our understanding of the impacts of shoreline modification on estuarine systems. Hardened shorelines and invasive marsh plants represent an important spatial feature which influences spawning in the intertidal zone. Despite the magnitude of this issue, there is little information on the impact of shoreline habitat alteration effects on spawning by intertidally spawning fishes.

Menidia menidia is among the most abundant forage fish species in US Mid-Atlantic estuaries (De Sylva et al. 1962; Richards and Castagna 1970; Able and Fahay 2010). The importance of *M. menidia* as a food source for piscivores such as striped bass (*Morone saxatilis*), Atlantic mackerel (*Scomber scombrus*), bluefish (*Pomatomus salatrix*) and other fishes (Merriman 1941; Bayliff 1950; Bigelow and Schroeder 1953; Schaefer 1970) is well documented. Spawning of *M. menidia* occurs between April and July in the Mid-Atlantic (Middaugh et al. 1981). Spawning occurs in the intertidal zone, where fish lay demersal, filamentous, adhesive eggs at high tide

(Kuntz and Radcliffe 1917; Hildebrand 1922; Wang 1974; Middaugh 1981). Just before high tide, schools of *M. menidia* swim along the shoreline until a suitable spawning substrate is encountered (Middaugh et al. 1981). The adhesive eggs are laid approximately 1.2 to 2.4 m above mean low water to reduce exposure to aquatic predators (Middaugh et al. 1981; Tewksbury and Conover 1987) and birds (Middaugh 1981). Numerous substrates for egg attachment have been noted in earlier studies, including: submerged vegetation (Bayliff 1950), eelgrass (Middaugh 1981), *S. alterniflora* (Middaugh et al. 1981), filamentous algae (Conover and Kynard 1984), sand (Wang 1974) and beach trash (Nichols 1908). Middaugh et al. (1981) reported *S. alterniflora*, detrital mats and abandoned crab burrows to be the most common sites of *M. menidia* egg attachment in an estuary in South Carolina. In a Massachusetts estuary *M. menidia* eggs were found to be attached only on filamentous algae (*Pilayella littoralis* and *Enteromorpha spp.*), despite the presence of many other substrates (Conover and Kynard 1984). *Menidia menidia* eggs have not been quantitatively collected for assessment of density across available egg attachment substrata.

The objectives of this study were to (1) compare egg deposition by *M. menidia* along six shoreline types (bulkhead, riprap, riprap-sill, sandy beach, *P. australis* and *S. alterniflora*), (2) compare substrate utilization for egg attachment, and (3) determine if a relationship exists among shoreline type and substrate type for egg deposition/attachment and therefore determine preferential spawning habitat for *M. menidia* in a US Mid-Atlantic estuary.

Materials and Methods

Study Area

Sampling occurred inside Roosevelt Inlet, located ~ 5 km from the mouth of the Delaware Bay in Lewes, Delaware, USA (Figure 4.1). The sampling area is polyhaline (22-27 psu,) and tides are semidiurnal. Egg collections were made at stations along six shoreline types (bulkhead, riprap, riprap-sill, sandy beach, *P. australis* and *S. alterniflora*) (Figure 4.1). Low, mid and high intertidal elevations at each sampling site were demarcated using neap tide water level on March 23, 2010, with low intertidal marked at low water level, mid intertidal marked three hours after low tide, and high intertidal marked at high tide.

Bulkhead shorelines (constructed in 1974) consisted of the vertical bulkhead structure, constructed of wood or metal, depending on station. Riprap shorelines (constructed in 1974) were comprised of large rocky structure directly positioned along the shoreline extending from the low to high intertidal zone. The riprap-sill shoreline (constructed in 2009) was comprised of a riprap structure (composed of large rock) positioned low in the intertidal zone with *S. alterniflora* planted in the mid intertidal zone, landward of the riprap structure and *Spartina patens* (saltmeadow cordgrass) planted in the high intertidal zone. Plantings occurred at the time of construction; *S. alterniflora* had grown to 60 cm and occurred at about 75% of average natural density, *S. patens* had grown to 30 cm and occurred in approximately natural density at the time of the present study. *Phragmites australis* has been expanding its range within the sampling area and surrounding estuary for several decades (Bruce Campbell, personal communication). The subtidal zones at all shoreline types are characterized by generally featureless, slowly sloping, sandy to muddy substrate.

Substrate for egg attachment varied, in both variety and density, among shoreline types. All bulkhead stations were covered with large quantities of the green filamentous alga *Enteromorpha spp.* (most likely *Enteromorpha intestinalis*). Barnacle colonies and small amounts of the brown alga *Fucus spp*. occurred in the lower intertidal zone of most bulkhead stations. Most rock surfaces on riprap shorelines had extensive coverage by *Enteromorpha spp.*, with *Fucus spp*. found slightly lower in the intertidal zone. The green alga *Ulva lacuta* was present in low densities in both the mid and lower intertidal zones of riprap stations. Small tufts of *S. alterniflora* were present at many riprap stations. Blue mussels (*Mytilus edulus*) were present in the lower intertidal area of many riprap stations. Detritus (primarily dead *P. australis*, loose *Enteromorpha spp.* and beach trash) was found at the high tide wrack lines most sampling days at all riprap stations. Riprap-sill station's substrates included *Enteromorpha spp.* on the rocks of the riprap structure found in the mid to lower intertidal zone as well as on the sparse, recently planted *S. alterniflora* in the mid to upper intertidal zone. Large quantities of detritus were present in the *S. alterniflora* dominated mid to upper intertidal. Slightly higher in the intertidal zone, recently planted *S. patens* was present at all riprap-sill stations. Sandy beach stations offered the least variety of substrata for *M. menidia* spawning; all were primarily featureless, with a few stations having sparse shoots of *P. australis* in the upper intertidal zone. Loose *Enteromorpha spp.*, dead shoots of *P. australis* and beach trash were commonly present on the sand in the high tide wrack line. *Phragmites australis* stations included dense stands of *P. australis* with occasional shoots of *S. alterniflora* both of which had *Enteromorpha spp.* growing densely near the base of stems. Detritus was found at the high tide wrack line of all *P. australis* stations. Blue mussels occurred sparsely in the lower intertidal zone. Many tunnels from Atlantic marsh fiddler crab (*Uca pugnax*), were found in the peat below *P. australis* stands. *Spartina alterniflora* stations had nearly all substrate types found at other shoreline types. *Enteromorpha spp.* was abundant between *S. alterniflora* stems, and *U. lacuta* and *Fucus spp.* were dispersed throughout stations in low abundance. *Spartina patens* was found in the upper intertidal zone and dense blue mussel beds occurred in the lower intertidal zone of some *S. alterniflora* stations. Tunnels from *U. pugnax* were present in the intertidal zone of several *S. alterniflora* stations. Detritus was found at the high tide wrack line of all *S. alterniflora* stations. Although substrate availability differed greatly among shorelines and stations, *Enteromorpha spp.* was found, in some density, at all sampling stations.

Egg Collections

Eggs were collected from shoreline types with a minimum length of 40 m (mean = 88 m, SE= 11 m). Eight sampling stations were established parallel with the shoreline at each of the six shoreline types (Figure 4.1) (n=48 sampling stations). Stations were positioned >4 m apart from one another and >4 m from the edge of that shoreline type. Stations were 3 m wide and extended perpendicular to the waterline from the mean low water mark to the highest high tide elevation of the sampling day.

Stations were sampled from April 14th – June 10th, 2010 (n=120 for each shoreline type). *Menidia menidia* eggs were identified using morphological features and egg deposition location (Hardy 1978; Able and Fahay 1998). Three stations at each shoreline type were randomly selected each day, for a total of 18 stations sampled per day. Incubation time for eggs at the water temperatures of this study range from 10 to 28 days (Martin and Drewry 1978); no sampling station was sampled at a timeframe greater than seven days during the study, making it unlikely eggs were deposited and hatched in this interval. Sampling began at least 2 hours after the first daytime high tide. Density of *M. menidia* eggs at each shoreline type was assessed using a 3 m x ¼ m PVC quadrat. The sample quadrat was placed at the high tide mark (noted by wrack and/or direct observation), parallel lengthwise to shore, at the selected station and the enclosed area was visually examined for eggs. If no eggs were present, the quadrat was moved down in ¼ m steps, toward the waterline, thus covering the entire elevation of the station until eggs were encountered or the waterline was reached. If eggs were encountered (eggs were frequently deposited in distinct <1/4 m wide bands parallel to shore) the quadrat was positioned such that it enclosed the area of the station with the densest abundance of eggs. All shoreline types were sampled consistently by objectively selecting the most representative area of high egg density within the sampling station for collections. Attempts at randomizing collection area by elevation resulted in detrimentally reduced egg collections. All eggs within the quadrat were collected by hand removal and placed in a jar with seawater labeled by the substrate the eggs were collected from. Due to the filaments of *M. menidia* eggs, substrate was often removed along with the eggs during collections. Removal of substrate was minimized through careful, deliberate collection of eggs and the use of scalpels to carefully remove plant stems or algae when required. Prior to egg collection, the two-dimensional areal percent coverage of *Enteromorpha spp.* (to 10% intervals) was visually estimated within each quadrat. When walking in *P. australis* or *S. alterniflora*, care was taken to minimally disturb the marsh.

In the laboratory, eggs were placed in 95% ethanol for 15 minutes until they became opaque. Eggs were enumerated and the substrate they were attached to was noted. When excessively large numbers of eggs were collected, eggs were estimated using egg volume in group sizes of 200. Analysis of 30 estimated groups of 200 eggs proved this method to be within 8% of the actual value.

Maxim iButton® thermochrons were positioned (using the previously described elevations) at low, mid and upper intertidal elevations at bulkhead, riprap, sandy beach, *P. australis*, and *S. alterniflora* shorelines. Thermochrons collected atmospheric/water temperatures every 15 minutes from April 17th – June 18th, 2010.

Data Analyses

Non-parametric analysis of variance tests (Kruskal-Wallis) were used to test the effect of shoreline type and substrate type on egg density. Non-parametric analysis of variance tests (Kruskal-Wallis) were used to test the effect of tidal height, moon phase, daily water temperature, percentage *Enteromorpha spp.* coverage, and days since stations had been last sampled on egg density. Predicted tidal height and moon phase data were obtained from Tides and Currents Pro © (v. 3.3). Non-parametric tests were used due to the large variation in egg density from day to day, shoreline to shoreline, and station to station. Transformations were not successful in normalizing egg density data. Friedman's two way analysis of variance was used to test differences in intertidal atmospheric/water temperature among shoreline types over time. A stepwise step-down multiple comparison post-hoc test, adopted from a method from Campbell and Skillings (1985), was performed to determine homogeneous subsets in each non-parametric analysis. Statistical analyses were performed using SPSS (v. 18.0.2).

Results

Egg Deposition by Shoreline Type

Eggs were present at all six shoreline types during the study with greater than 93% of the over 3 million *M. menidia* eggs collected deposited on *S. alterniflora* shorelines (Table 4.1). Eggs were present within the sampling area from April 14th - May 27th, 2010. Significant differences in mean rank sum of egg density existed between shoreline types with *S. alterniflora* and riprap-sill shorelines having significantly greater mean rank sums of egg densities ($p<0.01$) than all other shoreline types (Table 4.1). Riprap shorelines had significantly greater mean rank sums of egg densities than beach, *P. australis* and bulkhead shorelines. The beach shoreline had significantly greater mean rank sums of egg densities than bulkhead shorelines. The lack of significant difference in egg density between *S. alterniflora* marsh and riprap-sill structures can be explained by the method in which the Kruskal-Wallis test computes similarity using mean rank sums. High variability in egg density among stations at *S. alterniflora* shorelines (Figure 4.2) generated a wide range of ranks for *S. alterniflora* shorelines. The riprap-sill shoreline had very low variability generating relatively even, mid level ranks, which contributed to the grouping of the two shoreline types. Within shoreline types, only *S. alterniflora* shorelines had different mean rank sums of egg densities among its respective sampling stations ($p<0.01$). Bulkhead ($p=0.483$), riprap ($p=0.799$), riprap-sill ($p=0.649$), beach ($p=0.970$) and *P. australis* ($p=0.450$) shorelines had no differences in egg density among sampling stations.

Egg Attachment by Substrate Type

Enteromorpha spp. was the most commonly utilized substrate for *M. menidia* egg deposition at all six shoreline types (Table 4.2). Over 95% (>2,900,000) of eggs collected had been deposited on *Enteromorpha spp.* Other substrates utilized for egg deposition by *M. menidia* included *S. alterniflora* stems (4%), detritus (<0.1%), *Fucus spp.* (<0.1%), and *Ulva lacuta* (<0.1%). *Enteromorpha spp.* had significantly greater mean rank sums of egg densities than all other substrates (p<0.01), and all other substrates were utilized similarly.

All eggs collected from bulkhead and beach shorelines were deposited on *Enteromorpha spp.* (Table 4.2). Utilization of *Enteromorpha spp.* for egg attachment was also very high at riprap (96% of eggs), riprap-sill (98%), *P. australis* (99%) and *S. alterniflora* (91%) shorelines, where egg densities were significantly greater than on any other substrate type (p<0.01). At riprap shorelines *Fucus spp.* was used in addition to *Enteromorpha spp.* At the riprap-sill and *P. australis* shorelines detritus was used in addition to *Enteromorpha spp.* At the riprap-sill shoreline, despite access to *Enteromorpha spp.* covered rock and *S. alterniflora* stems on the landward side of the riprap structure, eggs were only deposited on the open water side of the riprap structure. At *S. alterniflora* shorelines, a greater number of substrate types were used for egg deposition than at any other shoreline type. In addition to dominance of egg attachment on *Enteromorpha spp.*, the base of *S. alterniflora* stems (p<0.01) was utilized more than other substrates, including *U. lacuta*, detritus, and *Fucus spp.*

The areal coverage of *Enteromorpha spp.* differed among shoreline types and stations (Figure 4.3). *Enteromorpha spp.* coverage was significantly greatest at riprap-sill (75% mean coverage) and riprap (70%) shorelines (p<0.01). *Spartina alterniflora* (52%) had the next greatest coverage of *Enteromorpha spp.*, and this was significantly greater than at the remaining shoreline types. *Enteromorpha spp.* coverage's at bulkhead (39%), *P. australis* (31%) and beach (7%) shorelines were all significantly different from one another (Figure 4.3).

Egg density was not equal across the range of *Enteromorpha spp.* coverage. Stations with greater coverage by *Enteromorpha spp.* had significantly greater egg densities than station's with lower coverage's ($p<0.01$). Stations with *Enteromorpha spp.* coverage percentages of 50%, 60% and ≥70% grouped into a subset with the highest mean egg density ranks (Figure 4.4). All coverage's below 30% grouped into the lowest mean egg density subset. However, when analyzed within individual shoreline types, the distribution of egg density did not differ across *Enteromorpha spp.* coverage percentages at five of the six shoreline types: bulkhead ($p=1.0$), riprap ($p=0.289$), beach ($p=0.901$), *P. australis* ($p=0.935$), and *S. alterniflora* ($p=0.551$). Only at the riprap-sill shoreline did egg density differ significantly among *Enteromorpha spp.* coverage s ($p=0.041$), although post-hoc tests reveal that only the subset containing the 60% coverage was higher than the others. Mean percentage of eggs found attached to *Enteromorpha spp.* was greater than was the coverage percent of *Enteromorpha spp.* at all shoreline types. Though direct measurements of other substrate coverage's were not conducted, no other substrate had a greater percentage of eggs attached to it than that substrates coverage percentage within a quadrat.

The percentage of *Enteromorpha spp.* coverage within sampling stations did not change over time. Although Kruskal-Wallis analysis did find significant differences in *Enteromorpha spp.* coverage's among sample dates ($p<0.01$), a Spearman's rho correlation did not reveal any statistical trends ($\rho=0 .086$, $p=0.135$). Furthermore, visual inspection of plotted *Enteromorpha spp.* coverage percentages over time demonstrates that coverage was not reduced during sampling at any shoreline type.

Abiotic Characteristics

Egg removal during sampling did not reduce egg numbers at a given station on subsequent sampling days as no relationship existed between the number of days since a station had been previously sampled and egg density (p=0.822). Height of the high tide immediately prior to sampling was not significantly related to egg density across shorelines (p=0.052). When analyzed within shoreline type, high tide height immediately prior to sampling was not significantly related to egg density at bulkhead (p=0.995), beach (p=0.153) or *P. australis* (p=0.433) shorelines. However, tidal height was significantly related to egg density at riprap (p<0.01), riprap-sill (p<0.01), and *S. alterniflora* (p<0.01) shorelines, with higher tidal heights having greater egg densities. Similar results were found for maximum height of high tide at 24, 48 and 72 hours prior to sampling. Moon phase was not significantly related to egg density (p=0.867). Atmospheric/water temperature differed among the five shoreline types (bulkhead, riprap, beach, *P. australis*, *S. alterniflora*) at which thermochrons were present during the sampling period. Significant differences in mean ranks of temperature were found (p<0.01) when all 15 locations (low, mid, and upper intertidal elevations at 5 shoreline types) were analyzed. Upper intertidal locations had significantly greater atmospheric/water temperatures than those at the lower intertidal locations across all shoreline types.

82% of days when eggs were collected had maximum daily atmospheric/water temperatures < 19°C. The highest atmospheric/water temperature on a day when eggs were collected was 34.1°C. Temperature loggers positioned at the mid intertidal elevation most closely represented the elevation at which *M. menidia* eggs were deposited. Mean atmospheric/water temperature at the mid intertidal elevation on dates when eggs were found was significantly different ($p<0.01$) among shoreline types (Table 4.3). When weighting the mean atmospheric/water temperature by egg density on the day eggs were found, cooler temperatures were generally noted, except at the *S. alterniflora* shoreline: *S. alterniflora* (18.9°C), *P. australis* (14.7°C), beach (15.1°C), riprap (14.8°C), bulkhead (14.5°C). The maximum temperature recorded at the mid intertidal elevation when eggs were found was at the *P. australis* shoreline (Table 4.4).

Efforts were made to locate eggs for four weeks after the last egg was collected from the sampling area. These efforts were unsuccessful both in and around the sampling area; indicating that *M. menidia* egg deposition had ended within the sampling area for the spawning season.

Discussion

Egg Deposition by Shoreline Type

The present results demonstrate that hardened and *P. australis* invaded shorelines support substantially reduced densities of *M. menidia* eggs compared with *S. alterniflora* shorelines. Approximately 94% of eggs collected were found at *S. alterniflora* stations suggesting depositional preference for the native, vegetated, unhardened shoreline type. According to the description of *M. menidia* spawning behavior reported by Middaugh et al. (1981), schools of adults actively choose the location of egg deposition. This concept is well supported by data from the present study.

High variability of egg density was found within *S. alterniflora* stations, including at adjacent stations, in this present study (Figure 4.2). The degree of variability in egg density among *S. alterniflora* stations contrasts with that at other shoreline types. Physical differences in these *S. alterniflora* stations, particularly the elevation of the marsh plain, may have strongly contributed to this variability in egg density. *Menidia menidia* have been found to spawn at high tide on *S. alterniflora* marsh surface where water depth is only 0-30 cm (Middaugh et al. 1981).Tewksbury and Conover (1987) noted certain locations within in their sampling area to consistently be areas of intense *M. menidia* spawning activity, surrounded by areas of much lower spawning activity. This likely reflects the searching behavior exhibited by *M. menida* schools (Middaugh et al. 1981) as *S. alterniflora* marshes exhibit variable elevation and substrate availability.

Hardened shorelines homogenize the intertidal zone and reduce variability in substrate and elevation, potentially removing areas suitable for high *M. menidia* spawning activity. Riprap and riprap-sill stations were comparable physically, and biologically homogenous with one another (with respect to genera and coverage of macroalgae available in the mid intertidal zone). This similarity was reflected in a 4% difference in egg density between these two shoreline types. The physical structure of bulkhead shorelines is the most unique of any of the sampled shorelines, and had by far the fewest eggs deposited. *Menidia menidia* avoided the vertical structures, both wood and metal, of bulkheads for egg deposition. This result is in contrast with earlier studies in which *M. menidia* were reported to utilize the algae covered vertical structure of a floating dock as well as the vertical structure provided by crab burrows for egg deposition (Moore 1980; Middaugh et al. 1981). The sandy beach shoreline, like the bulkhead shorelines, consisted of very homogenous structure. Eggs were deposited evenly among the beach stations, with very low variability in egg density. All eight sandy beach stations had eggs present on a similar percentage of sampling days, however egg density was low across beach stations, likely due in part to limited occurrence of *Enteromorpha spp.*

Among the shoreline types sampled in the present study, marsh shorelines comprised of well-established invasive *P. australis* have physical and biological structure most similar to S. alterniflora marsh. Variability in shoreline structure and substrate availability was higher at *P. australis* shorelines than at hardened shorelines, and this is reflected in the high variability in egg density at the *P. australis* shoreline. However, total egg density along the *P. australis* shoreline was more similar to that at hardened shorelines than that at *S. alterniflora* shorelines. One factor which may have contributed to disparate egg densities between *S. alterniflora* marsh and *P. australis* marsh is elevation of the marsh plain. Larger areas of *S. alterniflora* marsh are inundated at high tide compared to *P. australis* stands. The higher elevation of established *P. australis* marshes has been shown to reduce fish movement into the marsh interior (Weinstein and Balletto 1999; Able and Hagan 2003; Able et al. 2003). *Phragmites australis* also grows less densely than does *S. alterniflora* at the marsh surface, potentially reducing predation protection for *M. menidia* embryos.

Spawning site selection is likely impacted by unique current flow and wave action among the shoreline types studied. Dense stands of *S. alterniflora* potentially provide the greatest amount of wave attenuation in the mid intertidal zone, while hardened shorelines provide the least amount of wave attenuation. Wave energy has been noted to be one of the most important physical factors affecting intertidal organisms on hardened shorelines (Southward and Orton 1954; Denny and Wethy 2001; Jonsson et al. 2006) and may be a factor in *M. menidia* spawning site selection. Middaugh and Takita (1983) hypothesize that the spawning runs of *M. menidia* occur in direct response to current velocities. At slack high tide, water movement decreases and sperm is less susceptible to dispersion. By selecting spawning sites with shallow-water over suitable substrate, water volume per unit of suitable substrate is reduced, and *M. menidia* can increase the concentration of milt at spawning sites (Middaugh et al. 1984).

By exhibiting searching behavior and actively selecting a shoreline location for egg deposition, schools of spawning *M. menidia* (Middaugh et al. 1981; Conover and Kynard 1984) may not reduce total egg deposition in response to encountering *P. australis* and hardened shorelines. Rather, egg deposition may be largely displaced and concentrated in areas along *S. alterniflora* shorelines, where preferred combinations of shoreline type and substrate for *M. menidia* oviposition are found. Whether greater egg densities affect hatching and larval survival is a matter for future research. As the area of hardened shoreline increases in urbanizing estuaries, preferential spawning habitat may eventually reach thresholds below which reproductive activities are critically impaired.

Egg Attachment by Substrate Type

Substrate availability differed at each shoreline type, but *Enteromorpha spp.* was utilized most frequently, across all shorelines for egg attachment by *M. menidia*. Over 95% of eggs collected had been attached on *Enteromorpha spp.* The variety of other substrates, including *Fucus spp*., *U. lacuta*, *S. alterniflora* stems and detritus, were used significantly less often. Previously cited substrates used for deposition such as sand, trash, and crab burrows were not found to be utilized in the present study. By utilizing the green alga *Enteromorpha spp.* for egg attachment at a greater percentage than the coverage percentage of *Enteromorpha spp.* at each shoreline type, *M. menidia* are displaying a preference for oviposition on the substrate. Various species of *Enteromorpha spp.* are found throughout the range of *M. menidia* (Schneider and Searles 1991) and several studies have found filamentous algae to be a preferred substrate to the north (Massachusetts, Conover and Kynard 1984) and south (South Carolina, Moore 1980) of the present study area.

The preference for *Enteromorpha spp.* by *M. menidia* as a spawning substrate raises the possibility that areal coverage of this alga is the driving factor in their preference of shorelines for egg deposition. Riprap and riprap-sill shorelines had significantly greater coverage of *Enteromorpha spp.* than other shorelines, yet they had the third and fourth greatest egg densities of the shoreline types examined. Bulkhead shorelines had greater *Enteromorpha spp.* coverage than *P. australis* and beach shorelines, but far fewer eggs were deposited on bulkheads. Thus, although *Enteromorpha spp.* is the preferred substrate for egg attachment by *M. menidia* across all shoreline types, shoreline type affects choice of spawning location as well. Were this not the case, 94% of all eggs would not have been deposited along *S. alterniflora* shorelines. Likewise, bulkhead shorelines would have had greater egg densities than the *P. australis* shoreline and far greater egg densities than sandy beach shoreline. Within *S. alterniflora* shorelines, no significant differences existed in mean rank sums of egg density by *Enteromorpha spp.* coverage percentage. So, across a range of

Enteromorpha spp. coverage's (10-90%), egg density did not differ. As such, in the highly variable environment of *S. alterniflora* marsh, *M. menidia* are utilizing their preferred substrate, even as this increases the density of eggs over a smaller area of substrate. This further supports the concept that *M. menidia* are very precisely choosing locations for egg deposition/attachment.

Abiotic Characteristics

Greater egg densities were found to be related to higher tidal height at the *S. alterniflora* shorelines in the present study. This agrees with many prior *M. menidia* studies (Middaugh et al. 1981; Middaugh et al. 1984; Conover and Kynard 1984; Tewksbury and Conover 1987). Higher tides allow more preferential shoreline/substrate to be available, at an elevation with preferential temperature and moisture conditions for egg deposition. Observations of the behavior of pre-spawning schools of *M. menidia* suggest a probable response to the availability of preferred v. non-preferred spawning substrates (Middaugh and Takita 1983). These investigators observed schools of *M. menidia*, on occasions of sub-maximum high tide heights, swimming past areas where spawning activity was commonly observed during occasions of higher high tide heights. Because these sites were not inundated, and were thus unavailable for spawning, these schools moved to a "secondary" spawning location lower in the intertidal, just before slack high tide. These observations are consistent with the concept of tidally (on all high tides), rather than lunar (only on spring high tides) spawning intensity concluded by Conover and Kynard (1984). Moon phase was not found to be significantly related to egg density in the present study, supporting a prior laboratory study by Conover and Kynard (1984).

No detrimental atmospheric/water temperature conditions for embryonic survival were found at any of the shoreline types in this study. Maximum atmospheric/water temperatures when eggs were present were never >34.1°C, well below the danger levels for survival indicated by critical thermal maxima tests (Hutchinson 1961). Atmospheric/water temperatures, both means and maximums, were relatively high at the *S. alterniflora* shoreline in comparison with the other shoreline types examined (Tables 4.3 and 4.4). Whether this difference is primarily due to vegetative cover and retained moisture is not known. These higher mean temperature decrease incubation time, reducing potential predation (Able and Castagna 1975; Conover and Kynard 1984; Middaugh et al. 1983).

Middaugh et al. (1983) found different substrate types (*S. alterniflora* stems, *Enteromorpha spp.* and mud crab burrows) offered *M. menidia* embryos differing degrees of protection from thermal and desiccation stresses. Surface temperatures and atmospheric moisture were found to be most favorable for *M. menidia* embryo survival within *Enteromorpha spp.* mats (13-34°C; no desiccation observed). Less favorable environmental conditions were observed at the base of *S. alterniflora* stems (maximum of 36°C; significantly lower atmospheric moisture than algal mats) in that South Carolina study. This is important, as substrate choice for oviposition is among the only forms of parental care *M. menidia* provide for embryos (DeMartini 1999). Given that desiccation is a major source of mortality during incubation for intertidally spawning fishes (DeMartini 1999), oviposition/substrate choice is critical for *M. menidia* embryo survival. *Enteromorpha spp.* mats present at *S. alterniflora* shorelines provide preferable warm substrate temperatures and high atmospheric moisture for *M. menidia* embryos.

Menidia menidia employ spawning behaviors to maximize the survival and health of their embryos. Findings from the present study are consistent with previous findings on *M. menidia* searching behavior which suggest schools of *M. menidia* actively choose specific sites for egg deposition. Schools of spawning *M. menidia* appear to select sites for egg deposition primarily based on shoreline type, and then proceed to choose specific locations for egg attachment based on substrate type. Shorelines dominated by *S. alterniflora* are utilized more frequently than hardened shorelines, sandy beach shorelines and shorelines dominated by *P. australis*. The green alga *Enteromorpha spp.* is utilized more frequently than all other available substrates for *M. menidia* egg attachment. 91% of all eggs collected in the present study were deposited along a *S. alterniflora* shoreline, attached to *Enteromorpha spp.* This combination of *S. alterniflora* shoreline and *Enteromorpha spp.* provides preferential characteristics (temperature, atmospheric moisture, predation protection) for health and survival of *M. menidia* embryos. Increasing areas of hardened shoreline structures in estuarine environments may have serious implications for *M. menidia* and other intertidally spawning fishes by reducing habitat used most frequently for egg deposition.

Table 4.1 *Menidia menidia* eggs collected and mean density (eggs/m^2) by shoreline type collected near Roosevelt Inlet, Delaware Bay, during spring 2010. Significant differences denoted by superscript letters ($p<0.05$).

Shoreline Type	Total Eggs	Mean Eggs (Eggs per m^2 per day) with SE	Percentage of Total Eggs Collected
S. alterniflora^a	2,922,150	32,468±10,400	93.8
P. australis^{c,d}	94,190	1,046±1,003	3.0
Riprap-Sill^a	49,840	553±196	1.6
Riprap^b	46,460	516±238	1.5
Beach^c	2,530	28±14	0.1
Bulkhead^d	4	0.04±0.04	<0.01
Total Eggs	3,115,174		

Table 4.2 *Menidia menidia* eggs collected by substrate type utilized for egg attachment and substrate utilization percentage at shoreline types near Roosevelt Inlet, Delaware Bay, during spring 2010. Significant differences among substrate types across all shoreline types, as well as within shoreline types, denoted by superscript letters ($p<0.05$).

	Substrate Utilization		Substrate Utilization Percentage by Shoreline Type					
Substrate Type	Total Eggs	Percentage of Total Eggs	Bulkhead	Riprap	Riprap-Sill	Beach	*P. australis*	*S. alterniflora*
Entero spp.^{a}	2,975,857	95.5	100	95.6^{a}	98.5^{a}	100	98.6^{a}	91.1^{a}
S. alterniflora^{b}	127,062	4.0	0	0	0	0	0	4.4^{b}
Detritus^{b}	10,663	<0.1	0	0	1.5^{b}	0	1.4^{b}	1.7^{c}
Fucus spp.^{b}	891	<0.1	0	4.4^{b}	0	0	0	2.8^{c}
U. lacuta^{b}	701	<0.1	0	0	0	0	0	<0.01^{c}

Table 4.3 Mean and maximum (in parentheses) water/air temperatures (°C) at upper, mid and lower intertidal elevations at shoreline types near Roosevelt Inlet, Delaware Bay on days when eggs were present (April 17th – May 27th). Significant differences within intertidal elevations denoted by superscript letters ($p<0.05$).

Intertidal Elevation	Bulkhead	Riprap	Beach	*P. australis*	*S. alterniflora*
Upper	17.3^{c} (37.7)	17.9^{b} (42.7)	17.2^{d} (34.7)	17.9^{b} (39.5)	19.2^{a} (39.2)
Mid	16.3^{d} (28.8)	16.4^{d} (28.3)	16.8^{bc} (32.6)	16.7^{c} (34.1)	17.0^{a} (33.1)
Lower	16.2 (27.3)	16.4 (34.0)	16.1 (30.4)	16.3 (32.0)	16.3 (33.3)

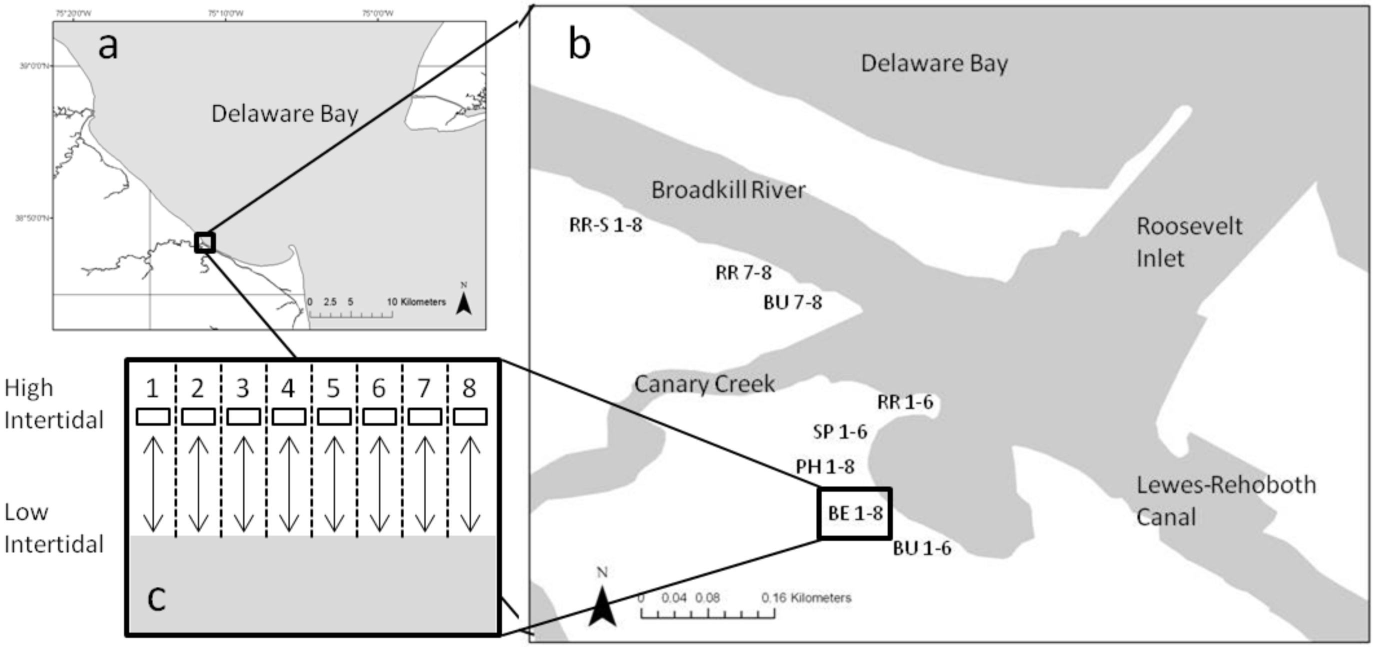

Figure 4.1 Sampling location for *Menidia menidia* eggs near the Roosevelt Inlet, Delaware Bay, during spring 2010. Eggs were collected from shoreline types (BU-bulkhead, RR-riprap, RR-S-Riprap-Sill, BE-Sandy Beach, PH- *Phragmites australis*, SP-*Spartina alterniflora*) with eight stations each.

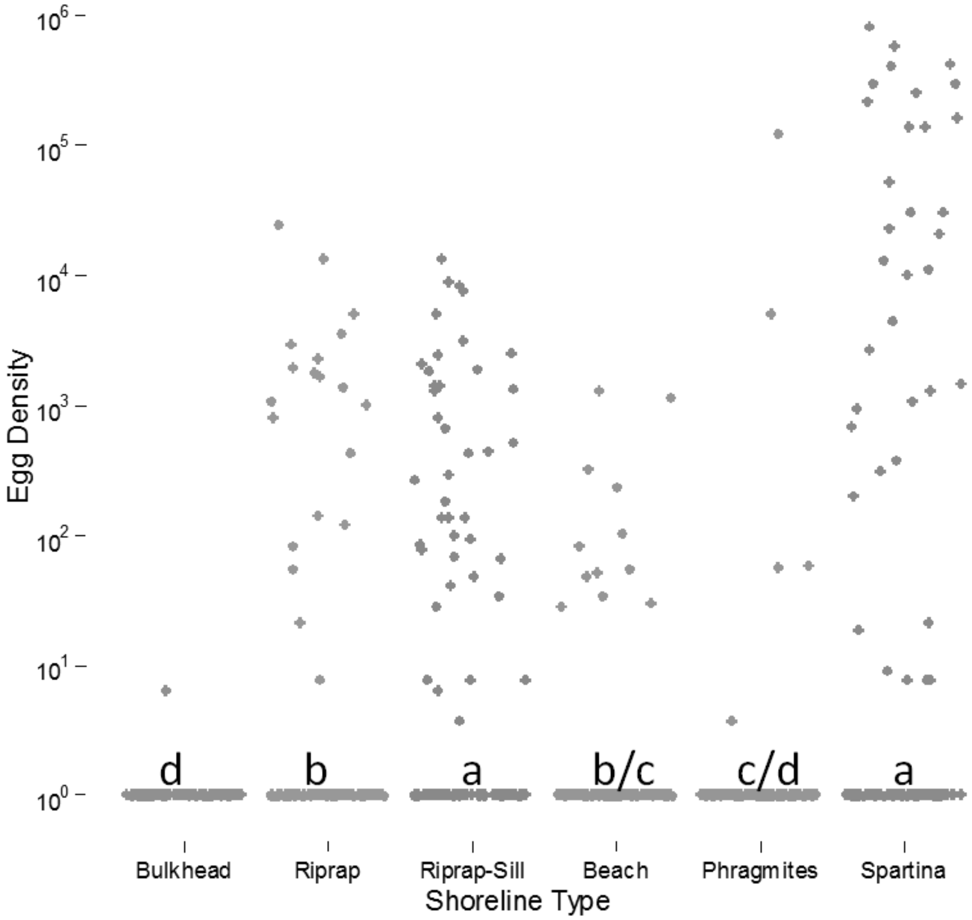

Figure 4.2 Daily *Menidia menidia* egg densities (log scale) by shoreline type. Points are individual, daily, station densities. Significant differences in mean rank sums denoted by superscript letters ($p<0.05$).

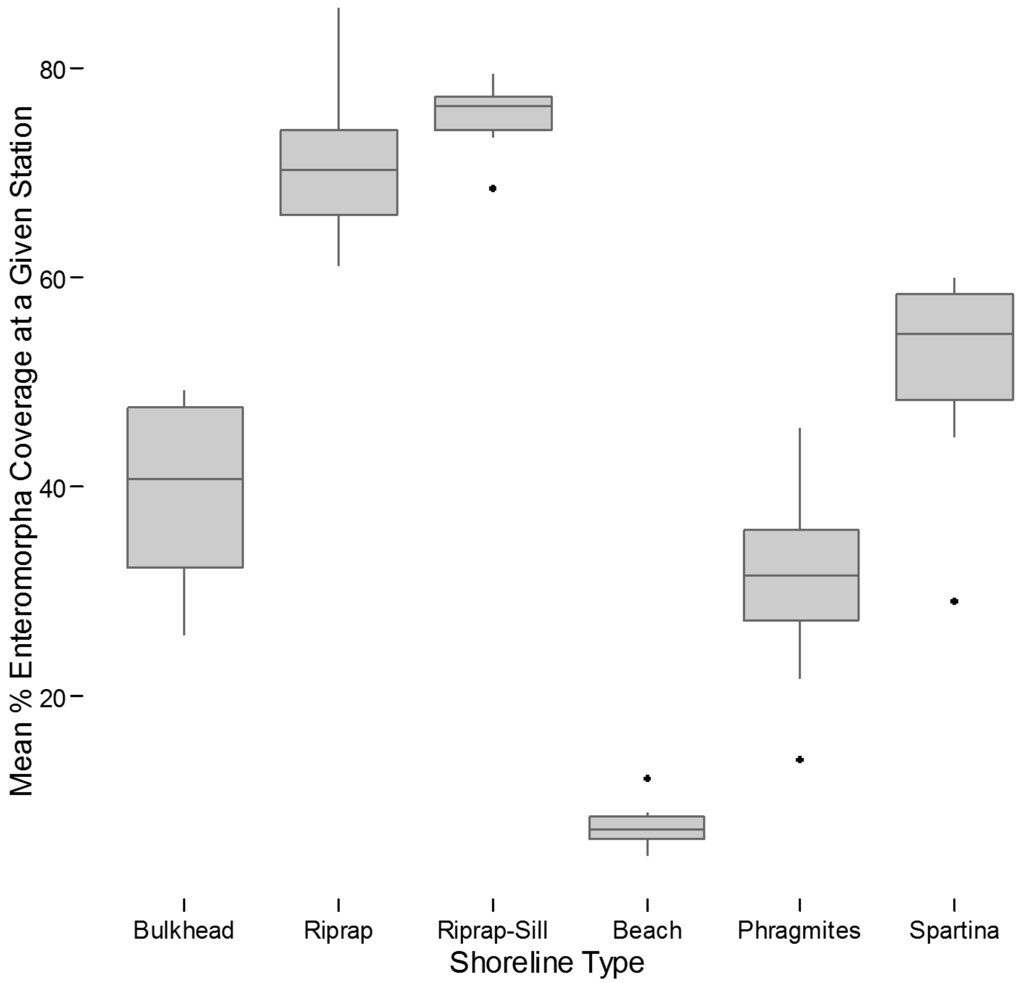

Figure 4.3 Mean percent *Enteromorpha spp.* coverage on sampling days at eight stations per shoreline type. The minimum and maximum observations are represented by the ends of the whiskers. The bottom and top of each box represent the 25th and 75th percentile respectively and the median is represented by the line in the box.

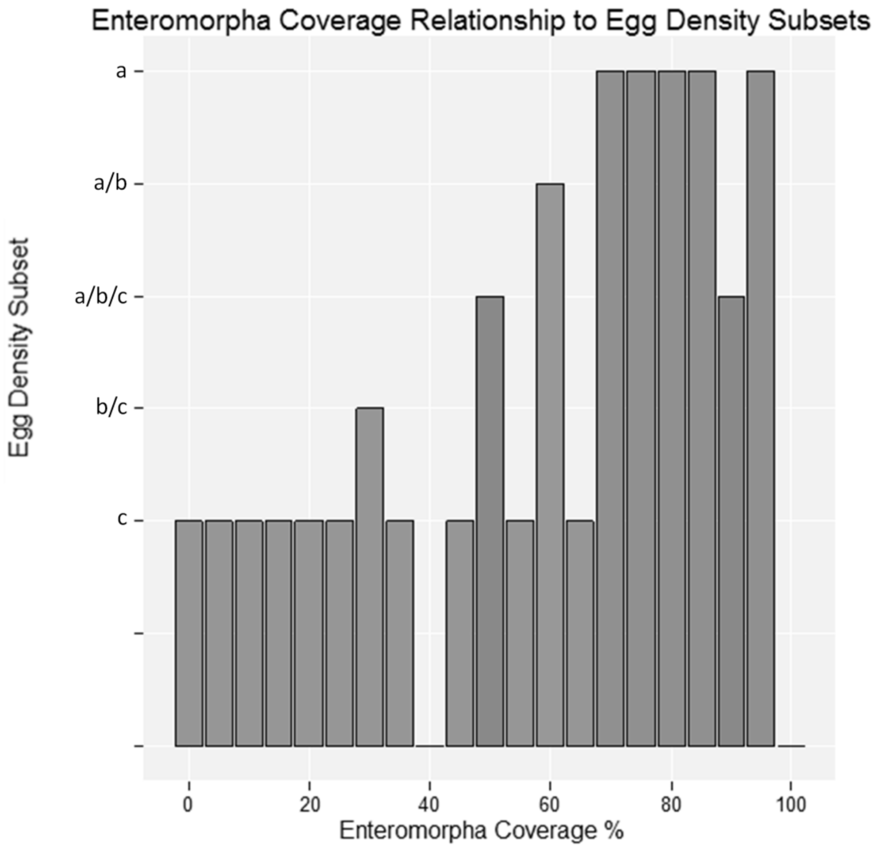

Figure 4.4 *Menidia menidia* egg density mean rank sum subsets by percent *Enteromorpha spp.* coverage across all six shoreline types. Egg density mean rank sums computed from Kruskal-Wallis test on mean egg densities by *Enteromorpha spp.* coverage at the station on the day each sample was taken. Mean rank sum subset denoted by bar height. Greater mean rank sums correspond with greater egg densities. Five subsets are presented, beginning with the greatest mean rank sums, a, a/b, a/b/c, b/c/ and c.

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