Bycatch associated with a horseshoe crab (*Limulus polyphemus*) trawl survey: identifying species composition and distribution

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Abstract

Horseshoe crabs (*Limulus polyphemus*) have been harvested along the east coast of the United States since the 1800s, however a Fishery Management Plan (FMP) was only recently created for this species. To date, there have not been any studies that have attempted to identify or quantify bycatch in the horseshoe crab trawl fishery. A horseshoe crab trawl survey was started in 2001 to collect data on the relative abundance, distribution, and population demographics of horseshoe crabs along the Atlantic coast of the United States. In the present study, species composition data were collected at sites sampled by the horseshoe crab trawl survey in 2005 and 2006. Seventy-six different taxa were identified as potential bycatch in the horseshoe crab trawl fishery. Non-metric multidimensional scaling (NMS) was used to cluster sites and identify the spatial distribution of taxa. Sites strongly clustered into distinct groups, suggesting that species composition changes spatially and seasonally. Species composition shifted between northern and southern sites. Location and bottom water temperature explain most of the variation in species composition. These results provide a list of species that are susceptible to this specific trawl gear and describe their distribution during fall months throughout the study area. Identifying these species and describing their distribution is a first step to understanding the ecosystem-level effects of the horseshoe crab trawl fishery.

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Introduction

Marine fisheries have traditionally been managed using a single-species approach, but in the past this type of management has often failed to create sustainable fisheries (NMFS 1999). Managers are now moving toward ecosystem-based fisheries management, not only focusing on sustaining fisheries, but also healthy ecosystems to support those fisheries (Pikitch et al. 2004). This holistic approach examines abiotic and biotic interactions between fisheries and the ecosystem and incorporates these interactions into management plans. To encourage managers to move toward an ecosystem-level approach of management, NOAA Fisheries suggested Fisheries Ecosystem Plans (FEPs) be developed for all major ecosystems in an effort to create sustainable fisheries for a greater number of species (Botsford et al. 1997; NMFS 1999). One objective of FEPs is to estimate the amount of non-target individuals that are incidentally caught by fishing gears within each fishery (NMFS 1999). The removal of these organisms (through harvest, death after release, or unobserved mortality due to gear encounter) can have negative impacts on populations, communities, and the ecosystem (Crowder and Murawski 1998).

Horseshoe crabs (*Limulus polyphemus*) have been harvested in many states along the Atlantic coast since the mid-1800s (ASMFC 1998). However, it was not until 1997 that a fishery management plan (FMP) was created for this species. Over the past 10 years, researchers have been collecting data to allow for more effective management of horseshoe crab populations. In 2001, a benthic trawl survey was developed to collect data on the abundance, distribution, and demographics of horseshoe crabs (Hata and Berkson 2004). Since the trawl survey uses the same gear as the commercial fishery, I was able to collect data to identify the non-target species that are susceptible to trawl gear used in the horseshoe crab commercial trawl fishery. Currently, the ecosystem-level impacts of this fishery are unknown. This study has identified the species that are commonly caught by horseshoe crab trawl gear, allowing managers to begin to understand the ecosystem-level effects of the horseshoe crab trawl fishery.

Ecosystem Management

Traditionally, marine fisheries have been managed from a single-species approach, with the goal of maximizing the catch of a single target species (Pikitch et al. 2004). However, this type of management is not always appropriate because it ignores the direct and indirect effects that harvesting a target species may have at the ecosystem level (Botsford et al. 1997; Pikitch et al. 2004), and the possible effects of distant ecosystem components on the target species itself. Oftentimes the harvest of one species can affect habitat characteristics, predator and prey relationships, and other ecosystem components (Pikitch et al. 2004). Today, managers are adapting a more holistic approach to management. Ecosystem management reverses the order of management priorities, starting with the ecosystem instead of the target species (Pikitch et al. 2004).

Although ecosystem management has been defined in many ways, all definitions have common themes including: developing system-wide perspectives; emphasizing the complex composition and processes of ecological systems; managing for ecological integrity and maintaining diversity; integrating across spatial and temporal scales and human ecological, economic and cultural concerns; using management techniques that remain flexible and can adapt to uncertainty; and including many participants in decision making (Grumbine 1994; Meffe et al. 2002). "Ecosystem management integrates scientific knowledge of ecological relationships within a complex sociopolitical and values framework toward the general goal of protecting native ecosystem integrity over the long term" (Grumbine 1994).

The overall objective of ecosystem-based fisheries management is to not only sustain fisheries, but also to sustain healthy ecosystems to support fisheries (Pikitch et al. 2004). To do this, researchers, managers, and politicians must work to understand the abiotic and biotic interactions between fisheries and the ecosystem and incorporate these interactions into management plans. Some of the most important aspects are managing ecosystems to avoid the degradation, minimizing the risk of irreversible changes to natural species assemblages and processes of the ecosystem, obtaining and maintaining long-term socioeconomic benefits, and generating knowledge of ecosystem processes to better understand the consequences of anthropogenic effects (Pikitch et al. 2004). However, developing a management plan that incorporates all interactions among physical, biological, and human components of the ecosystem is nearly impossible and would require a complete understanding of ecosystem dynamics and community structure (Botsford et al. 1997; NMFS 1999). Ecosystems are complex and adaptive systems that are constantly changing (NMFS 1999). They are widespread and open, which makes them hard to fully study or understand. Managing at the ecosystem level is an even more difficult task because living organisms are constantly adapting and evolving to their surrounding physical and biological environments (NMFS 1999).

Although challenging, strides are being made to manage marine fisheries at an ecosystem level. This type of management is favored over past strategies that have left many fishery stocks around the world overfished or depleted (Botsford et al. 1997; NMFS 1999). The Sustainable Fisheries Act of 1996 (SFA) was a first step in incorporating an ecosystem level of stock management in the United States. It encouraged the development of Fisheries Management Plans (FMP) that would identify and protect essential fish habitats (EFH), rebuild fisheries that were defined as overfished, reduce bycatch, and stop overfishing (NMFS 1999; Fluharty 2000). Rebuilding fisheries, reducing bycatch, and identifying EFHs stressed the importance of preserving habitat and sustaining valuable fish populations (NOAA 2005).

In recent years, regional fishery management councils have been moving toward an ecosystem level of management by incorporating multispecies and ecosystem assessments into single-species management (ASMFC 2003). Species such as summer flounder (*Paralichthys dentatus*), scup (*Stenotomus chrysops*), and black sea bass (*Centropristis striata*) are managed as a multispecies complex by the Mid-Atlantic Fisheries Management Council (MAFMC) and the Atlantic States Marine Fisheries Commission (ASMFC) (ASMFC 2003). This complex is based on fisheries interactions, but each species within the complex is still managed based on individual species stock assessments and other management measures (ASMFC 2003). Although this type of management begins to incorporate some direct effects of harvest on non-target species, there are still many ecosystem-level effects that are not incorporated in multispecies management plans.

To continue working toward an ecosystem level of management, the National Marine Fisheries Service (NOAA Fisheries) suggested that a Fisheries Ecosystem Plan (FEP) be developed by each regional fishery management council for all major ecosystems within their region (NMFS 1999). FEPs will provide background on the components and functions of the ecosystem, and guide the development of management strategies that take into account ecosystem-level effects. NOAA Fisheries suggested eight steps for the development of FEPs:

1. Delineate the geographic extent, characterize the biological, physical, and chemical dynamics, and define zones for alternative uses of the ecosystem(s) within each Council's authority;
2. Develop a model of the food web;
3. Describe the habitat needs for all stages of all animals and plants that are part of the food web and how they are involved in conservation and management;
4. Calculate total removals and relate these removals to standing biomass, production, optimal yields, natural mortality, and trophic structure;
5. Assess uncertainty and what kind of buffers can be used in conservation and management actions;
6. Develop targets for ecosystem health;
7. Describe available long-term monitoring data and how it will be used; and
8. Assess the ecological, human, and institutional elements of the ecosystem that most significantly affect fisheries.

Although difficult to create, FEPs will provide valuable information about ecosystems, which is crucial to fisheries management (NMFS 1999). While FMPs are still the basic tool used to manage fisheries, sustainability may be more easily achieved in fisheries by incorporating FEPs into management planning and acknowledging ecosystem-level effects.

One important aspect of ecosystem-based fisheries management is to calculate total removals within a fishery, including the total removal of target and non-target individuals that are landed, caught and released, eliminated by predation, or lost through incidental capture (NMFS 1999). Many individuals that are incidentally caught and thrown back into the water often die as a result of stress or trauma. These losses may affect the total size, age structure, and sex ratios of target and non-target populations, as well as the overall food web and habitat (Crowder and Murawski 1998; NMFS 1999; Horsten and Kirkegaard 2002). Therefore, it is critical to identify and estimate bycatch within each fishery. To do this adequately, NOAA Fisheries suggests that FEPs should list species that are caught as bycatch within each fishery and show how the distribution of these species changes spatially and temporally (NMFS 1999).

Bycatch

Before bycatch can be properly identified in each fishery, the term bycatch must be specifically defined. The Magnuson-Stevens Fishery and Conservation Act (MSFCA) describes bycatch as "…fish which are harvested in a fishery, but which are not sold or kept for personal use, and includes economic discards and regulatory discards…" (MSFCA 1996). However, some captured non-targeted species are harvested because they are profitable. Therefore, Crowder and Murawski (1998) divide bycatch into three main categories: kept bycatch, discarded bycatch, and unobserved mortalities. Kept bycatch includes non-target species that are harvested, sometimes providing an important source of income for the fisher (Crowder and Murawski 1998). Discarded bycatch includes individuals that are captured and then released (dead or alive) because they are not the species, size, sex, or condition that was targeted (Davis 2002). Individuals may also be discarded if there is a lack of space on board the vessel, or quotas have already been reached. Unobserved mortality, the last category of bycatch, includes individuals that encounter but are not retained by the gear (Crowder and Murawski 1998).

Biologically, bycatch that is discarded and suffers mortality is equivalent to harvested catch because, in both cases, individuals are eliminated as a living part of the system. Discarded bycatch can have positive effects on other ecosystem components by providing food for organisms, such as birds and benthic scavengers (Ramsay et al. 1997; Votier et al. 2004). However, the removal of individuals as bycatch can also impact the ecosystem in negative ways. Effects may be simple, resulting in the direct mortality of juvenile fishes, or more complex, causing changes at the population, community, and ecosystem levels (Alverson et al. 1994; Kennelly 1995). The mortality of smaller-sized individuals that have not reached reproductive age may result in lower recruitment to the spawning stock (Crowder and Murawski 1998). The removal of fish below the optimal or legal size limit may result in a reduction in overall efficiency within the fishery, because of a potential decrease in the yield-per-recruit of the stock (Crowder and Murawski 1998). Intense removal, through harvest or discard, can impact communities and the ecosystem by altering species abundance, predator/prey relationships, and competitive interactions (Alverson et al. 1994; Kennelly 1995; Crowder and Murawski 1998). Size distribution and prey abundance may shift as larger predators and grazers are removed from the ecosystem, leading to changes in the food web and trophic displacement (Ward and Myers 2005). Overall, scientists expect that if the rate of bycatch in a fishery is high, target and non-target species will be affected at some level (Alverson et al. 1994; Horsten and Kirkegaard 2002).

It is important to understand how fisheries are affecting non-target species. Identifying and quantifying bycatch species, allows managers to understand which non-target species are affected by each fishery and to incorporate total removals of non-target species into management plans. This is important for managed, unmanaged, and federally-listed species, all which may be susceptible to the gears used in fisheries. Many managed species are harvested by commercial fishers and recreational fisheries. The stock status of managed species is monitored, but this is not the case for all unmanaged species. Although not managed, these individuals may play important roles in the ecosystem. Without proper monitoring, populations of unmanaged species may plummet, undetected. Many federally-listed species are also susceptible to the gears used in some fisheries. These populations are already at low population sizes and removing even a few individuals can have detrimental effects on these species.

Monitoring programs provide data to identify bycatch occurring within each fishery, leading to a better understanding of some of the effects that these fisheries are having at the ecosystem level. However, in most fisheries, reliable methods to monitor bycatch have not yet been developed. Developing methods to calculate an unbiased and precise estimate of bycatch that can be used in all fisheries is a technical, logistic, and financial challenge (Crowder and Murawski 1998). Gear type varies among fisheries, ranging from fish traps to trawl nets to longlines, each of which fishes differently. Besides developing methods that can be used to estimate bycatch across all gear types, estimating the mortality of discarded bycatch is another challenge (Davis 2002).

Variable environmental conditions, such as capture and gear effects, light conditions, water temperature, air exposure, dissolved oxygen levels, and sea conditions all may affect the mortality rate of the individual (Davis 2002). Further, each species has a different vulnerability to stress. Some species may be able to endure longer time out of the water or tolerate more handling than other species.

Bycatch monitoring programs

Programs have been implemented to monitor bycatch, but many are inconsistent because they differ in methods, effort, and objectives (Crowder and Murawski 1998). Data are mainly collected using fishery-independent and fishery-dependent surveys (NOAA 2003). Fishery-independent surveys randomly sample sites using research or chartered fishing vessels (NOAA 2003). The objectives of fishery-independent surveys are to provide data for stock assessment (e.g., variation of abundance, spatial and temporal distribution patterns, age and size composition, and fecundity measures; NOAA 2003). These data are thought to accurately reflect the distribution and abundance of species because sites are selected based on a random sampling method (Fox and Starr 1996). However, these surveys do not always sample the same area or with the same fishing technique (e.g., gear, tow length) as do commercial fishers. Fishery-dependent surveys are more commonly used in bycatch-monitoring programs because these surveys collect data by directly observing the commercial fishery. This type of sampling may be more accurate in quantifying bycatch occurring in a fishery because it samples the same areas that are actively fished by commercial vessels. Some programs collect fishery-dependent data by requiring fishers to record bycatch in logbooks (NOAA 2003).

However, these data can be skewed due to varying economic conditions, changes in statutes, policies, and fishing methods, and the willingness of fishers to provide accurate data (NEAMAP 2003). It is preferred that scientific observers sort, identify, count, measure, and/or weigh bycatch aboard commercial vessels during fishing activities. These observers are usually trained by management agencies and provide a less-biased way to collect data aboard vessels. Still, there may be a lack of consistency because bycatch is quantified in different ways among various programs (Horsten and Kirkegaard 2002).

There are many fishery-independent and fishery-dependent surveys that monitor marine species along the Atlantic coast. The Northeast Fisheries Science Center (NEFSC), a part of NOAA Fisheries, conducts fall and spring bottom trawl surveys to monitor the abundance and seasonal distribution of adult and juvenile fish (Reid et al. 1999). The data from this survey show patterns of distribution along the northern and middle Atlantic coast of the United States, independent of state lines or other boundaries. Currently, NEFSC is the only organization that has conducted a repeated, consistent survey over both a long time-scale and broad geographical area. Surveys have been conducted since the early 1960s, extending throughout the Middle Atlantic Bight, southern New England Nantucket Shoals, Georges Bank, Gulf of Maine, and Scotian Shelf (Reid et al. 1999).

The Marine Resources Monitoring, Assessment and Prediction (MARMAP) program has been collecting fishery-independent data on groundfish, reef fishes, ichthyoplankton, and coastal pelagic fishes along the Southeast Atlantic Bight (SAB), from Cape Lookout, North Carolina to Cape Canaveral, Florida, for the past thirty years

(SEDAR 2003). MARMAP has conducted trawl and ichthyoplankton surveys, sampling and mapping of reefs, and life history, population, and tagging studies to provide data on the distribution, relative abundance, and critical habitat of fishes that are economically and ecologically important (SEDAR 2003).

Other fishery-independent surveys include the Southeast Area Monitoring and Assessment Program (SEAMAP) and the Northeast Area Monitoring and Assessment Program (NEAMAP). Since 1986, NOAA Fisheries has also sponsored SEAMAP, which surveys the Southern Atlantic Bight between Cape Canaveral, Florida and Cape Hatteras, North Carolina (Reid et al. 1999, SEAMAP 2000). In October 1997, a resolution was passed to develop NEAMAP in an effort to collect data north of Cape Hatteras, North Carolina (NEAMAP 2003). Pilot studies have been completed and this project is planned to be implemented in 2007 (NEAMAP 2003).

The Atlantic Coastal Cooperative Statistics Program (ACCSP) was started in 1995 to collect fishery-dependent data from commercial, recreational, and for-hire fishers along the Atlantic coast (ACCSP 2005). Although many programs are still in development, data will be collected for finfish, crustaceans, shellfish, live rock, coral, marine mammals, and federally-listed, discarded, protected, aquaculture, and internationally managed species (ACCSP 2005).

The collection of fishery-dependent and fishery-independent data is important for monitoring bycatch within a fishery. Hall (1996, 2000) suggests that these data can be used to identify the spatial and temporal patterns of species and provide a ratio of bycatch to catch based on area and season. When the ratio of bycatch to catch is high in a specific area or during a specific season, restrictions can be implemented to reduce the amount of non-targeted species that are caught. Fishers also can modify fishing location and gear type to target areas based on the spatial and temporal patterns of a target species. This technique can allow fishers to optimize catch, minimize effort, and reduce bycatch.

Data collected during fishery-dependent and fishery-independent surveys also can provide a better understanding of the ecology of an area. Changes in the species composition and distribution allow managers to gain a better understanding of community structure, relationships among species, and the effects of bycatch on an ecosystem (Greenstreet et al. 1999; Gomes et al. 2001; Beentjies et al. 2002). Abundance and distribution data can be used to identify species assemblages, or a group of species that commonly co-occur, and gain a better understanding of the direct and indirect effects that a fishery could have on non-target species. Examining co-occurrences allow researchers to gain a better understanding of predator-prey interactions, competition, and community structure (Greenstreet 1999; Gomes 2001). Identifying species assemblages also allows managers to predict which species may be caught as bycatch in a fishery. For example, past studies suggest that summer flounder along the Atlantic coast commonly associate with scup, black sea bass, northern searobin (*Prionotus carolinus*), and spiny dogfish (*Squalus acanthias*) (Colvocoresses and Musick 1984; Mahon 1998; Gabriel 1992). These species may all be caught as bycatch in the summer flounder trawl fishery because studies suggest they inhabit the same areas. By understanding the species that interact and associate with one another, gear modification, restrictions, and regulations can be placed on the fishery to reduce bycatch of non-target species.

The data provided by fishery-dependent and fishery-independent surveys are crucial to understanding ecosystem dynamics and community structure. Researchers can use these data to identify bycatch, species assemblages, and species distribution. The results from these studies allow managers to gain a better understanding of the complex interactions among all living organisms and begin to manage at an ecosystem level as they incorporate these biological interactions into management plans. Bycatch monitoring programs have begun to identify and quantify bycatch in many of the fisheries around the world; however, monitoring programs have not been established for all fisheries.

The horseshoe crab trawl survey

In 2001, the Horseshoe Crab Research Center (HCRC) benthic trawl survey was initiated to monitor horseshoe crab populations along the Atlantic coast (Hata and Berkson 2003). This trawl survey uses the same gear as in the commercial fishery and can provide information as to which species could be caught as bycatch in the horseshoe crab trawl fishery. Species composition data collected during the HCRC trawl survey can be used to identify species that are susceptible to the commercial trawl gear used in this fishery. These data also can identify species composition and distribution along the Atlantic coast. Identifying bycatch species and their distribution provides managers with information to begin to understand the possible ecosystem-level effects of the horseshoe crab trawl fishery.

The horseshoe crab fishery

Horseshoe crabs have supported a fishery along the Atlantic coast since the mid-1800s; however it was only recently that an FMP was developed for this fishery (Walls et al. 2002). To date, there have not been any studies that have attempted to identify or quantify bycatch in the horseshoe crab trawl fishery. Identifying the species caught as bycatch allows a better understanding of which non-target species may be affected by the horseshoe crab trawl fishery.

Horseshoe crabs are considered a multiple-use resource. They are valued by many stakeholders, including commercial fishers, environmental interest groups, and biomedical companies (Berkson and Shuster 1999). Horseshoe crabs are commercially harvested by hand and by fishing gears such as trawl, dredge, and gillnet (HCTC 1998). Harvested horseshoe crabs are sold as bait for American eel (*Anguilla rostrata*) and whelk (*Busycotypus* spp.) fisheries (ASMFC 1998). Based on estimates from 1999, the eel and whelk fisheries that rely on horseshoe crabs as bait contribute about $15 million to local economies and create over 400 jobs (Manion et al. 2000). Besides supporting a commercial fishery, horseshoe crabs are important to biomedical companies. The copper-based blood of horseshoe crabs contains a unique clotting agent, called Limulus Amoebocyte Lysate (LAL). This compound is used to detect pathogenic endotoxins in injectable drugs or on medical devices (Novitsky 1984; Mikkelsen 1988). Currently, the biomedical industry is dominated by three U.S. firms that provide a combined total of $73 to $96 million dollars to their local economies (Manion et al. 2000). Horseshoe crabs are ecologically important because their eggs serve as a food source to migrating shorebirds. During spawning season, each female horseshoe crab lays about 4,000 eggs on intertidal sandy beaches during each nesting trip, resulting in about 88,000 eggs per female per year (Shuster and Botton 1985; Mikkelsen 1988). Species of birds [e.g., Red Knots (*Calidris canutus*), Ruddy Turnstones (*Arenaria interpres*), Sanderlings (*Calidris* *alba*), and Semipalmated Sandpipers (*Calidris pusilla*)] rely on these eggs to provide vital energy during the 3,000 – 4,000 mile migration route to breeding grounds (Clark et al. 1993). Many people gather to witness the masses of horseshoe crabs and shorebirds on many beaches, creating an ecotourism industry that provides $3 to $4 million a year to local communities (Manion et al. 2000).

Despite their importance to many users, prior to 1997 the horseshoe crab was not a managed species. In the fall of 1997, the Atlantic States Marine Fisheries Commission (ASMFC) developed an FMP to manage horseshoe crabs within state (0-3 nautical miles) and federal waters (3 or more nautical miles) (ASMFC 1998). Prior to this management plan, little was known about the status of horseshoe crab populations. This species was thought to be sensitive to overfishing because it is slow to sexually mature and is easily harvested with minimum financial investment (ASMFC 1998). However, due to insufficient information, it was unclear whether overfishing was occurring. New efforts have been made to provide information for effective management, but many data are still lacking. The gear used in this fishery is unselective; catching target and non-target individuals. Therefore, it is important to identify bycatch species to better understand the effects that the horseshoe crab trawl fishery has on non-target species.

The HCRC at Virginia Tech designed an annual benthic trawl survey to provide data of horseshoe crab abundance and distribution for future stock assessments (Walls et al. 2002; Hata and Berkson 2003; Hata and Berkson 2004). This survey could also identify species that are susceptible to horseshoe crab trawl gear along the Atlantic coast. Although many fishery-independent trawl surveys have sampled this area, there have not been any studies that have identified species that commonly associate with horseshoe crabs. Most previous surveys that sample the Atlantic coast use benthic trawl nets that are equipped with rollers to prevent damage to the sea floor. These data could not be used to assess horseshoe crab populations because the gear has potential to roll over, and undersample, buried crabs. Commercial fishers have developed a method for targeting buried horseshoe crabs; they do not use rollers, instead their nets are modified with heavy chains along the footrope to dig into the sediment (Hata and Berkson 2003). The horseshoe crab trawl survey uses the same gear as in the commercial trawl fishery, thereby providing an efficient and representative method of capture (Hata and Berkson 2003; Hata and Berkson 2004).

The Middle Atlantic Bight

The Middle Atlantic Bight (MAB) includes all shelf and coastal waters in the North Atlantic Ocean from Nantucket Shoals, USA (east of Cape Cod, Massachusetts) to Cape Hatteras, North Carolina, USA (Muir et al. 1989; Figure 1.1). The Gulf Stream is the major current in offshore MAB waters, flowing in a northward-offshore direction. Storm winds and weather events can move shelf waters over the entire MAB, sometimes causing currents to shift from an along-shore direction to a toward-shore direction (Chant et al. 2004). This shift has also been documented to cause upwelling events in some areas (Song et al. 2001; Chant et al. 2004; Glenn et al. 2004). The U.S. Geological Survey (USGS) put together a data set of surficial sediment type for the entire Atlantic Coast (usSEABED, version 1.0; Reid et al. 2005). Based on these data, bottom sediment composition within the study area is composed mostly of sand.

Within the MAB, the waters between Cape May, New Jersey and Montauk, New York compose the New York Bight (Muir et al. 1989; Figure 1.1). Water movements in the New York Bight are dominated by local winds and river discharge (Bowman and Wunderlich 1976). Density structure is dominated by salinity effects in the fall and winter and by local heating in the spring and summer (Bowman and Wunderlich 1976).

The horseshoe crab trawl survey samples sites within the MAB, specifically between New York and Virginia. Although horseshoe crabs are found in areas outside of the MAB (i.e., along the shores of Maine to the Gulf of Mexico: Shuster 1982), the MAB is the target of the HCRC trawl survey because the largest populations, and therefore the majority of horseshoe crab harvests, occur in this area (Shuster 1982; ASMFC 1998).

Potential bycatch species

Bycatch is monitored in other trawl fisheries along the MAB [e.g., summer flounder, black sea bass, scup, squid (*Loligo pealei*), and butterfish (*Peprilus triacanthus*); NOAA 2003], but to date a bycatch monitoring program does not exist for the horseshoe crab trawl fishery. We would expect the bycatch in the horseshoe crab trawl fishery to be similar to that caught in other trawl fisheries because the nets fish in similar ways. However, since the trawl gear used in the commercial horseshoe crab trawl fishery is designed to efficiently capture horseshoe crabs that are on or within bottom sediment, it is possible that this gear would also be effective at capturing other benthic species. Bycatch monitoring programs have been developed for other benthic trawl fisheries, but many of these fisheries use gears that are equipped with rollers and may not catch all benthic species.

Benthic trawl gear has the potential to capture benthic species such as summer flounder, winter flounder (*Pseudopleuronectes americanus*), blue crab (*Callinectes sapidus*), and American lobster (*Homarus americanus*) and demersal species such as Atlantic croaker (*Micropogonias undulatus*), black sea bass, scup, spot (*Leiostomus xanthurus*), and striped bass (*Morone saxatilis*). All of these species support commercial and recreational fisheries along the Atlantic coast. The total removals as bycatch of these species should be quantified and included in stock assessments for proper management of these fisheries.

Clearnose skate (*Raja eglanteria*) and little skate (*Leucoraja erinacea*) are two species that are incidentally caught in many commercial and recreational fisheries (Packer et al. 2003a, 2003b) and have potential to be captured by horseshoe crab trawl gear. In the past, some skate species have drastically declined because populations were not monitored or managed [e.g., barndoor skate (*Raja laevis*): Casey and Myers 1981]. Starting in the 1960s, barndoor skate populations began to decline due to heavy harvest as bycatch in the groundfish fishery off the coast of New England (Casey and Myers 1981). Populations continued to plummet through the early 1980s, then increased slightly in the 1990s, but were still only 5% of the peak abundance that occurred in 1963 (Sosebee 2000). This example shows the importance of long-term monitoring for all species over a large spatial scale (Casey and Myers 1981). It was not until 2003 that the New England Fishery Management Council developed a management plan for skate (Sosebee 2006). Prior to this plan, most landings reported were not species-specific, categorizing about 99% as "unclassified skates" (Packer et al. 2003a, 2003b). Currently, clearnose and little skate are not considered overfished (Sosebee 2006). The abundance of these species should be carefully watched to ensure that excessive harvest as bycatch is not affecting population sizes.

There are four sea turtles that are commonly found in the northern and middle Atlantic off the coast of the United States [i.e., the green sea turtle (*Chelonia mydas*), loggerhead sea turtle (*Caretta caretta*), leatherback sea turtle (*Dermochelys coriacea*), and Kemp's ridley sea turtle (*Lepidochelys kempii*)], all of which are federally listed (Alverson et al. 1994). Past studies have shown that fisheries, especially southern shrimp trawl fisheries, have been a source of mortality for these species of sea turtles (Crowder and Murawski 1998). The nets used in shrimp trawl fisheries are now equipped with turtle excluder devices (TEDs), which have been shown to reduce bycatch mortality by allowing turtles to escape with minimum damage (Crowder and Murawski 1998). In 2001, NOAA Fisheries required all shrimp trawlers in the United States that fished Atlantic and Gulf of Mexico waters to use TEDs to reduce the bycatch of sea turtles (NMFS 2001). This ruling does not, however, require TEDs on the trawl nets of vessels that use a larger mesh size in these waters. Horseshoe crab trawl vessels are currently allowed to fish waters that are inhabited by sea turtles without using TEDs. If bycatch of turtle species is occurring in the horseshoe crab trawl fishery, it may be necessary to require TEDs on these vessels as well.

Sturgeon populations along the Atlantic coast of the United States are also susceptible to many fishing gears. Atlantic sturgeon (*Acipenser oxyrhynchus*) are very sensitive to overfishing because they mature at age 20 and spawn only once every few years (Bain 1997). This species was harvested for its flesh and caviar until 1998 when a coastwide moratorium was initiated due to low population sizes (ASMFC 2005a).

Despite the moratorium, Atlantic sturgeon are still caught as bycatch in many fisheries along the Atlantic coast (Stein et al. 2004). Of the coastal states, Virginia, Maryland, and North Carolina have the highest bycatch rates associated with landings from otter trawl and gill net fisheries (Stein et al. 2004). Shortnose sturgeon (*Acipenser brevirostrum*) is another species that could be caught as bycatch in trawl fisheries along the Atlantic coast. This species is federally listed as endangered and is greatly affected by excessive harvest due to late maturation and infrequent spawning (Bain 1997). Although the harvest of Atlantic and shortnose sturgeon are prohibited, removal as bycatch in trawl fisheries, combined with the life history characteristics and low population sizes of sturgeon species, can contribute to the decline of these species.

There are many species found along the MAB that could be susceptible to horseshoe crab trawl gear. Collecting species composition data at sites sampled during the horseshoe crab trawl survey will identify species that are susceptible to this specific type of benthic trawl gear. This will assist managers to understanding some of the ecosystem-level effects of the horseshoe crab trawl fishery.

Project objectives

The purpose of this study was to collect data that will assist managers in working toward an ecosystem-level approach to managing the horseshoe crab fishery. The main objective was to identify the non-target species that may be negatively affected by the horseshoe crab trawl fishery. To do this, data were collected and analyzed to:

1. identify species that are susceptible to horseshoe crab trawl gear within the study area,
2. compare species composition and distribution among sites within the study area, and

3. identify the abiotic (geographical and environmental) factors that contribute to differences in species composition and distribution.

A secondary purpose of this study was to test the methods currently used by state agencies to convert landing data from numbers of crabs to pounds of crabs. I also developed a model to predict the weight of a horseshoe crab based on a measured prosomal width in order to determine if this method would provide a more accurate estimation of weight than the current conversion factors do.

Conclusion

Horseshoe crabs are important to many stakeholders. This species supports commercial fisheries along the Atlantic coast, is used by biomedical companies to improve human health, and provides vital nutrients to migrating shorebirds. Over the past ten years, strides have been made to improve the management of horseshoe crabs. An FMP was developed for this species and numerous research projects have been initiated to provide crucial information about this species. However, to date, the species that are commonly caught as bycatch in the horseshoe crab trawl fishery have not been identified. A benthic trawl survey was developed to collect data on horseshoe crab populations along the Atlantic coast. I collected species composition data aboard the horseshoe crab trawl survey and used these data to identify non-target species that are susceptible to the gear used in the commercial fishery. I also used these data to identify the distribution of these species. These results from my study identify non-target species that may be affected by the horseshoe crab trawl fishery and assist managers in understanding some of the potential effects that this fishery may have at an ecosystem-level.

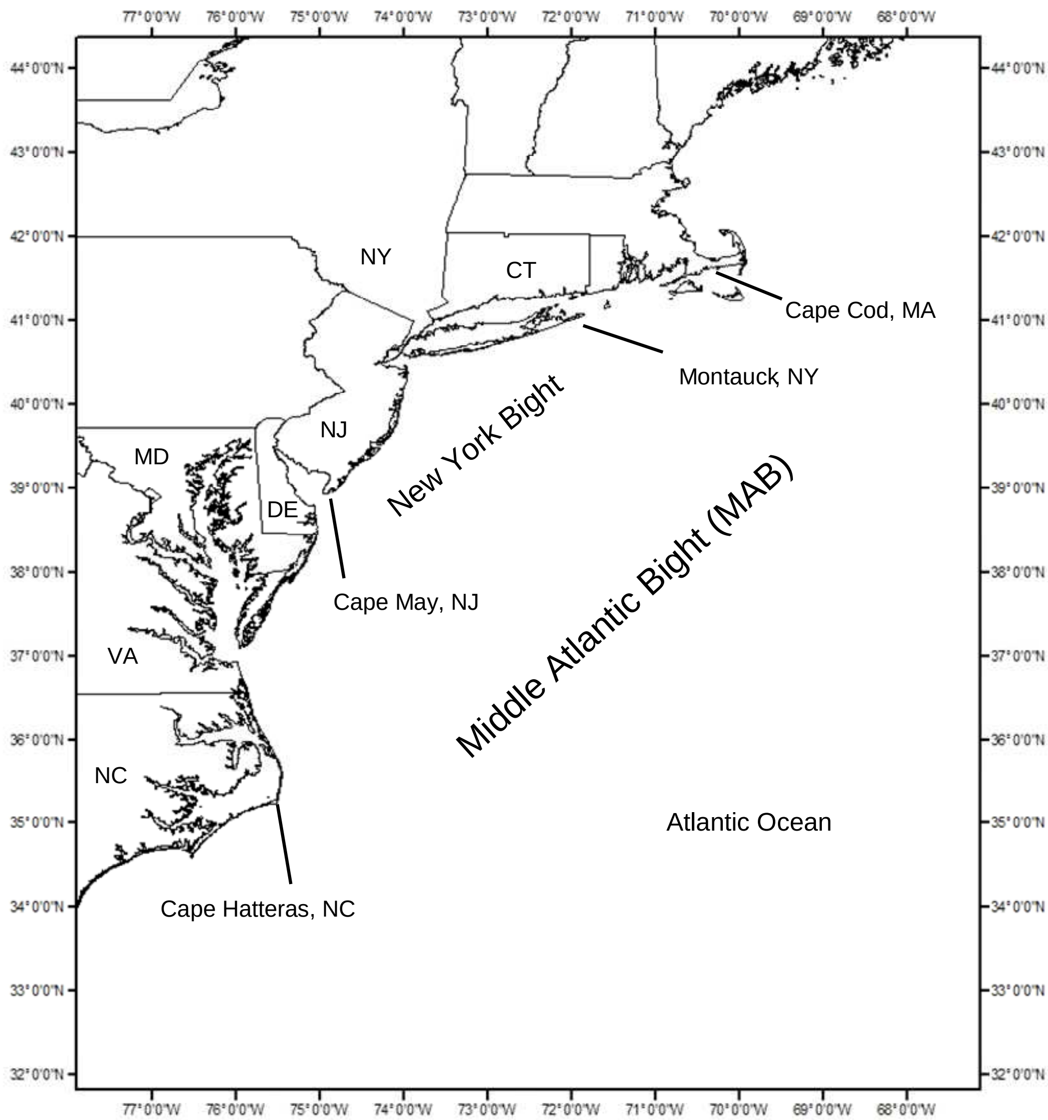


Figure 1.1. A horseshoe crab trawl survey sites along the Middle Atlantic Bight (MAB). The MAB extends from Georges Bank (offshore of Cape Cod, Massachusetts) to Cape Hatteras, North Carolina. The New York Bight is included within the MAB, extending from Montauk, New York to Cape May, New Jersey.

Chapter 2.

Identifying species composition and distribution of bycatch associated with a horseshoe crab (*Limulus polyphemus*) trawl survey along the Middle Atlantic Bight USA.

Abstract

Horseshoe crabs (*Limulus polyphemus*) have been harvested along the east coast of the United States since the 1800s, however a Fishery Management Plan (FMP) was only recently created for this species. To date, there have not been any studies that have attempted to identify or quantify bycatch in the horseshoe crab trawl fishery. A horseshoe crab trawl survey was started in 2001 to collect data on the relative abundance, distribution, and population demographics of horseshoe crabs along the Atlantic coast of the United States. Species composition data were collected at sites sampled by the horseshoe crab trawl survey in 2005 and 2006. Seventy-six different taxa were caught and identified. These species are susceptible to the trawl gear and may have potential to be caught as bycatch in the horseshoe crab trawl fishery. Non-metric multidimensional scaling (NMS) was used to cluster sites and identify the spatial distribution of taxa. Sites strongly clustered into distinct groups, suggesting that species composition shifts between northern and southern sites. Location and bottom water temperature explain most of the variation of species composition among sites. These results identify species that are susceptible to the gear used in the horseshoe crab trawl fishery and describe the distribution of these species during fall months throughout the study area. They also provide researchers with a better understanding of which non-target populations may be affected by the horseshoe crab trawl fishery. Identifying non-target species is a first step

in beginning to understand the effects that the horseshoe crab trawl fishery may have at the ecosystem level.

Introduction

Traditionally, marine fisheries have been managed using a single-species approach. This type of management is not always appropriate because it ignores the direct and indirect effects that harvesting a target species may have at the ecosystem level (Botsford et al. 1997; Pikitch et al. 2004), and the possible effects of distant ecosystem components on the target species itself. Oftentimes the harvest of one species can affect habitat characteristics, predator and prey relationships, and other ecosystem components (Pikitch et al. 2004). Today, managers are adapting a more holistic approach to management. The overall objective of ecosystem-based fisheries management is to not only sustain fisheries, but also to sustain healthy ecosystems to support fisheries (Pikitch et al. 2004). To do this, researchers, managers, and politicians must work to understand the abiotic and biotic interactions between fisheries and the ecosystem, and incorporate these interactions into management plans. Some of the most important aspects of ecosystem management are managing ecosystems to avoid the degradation, minimizing the risk of irreversible changes to natural species assemblages and processes of the ecosystem, obtaining and maintaining long-term socioeconomic benefits, and generating knowledge of ecosystem processes to better understand the consequences of anthropogenic effects (Pikitch et al. 2004).

In the late 1990s, The National Marine Fisheries Service (NOAA Fisheries) encouraged each Fishery Management Council to develop a Fisheries Ecosystem Plan

(FEP) for all major ecosystems within their regions, describing interactions among physical, biological, and human components of the ecosystem (NMFS 1999). One step in developing an FEP is to calculate the total removals within each fishery. This measurement includes all target and non-target species, whether the individuals are landed or caught and released (NMFS 1999). Intense removal of target and non-target individuals can impact communities and the ecosystem by altering population abundance, changing predator and prey relationships, and competitive interactions (Alverson et al. 1994; Kennelly 1995; Crowder and Murawski 1998). NMFS suggests that FEPs should list all species that are susceptible to incidental mortality within each fishery and describe the spatial and temporal distribution of these species (NMFS 1999).

Horseshoe crabs (*Limulus polyphemus*) have supported a commercial fishery along the Atlantic coast since the mid-1800s (Shuster and Botton 1985). Fishers harvest horseshoe crabs for use as bait for American eel (*Anguilla rostrata*) and whelk (*Busycotypus* spp.) fisheries (ASMFC 1998). Additionally, horseshoe crabs are harvested for biomedical companies, which use a clotting agent found in horseshoe crab blood (LAL, Limulus Amoebocyte Lystate) to test for pathogenic endotoxins on medical devices (ASMFC 1998). In the 1990s, fishing effort increased, some local populations of horseshoe crabs began to decline, and many stakeholders became concerned that the fishery was being overexploited (ASMFC 1998). In 1998, an FMP was developed to provide consistent management and regulation for the horseshoe crab fishery (ASMFC 1998). Managers believed a coordinated management strategy along the Atlantic coast could preserve the viability of this resource (ASMFC 1998). However, the data needed for proper management of the stock were lacking (Berkson and Shuster 1999).

In 2001, a benthic trawl survey was initiated by the Horseshoe Crab Research Center (HCRC) at Virginia Tech to collect data on the abundance, distribution, and population demographics of horseshoe crabs along the Atlantic coast of the United States. Although many fishery-independent trawl surveys had sampled this area, most were inefficient in sampling gear or design (ASMFC 1998). For example, the National Marine Fisheries Service (NMFS) has been conducting annual fall and spring surveys along the middle and northern Atlantic coast of the United States since the 1960s, using benthic trawl nets that are equipped with rollers (Reid et al. 1999). Data from these surveys are not appropriate for determining the status of horseshoe crab populations because the gear has the potential to roll over, and thus undersample, buried crabs. Commercial fishers have developed a method for targeting buried horseshoe crabs. Instead of using rollers, commercial fishing nets are modified with heavy chains along the footrope of the trawl to dig into the sediment (Hata and Berkson 2003). The horseshoe crab trawl survey, developed by the HCRC, uses the same gear deployed in the commercial trawl fishery.

I collected species composition data and environmental data during the HCRC trawl survey in the fall season of 2005 and 2006. NOAA Fisheries suggests that managers identify species that are caught as bycatch in each fishery and show how bycatch changes spatially and temporally (NMFS 1999). The data collected during these surveys will assist FEP development for this region. I have created a list of species that are susceptible to commercial horseshoe crab trawl gear and have the potential to be caught as bycatch in the fishery. My results also identify the spatial and seasonal changes of species composition throughout the study area and the environmental variables influence these changes.

Methods

Data collection

I conducted my sampling throughout the Middle Atlantic Bight, from the eastern tip of Long Island, New York, USA (approximately 71 50’W and 41 04’N) to the southern tip of the Delmarva Peninsula, Virginia, USA (approximately 75 55’W, and 37 05’N) (Figure 2.1). I sampled sites that were randomly selected using a stratified sampling approach based on the methods of Hata and Berkson (2004). Sites between Atlantic City, New Jersey (approximately 74 5’W and 39 20’N) and the southern tip of the Delmarva Peninsula, Virginia were randomly selected based on the site’s distance from shore (inshore and offshore) and bottom topography (trough and non-trough). Sites within state waters (0 – 3 nautical miles) were defined as inshore sites, and those within federal waters (3 – 12 nautical miles) were defined as offshore sites. Troughs were defined as areas where bottom depressions were at least 2.4 m deep, no more than 1.8 km wide, and more than 1.8 km long. All other sites were defined as non-trough. Sites north of Atlantic City, New Jersey were randomly selected based on their distance from shore. Sites were also designated as northern or southern sites. All sites sampled north of Atlantic City, New Jersey were classified as northern sites and all sites sampled south of Atlantic City, New Jersey were classified as southern sites.

Commercial stern trawlers sampled each site using a flounder trawl net with modified ground gear: a Texas sweep that consisted of a chain line instead of a footrope (Hata and Berkson 2003; 2004). This type of net is commonly used in the commercial horseshoe crab trawl fishery (Hata and Berkson 2003). Each site was towed for 15 min at a speed of 4.6 to 5.6 km/h.

I sampled 156 sites during two fall seasons (September – November): 73 sites in 2005 and 83 sites in 2006 (Figure 2.1). I recorded the time, depth, and geographic coordinates (latitude/longitude) at the start and end of each tow. Bottom water temperature and bottom salinity were measured at the beginning of each tow. After the net had been towed for 15 min, I identified and sorted the catch. Jellyfish and sea anemones (Phylum Cnidaria) were classified to the lowest identifiable group. I identified all other taxa to the species level, except little and winter skates (*Leucoraja* spp.), bullnose and southern eagle rays (*Myliobatis* spp.), seastars (*Asterias* spp. and *Astropecten* spp.), moon snails (*Lunatia* spp.), hermit crabs (*Pagurus* spp.), and spider crabs (*Libinia* spp.), which were grouped by genus. The total count and biomass (to the nearest 0.01 kg) were recorded for all taxa. For large catches, I used a haphazard method to collect a subsample of all taxa that were in high abundance. I weighed the subsample and counted the remaining individuals not included in the subsample. I estimated the total weight of the counted individuals based on the weight of the subsample, and summed these values to calculate the total weight for the subsampled taxa. Because some fishes were too large to be weighed, I estimated the mass. I used the body length to estimate their weight using published length-weight relationships (Franks et al 1998; NMFS 2003). I used prosomal-width-to-crab-weight relationships to estimate the total biomass of horseshoe crabs from each tow based on sex and maturity stage (Chapter 3, this thesis). For each trawl, I measured the total length (to the nearest 1 mm) for all or a subsample of at least 15 individuals of most taxa. I also recorded the weight of each individual when time was available.

Data analysis

Descriptive analysis

For each survey, I calculated total count and biomass, percent occurrence, and percent total catch for each taxa. The distance towed (km) was calculated using the beginning and ending coordinates for each sites. I used the distance towed at each site to calculate the catch per unit effort (CPUE) for the count (number of individuals/km) and biomass (kg/km) of each taxa for 2005 and 2006.

Community analysis

I analyzed CPUE of count and biomass data for sites sampled in 2005, 2006, and both years combined. I normalized these data using a log transformation and removed rare species (i.e., those present in less than 5% of tows) from the data set (Gomes et al. 1992; Mahon et al. 1998; McCune and Grace 2002; Sousa et al. 2005). I used non-metric multidimensional scaling (NMS; PC-ORD version 5) to group sites based on similarity of species composition and local abundance. NMS, an ordination method that ranks samples based their on similarities and is considered the most generally effective of all ordination methods (McCune and Grace 2002). Many studies have used similar methods to identify fish assemblages in marine and estuarine systems (Greenstreet et al. 1999; Able et al. 2001; Beentjes et al. 2002; Layman 2000; Massuti and Moranta 2003; Fock et al. 2004; Stergiou et al. 2006). The main advantages of NMS are that this method does not assume a linear relationship among variables, works well with data sets that contain many zeroes because distance measures are rank transformed, and allows the use of any distance measure (McCune and Grace 2002). I used a Bray-Curtis distance measure to calculate the dissimilarity between sites. The Bray-Curtis distance measure is often used in community analyses because it not affected by zeroes and weights more abundant species more heavily than less abundant species (Bray and Curtis 1957; McCune and Mefford 2006). NMS provided a stress value for each ordination plot which measures the departure from monotonicity in the plot of dissimilarity among points in original space compared to the distance among points in ordination space (McCune and Grace 2002). The closer the plot in ordination space fits the original data, the better the fit and the lower the stress (McCune and Grace 2002). According to Clarke (1993) an ordination plot with a stress value less than 0.20 provides an accurate representation of the data.

Community analysis: influence of abiotic factors

To determine whether bottom sediment type contributed to the change in species composition throughout the study area, I used a data set from the United States Geological Service (USGS, usSEABED, version 1.0; Reid et al. 2005) to estimate the bottom composition (percent clay, mud, sand, and gravel) at each site trawled (Reid et al. 2005). I used Geographic Information System (GIS; ArcMap, version 9.1) to interpolate point data in the usSEABED database and estimate bottom composition at each of the sites towed. I used a cross-validation method to determine which interpolation method was best for this data set. I randomly selected 10% of points from the original data set and removed them from interpolation (n = 4271). I used the remaining 90% to interpolate bottom type throughout the study area using 1000 meter plots (n = 41557).

Each interpolation method was cross-validated by comparing the estimated values of bottom composition to the observed values. I determined that inverse distance weighting (IDW) provided the closest estimates of bottom type for the USGS data set within the study area and then used all data points to determine across-site estimates of bottom type.

I used PC-ORD (version 5) to incorporate abiotic factors, including bottom temperature, bottom salinity, depth, and bottom composition, in each of the final NMS solutions. The direction and length of each resulting vector indicated how strongly a variable was correlated with a group of sites. An R^2 value was calculated for each abiotic factor, indicating the amount of variation explained along each NMS axis.

I also used PC-ORD (version 5) to perform Multi-Response Permutation Procedures (MRPP) to determine if changes in species composition were due to location (north or south of Atlantic City, New Jersey), distance from shore (inshore or offshore), or time of tow (day or night). MRPP output provided a p-value to indicate statistical significance, however with a large sample size, statistical significance can occur even if the groups being tested do not have a large effect on species composition (McCune and Grace 2002). To determine whether groupings had an effect on species composition, a chance-corrected within-group agreement index, *A*, was given to describe the homogeneity of species composition within groups compared to what is expected at random (McCune and Grace 2002). When $A = 1$, all items are identical within the groups and when $A = 0$, all items within the groups are the same as expected by chance. A fairly high *A*-value ($A > 0.30$) is usually associated with an ecologically significant difference between the groups (McCune and Grace 2002).

Community analysis: species composition

Differences in species composition among sites were identified using PC-ORD (version 5). I overlaid the main matrix which displays the relative abundance of species at each site on the plot in ordination space. This also allowed me to identify the species that were abundant at each group of sites.

Results

Descriptive analysis

In 2005 and 2006, distances of 88.66 km and 104.95 km were trawled, respectively. Over both years, 46,415 individuals from seventy-six different taxa were caught (n = 60 in 2005, n = 69 in 2006), including 47 finfish species from 33 families (Table 2.1). Skates comprised more than half of the total biomass and horseshoe crabs comprised one-third of the total biomass caught during each year surveyed (Figure 2.2). CPUE (kg/km) was highest for little (*Leucoraja erinacea*) and winter (*Leucoraja ocellata*) skate, horseshoe crab, and clearnose skate (*Raja eglanteria*) (Table 2.1). Clearnose skate, horseshoe crab, summer flounder (*Paralichthys dentatus*), spider crab, and windowpane flounder (*Scophthalmus aquosus*) were caught throughout the entire survey area and were present in over half of tows (Table 2.2; Figure 2.3). Horseshoe crabs were found at sites throughout the study area, but were more abundant at southern sites (Figure 2.4).

Community analysis

NMS for count (number of individuals/km) and biomass (kg/km) data for 2005, 2006, and both years combined resulted in two well-supported groups of sites based on species presence and abundance. These two groups were consistent throughout analyses: one group was comprised of all southern sites (sites south of Atlantic City, New Jersey) and the other group of mostly northern (sites north of Atlantic City, New Jersey) sites. NMS of biomass data collected in 2005 and 2006 both resulted in 2-dimensional solutions with stress values of 0.14 in 2005 and 0.15 in 2006 (Figure 2.5; Figure 2.6). NMS of biomass data from both years combined resulted in a 3-dimensional solution with a stress value of 0.11 (Figure 2.7). When the sites sampled in 2005 and 2006 were combined for analyses, three subgroups were formed within the larger group consisting of mostly northern sites. These subgroups consisted of an intermediate subgroup, comprised of northern and southern groups; a subgroup consisting of all northern sites; and a subgroups consisting of all deep water, northern sites (Figure 2.7).

Community analysis: influence of abiotic factors

Species composition varied with location, specifically whether a site was located south or north of Atlantic City, New Jersey. The difference in species composition due to location is evident from the NMS ordination plots in which most southern and northern sites grouped separately. MRPP results indicated that the grouping of north versus south was statistically and ecologically significant for all data sets, with *A*-values ranging from 0.28 to 0.37 (Table 2.3).

Bottom temperature and salinity explained most of the observed variation in the differences in species composition of sites sampled in 2005 and both years combined (Table 2.4, Figure 2.5, Figure 2.7). These abiotic factors were inversely correlated; higher bottom temperatures were more common at southern sites and higher bottom salinities at northern sites. Bottom temperature and salinity did not explain much of the variation in species composition among sites sampled in 2006 (Table 2.4; Figure 2.6).

Depth explained the most variation in species composition among sites sampled in 2006 ($R^2 = 0.51$; Table 2.4). Other factors, including depth, distance from shore, time that a site was sampled, and bottom type, did not greatly contribute to the predicted clustering of sites sampled in 2005, 2006, or when all data were combined (Table 2.4).

Community analysis: species composition

Species presence and abundance differed between the group comprised of southern sites and the group comprised of mostly northern sites (Table 2.1, Table 2.2). The differences in species composition between northern and southern sites seemed to be driven by the distribution of southern species; many southern species were not found at any northern sites. Many northern species were found at various sites throughout the study area, but were more abundant at northern sites.

Species composition differed among the three subgroups within the larger group comprised of mostly northern sites. The first subgroup was comprised of northern and southern sites ("Intermediate" subgroup, Figure 2.7). All of the southern sites within this subgroup were sampled in November and had colder bottom temperatures. The species composition at these sites was very different than at southern sites that were sampled earlier in the fall. Most species that were common to southern sites were not found at southern sites that were sampled in November. Channeled (*Busycotypus canaliculatus*) and knobbed (*Busycon carica*) whelks were found at all southern sites independent of bottom temperature. Migrating species were also found at the southern sites sampled in

November, including little skate, winter skate, bluefish (*Pomatomus saltatrix*), striped bass (*Morone saxatilis*), and spiny dogfish (*Squalus acanthias*).

The species composition of northern sites sampled in deeper, offshore waters differed slightly from most northern sites ("Deep" subgroup, Figure 2.7). Deep-sea scallop (*Placopecten magellanicus*) and jonah crab (*Cancer borealis*) were more abundant at deep water sites than at other sites.

Lastly, some species were common throughout the study area, independent of location (Table 2.1, Table 2.2). These species included clearnose skate, hermit crab, and windowpane flounder, sea star (*Asterias* spp.), spider crab, and summer flounder. Clearnose skate and horseshoe crabs were more abundant at southern sites and windowpane flounder and hermit crab at northern sites.

Discussion

Descriptive analysis

The gear used during this survey was effective at targeting horseshoe crabs and also at catching many non-target organisms. The majority of the species caught during the surveys were benthic fishes or invertebrates that live or feed on the bottom (Table 2.1). However, catch was not limited to benthic organisms. Since the benthic trawl net also fishes the a few meters of the water column, many demersal fishes were caught during the survey (Table 2.1).

Community analysis

NMS for all data sets resulted in two distinct groups of sites based on differences in species composition. I can be confident the ordination plots of each of these analyses are accurate based on their corresponding stress value. The stress values for each final solution of NMS were relatively low, ranging from 0.11 to 0.15. All of these stress values are less than 0.20, and therefore the ordination plots provide a useful interpretation of the data (Clarke 1993). The values I have reported are also within the range of stress values from previous published studies (Greenstreet et al. 1999; Layman 2000; Massuti and Moranta 2003; Stergiou et al. 2006).

Community analysis: influence of abiotic factors

Based on my results, the clustering of sites due to changes in species presence and abundance was mainly correlated with two factors: latitudinal location and bottom temperature. The shift in species composition due to location can be explained by the latitudinal distribution of species along the Middle Atlantic Bight. Most species found at southern sites were not found at any northern sites during fall months [e.g., knobbed whelk, channeled whelk, southern stingray (*Dasyatis americanus*), bullnose ray (*Myliobatis freminvillei*), and southern eagle ray (*Myliobatis goodei*); Figure 2.8]. Species more common to southern sites, such as Atlantic croaker (*Micropogonias undulatus*, ASMFC 2005b) and spot (*Leiostomus xanthurus*, ASMFC 1987) have a distribution that is centered at the southern portion of the study area. Other species were more common to northern sites including little skate (Packer et al. 2003a), spiny dogfish (McMillan and Morse 1999), and winter flounder (*Pseudopleuronectes americanus*, Pereira et al. 1999) and have distributions that are centered in northern portion of the study area during fall months (Figure 2.9).

Another explanation for the shift in species composition between northern and southern sites may be the wind-driven upwelling events that occur along the inner shelf off New Jersey (Munchow and Chant 2000; Chant et al. 2004). During summer months, water on the shelf becomes strongly stratified, with surface temperatures becoming much warmer than bottom temperatures (Chant et al. 2004). Southwesterly winds, driven by the Bermuda High, begin to move up the coast, counteracting the northeasterly water currents (Chant et al. 2004). Waters begin to flow offshore and cold, bottom water is pulled to the surface by advection (Munchow and Chant 2000). This flow reversal has been documented many times in the past, but only during summer months. However, this upwelling process could affect the shift in species composition between northern and southern sites by affecting the transport of larvae (Norcross and Shaw 1984; Cowen et al. 1993).

Differences in species composition also were correlated with bottom water temperature. Most species common to southern waters can only tolerate warmer temperatures [e.g., Atlantic croaker: Miglarese et al. 1982; striped burrfish (*Chilomycterus schoepfii*): Holmquist 1997] and are less likely to be found in northern waters during fall months. Species common to northern waters (e.g., little skate: Packer et al. 2003a; winter flounder: Perceira 1999) are able to tolerate cooler temperatures and persist in these areas when water temperatures drop. Other species can tolerate a wide range of temperatures (e.g., horseshoe crab: Shuster 1979, 1982; summer flounder: Packer et al. 1999; and windowpane flounder: Chang et al. 1999) and are common throughout the entire survey area independent of bottom temperature.

Other abiotic factors did not strongly explain the variation in species composition among sites. Changes in species composition were statistically significant, but not ecologically significant ($A < 0.30$; Table 2.3), due to distance from shore (inshore or offshore). This is not surprising since inshore and offshore sites had very similar characteristics. At most, offshore sites were 12 nautical miles from the shore and did not change much in depth or temperature from inshore sites. Changes in species composition also were not ecologically significant based on the time that a site was sampled (day or night). Research has shown that some species, such as little skate, are more active at night and remained buried in depressions during the day (Packer et al. 2003a). However, during the survey, little skate were caught at large numbers independent of time of day. It is possible that the heavy chains on the trawling gear allowed us to catch individuals that were buried in sediment, reducing variation between day and night sampling. Bottom type did not contribute largely to the grouping of sites, although research has shown that species are associated with certain bottom types and grain sizes (Methratta and Link 2006). This is probably because the categories lacked fine-scale characteristics of bottom type, resulting in very similar estimates of bottom type throughout the study area (Figure 2.10). If more variables were included in my analyses, such as grain size and benthic structure, I would predict that bottom type would have explained more variance in the grouping of sites.

Community analysis: species composition

Species presence and abundance were explained by the latitudinal distribution and preferred temperature of species found throughout the study area. Many species were only found at southern sites when bottom temperatures were between 19^{o} and 24.5^{o}C (Table 2.1, Table 2.2). These species were not found at sites of cooler temperatures (i.e., northern sites or southern sites that were sampled in November), suggesting that these species prefer warmer water temperatures. Atlantic croaker, spot, and southern kingfish

(*Menticirrhus americanus*) were common to southern sites. Spot and southern kingfish are commonly found within southern waters (Wenner and Sedbury 1989). Research has shown that Atlantic croaker are common to water temperatures between 9^{o}C and 31^{o}C, but most abundant at temperatures above 24^{o}C (Miglarese et al. 1982).

Within the larger group that was comprised of mostly northern sites, species composition varied among subgroups of sites. One subgroup was comprised of northern and southern sites ("Intermediate" subgroup; Figure 2.7). All of the southern sites that joined this subgroup were sampled in November. The bottom temperatures at these southern sites (13 - 15^{o}C) were very similar to most northern sites (14.5 - 19^{o}C). Due to the colder water temperatures, fewer southern species were found at these sites. Atlantic croaker numbers were lower at southern sites sampled in November than southern sites sampled early in the fall, probably because this species had migrated to warmer, southern waters (Miglarese et al. 1982; ASMFC 2005b). Other southern species [e.g., bullnose ray, smooth butterfly ray, southern stingray, southern eagle ray, southern kingfish *(Menticirrhus americanus*), and spiny butterfly ray (*Gymnura altavela*)] were not found at any southern sites sampled in November, most likely because these species prefer warmer water temperatures.

Less mobile species, such as channeled and knobbed whelk, were found at the southern sites independent of bottom temperature. Whelks are found throughout the study area, but are most abundant in the southern region of the survey area, south of New Jersey (Gosner 1978). It seems that these species can tolerate cool, winter temperatures at southern sites because they did not migrate as water temperatures dropped. Some research has shown that whelks move along intertidal and subtidal flats, but do not migrate long distances (Walker 1988). When water temperatures drop, whelks bury themselves instead of migrating (Walker 1988). This may explain why channeled and knobbed whelks were found at southern sites even in November when temperatures had dropped.

Species composition at southern sites sampled in November was very similar to the species composition at most northern sites. This suggests that many of these species had migrated, moving from northern sites to southern sites as water temperatures drop. Studies have shown that bluefish (Shepherd and Packer 2006), striped bass (ASMFC 1981), and spiny dogfish (Bigelow and Schroeder 1948) migrate in a southern direction in the winter as water temperature changes.

Many species that were found at northern sites can tolerate, but are not restricted to, cooler northern waters during fall months. All northern sites were sampled when bottom temperature ranged from 14.5^{o} to 19^{o}C. Past surveys and studies have shown that winter flounder (Perceira et al. 1999), spiny dogfish (Bigelow and Schroeder 1948), little skate (Packer et al. 2003a), smooth dogfish (*Mustelus canis*; Compagno 1984), and black sea bass (*Centopristis striata*; Drohan et al. 2007) are all common to temperatures between 6 and 15^{o}C. Rock crabs (*Cancer* spp.) have a wide distribution throughout the Mid-Atlantic bight and are commonly found at temperatures of 8^{o}C to 18^{o}C during fall months (Stehlik et al. 1991). Hermit crabs are also common throughout the Atlantic coast (Gosner 1978) within an optimal temperature from 15^{o}C to 17^{o}C (Young 1991).

Changes in species composition were also found at northern sites sampled in deep waters, when temperatures ranged from 16^{o}C to 18^{o}C. Northern species at these sites included spiny (Bigelow and Schroeder 1948) and smooth dogfish (Bigelow and

Schroeder 1948), rock crab (Stehlik et al. 1991), little skate (Packer et al. 2003a), and winter flounder (Perceira 2004), all which can tolerate cooler water temperatures. Some of these species are also common to deeper waters. Spiny dogfish also have been documented to move offshore during winter months (Bigelow and Schroeder 1953; Colvocoresses and Musick 1980). Deep sea-scallops are commonly found in deep waters (10 to 20 meters) along the middle Atlantic coast (Stewart and Arnold 1994). Jonah crabs are typically found at temperatures of 3^{o}C to 23^{o}C, migrating to offshore zones of the Mid-Atlantic Bight during winter months (Stehlik et al. 1991).

Finally, there were many species that were found throughout the entire study area (e.g., clearnose skate, horseshoe crab, summer flounder, and windowpane flounder). These species are common throughout the Middle Atlantic Bight and can tolerate a wide range of bottom temperatures. Studies have shown that clearnose skate are most commonly caught at bottom temperatures between 18^{o}C and 22^{o}C, but have been found at temperatures as low as 10^{o}C (Packer et al. 2003b). Surveys have consistently caught most adult summer flounder at temperatures between 9^{o}C and 26^{o}C (Packer et al. 1999) and adult windowpane flounder at temperatures between 12^{o}C and 18^{o}C (Chang et al. 1999). Horseshoe crabs are considered ecologic generalists that can live in a wide range of environmental conditions (ASMFC 1998). This species is found along the Atlantic coast and is capable of surviving temperatures from below 0^{o}C to 41^{o}C (Shuster 1979, 1982). These species can tolerate a wide range of temperatures, allowing them to live throughout the study area during all months.

Species assemblages

Many studies have analyzed NMFS fall trawl survey data to identify species assemblages along the Middle Atlantic Bight (Colvocoresses and Musick 1984; Phoel 1985; Gabriel 1992; Mahon et al. 1998). Species assemblages that included horseshoe crabs were not identified during these studies because data were collected using benthic trawl nets equipped with rollers, a type of gear that has potential to under-sample horseshoe crabs. The results from these studies, however, could be used to compare the co-occurrences of benthic and demersal species. During the horseshoe crab trawl survey, winter flounder, little skate, and spiny dogfish were common at northern sites. These co-occurrences are similar to results reported by Colvocoresses and Musick (1984), Phoel (1985), and Mahon (1998). Smooth dogfish, summer flounder, and windowpane flounder also commonly co-occurred throughout the study area, a result that is similar to past studies (Colvocoresses and Musick 1984; Phoel 1985; Gabriel 1992; Mahon 1998).

Data collected from the HCRC trawl survey could not be used to identify species assemblages due to the short time span of the study. Studies that persist over long periods of time are more appropriate for determining species assemblages because the chance of incidental co-occurrence is reduced (Tyler et al. 1982; Gomes et al. 2001). With only two years of data, I could not determine whether species co-occurred by chance alone. I did, however, identify species that were commonly caught with horseshoe crab throughout the survey area. Horseshoe crab were commonly caught with knobbed and channeled whelks, clearnose skate, southern stingray, spider crab, and summer flounder at southern sites; and clearnose skate, hermit crabs, little and winter skates, spider crab, windowpane flounder, striped searobin (*Prionotus evolans*), and summer flounder at northern sites. This does not necessarily mean that these species compose an assemblage; all of these species may have just been very abundant throughout the study area. Over both years, horseshoe crabs were caught at only 32 out of 156 sites. To understand whether these species do commonly occur with horseshoe crabs, more sites where horseshoe crabs are not present must be sampled and data should be collected over numerous years. This will give us a better understanding of whether these species are commonly associated with horseshoe crabs.

Management Implications

In recent years, managers have been working toward an ecosystem-level of management; developing FEPs to incorporate interactions among physical, biological, and human components of the ecosystem into fisheries management plans (Botsford et al. 1997). One objective in developing FEPs is to calculate the total removal of target and non-target species in each fishery. NOAA Fisheries suggests that to achieve this objective, managers must identify a list of species that are caught as bycatch in each fishery and show how bycatch changes spatially and temporally (NMFS 1999).

My study identified 76 taxa that are susceptible to the trawl gear that is used in the commercial horseshoe crab trawl fishery. Based on life history characteristics, conservation status, and current management strategies it seems that some species may be more heavily affected by removal as bycatch in the horseshoe crab trawl fishery than others (Table 2.5). By examining these categories, we can prioritize the species susceptible to the trawl gear used in the horseshoe crab fishery and give managers a better understanding of which species may be in need of the most protection.

Based on this study, it seems that skates have the highest potential to be caught as bycatch in the horseshoe crab trawl fishery. Skates comprised over half of the biomass during both sampling years, and clearnose, little, and winter skates had high CPUE values. Clearnose, little, and winter skates also have low population doubling times (4.5 – 14 years, Table 2.5; Fishbase 2007) suggesting that the populations of these species may be affected by high removal as bycatch. The commercial harvest of skate species is currently monitored and managed by the New England Fishery Management Council (NEFSC 2007). Clearnose and little skate populations are not considered to be overfished (NEFSC 2007); however overfishing is occurring on winter skate populations (NEFSC 2007). Therefore, it is essential that the bycatch of skates within this fishery be quantified and included in management plans.

Many other managed species, such as summer flounder and spiny dogfish, were also commonly caught throughout the study area (Table 2.1; Table 2.2). Summer flounder currently support a large fishery along the Atlantic coast (Packer et al. 1999). Although this fishery is not currently overfished, overfishing is occurring (ASMFC 2006a). By quantifying bycatch in the horseshoe crab trawl fishery, managers can include total removals in stock assessments for summer flounder.

Spiny dogfish were also identified as susceptible to horseshoe crab trawl gear, although they were not caught in high abundance throughout the study area (Table 2.5). This species is of concern because it is currently listed on the IUCN (The World Conservation Union) Red List as vulnerable, which is defined as facing a high risk of extinction in the wild (IUCN 2006). Spiny dogfish also have a very low population doubling time (>14 years, Table 2.5; Fishbase 2007), suggesting that populations may not be quick to recover when individuals are removed. Although spiny dogfish are currently listed on the IUCN Red List, this species is still fished along the eastern coast of the United States (NEFSC 2007) and the stock is not considered to be overfished (NEFSC 2007).

Many protected species, including Atlantic angel shark (*Squatina dumeril*), Atlantic sturgeon (*Acipenser oxyrinchus*), and dusky shark (*Carcharhinus obscurus*), also were identified as susceptible to horseshoe crab fishery trawl gear. Harvest of these species is currently restricted within United States waters (Fishbase 2007; NEFSC 2007). Atlantic sturgeon and dusky shark are listed as near threatened on the IUCN Red List (IUCN 2006). Atlantic angel shark is not currently listed on the IUCN Red List because not enough data are available to properly access the status of this species IUCN 2006). All of these species have low population sizes and life history characteristics that do not allow quick recovery of populations when individuals are removed (Fishbase 2007; NEFSC 2007; Table 2.5). Therefore, it is important to quantify the number of individuals being removed as bycatch in the horseshoe crab trawl fishery.

Lastly, there were many species that were caught during the horseshoe crab trawl survey that are not protected or managed within U.S. waters, including smooth dogfish, cownose ray (*Rhinoptera bonasus*), and southern stingray. Smooth dogfish, for example, is commercially and recreationally fished in U.S. waters (Fishbase 2007), but is not currently managed. The same is true for cownose ray, although no directed fishery exists in the U.S. (Fishbase 2007). Both of these species have low population doubling sizes and are listed as near threatened on the IUCN Red List (Fishbase 2007; IUCN 2006; Table 2.5). Lastly, southern stingray is commercially harvested and has a very low population doubling size (Table 2.5; Fishbase 2007). This species is not currently listed on the IUCN Red List because the data is lacking to properly assess its population status (IUCN 2006).

Overall, it seems some of the species identified as susceptible to horseshoe crab trawl gear may be more vulnerable than others. A small number of species caught during this survey have life history characteristics or low population sizes that suggest populations will not be quick to rebound. Other species are already listed as near threatened or protected by management regulations. Managers may want to focus conservation actions on these species instead of other populations that seem to be relatively stable, currently managed, and considered not to be overfished.

This study also provides data on the spatial and temporal distribution of the species that are susceptible to horseshoe crab trawl gear during fall months. My results suggest that species composition shifts due to location and bottom temperature. Although bottom temperature seems to be the driving force in determining species distribution (e.g., southern waters have warmer temperatures and northern waters have cooler temperatures), setting regulations based on location is much easier for management purposes. My study suggests that during the fall season species composition will shift north and south of Atlantic City, New Jersey. My study also identified which species have potential to be caught in each of these areas. By documenting and understanding species distribution and migration patterns, fishing areas can be closed to protect species if needed.

Although, it may not be feasible for managers to set regulations based on water temperature, this result is still important for management purposes. Preferred temperature ranges have been published for many species along the Atlantic coast. My data suggests that, during the fall season, species are commonly found in areas where water temperatures are within the preferred temperature ranges identified by past studies. By collecting bottom temperature data along the Atlantic coast, managers can use previously published data on preferred temperature ranges to predict species that are likely to be caught as bycatch in the horseshoe crab trawl fishery during other seasons.

Conclusion

I have identified species that are susceptible to horseshoe crab trawl gear in waters off the coasts of New York to Virginia, where the majority of the commercial trawl fishery occurs (ASMFC 1998). This may not be a complete list because data were only collected during fall months over a two-year time span; however it is a first step in identifying bycatch within the commercial horseshoe crab trawl fishery. My study suggests that species composition shifts by location, specifically north and south of Atlantic City, New Jersey, and bottom temperature. These results allow managers to better understand the distribution of species along the Atlantic coast. To fully identify and quantify bycatch occurring in the commercial fishery along the Atlantic coast, data need be collected directly from the commercial fishery. This will allow managers to calculate total removals of all species which is a critical component to the development of ecosystem-level management in the horseshoe crab trawl fishery.

Table 2.1. Catch per unit effort (CPUE) for all species caught during the 2005 and 2006 horseshoe crab trawl survey. Sites between Montauk, New York and Atlantic City, New Jersey were defined as northern sites and sites between Atlantic City, New Jersey and the southern tip of the Delmarva Peninsula were defined as southern sites.

		2005 CPUE (kg/km)			2006 CPUE (kg/km)		
Taxa- common name	**Scientific name**	**Southern Sites**	**Northern Sites**	**All sites**	**Southern Sites**	**Northern Sites**	**All sites**
American lobster	*Homarus americanus*	0.01	0.08	0.04	<0.01	0.04	0.02
Asteriid sea star	*Asterias* spp.	0.30	1.18	0.65	0.15	0.56	0.39
Atlantic angel shark	*Squatina dumeril*	2.25	0.00	1.35	0.50	0.00	0.21
Atlantic croaker	*Micropogonias undulatus*	5.22	0.04	3.15	1.12	0.01	0.47
Atlantic sharpnose shark	*Rhizoprionodon terraenovae*	0.00	0.00	0.00	0.09	0.00	0.04
Atlantic sturgeon	*Acipenser oxyrinchus*	0.00	1.73	0.69	1.11	0.87	0.97
Atlantic thread herring	*Opisthonema oglinum*	<0.01	0.00	<0.01	0.00	0.00	0.00
Black drum	*Pogonias cromis*	<0.01	0.00	<0.01	0.03	0.00	0.01
Black sea bass	*Centopristis striata*	0.01	0.03	0.02	0.00	0.07	0.04
Blue crab	*Callinectes sapidus*	0.45	0.12	0.32	0.09	0.02	0.05
Blue mussel	*Mytilus edulis*	0.00	0.00	0.00	0.00	0.03	0.02
Bluefish	*Pomatomus saltatrix*	0.32	0.64	0.45	0.10	0.36	0.25
Bullnose/Southern eagle ray	*Myliobatis freminvillei/goodei*	3.29	0.00	1.97	0.28	0.00	0.11
Butterfish	*Peprilus triacanthus*	0.01	<0.01	0.01	0.04	<0.01	0.02
Channeled whelk	*Busycotypus canaliculatus*	1.12	0.00	0.67	1.12	0.00	0.46
Clearnose skate	*Raja eglanteria*	49.15	2.20	30.37	16.19	2.83	8.35
Cobia	*Rachycentron canadum*	0.00	0.00	0.00	1.13	0.00	0.46
Conger eel	*Conger oceanicus*	0.00	0.00	0.00	0.01	0.00	0.00
Cownose ray	*Rhinoptera bonasus*	0.77	0.11	0.50	0.58	0.00	0.24
Deep-sea scallop	*Placopecten magellanicus*	0.00	3.09	1.24	0.00	1.12	0.66
Dusky shark	*Carcharhinus obscurus*	0.40	0.00	0.24	0.28	0.00	0.12
Fourspot flounder	*Paralichthys oblongus*	0.00	<0.01	<0.01	0.00	0.00	0.00
Gray triggerfish	*Balistes capriscus*	0.00	0.03	0.01	0.00	0.00	0.00

Table 2.1 continued.

Taxa- common name	Scientific name	2005 CPUE (kg/km) Southern Sites	2005 CPUE (kg/km) Northern Sites	2005 CPUE (kg/km) All sites	2006 CPUE (kg/km) Southern Sites	2006 CPUE (kg/km) Northern Sites	2006 CPUE (kg/km) All sites
Hermit crab	*Pagurus* spp.	0.03	0.32	0.15	0.29	0.46	0.39
Hogchoker	*Trinectes maculatus*	0.00	0.00	0.00	<0.01	0.00	<0.01
Horseshoe crab	*Limulus polyphemus*	80.61	18.66	55.82	131.95	8.10	59.24
Jellyfish (unk)	n/a	0.00	0.00	0.00	0.04	0.01	0.02
Jonah crab	*Cancer borealis*	<0.01	1.29	0.51	<0.01	0.08	0.05
Knobbed whelk	*Busycon carica*	7.31	0.00	4.39	1.96	0.00	0.81
Lady crab	*Ovalipes ocellatus*	<0.01	<0.01	<0.01	0.01	<0.01	0.01
Lesser blue crab	*Callinectes similis*	0.00	0.00	0.00	<0.01	0.00	<0.01
Lightning whelk	*Busycon contrarium*	0.03	0.00	0.02	0.01	0.00	0.00
Little/Winter Skate	*Leucoraja erinacea/ocellata*	3.58	95.86	40.50	19.72	134.87	87.32
Long-finned squid	*Loligo pealei*	0.01	0.02	0.01	0.07	0.01	0.03
Margined sea star	*Astropecten* spp.	0.00	0.05	0.02	0.00	0.01	0.00
Northern stargazer	*Astroscopus guttatus*	0.06	0.08	0.07	0.52	0.00	0.21
Monkfish	*Lophiodes americanus*	0.00	0.24	0.09	0.00	0.00	0.00
Moon snail	*Polinices heros*	0.00	0.24	0.10	<0.01	0.11	0.07
Mud crab	*Panopeus* spp.	<0.01	0.00	<0.01	0.00	0.00	0.00
Northern puffer	*Sphoeroides maculatus*	0.01	0.03	0.02	0.02	<0.01	0.01
Northern searobin	*Prionotus carolinus*	0.10	0.03	0.07	0.01	0.11	0.07
Octopus (unknown)	n/a	0.00	0.00	0.00	<0.01	<0.01	<0.01
Pigfish	*Orthopristis chrysoptera*	0.00	0.00	0.00	<0.01	0.00	<0.01
Purple sea urchin	*Arbacia punctulata*	0.02	<0.01	0.01	0.02	0.02	0.02
Quahog	*Mercenaria mercenaria*	0.00	0.00	0.00	0.00	<0.01	<0.01
Red drum	*Sciaenops ocellatus*	0.20	0.00	0.12	0.00	0.00	0.00
Red hake	*Urophycis chuss*	0.00	0.01	<0.01	0.00	0.01	0.00
Rock crab	*Cancer spp.*	0.02	0.24	0.10	0.02	0.13	0.08
Sand dollar	*Echinarachnius parma*	<0.01	<0.01	<0.01	0.00	<0.01	<0.01

Table 2.1 continued.

Taxa- common name	Scientific name	2005 CPUE (kg/km) Southern Sites	Northern Sites	All sites	2006 CPUE (kg/km) Southern Sites	Northern Sites	All sites
Scup	*Stenotomus chrysops*	0.02	0.48	0.21	0.04	0.03	0.03
Sea anemone (unk)	n/a	0.00	0.00	0.00	<0.01	0.00	<0.01
Sea cucumber	*Sclerodactyla* spp.	0.09	0.00	0.06	0.16	0.00	0.06
Sea mouse	*Aphrodita aculeata*	0.00	0.00	0.00	0.00	0.02	0.01
Seahorse	*Hippocampus spp.*	0.00	0.00	0.00	0.00	<0.01	<0.01
Sheepshead	*Archosargus probatocephalus*	0.00	0.00	0.00	0.22	0.00	0.09
Shrimp (unk)	n/a	<0.01	0.00	<0.01	0.00	<0.01	<0.01
Silver hake	*Merluccius bilinearis*	0.00	<0.01	<0.01	0.00	0.01	<0.01
Smallmouth flounder	*Etropus microstomus*	0.00	0.00	0.00	0.00	<0.01	<0.01
Smooth butterfly ray	*Gymnura micrura*	3.04	0.00	1.82	0.19	0.00	0.08
Smooth dogfish	*Mustelus canis*	2.01	1.35	1.75	0.11	0.24	0.18
Southern kingfish	*Menticirrhus americanus*	0.07	0.00	0.04	0.11	0.00	0.05
Southern stingray	*Dasyatis americanus*	18.85	0.00	11.31	1.35	0.00	0.56
Spider crab	*Libinia* spp.	0.59	1.30	0.87	0.31	0.32	0.31
Spiny butterfly ray	*Gymnura altavela*	5.49	0.00	3.30	1.77	0.00	0.73
Spiny dogfish	*Squalus acanthias*	0.00	3.95	1.58	0.67	1.40	1.10
Spot	*Leiostomus xanthurus*	0.13	0.00	0.08	0.02	0.00	0.01
Spotted hake	*Urophycis regia*	0.00	0.00	0.00	0.00	<0.01	<0.01
Striped bass	*Morone saxatilis*	0.33	1.40	0.76	0.33	0.24	0.28
Striped burrfish	*Chilomycterus schoepfii*	0.05	0.00	0.03	0.04	0.00	0.02
Striped searobin	*Prionotus evolans*	0.05	1.22	0.52	0.13	1.98	1.22
Summer flounder	*Paralichthys dentatus*	2.98	0.86	2.13	1.55	4.99	3.57
Surf clam	*Spisula solidissima*	0.01	1.45	0.59	0.03	0.36	0.22
Weakfish	*Cynoscion regalis*	0.05	0.04	0.05	0.07	0.05	0.06
Windowpane flounder	*Scophthalmus aquosus*	0.31	1.35	0.73	1.43	1.54	1.49
Winter flounder	*Pseudopleuronectes americanus*	0.00	0.27	0.11	0.00	0.19	0.11
Witch flounder	*Glyptocephalus cynoglossus*	0.00	0.04	0.01	0.00	0.00	0.00

Table 2.2. Percent occurrence of most commonly caught taxa during the 2005 and 2006 horseshoe crab trawl surveys. Only taxa that were present in at least 10% of tows for at least one survey are listed.

	2005 Percent occurrence			2006 Percent occurrence		
Common name	**Northern Sites**	**Southern Sites**	**All sites**	**Northern Sites**	**Southern Sites**	**All sites**
Sea star (*Asterias* spp.)	54%	27%	33%	76%	29%	51%
Atlantic croaker	4%	51%	29%	3%	33%	19%
Blue crab	18%	38%	27%	16%	31%	24%
Bluefish	25%	4%	11%	11%	2%	6%
Bullnose/Southern eagle ray	0%	47%	25%	0%	11%	6%
Butterfish	4%	9%	6%	16%	7%	11%
Channeled whelk	0%	73%	40%	0%	84%	46%
Clearnose skate	75%	100%	80%	82%	91%	87%
Hermit crab	71%	13%	31%	68%	51%	59%
Horseshoe crab	68%	93%	73%	55%	93%	76%
Jellyfish	0%	0%	0%	18%	7%	12%
Jonah crab	29%	2%	11%	8%	4%	6%
Knobbed whelk	0%	82%	45%	0%	73%	40%
Lady crab	4%	2%	2%	8%	13%	11%
Little/Winter Skate	100%	16%	42%	100%	47%	71%
Long-finned squid	18%	11%	12%	21%	18%	19%
Moon snail	43%	0%	14%	32%	4%	17%
Northern Puffer	18%	7%	10%	5%	2%	4%
Northern searobin	18%	27%	20%	32%	4%	17%
Rock crab	50%	29%	33%	63%	22%	41%
Scup	29%	20%	20%	45%	11%	27%
Smooth butterfly ray	0%	31%	17%	0%	9%	5%
Smooth dogfish	39%	11%	19%	16%	2%	8%
Southern kingfish	0%	24%	13%	0%	16%	8%
Southern stingray	0%	67%	36%	0%	13%	7%
Spider crab	61%	73%	60%	58%	58%	58%
Spiny butterfly ray	0%	22%	12%	0%	7%	4%
Spiny dogfish	39%	0%	13%	8%	16%	12%
Spot	0%	51%	28%	0%	9%	5%
Striped searobin	68%	13%	30%	79%	20%	47%
Summer flounder	57%	78%	61%	71%	47%	58%
Surf clam	46%	4%	18%	47%	4%	24%
Weakfish	7%	20%	13%	5%	13%	10%
Windowpane flounder	86%	51%	57%	97%	44%	69%

Table 2.3. Multiple response permutation procedure (MRPP) tests among changes in species composition at sites due to location, time of tow, and distance from shore. Tests were preformed using rank transformed Bray-Curtis distance measures. A chance-corrected within-group agreement value (*A*) greater that 0.30 indicates ecological significance.

	Fall 2005		**Fall 2006**		**2005 and 2006 combined**	
Abiotic factor	**p-value**	***A***	**p-value**	***A***	**p-value**	***A***
Location (north/south)	< 0.001	0.37	< 0.001	0.29	< 0.001	0.34
Time of tow (day/night)	< 0.001	0.10	0.007	0.03	< 0.001	0.05
Distance from shore (inshore/offshore)	0.001	0.04	0.009	0.03	< 0.001	0.04

Table 2.4. Non-metric multidimensional scaling (NMS) environmental loadings on axes 1 and 2. NMS analyses were conducted for horseshoe crab trawl survey data collected in 2005, 2006, and both years combined.

	Fall 2005		**Fall 2006**		**2005 and 2006 combined**	
Abiotic factor	**NMS axis 1 (R^2)**	**NMS axis 2 (R^2)**	**NMS axis 1 (R^2)**	**NMS axis 2 (R^2)**	**NMS axis 1 (R^2)**	**NMS axis 2 (R^2)**
Bottom temperature	-0.840	0.625	-0.369	0.333	-0.686	-0.294
Bottom salinity	0.741	-0.541	0.219	-0.204	0.554	0.170
Depth	0.457	0.070	0.510	-0.108	0.395	-0.391
Percent gravel	0.256	-0.069	0.277	-0.201	0.270	-0.060
Percent sand	-0.296	0.108	-0.145	0.123	-0.189	0.092
Percent clay	0.233	0.034	-0.045	-0.147	0.057	-0.197
Percent mud	0.135	-0.093	-0.088	0.091	-0.002	-0.019

Table 2.5. List of species caught as bycatch during the horseshoe crab trawl survey. A measure of resilience and the status of the fishery, management, and conservation are listed for each species.

Common name[a]	Occurrence[b] (within present study)	Abundance[c] (within present study)	Resilence[d]	Fishery status worldwide[e]	Management status	IUCN listing[f]
American lobster	Rare	Very Low	Unknown (Blue Ocean Institute 2005)	Fished (C/R) (NEFSC 2007)	Managed (NEFSC 2007)	None
Asteriid sea star	Common	Very Low	No info available	No info available	No info available	None
Atlantic angel shark	Rare	Very Low	Low	Fishery closed in U.S.	Managed (Fishbase 2007)	Data deficient
Atlantic croaker	Common	Low	Medium	Fished (C/R)	Managed; Not overfished/ no overfishing (ASMFC 2007; MD DNR 2007)	None
Atlantic sturgeon	Rare	Very Low	Very Low	Fished (C/R); Fishery closed in U.S.	Managed; Current moratorium (NEFSC 2007)	Near thr.
Black drum	Rare	Very Low	Medium	Fished (C/R)	Managed (MD DNR 2007)	None
Black sea bass	Uncommon	Very Low	Medium	Fished (C/R)	Managed; Not overfished, overfishing status unknown (ASMFC 2007; MDDNR 2007)	None
Blue crab	Common	Very Low	High (Blue Ocean Institute 2005)	Fished (C/R)	Managed (MD DNR 2007)	None
Blue mussel	Rare	Very Low	No info available	Fished (C) (ME DMR 2007)	Managed (ME DMR 2007)	None

Table 2.5 continued.

Common name[a]	Occurrence[b] (within present study)	Abundance[c]	Resilience[d]	Fishery status worldwide[e]	Management status	IUCN listing[f]
Bluefish	Uncommon	Very Low	Medium	Fished (C/R)	Managed; Not overfished/ no overfishing (ASMFC 2007; MD DNR 2007)	None
Bullnose ray	Uncommon	Low	Very Low	Fished (C)	No info available	None
Butterfish	Uncommon	Very Low	High	Fished (C/R)	Managed; Not overfished, overfishing status unknown (NEFSC 2007)	None
Clearnose skate	Abundant	High	Low	Fished (C)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None
Cownose ray	Uncommon	Very Low	Low	Fished (C); No direct fishery in U.S. waters	No info available	Near thr.
Deep-sea scallop	Uncommon	Low	Medium (Blue Ocean Institute 2005)	Fished (C)	Managed; Not overfished, no overfishing (NEFSC 2007)	None
Dusky shark	Rare	Very Low	Very Low	Fished (C); Fishery closed in U.S. waters (NMFS 2007a)	Managed; Listed as "species of concern" (NMFS 2007b)	Near thr.
Hermit crab	Common	Very Low	No info available	No info available	No info available	None

Table 2.5 continued.

Common name[a]	Occurrence[b] (within present study)	Abundance[c] (within present study)	Resilience[d]	Fishery status worldwide[e]	Management status	IUCN listing[f]
Jellyfish (unk)	Uncommon	Very Low	No info available	No info available	No info available	None
Jonah crab	Uncommon	Very Low	Unknown (Monterey Bay Aquarium: Seafood Watch 2007)	Not currently managed (Monterey Bay Aquarium: Seafood Watch 2007)	Not currently managed (Monterey Bay Aquarium: Seafood Watch 2007)	None
Knobbed whelk	Common	Medium	No info available	Fished (C/R) (Monterey Bay Aquarium: Seafood Watch 2007)	Managed (VA MRC 2007)	None
Lady crab	Uncommon	Very Low	No info available	No info available	No info available	None
Little skate	Abundant	High	Low	Fished (C)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None
Long-finned squid	Uncommon	Very Low	High (Blue Ocean Institute 2005)	Fished (C) (NEFSC 2007)	Managed; No overfishing (NEFSC 2007)	None
Northern Puffer	Uncommon	Very Low	High	No info available	Not currently managed	None
Northern searobin	Uncommon	Very Low	Medium	Fished (C)	No info available	None
Northern stargazer	Uncommon	Very Low	Medium	Fished (R)	No info available	None
Red hake	Rare	Very Low	Medium	Fished (C/R)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None

Table 2.5 continued.

Common name[a]	Occurrence[b] (within present study)	Abundance[c] (within present study)	Resilience[d]	Fishery status worldwide[e]	Management status	IUCN listing[f]
Rock crab	Common	Very Low	No info available	No info available	No info available	None
Sand dollar	Uncommon	Very Low	No info available	No info available	No info available	None
Scup	Common	Very Low	Medium	Fished (C/R)	Managed; Not overfished, overfishing status unknown (ASMFC 2007)	None
Silver hake	Rare	Very Low	Medium	Fished (C)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None
Smooth butterfly ray	Uncommon	Low	Very Low	Fished (C); No direct fishery in U.S. waters	No info available	Data deficient
Smooth dogfish	Uncommon	Low	Low	Fished (C/R)	Not currently managed	Near thr.
Southern eagle ray	Uncommon	Low	Very Low	Fished (C)	No info available	None
Southern kingfish	Uncommon	Very Low	Medium	Fished (C/R)	Managed (GA DNR 2007)	None
Southern stingray	Common	Medium	Very Low	Fished (C/R)	No info available	Data deficient
Spider crab	Abundant	Very Low	No info available	No info available	No info available	None
Spiny butterfly ray	Uncommon	Low	Very Low	Fished (C/R)	No info available	None

Table 2.5 continued.

Common name[a]	Occurrence[b] (within present study)	Abundance[c]	Resilience[d]	Fishery status worldwide[e]	Management status	IUCN listing[f]
Spiny dogfish	Uncommon	Low	Very Low	Fished (comm.)	Managed; Not overfished/ no overfishing (NEFSC 2007)	Vul.
Spot	Uncommon	Very Low	High	Fished (C)	Managed; Overfished/ overfishing unknown (ASMFC 2007)	None
Striped bass	Uncommon	Very Low	Low	Fished (C/R)	Managed; Not overfished, overfishing status unknown (MD DNR 2007; NEFSC 2007)	None
Striped burrfish	Uncommon	Very Low	No info available	Fished (R)	No info available	None
Striped searobin	Common	Low	Medium	Fished (C/R)	No info available	None
Summer flounder	Abundant	Medium	Medium	Fished (C/R)	Managed; Not overfished, overfishing is occurring (MD DNR 2007; NEFSC 2007)	None
Surf clam	Common	Very Low	No info available	Fished (C) (NEFSC 2007)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None

Table 2.5 continued.

Common name[a]	Occurrence[b] (within present study)	Abundance[c]	Resilience[d]	Fishery status worldwide[e]	Management status	IUCN listing[f]
Weakfish	Uncommon	Very Low	High	Fished (C)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None
Windowpane flounder	Abundant	Low	Medium	Fished (C)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None
Winter flounder	Uncommon	Very Low	Medium	Fished (C/R)	Managed; Not overfished/ no overfishing (NEFSC 2007)	None
Winter skate	Abundant	High	Low	Fished (C)	Managed; Not overfished, overfishing is occurring (NEFSC 2007)	None

a. Species caught during HCRC trawl survey

b. Percent of tows in which species was present; Abundant: > 50%, common: 21-50%, uncommon: 6-20%, rare: <5% of tows (Data from present study).

c. Percent of total biomass that species comprised; High: >10%, Medium: 2-10%, Low: 1%, Very Low <1% of biomass (Data from present study).

d. Population doubling time; High: <15 months, Medium: 1.4-4.4 years, Low: 4.5-14 years, Very low: >14 years (FishBase 2007 unless noted otherwise).

e. Fishery status worldwide; C: commercial fishery, R: recreational fishery (FishBase 2007 unless noted otherwise).

f. IUCN Listing; Vul: vulnerable, facing high risk of extinction in the wild, Near thr.: near threatened, close to qualifying for threatened category in the future, Data deficient: appropriate data are lacking (IUCN 2006).

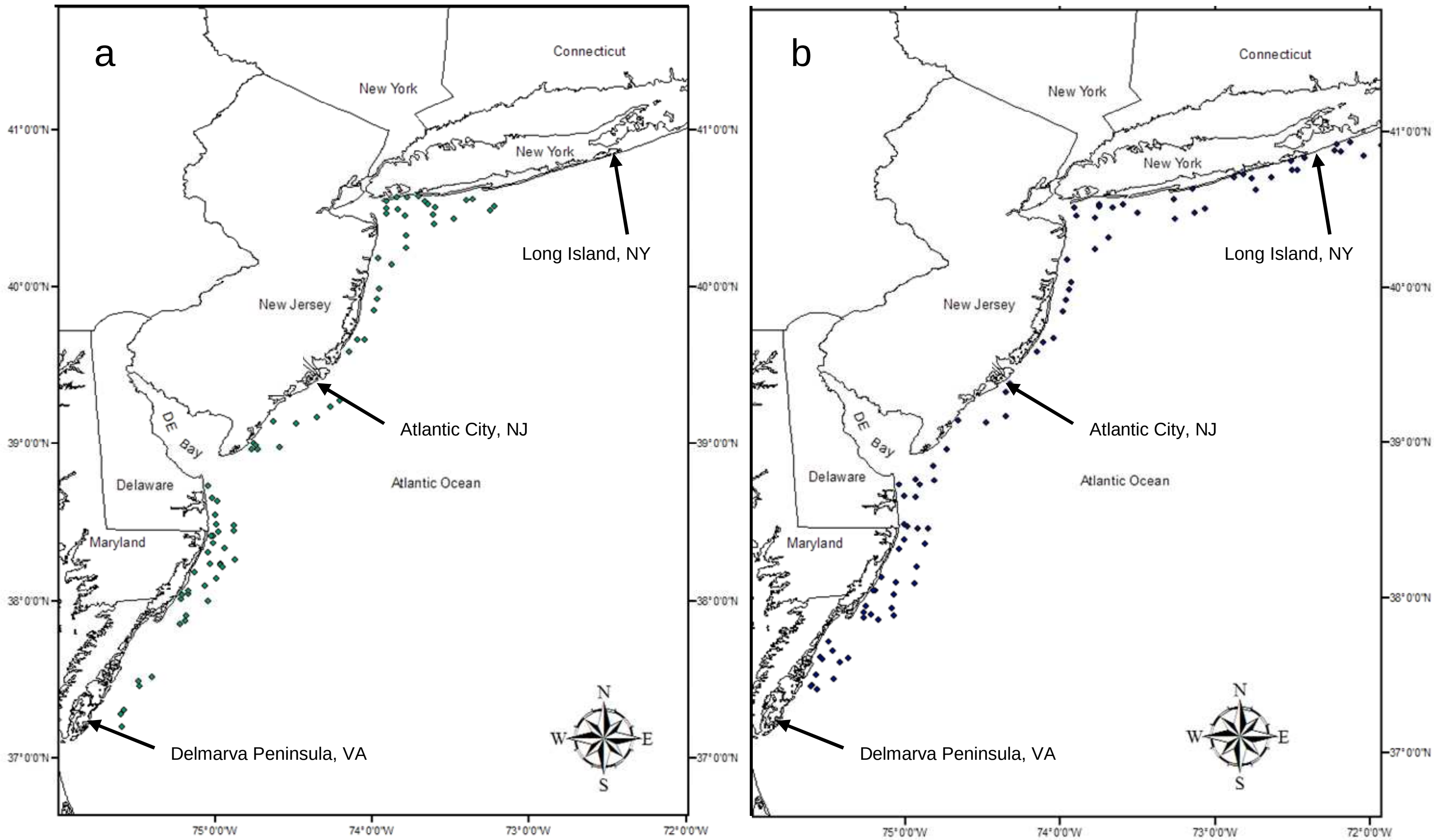
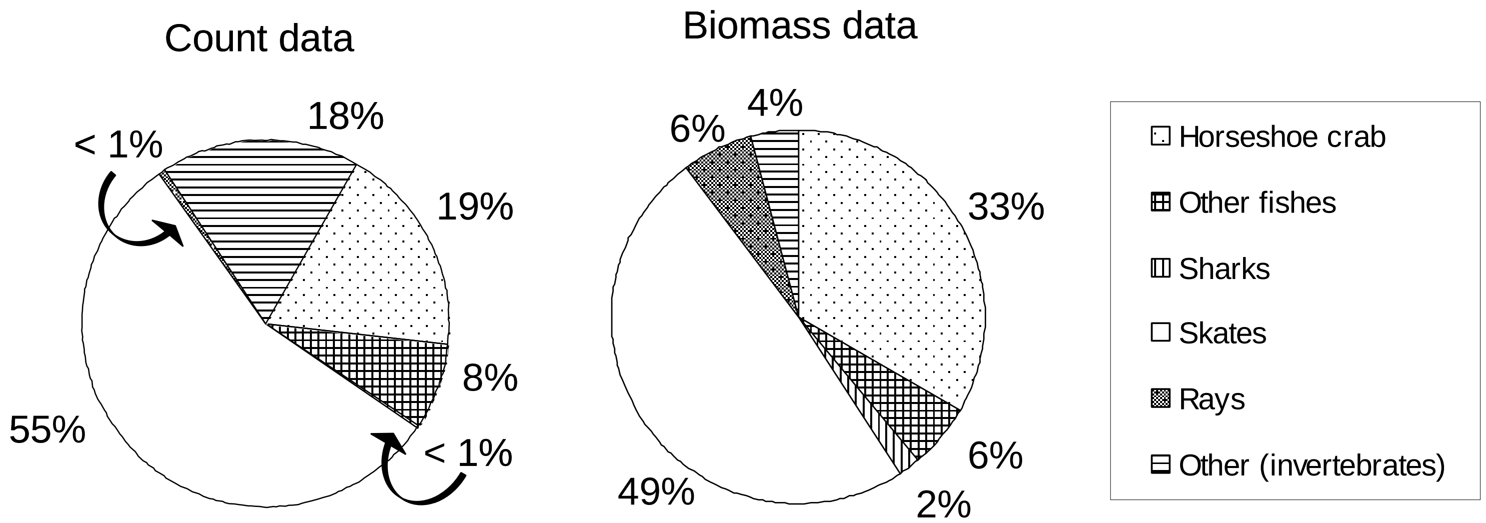


Figure 2.1. Sites sampled during the horseshoe crab trawl survey during the fall season of 2005 (a) and 2006 (b). Northern sites were defined as sites north of Atlantic City, New Jersey and southern sites as sites south of Atlantic City, New Jersey.

Figure 2.2. Percentage of total biomass and count data for major taxa caught during the 2005 and 2006 horseshoe crab trawl surveys. Skates and horseshoe crabs comprised the majority of catch.



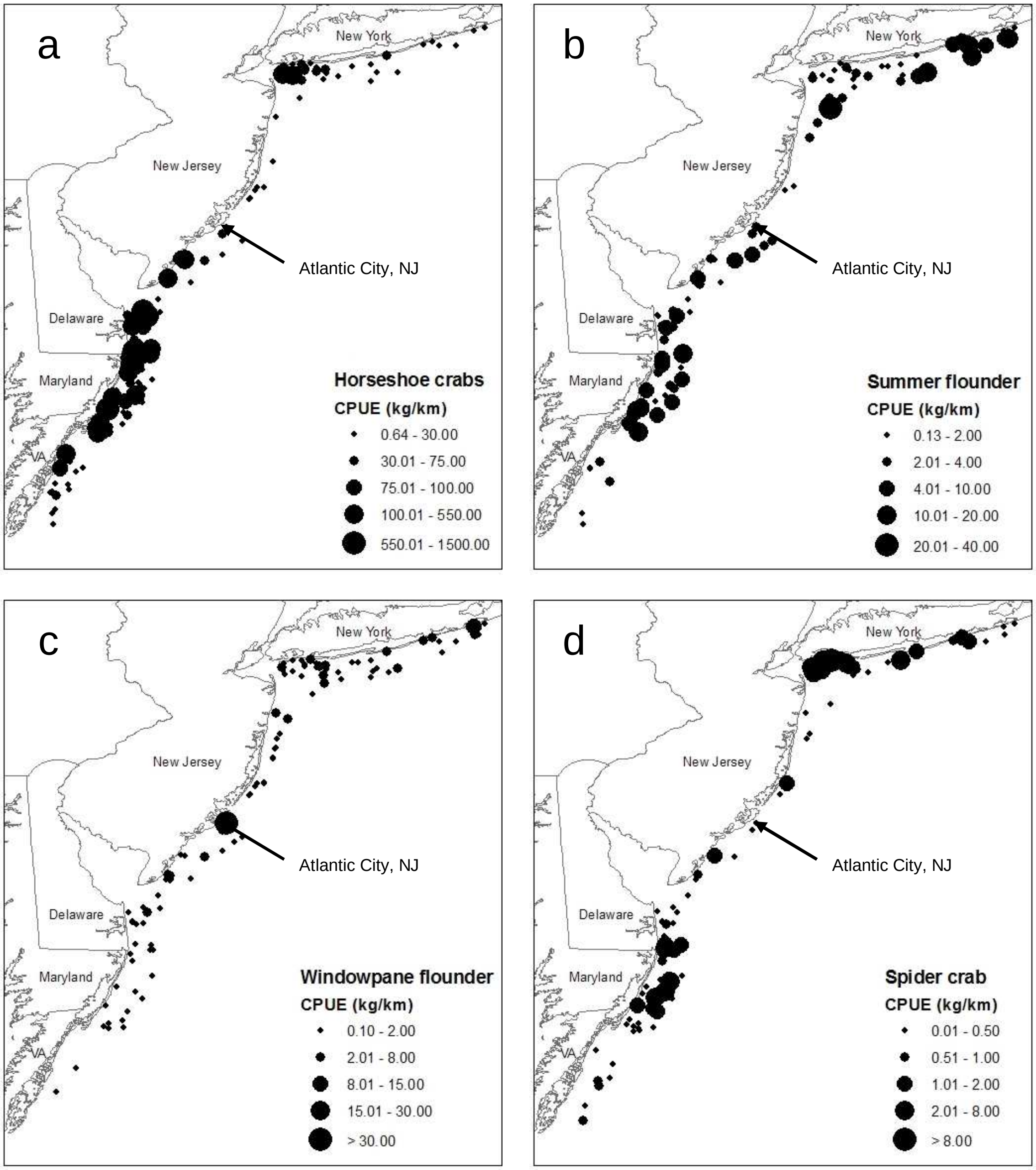


Figure 2.3. Distribution and catch per unit effort (CPUE, kg/km) for species commonly caught during the horseshoe crab trawl survey. Summer flounder (b), windowpane flounder (c), and spider crab (d) were often caught in the same areas as horseshoe crabs (a).

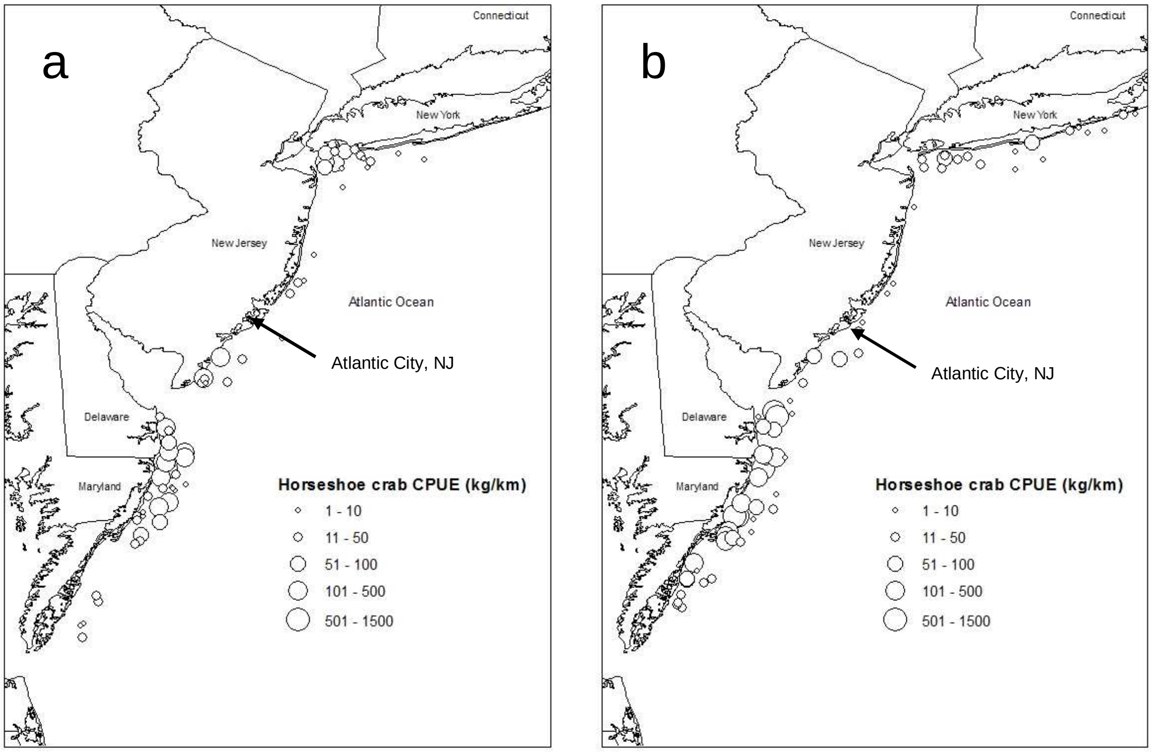


Figure 2.4. Catch per unit effort (CPUE, kg/km) for horseshoe crabs during the 2005 (a) and fall 2006 (b) horseshoe crab trawl surveys. Horseshoe crabs were caught in higher abundance at southern sites during both years.

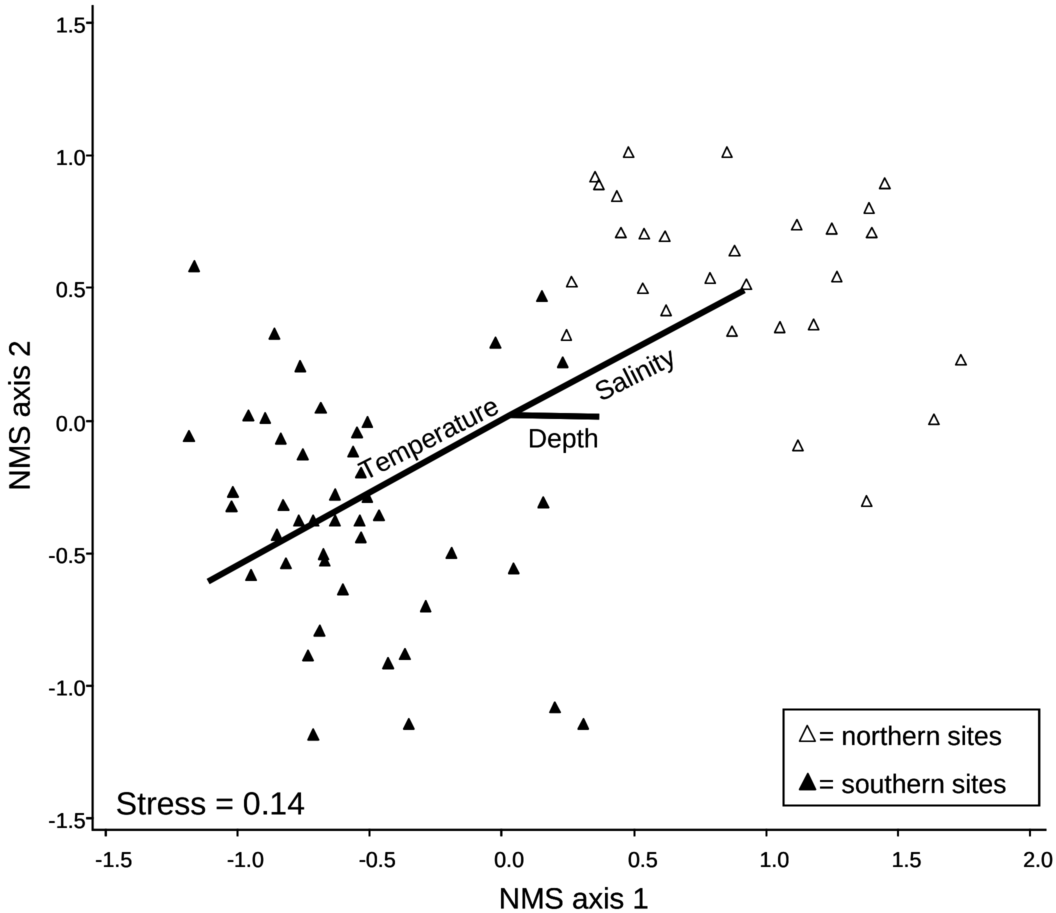


Figure 2.5. Cluster ordination using non-metric multidimensional scaling (NMS) with a Bray-Curtis distance measure for species composition data collected during the 2005 horseshoe crab trawl survey. Sites formed two distinct groups: one group of all southern sites and one group of mostly northern sites. Temperature and salinity explained most of the variation in species composition among sites.

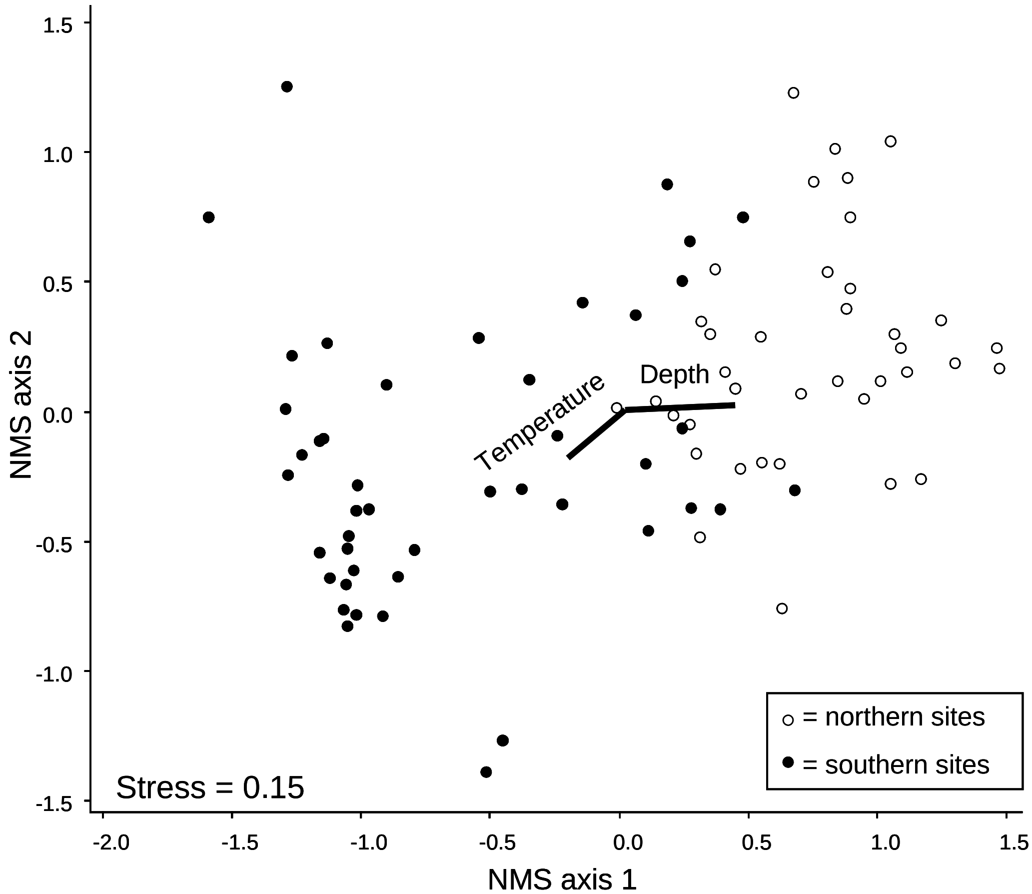


Figure 2.6. Cluster ordination using non-metric multidimensional scaling (NMS) with a Bray-Curtis distance measure for species composition data collected during the 2006 horseshoe crab trawl survey. Sites formed two distinct groups: one group of all southern sites and one group comprised of northern and southern sites. Temperature and depth explained most of the variation in species composition among sites.

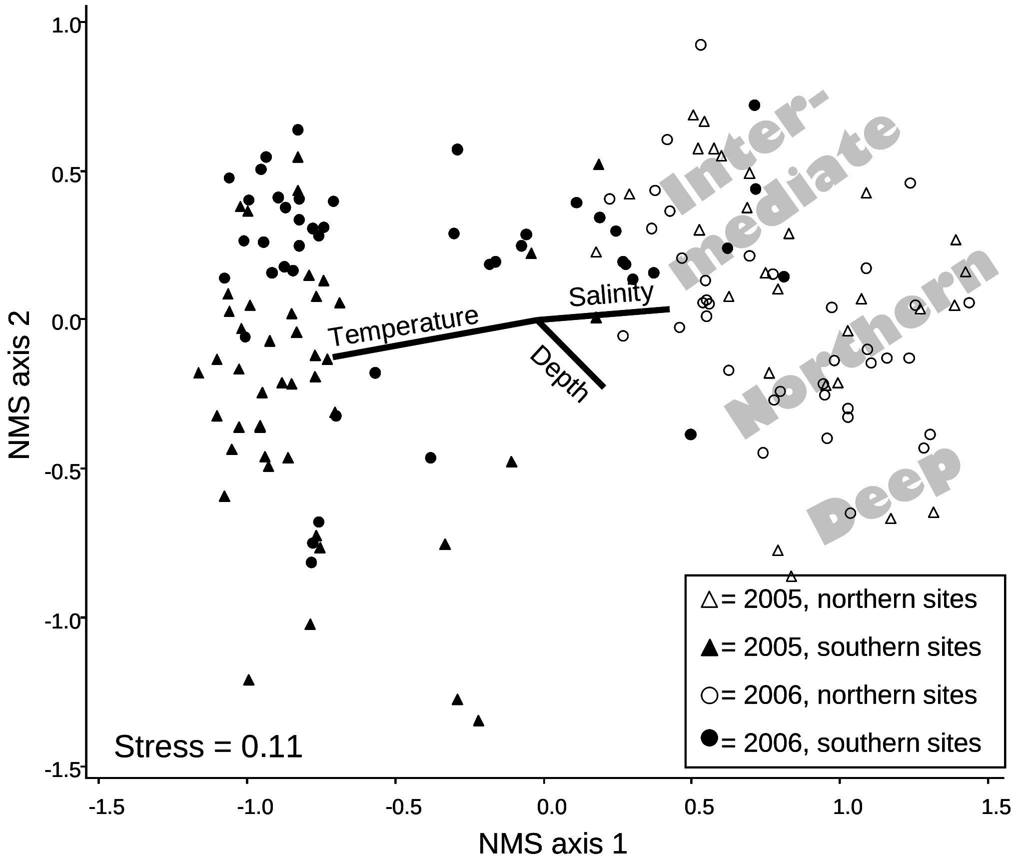


Figure 2.7. Cluster ordination using non-metric multidimensional scaling (NMS) with a Bray-Curtis distance measure for species composition collected at sites during the 2005 and 2006 horseshoe crab trawl survey. Sites formed two distinct groups: one group of all southern sites and one group comprised of mostly northern sites. The group comprised on mostly northern sites contained three sub-groups: one subgroup comprised of northern and southern sites ("Intermediate"), one sub-group of all northern sites ("Northern"), and the last sub-group comprised of northern, deep-water sites ("Deep"). Temperature explained most of the variation in species composition among sites.

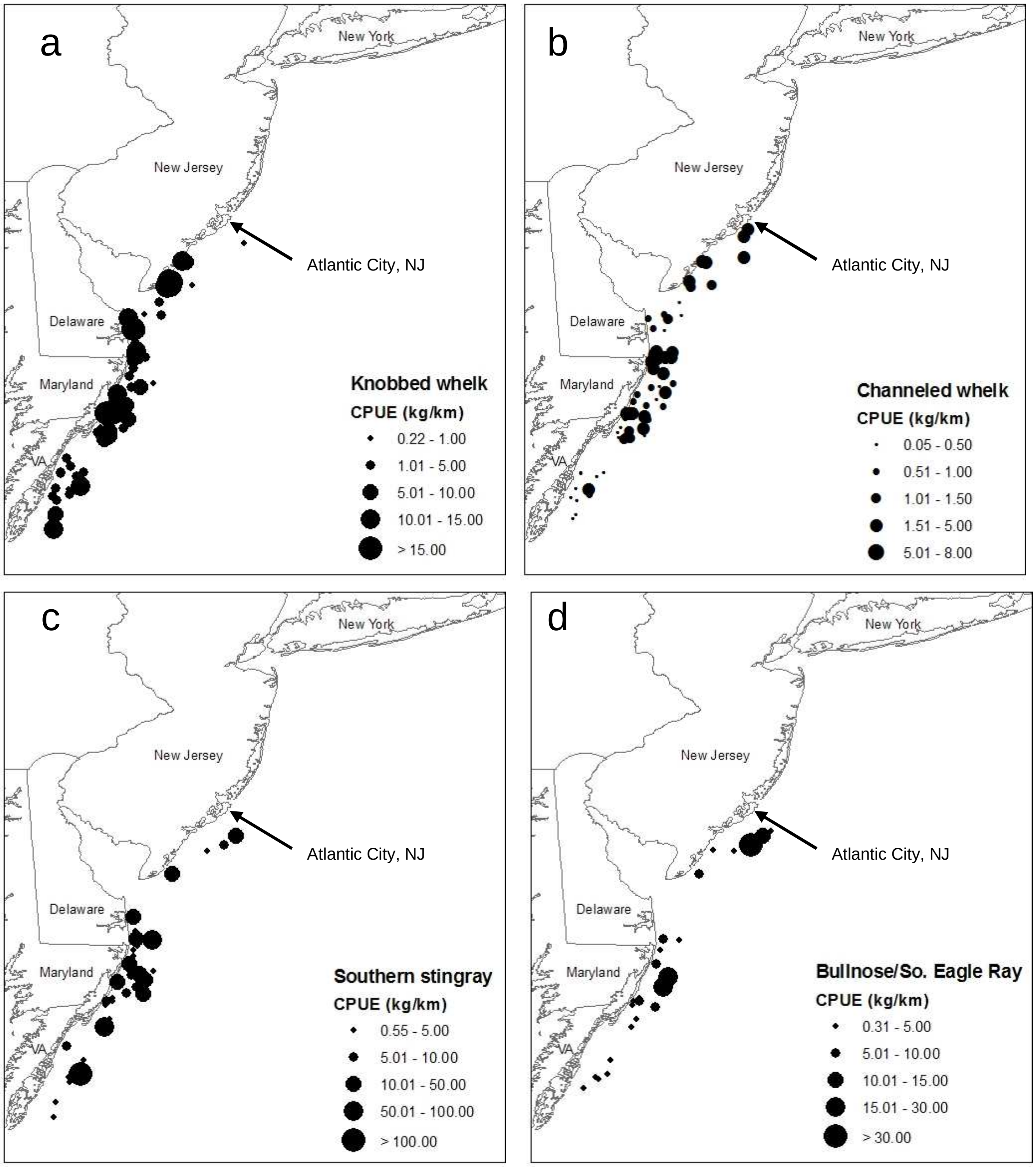


Figure 2.8. Distribution and catch per unit effort (CPUE, kg/km) of species commonly caught at southern sites during the 2005 and 2006 horseshoe crab trawl survey. Maps are shown for knobbed whelk (a), channeled whelk (b), southern stingray (c), bullnose and southern eagle ray (d).

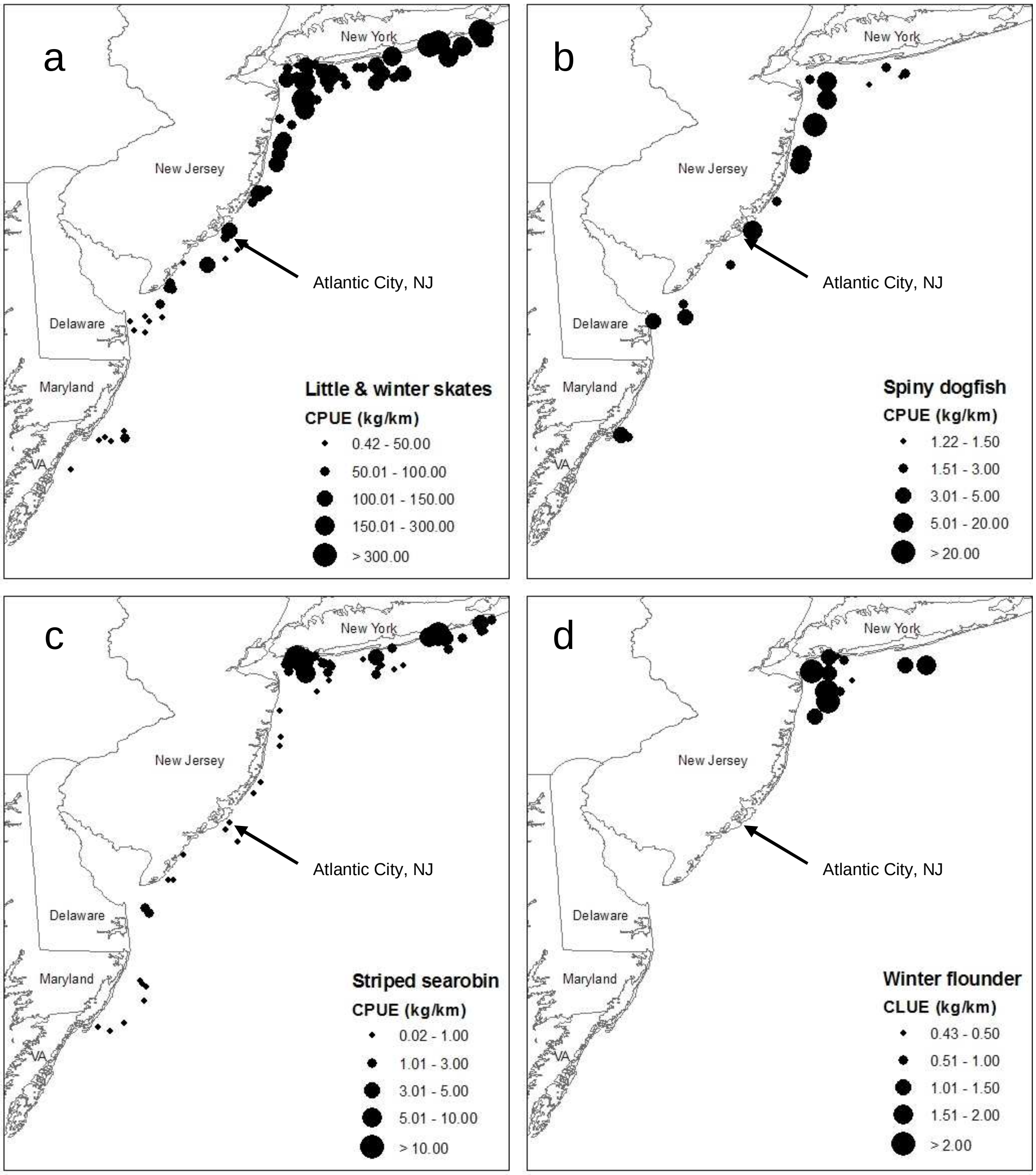


Figure 2.9. Distribution and catch per unit effort (CPUE, kg/km) of species commonly caught at northern sites during the 2005 and 2006 horseshoe crab trawl survey. Maps are shown for little and winter skate (a), spiny dogfish (b), striped searobin (c), winter flounder (d).

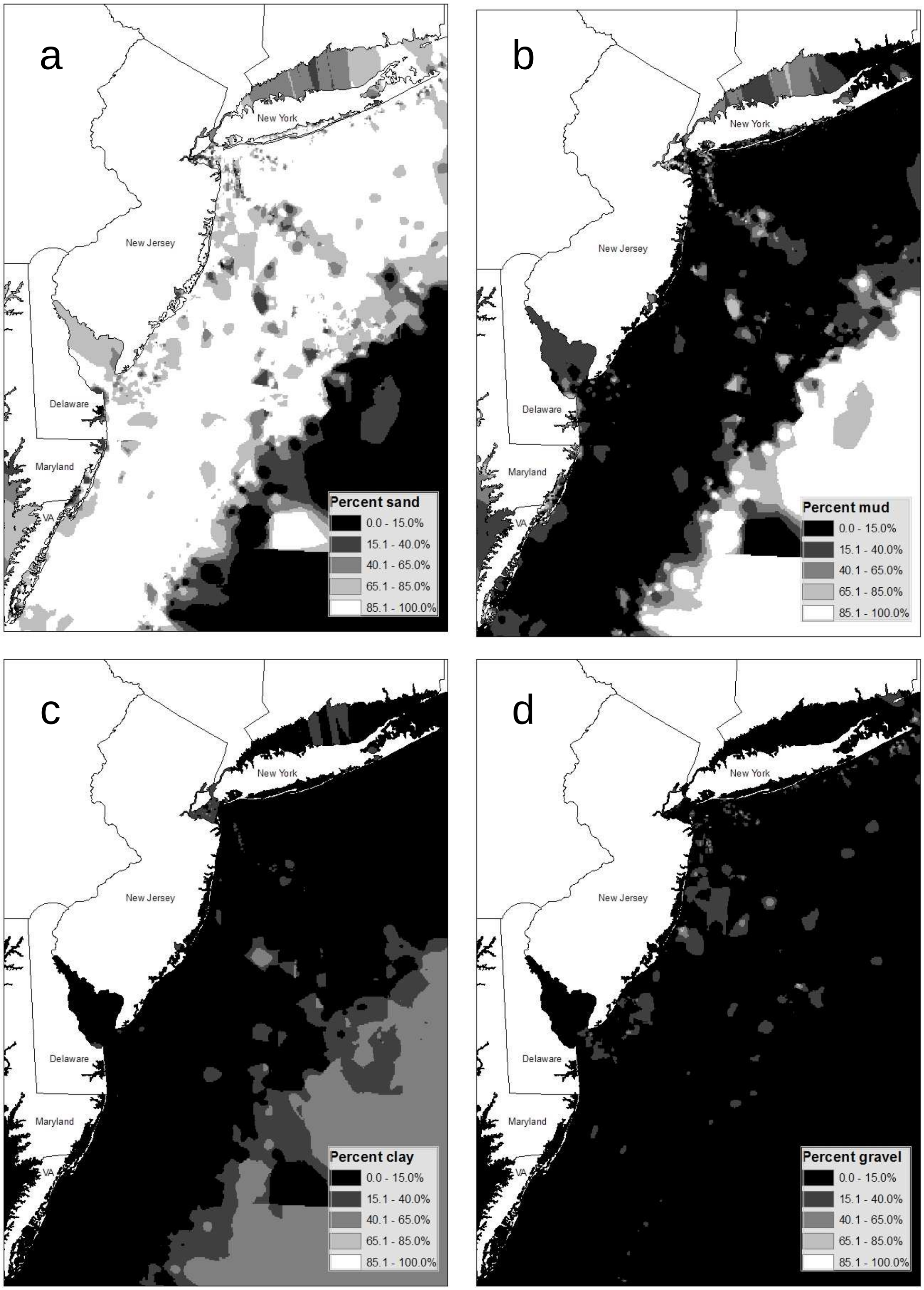


Figure 2.10. Percent sand (a), mud (b), clay (c), and gravel (d) throughout the study area. Bottom composition throughout most of the study area consisted of more than 80% sandy substrate.

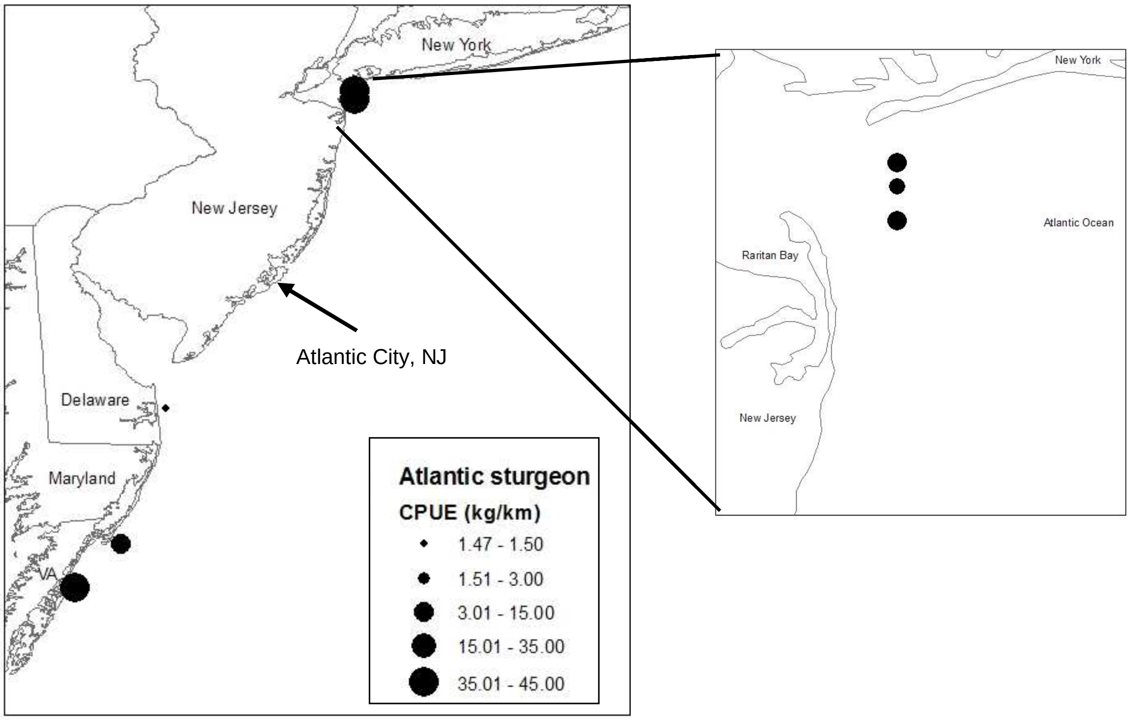


Figure 2.11. Distribution and catch per unit effort (CPUE, kg/km) of Atlantic sturgeon caught during the horseshoe crab trawl survey.

Chapter 3.

Examining current and alternative approaches to estimating biomass of horseshoe crab (*Limulus polyphemus*) landings along the Middle Atlantic Bight.

Abstract

Conversion factors are currently used by states along the Atlantic coast to convert landing data from numbers of horseshoe crabs to biomass (pounds) of crabs. However the accuracy of these conversion factors has not been thoroughly investigated. These conversion factors vary by state, for example Virginia uses one conversion factor to estimate the biomass of harvested horseshoe crabs, whereas New Jersey and Delaware each use two conversion factors, one for males and one for females, to estimate biomass. I designed a study to test the accuracy of the conversion factors used by the states of Maryland, New Jersey, Delaware, and Virginia and to develop and test an alternative approach, using prosomal-width-to-crab-weight equations, to estimate the biomass of landing data. I derived two width-weight equations from individual prosomal width and weight measurements collected in trawlable waters off the coasts of New York, New Jersey, Delaware, Maryland, and Virginia. I tested these two approaches (i.e., using conversion factors versus using width-weight equations) with four independent data sets. The calculated width-weight equation provided a relatively accurate prediction of weight, but not more accurate than the conversion factors that are currently used by New Jersey, Delaware, and Maryland. The width-weight equations and the conversion factors used by New Jersey, Delaware, and Maryland predicted biomass much more accurately than conversion factor that is currently used by the state of Virginia. On average, the Virginia

conversion factor underestimated biomass 45% more than the conversion factors used by New Jersey, Delaware, and Maryland. This study suggests that of the approaches I tested, using two conversion factors, based on sex, is the best method to estimating biomass from landing data reported in numbers of horseshoe crabs.

Introduction

Horseshoe crabs (*Limulus polyphemus*) are considered a multiple-use resource. They are valued by many stakeholders, including commercial fishers, biomedical companies, and environmental interest groups (Berkson and Shuster 1999). Horseshoe crabs are commercially harvested and sold as bait for American eel (*Anguilla rostrata*) and whelk (*Busycotypus* spp.) fisheries (ASMFC 1998). This species is also harvested for biomedical companies because its copper-based blood contains a unique clotting agent (LAL, Limulus Amoebocyte Lysate) that is used to detect pathogenic endotoxins on medical devices and in injectable drugs (Novitsky 1984; Mikkelsen 1988). Horseshoe crabs are ecologically important because their eggs serve as a food source for migrating shorebirds (Shuster and Botton 1985; Mikkelsen 1988).

Despite their importance to many stakeholders, it was not until 1998 that a Fishery Management Plan (FMP) was developed for this species. Current regulations require commercial fishers to report the harvest of horseshoe crabs for bait or biomedical companies to the National Marine Fisheries Service (NMFS) (ASMFC 1998). These landings must include the sex, harvest method, number, and pounds of all horseshoe crabs harvested (ASMFC 1998). However, in many cases, fishers do not report landings in pounds, but only in numbers of crabs landed. Conversion factors are then used by state agencies to convert number of crabs to pounds (ASMFC 1998). Due to the uncertainty in these conversion factors and resulting biomass estimates, managers only use numbers of crabs (not biomass) when assessing the status of the stock (S. Michels, Delaware Division of Fish and Wildlife, personal communication).

The conversion factors used to convert landings from numbers to pounds of horseshoe crabs vary by state and are based on previously collected fishery-independent and fishery-dependent data (J. Brust, New Jersey Department of Fish and Game, personal communication; S. Michels, Delaware Division of Fish and Wildlife, personal communication; S. Doctor, Maryland Department of Natural Resources, personal communication). Some states, such as Virginia, use only one conversion factor (2 lbs/crab; L. Gillingham, Virginia Marine Resource Commission, personal communication) to calculate the weight of each crab harvested. Other states, such as New Jersey, Delaware, and Maryland, use two conversion factors, taking the sex of the horseshoe crab into account (New Jersey/Delaware: 5.12 lbs/female crab, 2.32 lbs/male crab; S. Michels, Delaware Division of Fish and Wildlife, personal communication; Maryland: 4.97 lbs/female crab, 2.12 lbs/male crab; S. Doctor, Maryland Department of Natural Resources, personal communication).

The accuracy of these conversion factors has not been thoroughly investigated. I designed a study to test the accuracy of the conversion factors currently used by the states of New Jersey, Delaware, Maryland, and Virginia and to test an alternative approach to estimating the biomass of landing data (using prosomal-width-to-crab-weight equations).

Methods

Calculating the width-weight equations

To develop prosomal-width-to-crab-weight equations, width and weight measurements were collected from horseshoe crabs along the Atlantic coast from the eastern tip of Long Island, New York (approximately 71^{o}50'W and 41^{o}04'N) to the southern tip of the Delmarva Peninsula, Virginia (approximately 75^{o}55'W and 37^{o}05'N) during September, October, and November of 2005 and 2006, and June of 2006 (Figure 3.1). Trawling gear, specifically a flounder net, was used to collect all crabs. The ground gear was modified with a Texas sweep, which has a chain line instead of a footrope (Hata and Berkson 2003; Hata and Berkson 2004). I recorded prosomal width (to the nearest 1 mm), weight (to the nearest 0.01 kg), sex, and maturity for all or a subsample of horseshoe crabs at each site. Maturity was classified into three groups: immature, newly mature (primiparous), and mature (multiparous). Maturity stages in females were not always morphologically distinct, so some individuals had to be probed with an awl to look for eggs and determine the stage of maturity. Females with eggs but no rub marks, which are indicative of amplexus, were categorized as newly mature and those with eggs and rub marks were categorized as mature (D. Hata, Virginia Tech, personal communication). During the fall seasons of 2005 and 2006, I collected measurements during the horseshoe crab trawl survey conducted by the Horseshoe Crab Research Center (HCRC) at Virginia Tech (for methods see Hata and Berkson 2004). All horseshoe crabs were collected within 12 nautical miles off the states of New York, New Jersey, Delaware, Maryland, and Virginia. In the summer of 2006, I sampled horseshoe crabs aboard a commercial trawl vessel harvesting crabs for a biomedical company off the coast of Ocean City, Maryland.

I normalized the width and weight measurements using a log transformation. Width-weight regression equations were calculated to estimate the parameters of conversion between width and weight, using the form

$$\log_e(WT) = \log_e(PW) \cdot b + \log_e(a),$$

where WT = weight of a horseshoe crab (kg), PW = prosomal width (mm), b = slope, and a = y-intercept (EXCEL, 2003). I tested the slopes of the linear regressions using a two sample t-test.

Choosing the conversion factors

The conversion factors currently used by each state to estimate the biomass of landing data were reported in the horseshoe crab fishery management plan (ASMFC 1998) and verified by researchers from state agencies. I tested the conversion factors used by New Jersey, Delaware, Maryland, and Virginia; the conversion factor used by Virginia is independent of sex, whereas New Jersey, Delaware, and Maryland use separate conversion factors for males and females. I only wanted to test the conversion factor(s) used by the Atlantic states between New York and Virginia to stay within the same geographic range of the data collected to calculate the width-weight equations. The state of New York does not use conversion factors; instead, all management actions are based on number of horseshoe crabs (R. Burgess, New York Department of Environmental Conservation, personal communication).

Obtaining the independent data sets

I used independent data sets containing individual width and weight measurements of horseshoe crabs to test and compare the predictive accuracy of the state conversion factors and the calculated width-weight equations. I obtained these data sets from researchers from universities and state agencies (M. Botton, Fordam University, personal communication; S. Michels, Delaware Division of Fish and Wildlife, personal communication; L. Gillingham, Virginia Marine Resource Commission, personal communication). Four sets of data were used: three from spawning surveys, each along a different beach (i.e., the Delaware side of the Delaware Bay, the New Jersey side of the Delaware Bay, and the Raritan Bay in New Jersey), and one from commercial harvests from Chincoteague, Virginia (i.e., conch dredge and hand harvest).

Estimating total biomass

I estimated the total biomass of each data set using the conversion factors from New Jersey, Delaware, Maryland, and Virginia. I also used the appropriate width-weight equation (i.e., based on the specific state, sex, and maturity stage) to estimate the total biomass of each data set by summing the estimated weight of each horseshoe crab within the data set. Lastly, to determine which method provided the most accurate estimate of biomass, I compared the two methods used to estimate weight to the total observed biomass for each data set (i.e., conversion factors and calculated width-weight equations).

Results

Calculated width-weight equations: estimates of biomass

In total, 925 horseshoe crabs were sampled throughout the study area (Figures 3.1 and 3.2, Table 3.1). The slopes of the relationship between prosomal width and horseshoe crab weight were significantly different between males and females (t = 177.43; $P \leq 0.05$). I therefore developed two prosomal-width-to-weight equations for testing: one for female horseshoe crabs and one for male horseshoe crabs (Table 3.2). Maturity stage was not incorporated into these equations because it is not feasible for fishers to report the maturity stage of every horseshoe crab harvested.

These calculated with-weight equations slightly overestimated or underestimated the weights of horseshoe crabs from the spawning survey data sets by 3 - 12% (Figure 3.3; Table 3.3). The calculated width-weight equations overestimated the weights of crabs collected from the Virginia conch dredge fishery by 23 - 29% (Figure 3.1).

Conversion factors: estimates of biomass

The conversion factors currently used by New Jersey, Delaware, and Maryland provided a more accurate estimate of biomass than the conversion factor used by Virginia. The New Jersey, Delaware, and Maryland conversion factors overestimated or underestimated the total observed biomass for the spawning survey data sets by 1 - 15% (Table 3.3). These conversion factors overestimated the biomass of the Virginia commercial harvest data set by 2.7 - 18% (Table 3.3). The conversion factor used by the state of Virginia consistently underestimated the biomass of horseshoe crabs from all data sets, in some cases by more than 50% (Table 3.3). The conversion factor used by Virginia provided a more accurate estimate of the biomass of male horseshoe crabs, only underestimating biomass by 3 - 19% (Table 3.3). However, the biomass of female horseshoe crabs was not accurately estimated, being underestimated by 53 - 66% (Table 3.3).

Discussion

Calculated width-weight equations: estimates of biomass

In most cases, the width-weight equations slightly underestimated biomass of horseshoe crabs collected during Delaware and New Jersey spawning surveys (Table 3.3; Figure 3.3). This may be due to the time of year that horseshoe crabs were sampled. The measurements used to create the width-weight equations were only taken during summer (June) and fall (September, October, and November) months and therefore might not accurately predict the weight of horseshoe crabs that are sampled during the rest of the year. Measurements from the spawning surveys were collected during spring months, when female horseshoe crabs may have been heavier due to additional weight from eggs. Some of the measurements collected during the spawning survey may have been recorded after a female had laid her eggs. However, these horseshoe crabs would still have many eggs in their prosoma because female horseshoe crabs re-nest several times during spawning events (Brockmann and Penn 1992), laying clusters of a few thousand eggs each time (Shuster and Botton 1985). Males will also revisit beaches several times during spawning season (Brockmann and Penn 1992) and may be heavier due to the weight of sperm during this time.

The width-weight equations did not predict the weight of crabs from commercial harvest in Virginia as accurately as they predicted weights from the New Jersey and

Delaware spawning surveys. I expected the calculated width-weight equations to accurately estimate the weight of the horseshoe crabs sampled from Virginia commercial harvests because all of these measurements were collected in trawlable waters. Instead, the horseshoe crabs in the commercial data set generally weighed less than crabs used to calculate the width-weight equations. Horseshoe crabs from Virginia may be morphologically distinct from other populations within our study area; however this is probably not the case. Past research has shown that populations along the Middle Atlantic Bight are very similar in morphology (Shuster 1979; Riska 1981). One possible explanation of the width and weight differences between horseshoe crabs from Virginia compared to other areas is that the measurements collected from Virginia commercial fishery were taken after spawning season, when the crabs had left the beaches and returned to deeper waters. Body weight may have decreased due to the loss of eggs and sperm. The majority of horseshoe crabs that were measured to create the width-weight equations were sampled in the fall season, a few months after spawning season. These horseshoe crabs may have weighed more due to new batches of eggs and sperm being produced for the next spawning season. Measuring error of the observed weights of the horseshoe crabs from the commercial data may have also been a factor.

Conversion factors: estimates of biomass

The conversion factors currently used by New Jersey, Delaware, and Maryland were more accurate at estimating the biomass of horseshoe crabs than the conversion factor used by Virginia. Using two conversion factors- one for females and one for males, provided a more accurate estimate of biomass than using one conversion factor that was independent of sex. Using two conversion factors that are based on sex is more appropriate because, on average, mature female horseshoe crabs have wider prosomal width and associated weight than male horseshoe crabs (Yanasaki 1988; Figure 3.2). A mature female horseshoe crab also has additional weight due to the many eggs (88,000 eggs/mature female; Mikkelsen 1988) within her prosoma. These factors increase the weight of a female horseshoe crab, generally making it much heavier than a male horseshoe crab.

According to the data collected to calculate the width-weight equations, significant differences exist in the width-weight relationship of mature female horseshoe crabs compared to mature male and all immature horseshoe crabs. The larger prosomal width and greater weight of mature female horseshoe crabs were most likely the major factors contributing to the difference in the width-weight relationship. The results from this study suggest that the average weight per mature female horseshoe crab is much greater than for mature male horseshoe crabs. Therefore, it is appropriate that different conversion factors should be used when estimating the weight of female and male horseshoe crabs.

Management implications

Before this study, there had not been an assessment of the accuracy of the conversion factors that are currently used by managers to calculate the total biomass of horseshoe crabs landed. Although landings reported as pounds currently are not used to manage stocks, it may be important to provide accurate biomass estimates of harvest data for future management purposes. Therefore, an accurate conversion method should be developed.

At best, width-weight equations provide biomass estimates that are not notably more accurate than properly used conversion factors. The width-weight equations calculated in this study may be helpful for researchers who want to estimate the weight of an individual horseshoe crab based on a measured prosomal width, but are probably not feasible for management purposes. To calculate the weight of a crab using the width-weight equations, the prosomal width must be measured. Currently, fishers are not required to report the prosomal width of each harvested horseshoe crab. This would be extremely time-consuming for fishers, especially those that are harvesting hundreds of horseshoe crabs in a matter of hours. An observer program could be implemented to measure prosomal widths for a subsample of commercial catch and the measurements could be converted to biomass using the width-weight equations. This might provide managers with an estimate of harvested biomass; however developing accurate conversion factors may be a more efficient approach to estimating total biomass of landing data based on numbers of horseshoe crabs harvested.

It is clear that some conversion factors used by state agencies are better than others in their accuracy of calculating biomass. The conversion factor used by Virginia assumes that the average weight of a sample of horseshoe crabs is two pounds. However, the conversion factor used by Virginia consistently underestimated the total biomass of a sample, especially of female horseshoe crabs. In some cases biomass was underestimated by more than 50%, suggesting that an estimate of two pounds for each female horseshoe crab is far too low (Table 3.3).

Currently there are no size restrictions on the commercial fishery, thus allowing fishers to harvest horseshoe crabs of all sizes. However, biomedical companies require harvests of adult horseshoe crabs (> 8” prosomal width) because they contain more blood than smaller horseshoe crabs (Hurton 2003; ASMFC 2006b). These catches, comprised of larger, heavier horseshoe crabs, that are harvested for biomedical companies would most likely be underestimated using the Virginia conversion factor.

The conversion factors currently used by New Jersey, Delaware, and Maryland seem to provide accurate estimates of horseshoe crab biomass. Currently, all horseshoe crab landings along the Atlantic coast are required to be reported by sex (ASMFC 1998). Since these data are already being collected, it seems only reasonable that two conversion factors, based on sex, be used to convert landing data from number of horseshoe crabs to biomass of horseshoe crabs. Using two conversion factors is also more feasible for management purposes because, currently, many Atlantic coast states are regulating the harvest of horseshoe crabs based on sex. New Jersey and Delaware currently only allow male horseshoe crabs to be harvested (ASMFC 2006b). Maryland allows both male and female horseshoe crabs to be harvested, but enforces a sex ratio of 2:1 (males:females) (ASMFC 2006b). Using two conversion factors, based on sex, should provide an accurate estimate of biomass despite the differing sex ratios among states.

Conclusion

Based on this study, an accurate estimate of biomass can be obtained using a conversion method that incorporates sex and prosomal width of each individual horseshoe crab. However, this method may not be feasible for management. This method would be very difficult to implement and did not provide an estimate of biomass that was more accurate than the conversion factors that are currently used by New Jersey, Delaware, and Maryland. It seems that the most practical approach to estimating biomass from data reported by number of horseshoe crabs landed is to use two conversion factors: one for females and one for males to account for the variation of weight between the sexes. However, I also suggest that researchers from these states should continuously collect data on the average weight of female and male horseshoe crabs in order to fine tune these conversion factors. In this way, the accuracy of these conversion factors could be improved, thereby providing better data for future management assessments.

Table 3.1. Prosomal width and horseshoe crab weight characteristics for female, male, and combined sexes of horseshoe crabs.

		Width characteristics (mm)				Weight characteristics (kg)			
	n	Mean	SE	Min	Max	Mean	SE	Min	Max
Female	482	227	± 2.36	82	321	1.80	± 0.05	0.10	3.94
Male	443	194	± 1.43	80	292	0.87	± 0.01	0.10	3.14
Both	925	211	± 1.51	80	321	1.36	± 0.03	0.10	3.94

Table 3.2. Parameters of the relationship ($\log_e(WT) = \log_e(PW) \cdot b + \log_e(a)$) between prosomal width and weight for female, male, and combined sexes of horseshoe crabs.

	n	a	b	SE(a)	SE(b)	R^2
Female	482	3.0246	-15.974	± 0.024	± 0.130	0.971
Male	443	2.6378	-14.090	± 0.041	± 0.215	0.904
Both	925	3.0134	-15.985	± 0.024	± 0.126	0.947

Table 3.3. Two methods of estimating biomass were tested (width-weight equations calculated from measurements taken during a horseshoe crab trawl survey and conversion factors currently used by various state agencies) using independent data sets that contained width and weight measurements from individual horseshoe crabs. The percentage that each method over- or under-estimated biomass of female (F), male (M), and all (T) horseshoe crabs is listed in the table below.

	Total observed weight (kg)			Percent total weight (kg) was overestimated (+) or underestimated (-) using:											
				Width-weight equations			NJ/DE conversion factors[a]			MD conversion factors[b]			VA conversion factor[c]		
Data set	F	M	T	F	M	T	F	M	T	F	M	T	F	M	T
Spawning survey Delaware Bay, New Jersey (*n* =379)	446	237	683	-7%	-12%	-9%	-13%	-6%	-10%	-15%	-15%	-15%	-66%	-19%	-50%
Spawning survey Raritan Bay, New Jersey (*n* = 303)	231	192	423	-3%	-6%	-4%	+2%	+7%	+4%	-1%	-2%	-1%	-60%	-8%	-36%
Spawning survey Delaware Bay, Delaware (*n* = 348)	631	90	721	5%	-5%	+4%	-4%	+1%	-3%	-7%	-7%	-7%	-63%	-13%	-56%
Conch fishery Atlantic Ocean, Virginia (*n = 252)*	378	55	433	29%	+24%	+29%	+18%	+13%	+18%	+15%	+3%	+13%	-54%	-3%	-47%

Conversion factor(s) used by each state: a. Male = 2.32 lbs/crab, Female = 5.12 lbs/crab (NMFS 1998); b. 2.0 lbs/crab (L. Gillingham, Virginia Marine Resource Commission, personal communication); c. Male = 2.12 lbs/crab, Female = 4.97 lbs/crab (S. Doctor, Maryland Department of Natural Resources, personal communication)

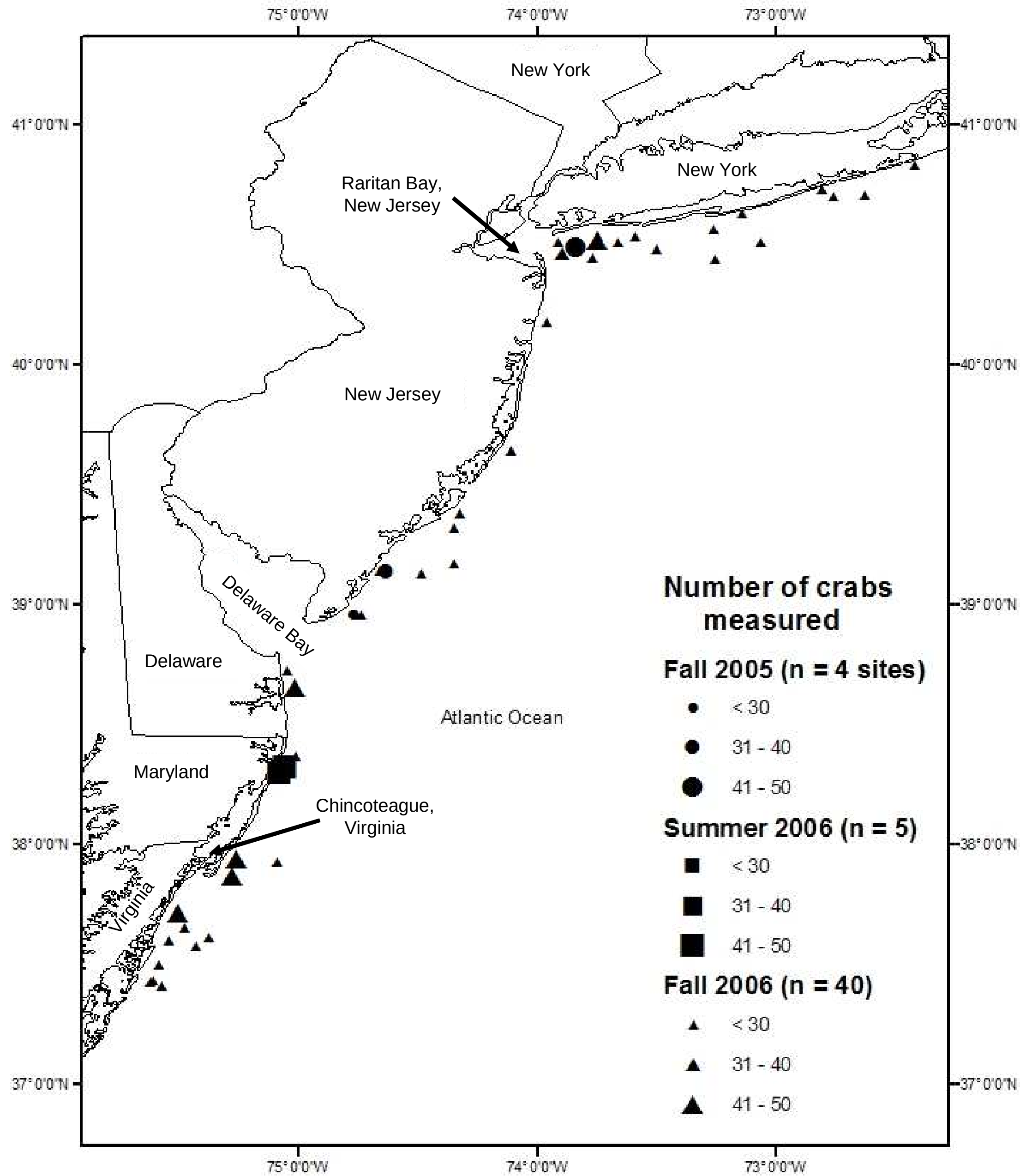


Figure 3.1. Sampling location and number of horseshoe crabs measured throughout the study area. Prosomal width and horseshoe crab weight were collected for 925 horseshoe crabs along the Atlantic coast of the United States.

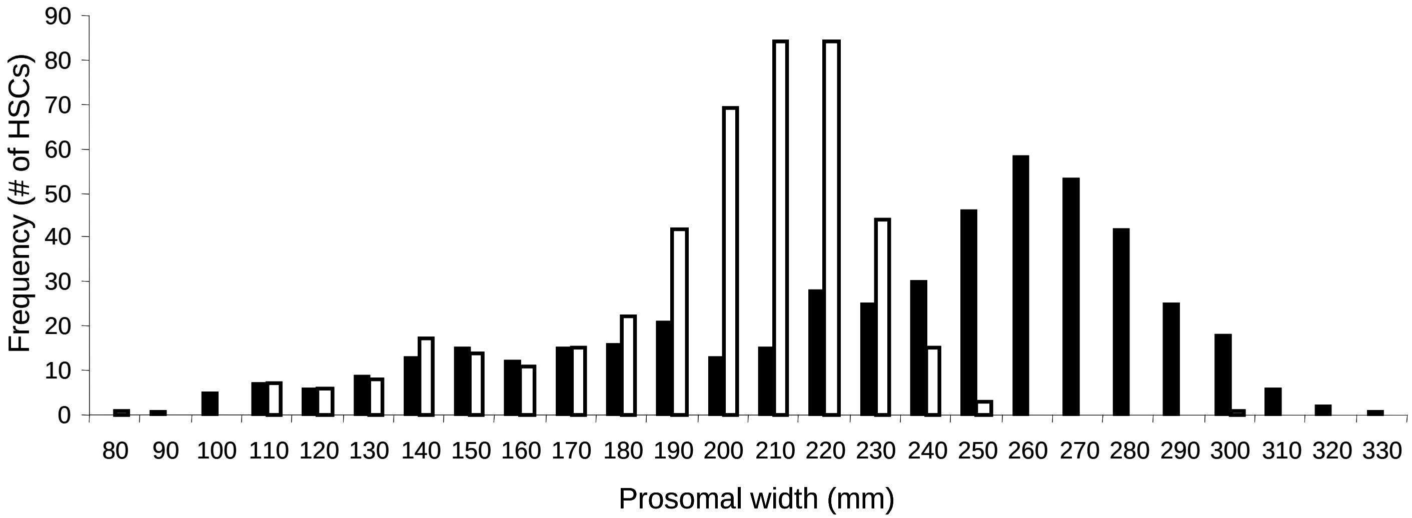


Figure 3.2. The prosomal-width-to-frequency distribution of female (solid bars) and male (open bars) horseshoe crabs collected along the Atlantic Coast.

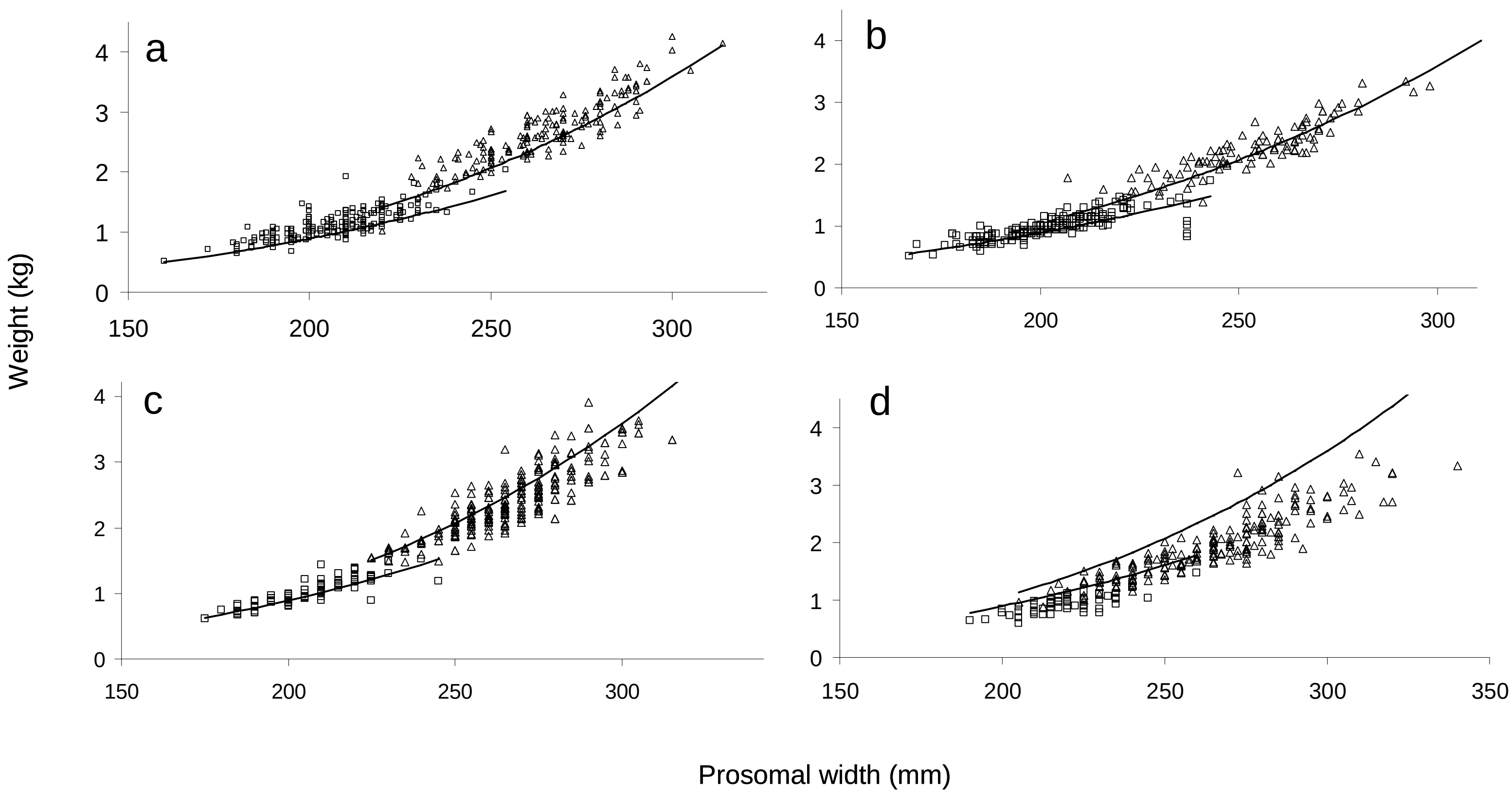


Figure 3.3. Observed and estimated horseshoe crab prosomal widths and weights. Observed measurements were collected from spawning surveys on the Delaware Bay, New Jersey (a); Raritan Bay, New Jersey (b); and Delaware Bay, Delaware (c); and from commercial harvest (hand and dredge) off the coast of Virginia (d). The observed measurements of male horseshoe crabs are represented by open squares and female horseshoe crabs are represented by open triangles. Two width-weight equations were derived using measurements collected from horseshoe crabs along the Atlantic coast. Weight was estimated based on sex and prosomal width of each individual crab, and is represented by the solid line.

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Vita

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