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LAND USE PRACTICE AND WATER QUALITY, BLUE CRAB POPULATION DYNAMICS AND FISH BIODIVERSITY IN BLACKBIRD CREEK, DELAWARE

by

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A THESIS

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DEDICATION

This work is dedicated to my wife, Tina. Thank you for all of your encouragement and understanding throughout this project.

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ABSTRACT

Blackbird Creek is a small watershed in north central Delaware that is comprised of about 36% agricultural land cover. In order to continue ongoing research and to expand existing knowledge of water chemistry and biotic responses to land use practices, two sets of data were collected in 2014 and 2015 from the aquatic ecosystem of the creek. The parameters studied included inorganic water quality variables (nitrogen, phosphorous, alkalinity, turbidity, temperature, salinity, pH, dissolved oxygen), blue crab population dynamics (size, gender, life history stage, molting stage), and fish biodiversity (Species Richness, Shannon-Weiner Index, Simpson Inverse, Margalef's Index), and basic fish trophic dynamics. Each of these variables were related to adjacent land use practices under the hypothesis that such practices could be impacting the aquatic biota in a bottom-up model. It is proposed that application of nutrient-heavy fertilizer and manure onto crop land in the watershed could run off into the waterway at specific times and locations as they relate to agricultural land use, which would increase the nutrient load in the system. The possible increased concentrations of nitrogen and phosphorous, particularly, could

increase phytoplankton production to the point of eutrophication, which could draw down the dissolved oxygen concentration via excessive decomposition following an algal bloom and, thus, force higher level biota to change their behavior and location within the waterway – possibly being forced to remove themselves from the watershed entirely. The data were run through generalized linear models in order to capture any significant differences across years, seasons, and percent agricultural land use. The data collected suggested that there were no water quality differences between agricultural land cover and forested wetland. There were also no significant differences in upper trophic level dynamics. There were seasonal changes, but it is unlikely that these differences are a function of adjacent land use, but rather the natural seasonal fluctuations of the water chemistry variables and life history stages of blue crabs and fish species. Based on the data collected and the analysis of variance, agriculture throughout the Blackbird Creek watershed does not appear to be influencing in-stream inorganic nutrient dynamics or upper level biota. However, there are some opportunities for improvement in future research. Additional parameters that could be included are organic and particulate nutrients (i.e., dissolved organic nitrogen, total particulate phosphorous, dissolved organic carbon, etc.), in addition to the continued long-term monitoring of the system as the climate continues to change and human populations continue to put pressure on coastal ecosystems.

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COMMON ABBREVIATIONS

CPUE	Catch per Unit Effort
GLM	Generalized Linear Model
NH_3	Ammonia
NO_3	Nitrate
NO_2	Nitrite
PO_4^{3-}	Orthophosphate
DO	Dissolved Oxygen
NTU	National Turbidity Unit
N	Total Abundance
n	Sample Abundance
R	Species Richness
M	Margelef's Diversity Index
H	Shannon-Weiner Diversity Index
GIS	Geographic Information Systems
sd	Standard Deviation
df	Degrees of Freedom
USEPA	United State Environmental Protection Agency

CHAPTER 1

GENERAL INTRODUCTION

Water resources throughout the Mid-Atlantic coastline of the United States are facing constant and increasing threats from growing human population (Postel *et al.*, 1996, Pimentel *et al.*, 2004). Water resource management needs to be a priority as land use is modified to accommodate this population increase. The change from a native ecosystem to an agricultural landscape, for example, has far-reaching implications, including increased soil erosion, fertilizer runoff, removal of critical wildlife habitat, and loss of ecosystem services which help mitigate the effects of climate change (i.e., carbon sequestration) (Schempf and Cox, 2007). If, for example, forested wetlands are drained and used instead for agricultural practices, the wetland habitat available for use by aquatic biota is reduced due to altered hydrology, affecting the quality of the water directly (Forney *et al.*, 2001). Marsh grasses and forested buffers have been shown to decrease nutrient and pollutant runoff into waterways (Castelle *et al.*, 1994; Spruill, 2000), and removal of these may allow for increased runoff (Bingham *et al.*, 1980).

The Blackbird Creek watershed in New Castle County, Delaware is a model catchment to study the utility of natural riparian buffers on water quality. The basin drains about 80.29 square kilometers of north-central Delaware into the Delaware Bay. The main waterway is about 44.10 km long with an additional 2.57 km of tributaries (DNREC, 2006). Roughly 36% of the watershed is designated as use for agriculture and the main waterway is about 44.10 km long with an additional 2.57 km of tributaries (DNREC, 2006). Roughly 36% of the watershed is designated as use for agriculture and an additional 13% as urban use. There are no known point sources of pollution throughout the watershed (DNREC, 2006). Therefore, excess nutrient concentrations are either autochthonous or from human-induced runoff.

Blackbird Creek is a tidal system for the lower 22.2 km of the waterway, making it an extremely important ecosystem for many ecologically and economically important species. As a nursery, it is home to several fish species especially in their juvenile stages, including weakfish (*Cynoscion regalis*), channel catfish (*Ictalurus punctatus*), black drum (*Pogonias cromis*), white catfish (*Ameiurus catus*), Atlantic menhaden, alewife (*Alosa pseudoharengus*), American eel (*Anguilla rostrata*), white perch (*Morone americana*), striped bass (*Morone saxatilis*), and many others. It is also home to the blue crab (*Callinectes sapidus*).

The blue crab is an ecologically and economically important species with a range that expands from southern Canada to the shores of Argentina (Williams, 1974) (Figure 1-1). Individuals within this species grow quickly, reproduce early, and live short lives

(Van Den Avyle, 1984). The blue crab fishery drew 60,6045 metric tons of saleable stock valued at $191.9 million in the United States in 2013 alone with the Mid-Atlantic region accounting for 24,811 metric tons, or $84.7 million. However, this condition reflected over a 25 percent decrease in landings from the previous year in the country and 36 percent drop in the Mid-Atlantic (NOAA, 2014). This is a trend that started in the early 1990s (Figure 1-2). This reduction in landings is a concern because the effort to capture and sell this product has not dropped considerably (NOAA, 2014).

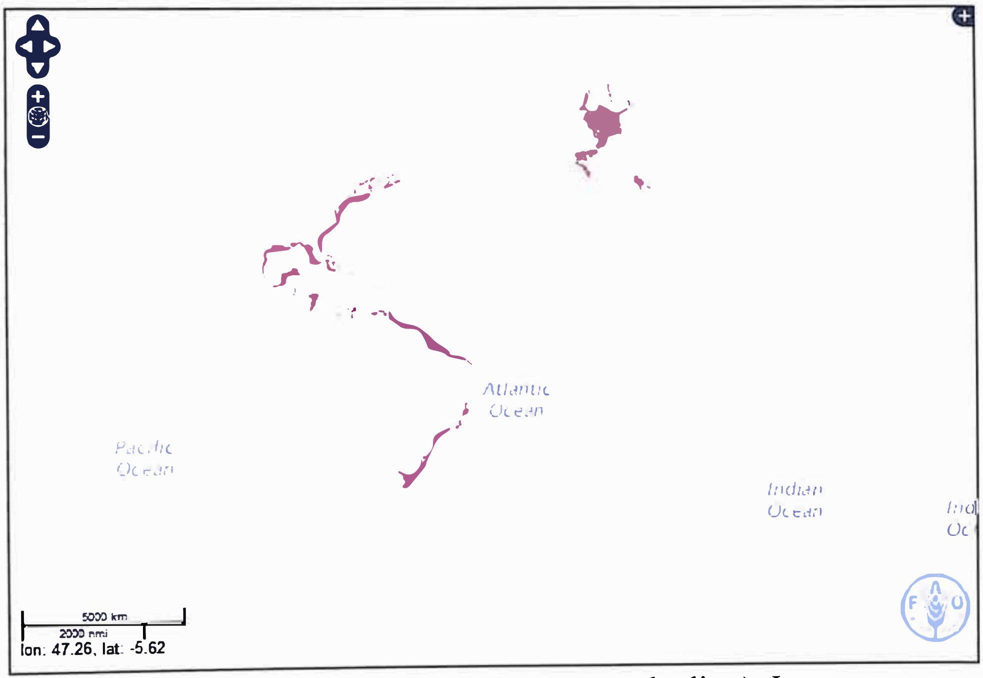

Figure 1-1. Blue crab population distribution (maroon shading). Image generated via Food and Agriculture Organization of the United States. (http://www.fao.org/figis/geoserver/factsheets/species.html?species=CRB-prj=4326).

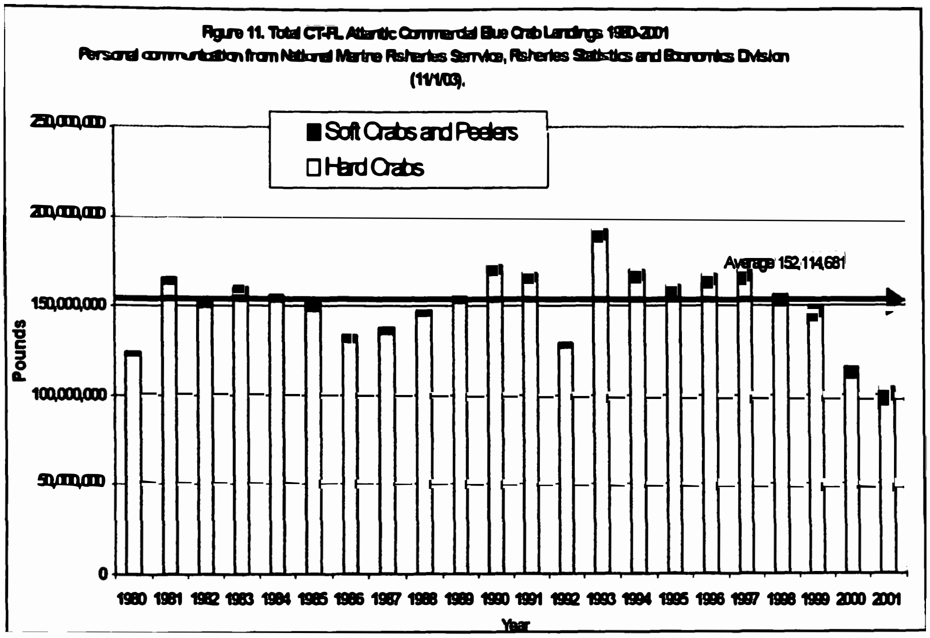

Figure 1-2. Blue crab landings (1980-2001) along the United States Atlantic coast. Image courtesy: Atlantic States Marine Fisheries Commission (2001).

There may be more to this story, however. The general understanding of blue crab population dynamics identifies heavy use of tidal creeks in the life history of the blue crab (Hill *et al.*, 1989). The Delaware Bay represents the northernmost portion of the commercial fishery for blue crabs (Kahn, 2003), where a number of pressures are placed on the populations, particularly along coastal waterways. These strains include habitat loss, extreme weather events, climate change, and compromised water quality due to development and agriculture.

Of particular importance is habitat change of the ecosystem due to anthropogenic impacts. Land use changes from native ecosystems to farmland can create temperature extremes due to lack of shading along the waterway (Blann *et al.*, 2001), can adjust salinity due to altered hydrology (Jayawickreme *et al.*, 2011), and can increase nutrient concentrations if a rain event occurs following fertilization of the crop fields in the watersheds (Johnson *et al.*, 1997). Blue crabs, particularly in their post-larval stages, are sensitive to changes in temperature and salinity (Mazzotti *et al.*, 2006; Mistiaen *et al.*, 2003). Costlow (1967) found that the most pronounced post-larval growth was at 30°C in salinities of 10 parts per thousand (ppt) to 40 ppt and that salinity greater than 30 ppt is optimal for growth. He also concluded that water temperatures ranging from 21.5°C to 34.5°C are suitable for survival and growth, and that 25°C is optimal (Costlow, 1967). Bucci *et al.* (2007) concluded that tissue enrichment of nitrogen in blue crabs and bivalves has a significant inverse relationship to water quality parameters and that there may be a relationship between nutrient sources (i.e., fertilizer runoff) and subsequent energy transfer through higher trophic levels (Bucci *et al.*, 2007). Thus, it is paramount to manage and maintain the nutrient dynamics of the tidal creeks in the Delaware Bay basin in order to preserve the blue crab populations.

A pervasive issue with regard to human impact is eutrophication – the introduction of large amounts of nutrients into ecosystems where blue crabs and other nekton exist. As evidenced in the Gulf of Mexico (Kling *et al.*, 2014) and the Chesapeake Bay (Boesch *et al.*, 2001), the nutrient concentrations can become high enough to seed phytoplankton blooms, which can draw dissolved oxygen down via decomposition. This causes a hypoxia episode, and could lead to death of sessile organisms and movement of motile organisms from the area (Diaz and Rosenberg, 2008).

Blackbird Creek is a watershed where there is great potential for eutrophic conditions, with up to 36% of the basin comprised of agriculture land cover (DNREC, 2006). These lands are fertilized to accommodate the timely growth of crops (Lu *et al.*, 2012). Nutrient loading can lead to eutrophication, a common problem for wetlands adjacent to agricultural lands that tends to favor the growth of aggressive, invasive, often weedy, plant species which displace native wetland plants. Drexler and Bedford (2002), for example, showed that inflow of nutrient rich water into a New York wetland led to growth of monotypic stands of native bluejoint (*Calamagrostis canadensis*) and broadleaf cattail (*Typhalatifolia*) in an otherwise diverse ecosystem. In extreme cases, such as the Chesapeake Bay, nitrogen and phosphorous loading causes algae growth so great that the blooms block out sunlight to submerged aquatic vegetation, a crucial habitat for larval and juvenile fish and crabs (Boesch *et al.*, 2001).

The ecosystem services provided by wetlands are wide-ranging. These include, but are not limited to, maintenance of water quality, regulation of atmospheric gases, and protection of shorelines (Dise, 2009; Clarkson *et al.*, 2013). It is becoming more apparent that wetland resilience is the key to predicting how such an ecosystem will accommodate long-term issues such as climate change, sea level rise, and permanent change due to human use.

A significant portion of the Blackbird Creek watershed is comprised of agricultural use, but only a small percentage is characterized by urban areas (DNREC, 2006; Pomilio, 2015). However, as time continues to pass, there is more pressure placed on the ecosystem as human population grows locally. There is a possibility that human impacts on fisheries will increase, reducing the habitat suitability of blue crabs (Engel and Thayer, 1998) and other fish species (Deegan and Buschbaum, 2005). Therefore, it is **the goal of this project** to examine potential effects of agricultural land use and identify baseline data of water quality parameters, food web components, and population dynamics of the blue crab throughout the watershed in the event of land use change to agriculture from wetlands.

Engel and Thayer (1998) identified that fisheries productivity can be influenced by human activity at every life stage of the blue crab (Figure 1-3). According to this schematic, blue crabs are in danger of population changes influenced by overfishing, point- and non-point source pollution, chemical additions (i.e., fertilizers), and dredging in the estuarine environment. Previous work has shown that habitat alterations by humans can have significant impacts on the overall aquatic health of the system. Kennish (1994) outlined the impacts of inappropriate waste management on estuaries, which leads to chemical releases into the ecosystem which can degrade water quality and reduce habitat suitability for local biota. Hrodey *et al.* (2009) showed that the index of biological integrity of warmwater Indiana streams was reduced with increased human land use practices. Mistiaen *et al.* (2003) modeled impacts of humans on blue crab harvests in the Chesapeake Bay and showed that crab harvests are reduced due to dissolved oxygen drawdown due to eutrophication.

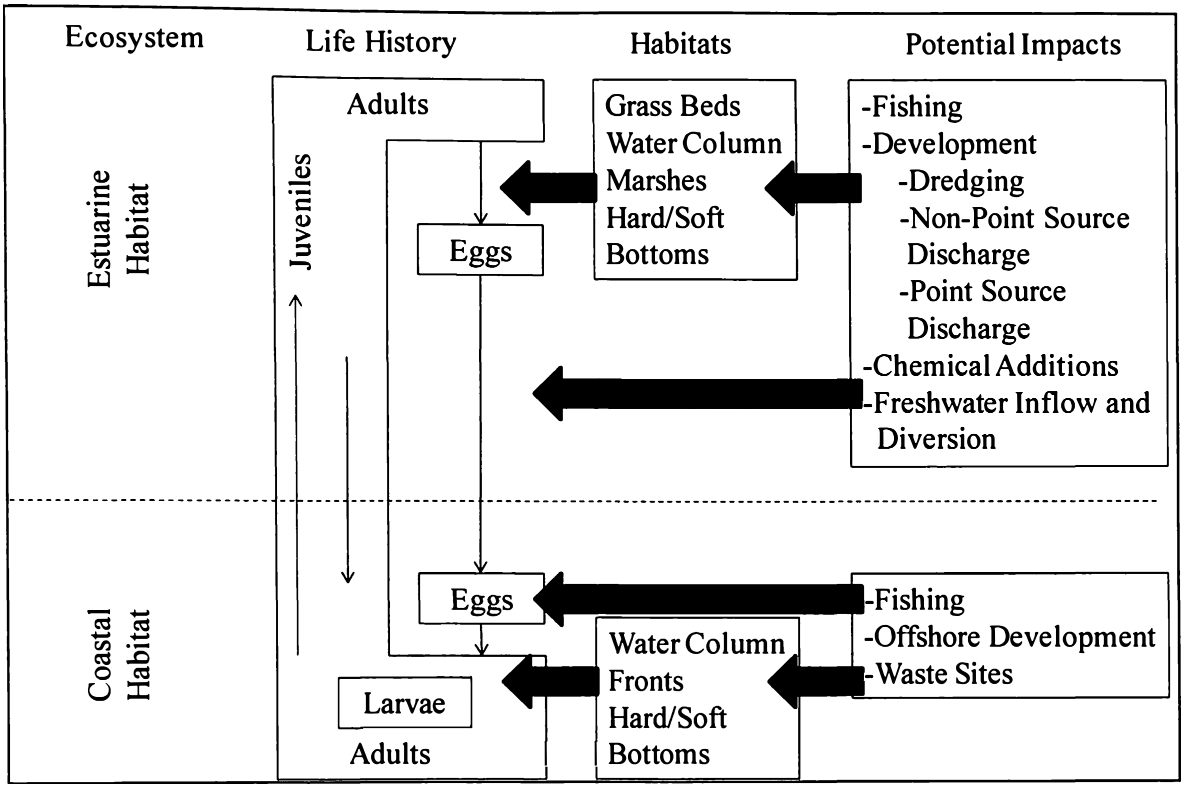

Figure 1-3. Life history of blue crabs and potential impacts on various life stages and habitats of the species as a function of human activity. Schematic adapted from Engel and Thayer (1998).

Substantial nitrogen is placed on cropland in order to maximize growth and productivity throughout the growing season throughout the Delmarva Peninsula (Bachman *et al.*, 1994). Extra nitrogen can accumulate in the soils, runoff into the waterway, migrate through the soil into groundwater channels, or enter the atmosphere by way of ammonia volatilization or nitrous oxide production. Of particular concern is the runoff into the waterway. Research has shown the detrimental effects of excessive nutrient input, as evidenced by work in the Gulf of Mexico (Rabalais *et al.*, 1996) and the Chesapeake Bay (Boesch *et al.*, 2001) because the land within these watersheds serve as alluvial plains for runoff into the waterway which, in turn, serves as a conduit to a centralized location where the nutrients can accumulate and cause a phytoplankton bloom (Carpenter *et al.*, 1998; Paul and Meyer, 2001; Valiela *et al.*, 2000).

Coupled with the pressures associated with changes in agricultural practices is the issue of how the fishery is managed as a whole. Given the commercial importance of the blue crab and its role in the ecosystem, it is important to be able to follow how the species will respond to adjustments to land use and land cover. Blue crabs are known to be cannibalistic, but they are also opportunistic carnivores and will consume whatever may be available. If there are changes to the environment which may favor one organism over another, this may mean that a crab will have to search more or less in order to meet its daily intake requirements. If sessile or slow-moving organisms such as mussels or clams, for example, cannot grow and reproduce due to reduced oxygen conditions, they will not be available for crab consumption. This would force the crab to find alternate food sources, perhaps those which are more difficult to obtain and, thus, require more energy use in order to capture the food item.

It was hypothesized for this project that Blackbird Creek, given its agricultural history, has the potential for nutrient runoff from crop fields and that there would be greater signals of certain parameters, particularly turbidity, inorganic nitrogen, and orthophosphate, in areas with adjacent cropland, and that there would be greater nutrient concentrations in the crop planting season (May-June) than the growth season because of field fertilization in early spring and lack of plants for nutrient uptake and soil erosion control. It was also hypothesized that there would be greater nitrogen, phosphate, and turbidity concentrations in the harvesting season (September-November) than the crop growth season (July-August) because there would not have been any uptake by plants of any remaining nutrients on the field, and because the lack of planted crops would reduce the stability of the soil landscape and allow for greater potential for runoff.

It was further hypothesized that if there was excessive nutrient runoff due to the proximity of crop fields to the waterway, there would be a greater abundance of small fish, which may increase secondary consumer abundances. Thus, there would be greater catch for both of these categories near agriculture than away from agriculture.

The final objective of this study was to determine fish biodiversity throughout the watershed with respect to adjacent land use practices. It was hypothesized that biodiversity would be greater in areas of natural land use than in agricultural areas where eutrophic conditions are possible because eutrophy-intolerant would be associated with areas of natural land use but avoid agricultural land use areas.

This nonpoint source pollution could have detrimental impacts on water quality, blue crab population dynamics, and fish biodiversity and trophic structure. On smaller spatial scales, little research has been done to understand how increased nutrient concentrations could impact higher trophic levels, particularly with respect to commercially important species and those which are directly or indirectly associated with that species. Therefore, three specific objectives are outlined for this thesis:

- Identify differences in water quality parameters in Blackbird Creek with varying degrees of adjacent agricultural land cover across years and agriculture seasons.
- Identify differences in blue crab population dynamics in Blackbird Creek with varying degrees of adjacent agriculture land cover across years and agriculture seasons.
- Identify differences in fish biodiversity and trophic structure in Blackbird Creek with varying degrees of adjacent agriculture land cover across years and agriculture seasons.

CHAPTER 2

LITERATURE REVIEW:

CHANGES IN LAND USE AND IMPACTS ON WATER CHEMISTRY, BLUE CRAB POPULATIONS AND NEKTON BIODIVERSITY

Over half of the United States population resides along the coastal and estuarine areas (Crossett *et al*., 2004), threatening local ecosystems due to changing land use and causing changes in soil and groundwater chemistry, watershed-level hydrology, and dissolved nutrients in waterways, particularly in the form of nitrogen and phosphorous species. As a whole, the United States has increased its use of commercial fertilizers from about 6.8 million metric tons in 1960 to roughly 20.0 million metric tons in 2011 in order to accommodate rapid population growth and need for increased food quantities, nearly a 3-fold increase (Nehrling, 2013).

The coastal state of Delaware has grown slightly faster than the United States as a whole. Delaware grew 4.2% between 2010 and 2014 while the country grew about 3.3% (http://www.indexmundi.com/facts/united-states/quick-facts/all-states/population-growth#map). Of particular interest is the Blackbird Creek watershed, located in southern New Castle County, Delaware. The Delaware Department of Natural Resources and Environmental Control (DNREC) identified that 36.1% of the watershed is designated for agricultural use, and an additional 13.2% for urban use (DNREC, 2006; Figure 2-1).

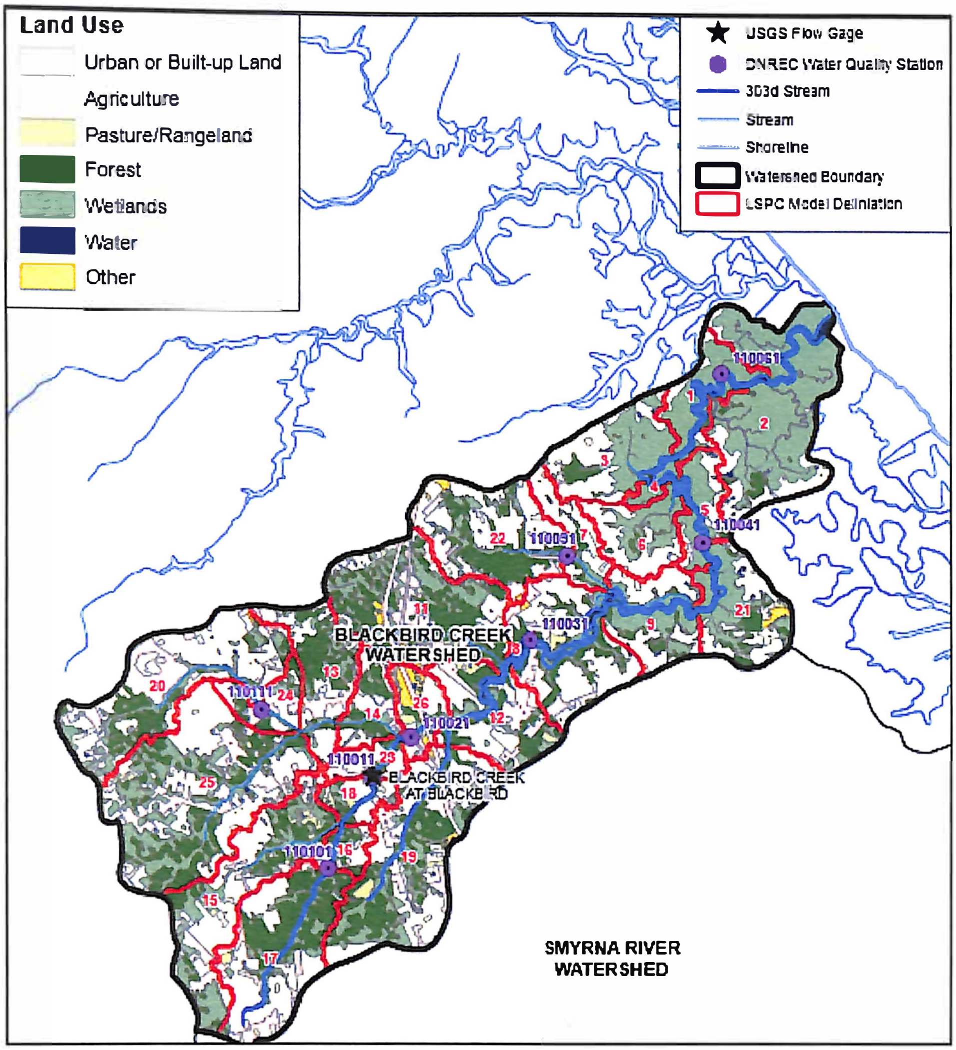

Figure 2-1. Land use practices in Blackbird Creek, as described by LSPC Model Segmentation (DNREC 2006).

Population growth and land use changes put pressure on ecosystems. As more people move into an area, more land is cleared for development and agricultural purposes. Roads, parking lots, golf courses, and other infrastructure can lead to more impervious surfaces, thus allowing sediments, harmful pollutants, and nutrients to enter local waterways. Research is available to provide information about how nutrient loading impacts streams, especially in large watersheds (Rabalais *et al.*, 1996; Boesch *et al.*, 2001). However, there is less information available on nutrient loading in smaller systems

(Gedan *et al.*, 2009). Nutrients entering estuarine systems such as Blackbird Creek often do so through both surface water and groundwater inputs. Some agricultural practices, including over-tilling soil and loosening particles unnecessarily, over-irrigating cropland, and over-applying fertilizers, can send more soil particles and nutrients into the creek.

This sort of input is characterized as nonpoint source pollution, whereby larger scale, more diffuse nutrient supplies are loaded into waterways, often due to human activity on modified surfaces. Unfortunately, this type of contamination is much more difficult to manage than point source pollution. Without a clear source which can identify where the runoff is coming from, it is challenging to place responsibility on any particular area of land and be able to prove that land use on that parcel is responsible for increased nutrient concentrations. Furthermore, it is difficult to persuade landowners to make adjustments to their land use practices if they cannot identify something obvious and tangible which places responsibility at their feet. Thus, while Delaware does have nutrient management regulations (Title 3; Section 1200) and rules governing the control of water pollution (Title 7; Section 7200), most on-site management that exists in the current landscape is that which reflects point sources of pollution.

Nitrogen is often a major contributor to eutrophication of lakes and rivers in agrarian systems. This is in part because nitrate is highly soluble and, therefore, is readily transported through sediments and across surfaces (Freeze and Cherry, 1979). In natural environments, nitrogen is a key component to sustaining productivity as a food source to algae and higher plants, and is thus important for maintaining biodiversity. Current input of nitrogen from human sources, mostly as agricultural contributions that include the planting of nitrogen-fixing crops and poor management of surface nutrient runoff into waterways, is equal to or greater than natural nitrogen input (Vitousek *et al.*, 1997). Rainfall and irrigation transport nitrate and other nutrients into adjacent waterways (Aelion *et al.*, 1997). The increased input leads to a reduction in dissolved oxygen, acidification in susceptible environments, and overall degradation of the water quality (Vitousek *et al.*, 1997). Long term responses by ecosystems to increased nitrogen loading tend to be harder to predict, but have been linked to shifts in plant and microbial communities due to nitrate saturation rather than nitrate limitation (Aelion *et al.*, 1997).

The potential for salt marshes to be able to intercept nutrients from draining into waterways is important to understand because the plants and soil composition within may act as buffers to the possible eutrophication of adjacent waters (Fisher and Acreman, 2004). Dentrification is a microbially-mediated process whereby nitrate is converted to nitrogen gas (N_2). In marsh ecosystems, this is an important component of the nitrogen cycle which removes nitrate from the system before it ends up in the waterway in dissolved form (Hopkinson and Giblin, 2008). There is a body of evidence to suggest that greater nitrogen input leads to higher rates of denitrification because it is presumed that the process is nitrate-limited (Aelion and Engle, 2010), but there is also published research which disproves such findings (Wigand *et al.*, 2003; Tuerk and Aelion, 2005). This is probably due to the wide range of denitrification rates at various horizons of the sediment profile. It is most likely that the greatest denitrification rates occur at the soil-air interface where (1) there is the greatest probability of microbes to meet surface dissolved nitrate from anthropogenic sources, and (2) nitrification is more prevalent, as it is an aerobic process which oxidizes ammonia to nitrite then to nitrate, again feeding the denitrification process if the dissolved nitrate becomes available in anaerobic conditions (Thompson *et al.*, 1995).

Even though nitrite is often quickly oxidized during denitrification, it remains an important constituent to measure in aquatic ecosystems. Pellerin *et al.* (2006) found that stream dissolved organic nitrogen (DON) was up to 4 times higher in agricultural watersheds than in forested sites. Higher DON has been correlated to increased nitrite in some ecosystems where otherwise it would be considered a rare form of nitrogen (Eddy and Williams, 1987). Previous research has shown that nitrite levels as low as 20 mg L^{-1} can lead to mortality in intermolt blue crabs, and as low as 2 mg L^{-1} for molting or post-molt crabs (Manthe *et al.*, 1984). Other studies have shown that nitrite accumulation greater than 0.45 mg L^{-1} can reduce the growth rate of shrimp in aquaculture ponds (Gross *et al.*, 2004). If this is also true in natural environments, nitrite loading could reduce shrimp populations in nursery systems such as Blackbird Creek, an important food component for many of the larger nekton. Indirectly, then, increased nitrite can impact higher trophic levels. Indeed, nitrite can also have direct detrimental effects, as it can be actively taken up across gill epithelia and can accumulate quickly to high concentrations in body fluids in fish and crustaceans (Kroupova *et al.*, 2005), which can become toxic (Lewis and Morris, 1986; Jensen, 2003).

Like nitrite, ammonia can disrupt the normal functions of internal organs such as damage to the gill epithelium (Lang *et al.*, 1987; Wilkie, 2002) and disruption to metabolic function of the liver and kidneys (Arillo *et al.*, 1981), often to the point of death to the organism. In aqueous solutions, ammonia (NH_3) exists in equilibrium with ionized ammonia (NH_4^+), and is correlated positively with both pH and water temperature (Emerson *et al.*, 1975). The United States Environmental Protection Agency (USEPA) has tried to limit the amount of total ammoniacal nitrogen in drinking water, setting the criterion at 0.0091 mg L^{-1} NH_3) for water at 20°C and a pH of 7.00 (USEPA, 1989). In saltwater, the restrictions are not as stringent, with maximum values set at 0.035 mg L^{-1} NH_3 for a four-day average concentration before there are any considerable effects to locally important species (USEPA, 1989). Because ammonia concentration is dependent on temperature, salinity, and pH, these values are adjusted to accommodate a particular aquatic system. In an ecosystem such as Blackbird Creek, where temperature ranges between 5°C and 35°C, and salinity ranges between about 2 ppt and 15 ppt, the maximum ammonia concentration criteria to minimize effects on extant species are between 0.69 mg L^{-1} NH_3 and 1.15 mg L^{-1} NH_3 (USEPA, 1989).

In addition to nitrogen runoff, Blackbird Creek is susceptible to phosphorous enrichment. Many crops that are grown adjacent to waterways are fertilized with products high in phosphate concentrations. Phosphorous in freshwater can be retained in sediments by interactions with cations such as Iron (Fe) and Aluminum (Al) while in seawater environments, deposited phosphorous is in large part returned to the overlying water through remineralization, making it biologically available for consumption of phytoplankton (Conley, 2000). In a system such as Blackbird Creek, most of the phosphorous is probably in the form of bound reactive phosphate attached to eroded particles that have run off into the surface water (Olli *et al.*, 2009). Phosphorous can be retained in water and sediments via adsorption, complex formation, chemical precipitation, and biogeochemical reactions, referred to as "cycling" (Correll, 1998; Reddy *et al.*, 1999). Once phosphorous inputs enter the water column, the compounds may be enzymatically hydrolyzed to form orthophosphate – the only form of phosphorous that can be taken up by algae, plants, and bacteria. Excessive fertilizer application can result in phosphorous buildup because the N:P ratio in plants is 8:1 while it is only 3:1 in some fertilizers (Sharpley *et al.*, 2007). The USEPA identified phosphorous concentrations of 0.31 mg L^{-1} or greater to be detrimental to aquatic organisms and is therefore the maximum value that should ever be in bioavailable form in any waterway (USEPA, 2001). Studies have shown the detrimental impacts that phosphorous has on aquatic ecosystems, including eutrophication (Schindler, 1974; Schindler, 1977; Havens, 2008), mortalities of fish and invertebrates (Anderson *et al.*, 2002), and stream community shifts from a heterotrophic to an autotrophic state (Peterson *et al.*, 1985).

Blue crab populations could be impacted indirectly by phosphate loading, mostly via oxygen reduction in the water. Reduced dissolved oxygen has been shown to increase blue crab hemolymph lactate activity (Lowery and Tate, 1986). Studies also found that crabs become agitated and tend to stand with their legs fully flexed, maximizing the space between the ground and their lower carapace when exposed to hypoxic conditions, where dissolved oxygen concentration is less than 2 mg L^{-1} (Batterton and Cameron, 1978; Lowery and Tate, 1986; USEPA, 2000).

While not necessarily acutely toxic, the overabundance of these reactive nutrients can be detrimental to the quality of waterways. This is also true of non-reactive constituents. Turbidity is a measure of how clear or cloudy a water sample is and, by proxy, a suitable measurement to determine suspended sediments which could settle out onto the riverbed. Sediments in suspension have the potential to smother benthic biota, irritate fish and crab gills, and transport adsorbed contaminants (Davies-Colley and Smith, 2001). It has already been documented here why this is important with respect to the role of phosphorous in receiving waters. It is also important to monitor because, while sediments do have the capability of settling out, there is also potential for them to remain in suspension and reduce the visual range of sighted organisms to seek out prey and/or members of the opposite gender for reproductive activity (De Robertis *et al.*, 2003; Kirk, 1994). High turbidity also reduces the light penetration into the water (Cloern, 1987). Less light availability reduces the amount of phytoplankton production, thus possibly impacting higher trophic levels in a bottom-up model (De Robertis *et al.*, 2003; Utne-Palm, 2001), as has been documented in the Delaware Bay (Pennock and Sharp, 1986). Reasons for high sediment concentrations in tidal creeks and estuaries include transport through the catchment basin, especially when land is modified to accommodate human uses, tidal activities, and wind-induced sediment resuspension (Talke and Stacey, 2007). If there is recently tilled land on a slope leading into the waterway, there is a strong chance of erosion of sediment due to rain and wind, and subsequent runoff into the waterway (Wischmeier and Smith, 1978). These phenomena lead to variability in light availability on scales of hours (tides) and days (calm versus windy days). In systems such as Blackbird Creek, primary production may be compromised due to diminished light penetration which could result in reduced zooplankton production and lead to lower recruitment of estuarine fish species (Steel and Henderson, 1991), thus altering the flow of energy through food webs since the autotrophic base would not be sufficient enough to sustain higher trophic levels (Perissinotto *et al.*, 2010).

Because tidal creeks and estuarine systems in general are extremely diverse and sustain many larval and juvenile individuals of fish and crabs, it is important to also monitor the alkalinity within the creek. This measurement provides information on the buffering capacity of the water, and is especially important in watersheds that could be impacted by land use changes by human activity. As atmospheric CO_2 diffuses passively into water, it forms carbon acid (H_2CO_3). This is a weak acid that dissociates rapidly to produce hydrogen ions (H^+) and bicarbonate ions (HCO_3^-). Increased H^+ concentration causes water pH to fall unless there are sufficient carbonate ions (CO_2^{3-}) which can react with the H^+ to form more HCO_3^-. This reaction ($CO_2 + H_2O \rightarrow H_2CO_3 \rightarrow HCO_3^- + H^+ \rightarrow CO_2^{3-} + 2H^+$) is called the carbonate buffering system and is what allows for a stable pH in aquatic systems. If this buffering system is compromised, there could be significant impacts on estuarine systems, especially shellfish populations. Since the shells of mollusks, crabs, clams, and oysters are made of calcium carbonate, any reduction in pH threatens these species because the shells could become softer, reducing their capability to provide sanctuary to the organisms within.

Temperature is another parameter of interest. When temperature is increased, up to a certain point, the growth of phytoplankton populations is exponential (Eppley, 1972). Blue crabs have shown increased levels of viruses and alterations in immune responses when subjected to chronically higher temperatures (Chung *et al*., 2015). Intermolt time periods and percentage of growth of the carapace width per molt are also temperature-dependent (Brylawski and Miller, 2006). Fish assemblages may also be impacted with elevated temperature. Often, fish migration is temperature-initiated (Boehlert and Mundy, 1988). As temperature rises, there is an increase in evapotranspiration from the surfaces of water bodies and from the plants nearby and within. When this occurs, salinity rises because salt will not form into a gaseous phase with water that is transformed into water vapor, thereby increasing salinity. This is augmented by changes in land use. If a forested buffer is removed from a landscape and replaced by an agricultural plot, two scenarios (or some combination of the two with varying degrees) could play out. Since the buffer is no longer in place, it is far less likely that surface water, especially during heavy rains, will remain on the land and infiltrate the soil. This would lead to greater runoff into waterways, decreasing the salinity. However, that newly available surface runoff likely going to include a ionized particles (Al^{+}, Fe^{+}, PO_4^{3-}, Na^{+}, Cl^{-}, SO_4^{2-}, K^{+}) that could react with each other to form salts, especially if fertilizers and pesticides are sprayed onto the adjacent crop fields. This would increase the salinity, which is a dominant factor regulating stratification, especially in slow-moving water bodies. Salinity changes influence development of algae blooms (Gallegos and Jordan, 2002) and the distribution of anammox bacteria (Bernhard *et al.*, 2005). As discussed earlier, these ammonia-oxidizing bacteria are important in the denitrification profile of marsh environments, which influence local flora utilized by upper trophic levels of both aquatic and terrestrial biota. While blue crabs seem to tolerate extreme salinities (Tagatz, 1971), there is evidence of increased respiration and excretion in environments with high salinity (Guerin and Stickle, 1992). The potential changes to their energy budgets may lower the range of prey items they would be able to obtain (e.g., perhaps a blue crab would be unable to prey upon a fast-moving fish). Fish populations, in turn, may experience some changes with changing salinity as well, particularly in nursery habitats such as Blackbird Creek. For example, Kraus and Secor (2005) showed a positive correlation between river discharge and juvenile abundances of white perch in the Chesapeake Bay estuary system. Martino and Able (2003) showed that species richness increased with increased salinity.

The evidence that land use practices could influence adjacent waterways is abundant in the literature. However, there is limited published research on how agriculture impacts water quality in tidal creeks. The changes in water quality may have a bottom-up effect on higher trophic levels, making a comprehensive study in such an ecosystem necessary. This is particularly true given that Blackbird Creek lies on the

Delaware coastal plain which is being influenced by population growth, land use changes, climate change, and sea level rise (Ross *et al.*, 2015).

CHAPTER 3

RESEARCH METHODOLOGY

3.1. Part I - Water Quality relationship to land use and land cover in Blackbird Creek

3.1.1. *Study Sites and Experimental Design*

Blackbird Creek is a tidal waterway located along the central Delaware that drains a watershed of about 80.29 square kilometers into the Delaware Bay. Roughly half of its 44.10 km in length is affected by tidal fluctuations while the upper portion of the waterway is comprised of freshwater only (DNREC, 2006). Several sites were selected in the tidal portion of the creek, representative of the lower 20.92 km of the waterway nearest to the bay. In 2014, 5 sites were monitored: three of which were in areas classified as forested wetland (referred to throughout as "non-agriculture") according to GIS (geographic information systems) layers collected from the Delaware Geospatial Exchange via the Office of State Planning, then layered into ArcMAP (Pomilio, 2015), and two that were in close proximity to agricultural fields adjacent to the waterway (referred to throughout as "agriculture") (Figure 3-1).

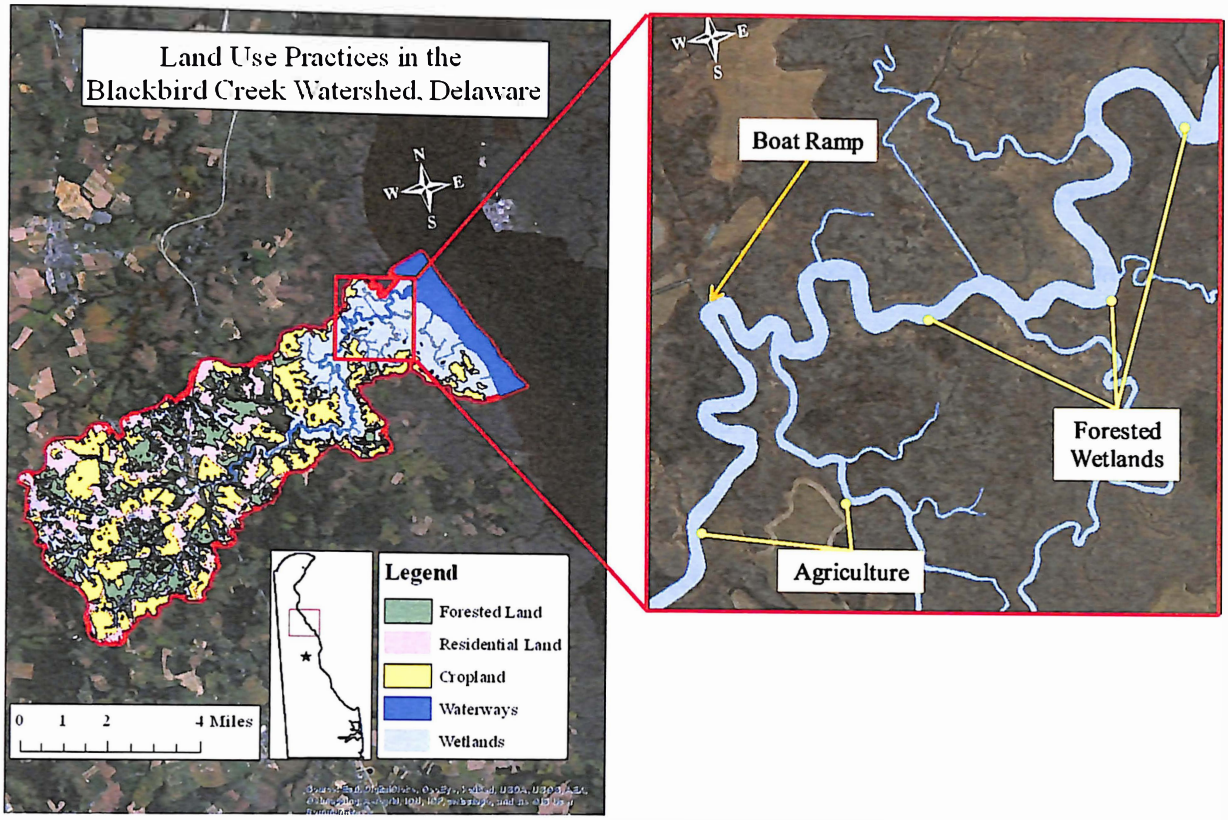

Figure 3-1. Site selection for 2014. Left side includes land use practices throughout Blackbird Creek, with an inset (right) to identify specific locations of each of the treatments.

The experimental design was continued into 2015, but there were some changes. The tidal portion of the creek was gridded out into 500 m x 500 m cells, as designated by aerial data collected in 2012 (Pomilio, 2015). Fourteen stations were selected throughout the tidal portion of the creek. Those where there was less than 5% agriculture land cover within the cell were designated as "non-agriculture," whereas those stations with greater than 5% agriculture land cover were designated as "agriculture." Therefore, for the 2014-2015 experimental design, there was a comparison of dependent variables between treatments of agriculture and non-agriculture.

A second experimental design was established in 2015 in order to obtain information on the remainder of the tidal portion of the waterway, and so that effects of dependent variables could be analyzed on wider range of percentage of adjacent cropland. In 2015, 14 stations were selected to encompass the portion of the creek upstream of the boat ramp (Figures 3-1 and 3-2). Each of the sites was labeled based on the percentage of agriculture within a 500 m x 500 m cell, as designated by aerial data collected in 2012 (Figure 3-2). Each cell was labeled based on a range of percent cropland in order to achieve appropriate replicate habitats. Thus, a cell labeled “20,” for example, represented a spatial percentage of agricultural land between 20% and 30%. In 2015, there were four treatments based on cropland percentage: 0%-10%, 10%-20%, 20%-30%, and 30%-40%. The 0%-10% treatment had seven replicates. The 10%-20% treatment had three replicate stations. Each of the remaining categories had two replicates. On any given date, 10 of the 14 sites were chosen using a random number generator in Microsoft Excel® for collection due to time and funding constraints. Therefore, in some instances, replicate samples may not have been obtained. This was accommodated for by repeated sampling within each of three seasons. The crop planting season was defined as May-June. The growth season was July-August. The crop harvesting season was defined as September-November. In 2014, samples were collected biweekly throughout the field season. In 2015, water was sampled weekly for four consecutive weeks in June (planting season), August (growth season), and October-November (harvesting season).

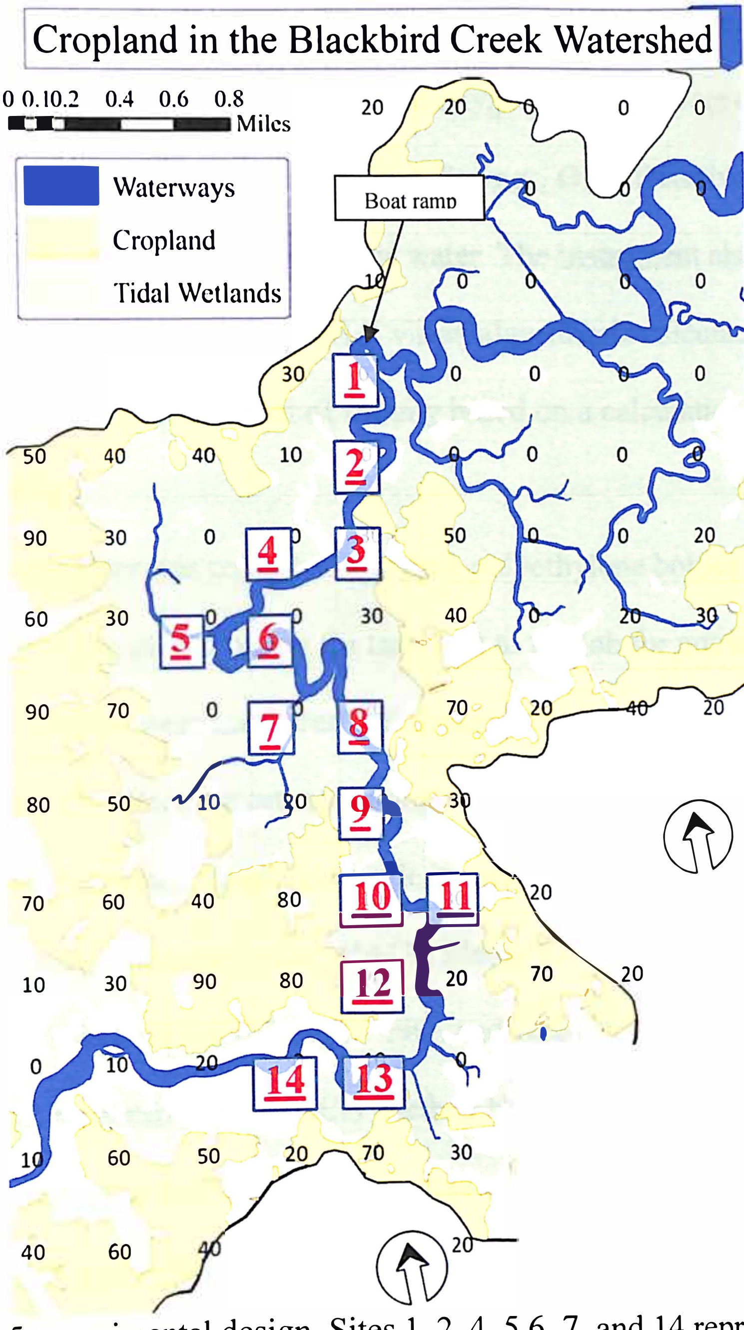

Figure 3-2. 2015 experimental design. Sites 1, 2, 4, 5 6, 7, and 14 represent 0%-10% agriculture. Sites 9 and 13 represent 10%-20% agriculture. Sites 8 and 12 represent 20%-30% agriculture. The remaining sites (3 and 11) represent 30%-40% agriculture.

3.1.2. *Surface Water Quality*

Physical data were collected *in situ* at the time of water collection. Temperature (^{o}C), pH, conductivity (mS cm^{-1}), and dissolved oxygen (mg L^{-1}) were obtained with a YSI 556 Multiprobe (YSI Incorporated, Yellow Springs, OH). Dissolved oxygen was based on the percent oxygen saturation of the water. The instrument also provided a measurement of dissolved oxygen in mg L^{-1} via an algorithmic calculation from the percent saturation and a measurement of salinity based on a calculation from the conductivity reading.

At each site, water was collected into dark polyethylene bottles (rinsed 3 times with sample water), then placed on ice for transport to the lab for nutrient analysis. For *ex situ* analysis, the samples were measured for nitrate+nitrite, ammonia, orthophosphate, turbidity, and alkalinity. Because orthophosphate is reactive in nature, a subsample of water was filtered on site into a separate bottle in order to minimize the potential for such reactions. Nitrate was determined using HACH Method No.8171, the cadmium reduction method to detect levels between 0.1 mg L^{-1} NO_3 and 10.0 mg L^{-1} NO_3. Nitrite concentration was evaluated using HACH Method No.8507, the diazotization method with detection levels up to 0.30 mg L^{-1}. Ammonia was measured using the Ammonia salicylate method, HACH Method No.8155, detecting ammonia concentrations between 0.01 mg L^{-1} NH_3 and 0.50 mg L^{-1} NH_3. Orthophosphate concentrations were determined using HACH Method No.8048, the ascorbic acid method. This reaction provided information on soluble reactive phosphate concentrations between 0.02 mg L^{-1} PO_4^{3-} and 2.50 mg L^{-1} PO_4^{3-}. Each of these parameters was measured on a HACH DR 3900 spectrophotometer (HACH, Loveland, CO). Palintest methods were used to determine alkalinity and turbidity concentrations in Blackbird Creek and measured on a YSI 9500 Photometer (YSI Incorporated, Yellow Springs, OH). Alkalinity was evaluated using an automatic wavelength selection, measuring between 0 mg L^{-1} $CaCO_3$ and 500 mg L^{-1} $CaCO_3$. Turbidity was also determined with an automatic wavelength selection and presented in Formazin Turbidity Units (FTU), which is generally equivalent to Nephelometric Turbidity Units and Jackson Turbidity Units (JTU). This method allowed for measurements between 5 FTU and 400 FTU.

3.1.3. *Statistical Analysis*

A general linear model (GLM) was conducted for the 2014-2015 experimental design to determine differences by the main effects of treatment, season, and year. Also within the model were interaction effects, which are dependent upon each other if proven to be significant: treatment × season, treatment × year, season × year, and treatment × season × year. Following analysis with a GLM, post-hoc pairwise comparisons were run using the Mann-Whitney U test to discern differences among levels of factors containing more than two levels.

The experimental design was adjusted in 2015, so additional statistical analyses were required. A GLM was conducted to identify differences in water quality parameters by the main effects of season and treatment, and by the treatment × season interaction term. Following analysis with a GLM, post-hoc pairwise comparisons were run using the

Mann-Whitney U test to determine differences among levels of factors containing more than two levels (represented in the results as *U* with a corresponding p-value).

To discern possible effects of treatment more clearly, each of the spatial treatments (ranged percentages of agriculture) was characterized as a single percent agriculture value within each of the 500m x 500m grid cells. This allowed for analysis of correlation between percentage of agriculture and each of the dependent variables.

3.2. Part II – Land use and blue crab population dynamics, fish biodiversity, and fish trophic structure in Blackbird Creek, Delaware

3.2.1. *Study Sites and Experimental Design*

See Section 3.1.1 for experimental designs. Blue crabs were collected using two methods: baiting pots and allowing a 24-hour soak, and pulling an otter trawl along the creek bed. Fish were collected using only otter trawling methodology.

3.2.2. *Blue Crab Collection and Processing in Crab Pots*

The pots were standard recreational Quonset hut style half-pots constructed of 38.1mm funnel black vinyl coated galvanized wire with dimensions of 60.96 cm × 25.4 cm × 35.5 cm (Figure 4-1). The pots were covered with a 9.525 mm polyethylene mesh and fastened to PVC pipes placed at site locations throughout the creek, as was the procedure followed in 2012-2013 by Roeske (2014).

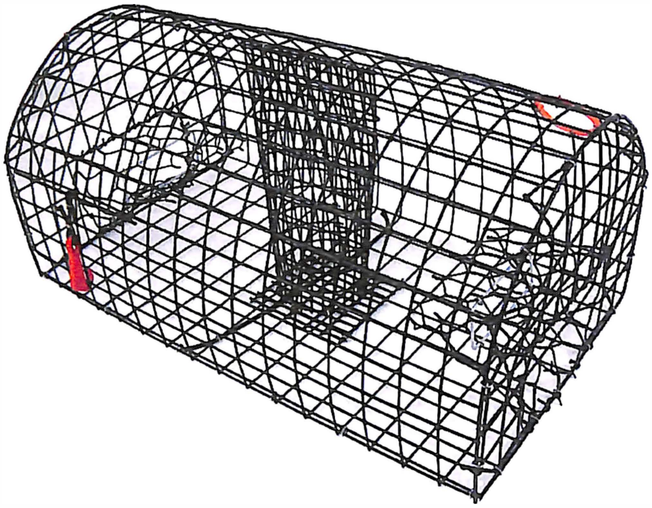

Figure 3-3. Diagram of Quonset hut style crab pots.

The pots were always placed on the shore of the creek that most closely represented the total percentage cropland identified by aerial imaging and GIS. The pots were always oriented in line with the flow of water and placed at or just below the intertidal zone. In 2014, a pot was placed at the beginning, middle, and end points of the 300m stretch identified at each site. Thus, for 5 sites per sampling date, 15 pots were set. In 2015, the effort was adjusted to include more pots, but fewer at each site. Pots were placed at 10 sites per sampling date, but only at the beginning and end points for a total of 20 pots per date.

Each pot was baited with an entire Atlantic menhaden that thawed overnight prior to placement. Each bait fish was cut open at three strategic locations: (1) just behind the head central to the body, (2) the central ventral area to open up the belly, and (3) near the anal opening. This should have minimized variation in the amounts of oil exuded from the bait to attract crabs. The pots soaked for 24 hours. If crabs were captured, the carapace width (CW) was measured (mm), molt stage (molting or not molting) was identified based on the hardness of the carapace and/or the color of the exoskeleton on the second swimmeret, and life history stage (recruit, juvenile, mature) was determined based on CW size (recruits < 30 mm CW; juveniles 30 mm – 115 mm CW; mature > 115 mm CW), in accordance with previous work done in Blackbird Creek by Roeske (2014). As part of a separate project in 2014, some male crabs larger than 100 mm were sacrificed for laboratory analysis. Otherwise, upon completion of the processing and data recording, all individuals were returned to the approximate location from which they were captured.

3.2.3. *Blue Crab and Fish Collection in Trawls*

Blue crabs and fish were both collected using an otter trawl. Otter trawling was conducted using two different nets between 2014 and 2015 (Figure 4-2). The first season featured a 3.048 m semi-balloon net made with 3.81 cm mesh throughout the body, 2.54 cm mesh in the bag, with a 0.95 cm knotless inner liner cod end. The trawl net was fully rigged with 38.1 cm x 76.2 cm doors, 3.81 cm x 6.35 cm spongex floats on the head rope and a loop-style tickler chain on the foot rope. The warp length was 19.65 m. In 2014, triplicate trawls were performed at each of the 5 sites in accordance with protocols identified in previous years, operating under the assumption that each subsequent trawl was pulling along the same track as every previous trawl (Roeske 2014).

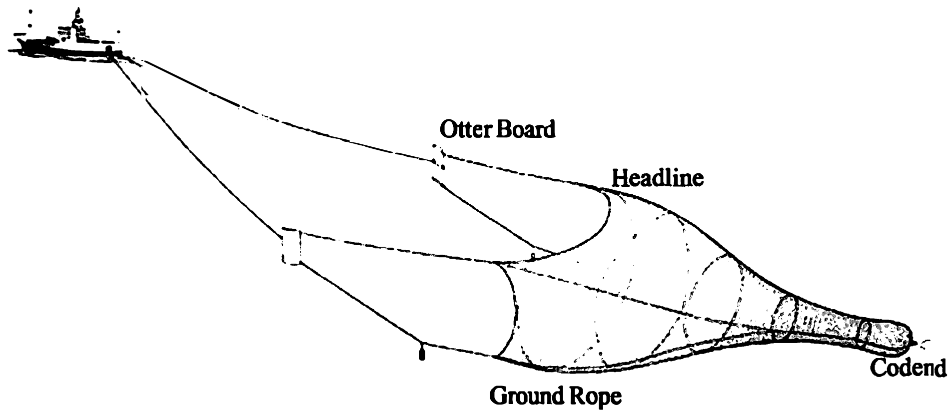

Figure 3-4. Diagram of basic components of an otter trawl. Photo courtesy: FIO Field Studies in Marine Science (http http://fiofieldstudies2013.blogspot.com/2013/06/usfsp-days-two-and-three.html).

The procedure and net were changed in 2015.The trawl net width was increased to a 4.88m and the warp length to 43.37 m. The larger net also required larger doors (70.0 cm x 91.4 cm) to accommodate greater drag. For each of the 10 sites chosen for a particular day's sampling, a single trawl was conducted. This allowed greater spatial resolution. Each trawl was carried out against the flow of the tidal regime between the PVC markers placed along the shoreline within each site. The boat operator maintained a consistent speed between 1500 rpm and 2500 rpm, depending on the tidal regime, for each tow in order to have a tow time between 4 and 7 minutes. The average depth was acquired using an SM-5 Portable Water Depth Sounder Gauge (Cole-Parmer Instrument Company, LLC; Vernon Hills, IL) in order to calculate the average scope. However, instantaneous adjustments to the scope could not be accommodated due to the many depressions that exist along the creek bed and the inability to change warp length mid-trawl. Thus, only average scope is available for each site (Table 4-1). Stream width varied between sites (roughly 50 m – 90 m) and tidal structure. Given the 3.048 m width of the trawl net in 2014, roughly 3% - 6% of the creek was sampled on any given trawl. The 4.88 m width of the trawl net in 2015 increased the percent sampled per trawl 5% - 10% of the creek.

Table 3-1. Warp lengths, average site depths, and scope ratio at each site in 2014 and 2015. Note: Site 4 between 2014 and 2015 was not changed.

Year	Station Number	Warp Length (m)	Average Depth (m)	Scope Ratio
2014	1	19.65	4.60	4.27
	2		5.85	3.36
	3		5.43	3.62
	4		6.74	2.92
	5		3.63	5.42
2015	1	43.37	4.91	8.84
	2		4.42	9.81
	3		5.24	8.27
	4		6.74	6.44
	5		5.15	8.42
	6		4.24	10.24
	7		6.13	7.08
	8		6.49	6.68
	9		7.01	6.19
	10		5.94	7.30
	11		6.16	7.04
	12		5.09	8.52
	13		5.30	8.18
	14		2.04	21.24

After every trawl, each crab was and processed on board to determine gender, CW, molt stage, and life history stage, as was done with crab pot captures. Fish were simply identified in the field and counted. In 2014, animals were released outside of the 300m trawling site in order to minimize the possibility of resampling them in subsequent trawls. In 2015, the animals were released back into the same site from which they were captured.

3.2.4. *Calculations and Statistical Analysis*

Blue crab data collected from pot sets was analyzed separately from data retrieved from trawling because there was no way to compare these two different types of gear empirically. However, the procedures for statistical analyses (GLMs and Spearman rank correlations) were the same, after accounting for catch per unit of effort (CPUE).

Each pot soaked for the same amount of time (24 hours), and had the same bait and set/retrieve procedures, so there was no need to accommodate for changes in gear type. The CPUE thus was simply the number of animals captured at each site, divided by the product of the number of pots set and the number of stations for each treatment, in a given season or year (Equation 3-1).

$$CPUE = \frac{n}{ht} \qquad \text{Equation 3-1}$$

where n is the number of crabs captured in a given treatment, h is the number of pots set, and t is the number of stations per treatment.

The calculation for CPUE was more complex for trawling. Because there were changes in both the experimental design and the gear type, both had to be accommodated. Therefore, in 2014, CPUE was calculated as the variable of interest divided by the product of the net area of 3.0480 m^2 and the number of trawls (3). In 2015, the variable of interest was divided only by the area of the new net (4.8768 m^2). Because single trawls were run in 2015, there was no need to account for the number of trawls (Equation 3-2).

$$CPUE = \frac{n}{dtA} \qquad \text{Equation 3-2}$$

where n is the number of crabs or fish captured in a given treatment, d is the number of trawls pulled, and t is the number of stations per treatment, and A is the area of the opening of the trawl net.

For fish trophic dynamics, upon completion of data collection, each species was assigned to a trophic position based on information from the literature (i.e., a primary consumer, a secondary consumer or predator, or an opportunistic omnivore). With these three categories, a trophic structure was ascertained.

Several fish species biodiversity indices were also calculated. Species richness (*R*), was a simple count of the number of species in a given treatment. Because this index was strongly dependent on sampling effort, despite effort being accounted for in the CPUE calculation, Margalef's Index (*M*) was also calculated to include the total abundance of the sample size (Equation 3-3). To include something more intuitive and include sample sizes of individual species, the Simpson Inverse (λ^{-1}) was determined using Equation 3-4. Because the Simpson Inverse is weighted more to dominant species than to evenness, the Shannon-Weiner Index ($H`$) was also calculated, which is based on the weighted geometric mean of the proportional abundances of species types (Equation 3-5).

$$M = \frac{R-1}{ln(N)} \qquad \text{Equation 3-3}$$

$$\lambda^{-1} = \frac{1}{\left(\frac{\Sigma(n(n-1))}{N(N-1)}\right)} \qquad \text{Equation 3-4}$$

$$H` = -\sum n[ln(n)] \qquad \text{Equation 3-5}$$

where, R is species richness, n is individual species sample size, N is total sample size.

Variables of interest included blue crab life history stages, molting stages, and genders, and also fish diversity indices in order to accommodate two experimental designs: the 2014-2015 design to discern differences in CPUE between agriculture and non-agriculture treatments across time and the 2015 design to identify differences in variables across several percentages of adjacent agriculture. To accomplish this, general linear models (GLMs) were conducted for each data set. Both spatial and temporal parameters needed to be considered, so the three main effects in the model for the 2014-2015 data set included: year, season, and treatment. If there were significant effects from the main factors, it was also important to determine if those factors were dependent upon each other. Therefore, each interaction term was also considered: year × season, year × treatment, season × treatment, and year × season × treatment. The GLM for the 2015 experimental design was much simpler because it did not include a main effect of year. Thus, the two factors in the 2015 data GLM included only season and treatment, with the season × treatment interaction term. Finally, following analysis with a GLM, post-hoc pairwise comparisons were run using the Mann-Whitney U test to discern differences among levels of factors containing more than two levels.

The experimental design in 2015 also allowed for an opportunity to separate each treatment cell from a range of agricultural percentage to a single percentage within each grid cell. With this alternate definition of treatments, percent agriculture was compared with each crab variable using a Spearman's Rank correlation to identify trends or significance along the land use gradient.

CHAPTER 4

RESEARCH FINDINGS

4.1. Part I – Water quality relationship to land use and land cover in Blackbird Creek, Delaware

4.1.1. *2014-2015 Experimental Design: Water Quality*

The mean values of the water quality data in the 2014-2015 experimental design were all well below the DNREC and EPA recommended levels, with the exceptions of orthophosphate and turbidity (Table 4-1).

Table 4-1. Average values and standard error of water quality parameters for both 2014 and 2015 with recommended values outlined by various agencies.

Variable	Recommended Concentration	Mean Concentration	Standard Error
NH_3 (mg L^{-1})	< 1.90 (USEPA, 1989)	0.09	0.003
NO_3 (mg L^{-1})	< 10.00 (USEPA, 2001)	0.45	0.059
NO_2 (mg L^{-1})	< 1.000 (USEPA, 2001)	0.019	0.0011
PO_4^{3-} (mg L^{-1})	< 0.31 (USEPA, 2001)	0.42	0.011
Alkalinity (mg $CaCO_3$ L^{-1})	> 20.00 (USEPA, 1986)	87.76	1.857
Turbidity (NTU)	< 20.00 (UMRCC, 2001)	63.36	2.094
Temperature (°C)	--	22.14	0.417
Salinity (ppt)	--	6.04	0.191
pH	--	7.13	0.056
Dissolved Oxygen (mg L^{-1})	> 5.00 (DNREC, 2006)	6.02	0.121

According to the GLM, there were no significant differences between years or agriculture treatments for ammonia, orthophosphate, or temperature, but there were significant differences across seasons (Appendix 4-A). There was a significant increase in water temperature from the planting season to the growth season, followed by a significant reduction into the harvesting season (Figure 4-1). Temperature was also significantly lower in the harvesting season than the planting season ($U = 533.50$, $p < 0.001$). There was a significant decrease in turbidity from the planting to the growth season ($U = 820.50$, $p < 0.001$), but no change from the growth season to the harvesting season. There was a significant drop in orthophosphate concentration from the planting to the growth season ($U = 713.00$, $p < 0.001$), followed by an increase into the crop harvesting season ($U = 538.00$, $p < 0.001$). Ammonia concentration was significantly greater during the planting season than the growth season ($U = 906.00$, $p = 0.001$) and the harvesting season ($U = 886.50$, $p = 0.002$).

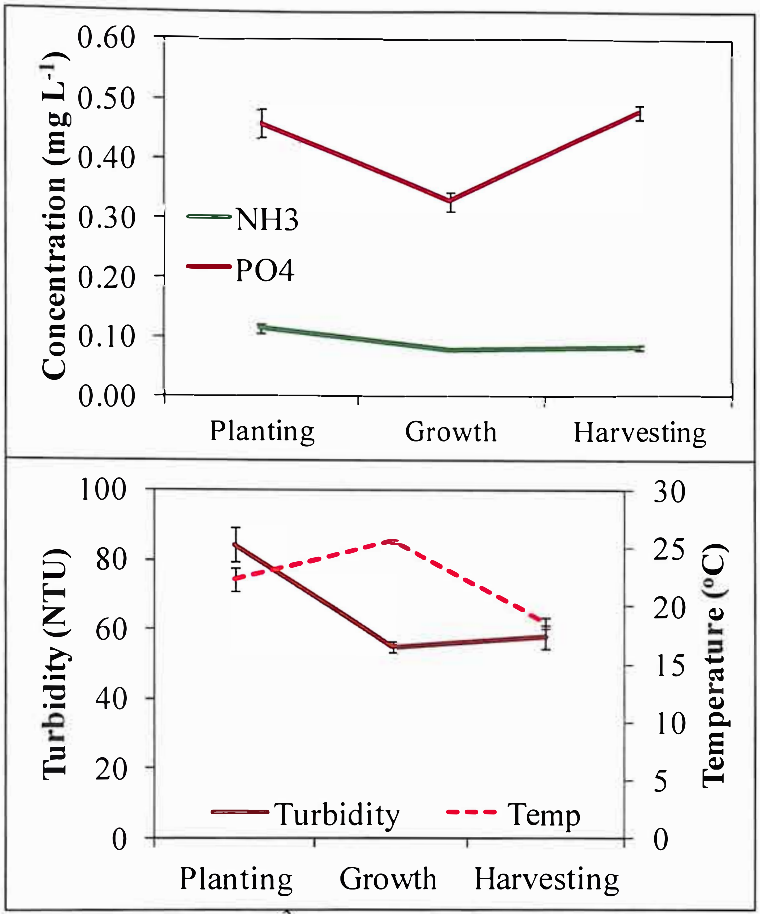

Figure 4-1. Mean levels of NH_3, PO_4^{3-}, temperature, and turbidity between agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November) for the 2014-2015 experimental design, ± 1 standard error.

According to the GLM, there were significant changes in nitrate concentration by year, season, and treatment, and in the year × treatment and year × season interaction terms (Appendix 4-A). There was a greater reduction in NO_3 concentration in agriculture treatments than non-agriculture sites from 2014 to 2015 ($F(1,96) = 4.912$; $p = 0.029$) (Figure 4-2).

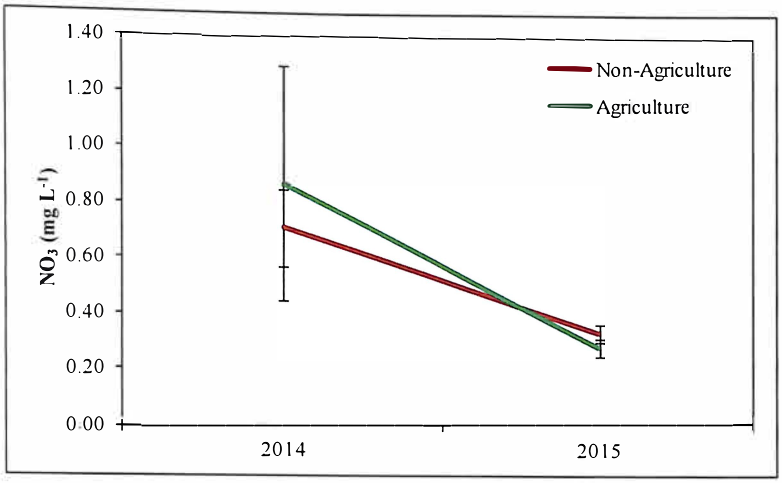

Figure 4-2. Mean nitrate concentration across years and adjacent land cover ± 1 standard error.

In 2014, there was a general decrease in nitrate concentration from the planting season through growth and the harvesting seasons ($U = 55.00$, $p = 0.002$), whereas in 2015, there was a significantly greater concentration in the growth season than the harvesting season ($U = 438.00$, $p = 0.001$) (Figure 4-3).

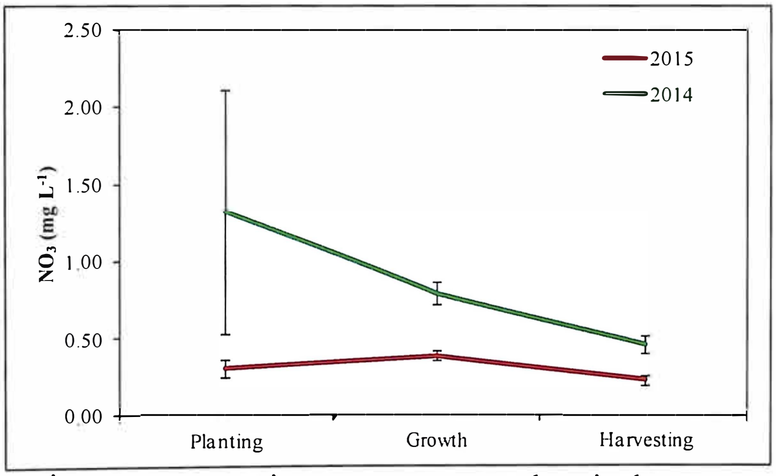

Figure 4-3. Mean nitrate concentration across years and agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November), ± 1 standard error.

Nitrite concentration varied across seasons and years, and within the season × year interaction, indicating that the two main effects were dependent on each other (Appendix 4-A). In both 2014 and 2015, there was a significant increase in concentration into the growth season ($U = 187.00$, $p < 0.001$), followed by a reduction into the harvesting season. In 2015, however, there was a significant reduction in concentration from the growth to harvesting seasons ($U = 44.50$, $p < 0.001$), and that reduction was greater than the reduction in the same time from in 2014 (Figure 4-4).

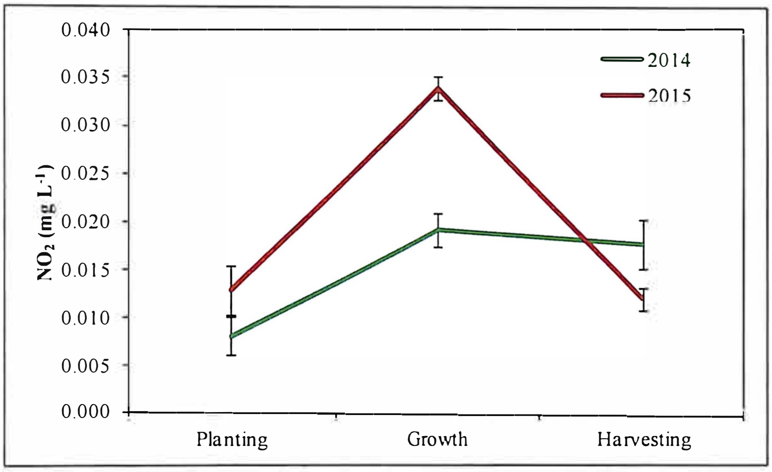

Figure 4-4. Mean nitrite concentration across years and agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November) ± 1 standard error.

Salinity varied across seasons and years, and within the season × year interaction, indicating that the two main effects were dependent on each other in order to ascertain the significance (Appendix 4-A). In 2015, there was a significant increase in salinity from the planting season to the growth season ($U = 120.50$, $p < 0.001$), but not in 2014 ($U = 72.00$, $p = 0.408$). For both seasons, there was a significant increase in salinity from the growth season to the crop harvesting season (2014: $U = 43.50$, $p < 0.001$; 2015 ($U = 438.00$, $p < 0.001$)), with a greater increase in 2015 (Figure 4-5).

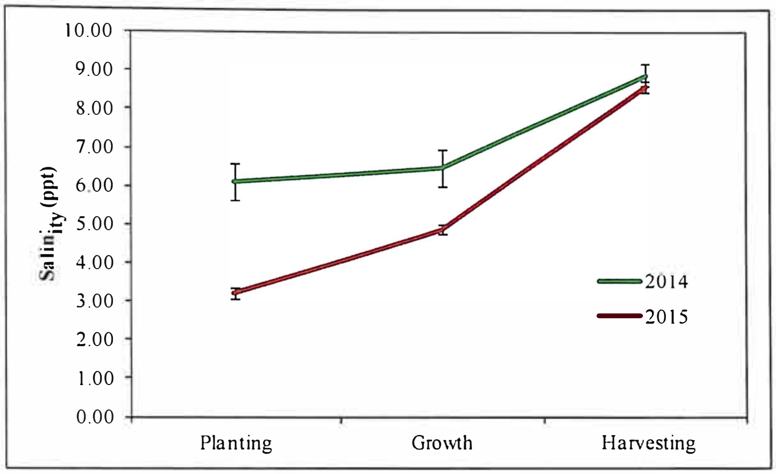

Figure 4-5. Mean salinity across years and agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November), ± 1 standard error.

There was a significant increase in alkalinity from the planting (82.20 mg $CaCO_3$ L^{-1} ± 3.214 SE) to the harvesting seasons (93.35 mg $CaCO_3$ L^{-1} ± 2.846 SE) ($U = 912.50$, $p = 0.003$), and significance in the year × season interaction term, indicating that the seasonal differences were dependent on the year (Appendix 4-A). Indeed, in 2014 there was a significant increase in alkalinity from the crop growth to the harvesting seasons ($U = 28.00$, $p < 0.001$), whereas in 2015, the opposite was true ($U = 584.50$, $p = 0.038$) (Figure 4-6).

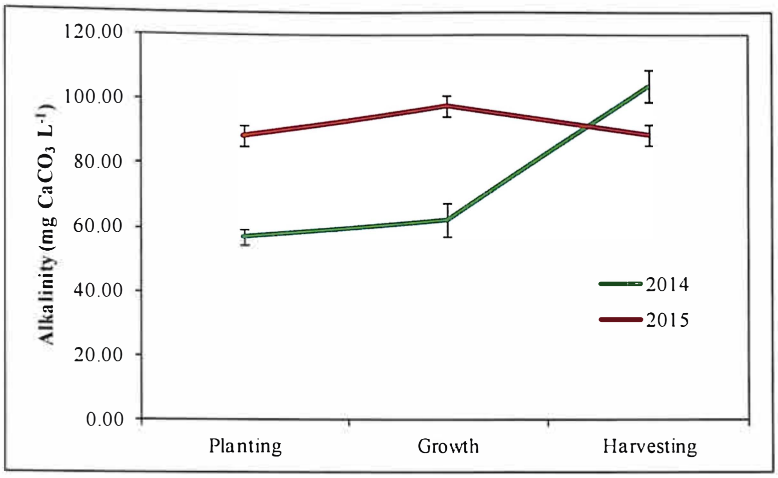

Figure 4-6. Mean alkalinity concentration across years and agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November), ± 1 standard error.

The GLM showed no main effects on pH between years, seasons, or agriculture treatments; there was however an interaction effect of season × treatment (Appendix 4-A), indicating that the two sources of variability are continent on each other to achieve a significant interaction. In non-agriculturally impacted areas, there was no significant difference in pH between the planting and harvesting seasons (U = 157.50, p = 0.013), and between the growth and harvesting seasons (U = 166.50, p = 0.031), increases in pH in both time stamps (Figure 4-7). At agriculture stations, pH was not significantly different between seasons.

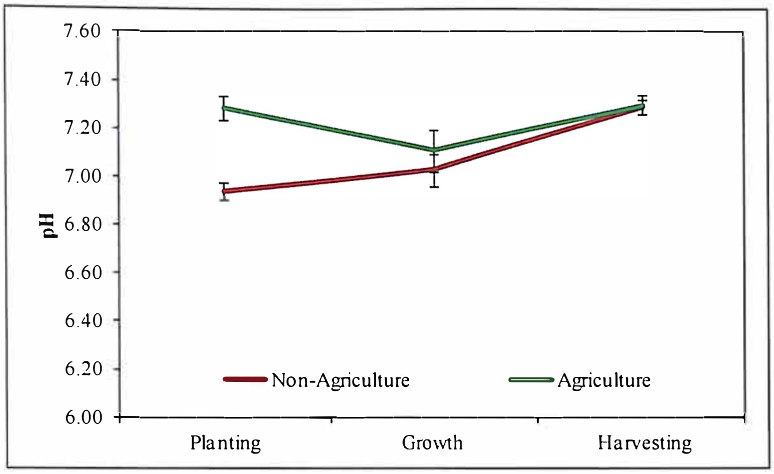

Figure 4-7. Mean pH values across agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November) and treatments ± 1 standard error.

Dissolved oxygen was significantly lower in 2014 (4.60 mg L^{-1} ± 0.192 SE) than 2015 (6.47 mg L^{-1} ± 0.121 SE) (Appendix 4-A), but there was also significance in the year × season interaction term, indicating that the yearly differences were dependent on season. In 2014, there was not a significant difference in dissolved oxygen between the growth and planting seasons ($U = 28.00$, $p = 0.496$), but there was a significant reduction in DO between the planting and harvesting seasons ($U = 8.00$, $p = 0.022$) and between the growth and harvesting seasons ($U = 79.00$, $p = 0.025$). In 2015, dissolved oxygen concentration during the planting and harvesting seasons were not significantly different ($U = 727.00$, $p = 0.603$), but during the growth season, the concentration was significantly lower than the other the planting ($U = 396.50$, $p < 0.001$) and harvesting seasons ($U = 275.50$, $p < 0.001$) (Figure 4-8).

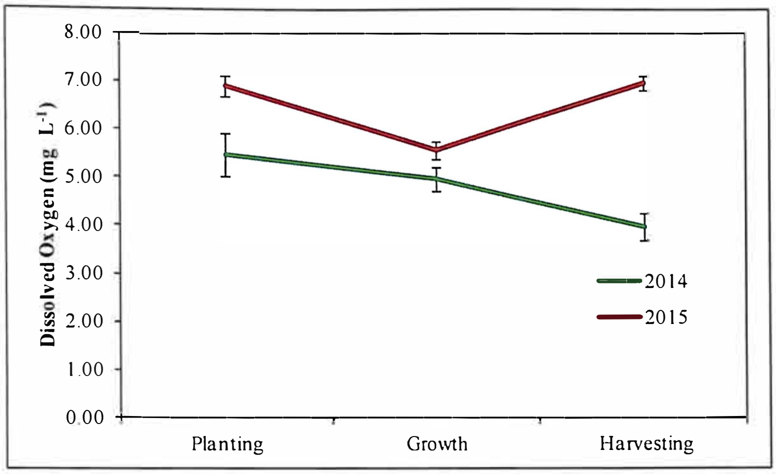

Figure 4-8. Mean dissolved oxygen concentration between years and agriculture (Planting: May-June; Growth: July-August; Harvesting: September-November) seasons ± 1 standard error in the 2014-2015 experimental design.

4.1.2 *Water Quality in 2015*

The GLM revealed no significant differences between treatments for any water quality variable (Appendix 4-B). There were, however, significant statistical differences between seasons for most water quality parameters, with the only exceptions of pH and NO_3 (Figure 4-9). There was a significant interaction of season × treatment for turbidity but no other variable (Appendix 4-B). The turbidity concentration in the 0%-10% and 30%-40% blocks was lower in the harvesting season than the growth season, whereas the other two blocks showed an increase in the same time frame (Figure 4-10).

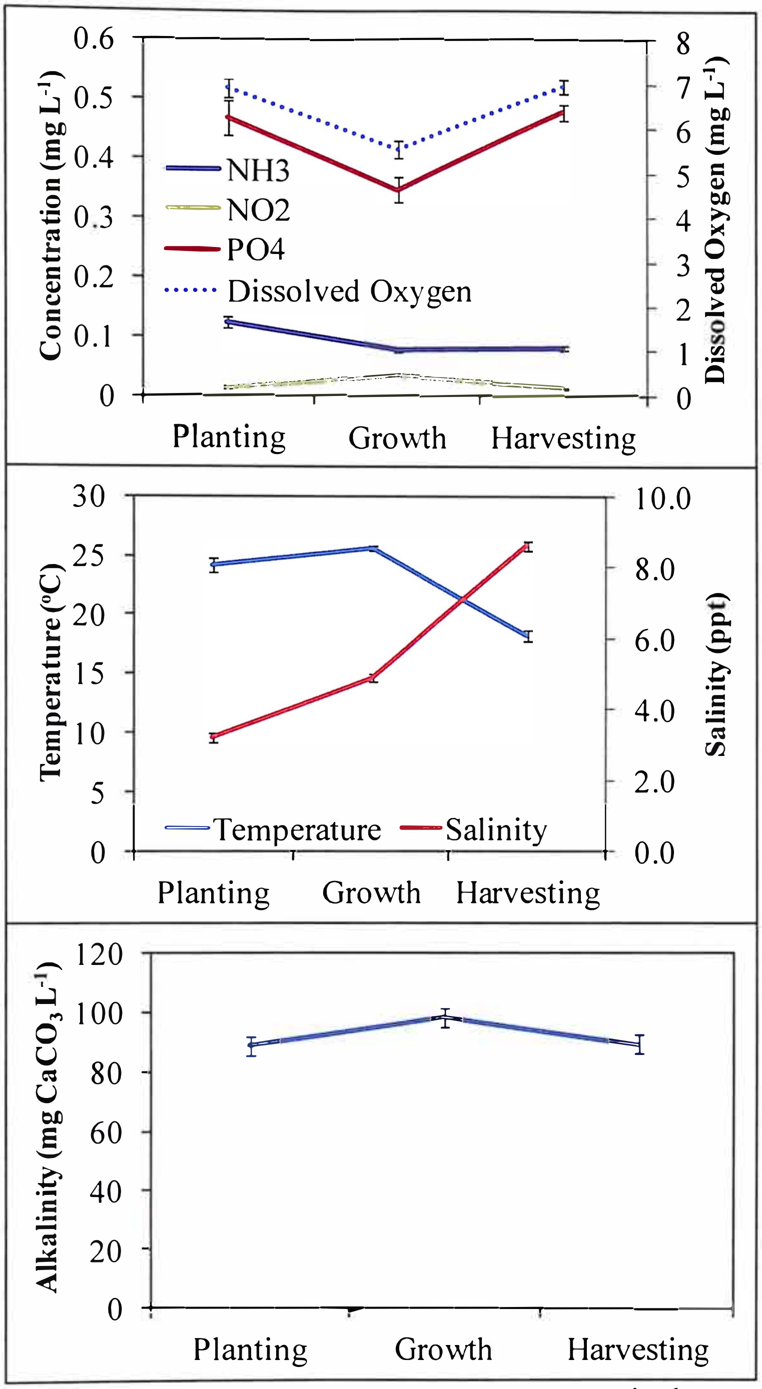

Figure 4-9. Mean values of water quality parameters across agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November) in the 2015 experimental design ± 1 standard error.

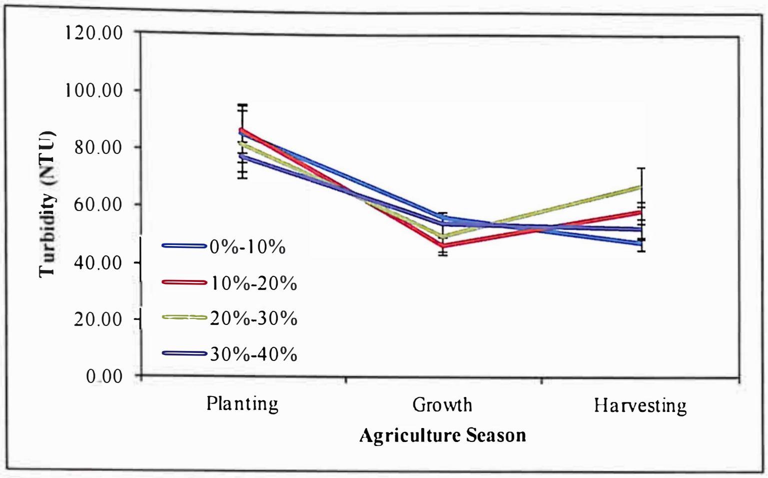

Figure 4-10. Mean turbidity across agriculture seasons (Planting: May-June; Growth: July-August; Harvesting: September-November) and treatments in the 2015 experimental design ± 1 standard error.

There were no significant correlations between percentage of agriculture and any water quality parameter, with the exception of alkalinity, which had a slightly negative correlation (Table 4-2).

Table 4-2. Spearman Rank correlations between percent agriculture and water quality parameters. Emboldened p-values represent a significant correlation.

Variable	Correlation Coefficient	Significance
NH_3	-0.017	0.853
NO_3	-0.113	0.254
NO_2	-0.047	0.629
PO_4^{3-}	0.003	0.977
Alkalinity	-0.183	**0.046**
Turbidity	0.081	0.382
Temperature	0.076	0.411
Salinity	-0.088	0.343
pH	0.038	0.710
Dissolved Oxygen	0.088	0.339

4.2. Part II – Land use and blue crab population dynamics, fish biodiversity, and fish trophic structure in Blackbird Creek, Delaware

There was a total capture of 59 blue crabs in pots in 2014 (0.97 crabs pot^{-1}) compared to 21 captures in 2015 (0.69 crabs pot^{-1}). In trawls, the total capture was much greater in both years. In 2014, 264 blue crabs were captured in trawls (29.7 crabs $trawl^{-1}$ m^{-2}) compared to 485 in 2015 (25.7 crabs $trawl^{-1}$ m^{-2}).

There were 1,231 total fish captures (144.7 fish $trawl^{-1}$ m^{-2}) in 2014 compared to 8,896 (615.4 fish $trawl^{-1}$ m^{-2}) in 2015. The top five most encountered species were the white perch (*Morone americana*), weakfish (*Cynoscion regalis*), Atlantic croaker (*Micropogonias undulatus*), hogchoker (*Trinectes maculatus*), and American eel (*Anguilla rostrata*), which made up over 92% of the catch. White perch, hogchoker, American eel, channel catfish (*Ictalurus punctatus*), and bay anchovy (*Anchoa mitchilli*) made up nearly 96% of the captures in 2015. Fourteen species were captured in 2014 compared to 17 the following year. In both years, the species most dominant, by far, was the white perch, which accounted for 55.8% and 74.1% of the total catch in 2014 and 2015, respectively. This species was encountered most for both experimental designs across years, seasons, and treatments.

4.2.1. *2014-2015 Experimental Design: Blue Crab Pots*

There were no significant differences between years or across treatments for any variables (Appendix 4-C). There was a significant difference in total capture of crabs ($F(2,28) = 5.752$; $p = 0.008$), total capture of adults ($F(2,28) = 8.790$; $p = 0.001$), and capture of both males ($F(2,28) = 5.296$; $p = 0.011$), and females ($F(2,28) = 4.497$; p =

0.020) per pot across seasons, with each variable showing greater catch in the harvesting season than the planting season (Figure 4-11; Appendix 4-C). There was also a significantly greater total catch in the growth season than the planting season (U = 537.50; p < 0.001), and a greater total catch in the harvesting season than the growth season (U = 54.00; p = 0.002). There was a significantly greater total catch of males in the growth season than the planting season (U = 78.00; p = 0.022), and a greater total catch of males in the harvesting season than the growth season (U = 8.50; p = 0.025). Total capture of adults was greater in the growth season than the planting season (U = 885.00; p < 0.001). There were no significant differences in any of the interaction terms for this data set (Appendix 4-C).

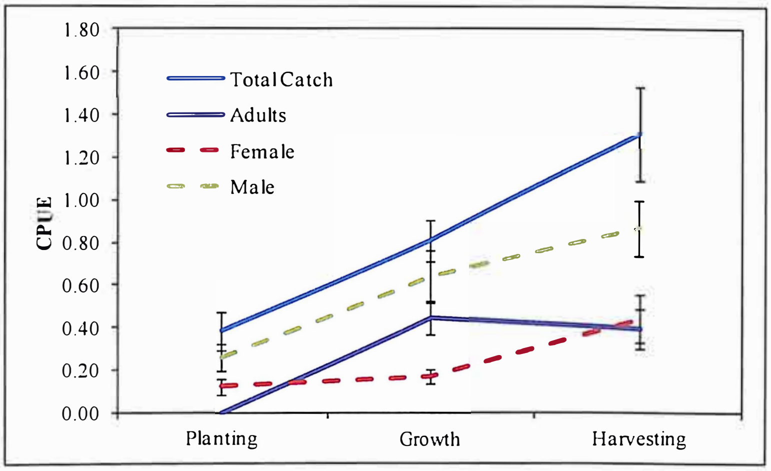

Figure 4-11. Mean total capture of blue crabs, mean total capture of adults, and mean total capture of males and females in pots (crabs trawl^{-1}) broken down by age class coupled with total capture ± 1 standard error.

4.2.2. *2014-2015 Experimental Design: Blue Crab Trawls*

See Appendix 4-D for complete results of the GLM. Total capture was significantly greater in the agriculture treatment (1.02 crabs $trawl^{-1}$ m^{-2} ± 0.158 SE) than the non-agriculture treatment (0.60 crabs $trawl^{-1}$ m^{-2} ± 0.131SE) ($F(1,14) = 5.049$; $p = 0.041$) (Appendix 4-D); there was also significance in the interaction term of year × season ($F(2,14) = 6.388$; $p = 0.025$), where in 2014, there was a greater total capture in the growth season than the planting season but no difference from capture in the harvesting season (Figure 4-12). In 2015, the greatest total capture per unit effort was in the planting season, which was greater than capture in the growth and harvesting seasons, which were not different from each other.

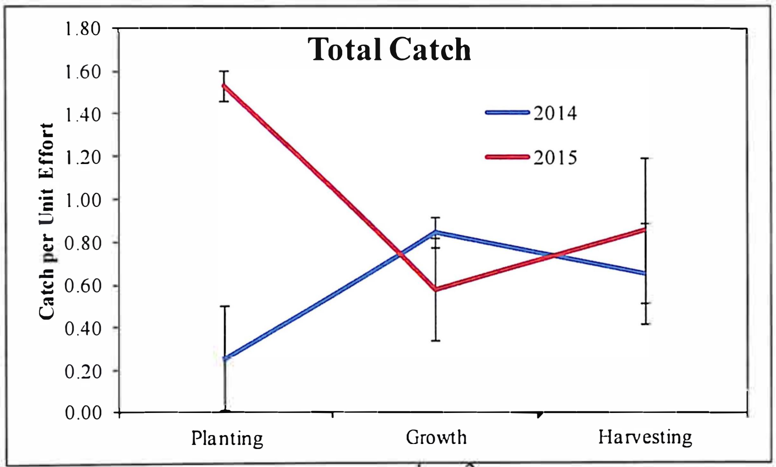

Figure 4-12. Mean catch of crabs (crabs $trawl^{-1}$ m^{-2}) across years and seasons ± 1 standard error.

There was a significant difference in total capture of recruit blue crabs (< 30 mm carapace width) across seasons ($F(2,14) = 5.773$; $p = 0.015$), where there were no captures in the planting season, and subsequent significantly greater captures in the growth ($U = 533.00$; $p < 0.001$), and the harvesting season ($U = 154.00$; $p = 0.012$).

There was no difference in recruitment capture across agriculture treatments or years, or in any interaction term (Figure 4-13; Appendix 4-D).

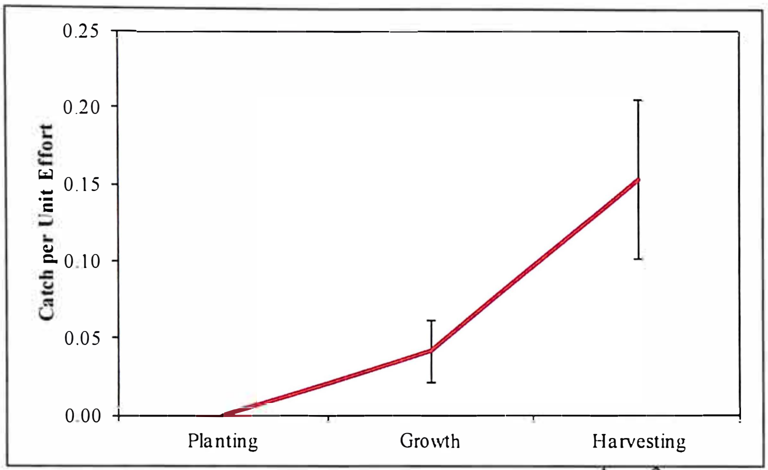

Figure 4-13. Mean capture of recruit blue crabs (crabs trawl^{-1} m^{-2}) across seasons ± 1 standard error.

Juvenile capture was significantly lower in 2014 (0.39 crabs trawl^{-1} m^{-2} ± 0.102 SE) compared to 2015 (0.67 crabs trawl^{-1} m^{-2} ± 0.162 SE) ($F(1,14) = 4.925$; $p = 0.044$). There was no significant difference between seasons for this variable (Appendix 4-D). However there was significance in the interaction term of year × season ($F(2,14) = 6.388$; $p = 0.011$), indicating that the effect of year was partially dependent on the effect of season (Appendix 4-D). In 2014, there was a lower catch per unit effort of juveniles during the planting season than the other two seasons, but in 2015, there was a greater capture in the planting season than the growth or harvesting seasons, and that there was a difference in catch in the growth season than the other two seasons (Figure 4-14).

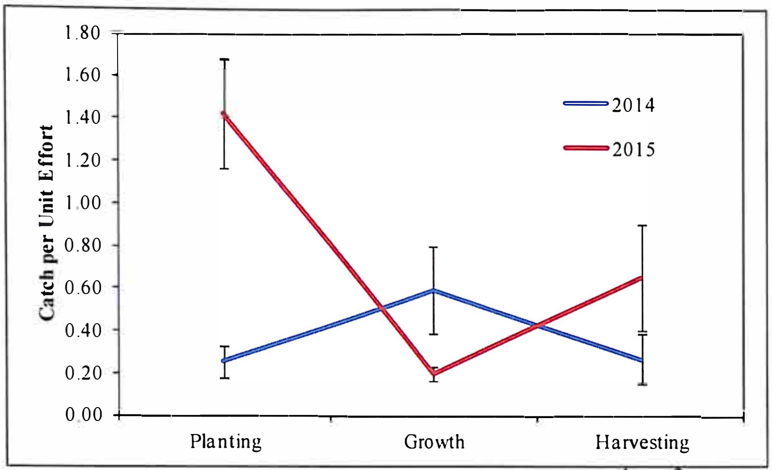

Figure 4-14. Mean capture of juvenile blue crabs (crabs trawl^{-1} m^{-2}) across seasons and years in the 2014-2015 experimental design ± 1 standard error.

For both adults and molting crabs, there was no significant difference in capture between years or treatments, nor was there significance in the interaction terms (Appendix 4-D). There was an effect of season, however. Capture per unit effort of adults ($F(2,14) = 4.176$; $p = 0.038$) was greater during the crop growth season than the other two seasons, in which there was no significant difference in adult capture (Figure 4-7). There was a significant difference in capture of molting crabs ($F(2,14) = 4.019$; $p = 0.042$) in the harvesting season than the other two seasons, when fewer were captured in trawls (Figure 4-15).

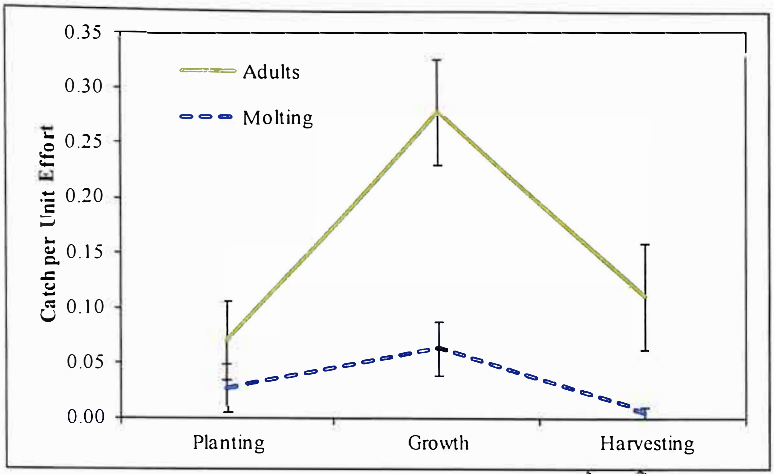

Figure 4-15. Mean capture of adult blue crabs (crabs $trawl^{-1}$ m^{-2}) and molting crabs across seasons in the 2014-2015 experimental design ± 1 standard error.

There was a significant difference in capture of females ($F(1,14) = 5.808$; $p = 0.030$) and non-molting crabs ($F(1,14) = 5.272$; $p = 0.038$) per unit effort between adjacent land uses, with greater captures recorded in areas used for agriculture than areas defined as forested wetland (Figure 4-16; Appendix 4-D).

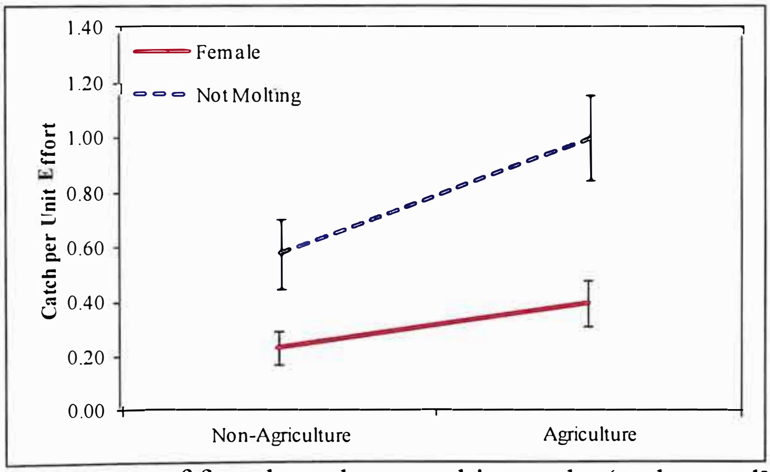

Figure 4-16. Mean capture of female and non-molting crabs (crabs $trawl^{-1}$ m^{-2}) between seasons in the 2014-2015 experimental design ± 1 standard error.

There was a statistically greater capture of non-molting crabs in trawls in 2015 (0.59 crabs $trawl^{-1}$ m^{-2} ± 0.025 SE) than 2014 (0.90 crabs $trawl^{-1}$ m^{-2} ± 0.142 SE) ($F(1,14) = 5.207$; $p = 0.039$). However, there was also significance in capture of non-molting crabs in the year × season interaction term, indicating that the yearly differences are dependent on seasonal variation ($F(2,14) = 11.413$; $p = 0.001$). In 2014, there was a decrease in non-molting crab capture between the planting season and the growth season, followed by a significant increase in capture from the growth to harvesting seasons. In 2015, there was an increase in capture between the planting and growth season, then a steady capture rate into the harvesting season (Figure 4-17).

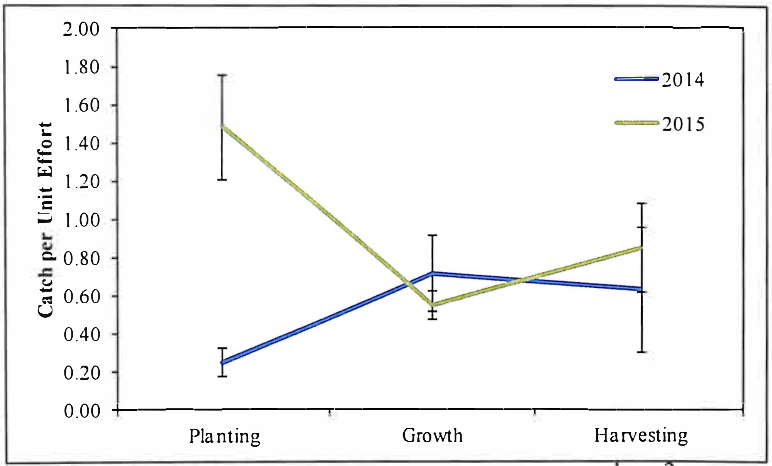

Figure 4-17. Mean capture of non-molting crabs (crabs $trawl^{-1}$ m^{-2}) across years and season in the 2014-2015 experimental design ± 1 standard error.

Finally, female crab capture was significant in the year × season interaction term ($F(2,14) = 11.413$; $p = 0.001$), but was not significant in either direct effect of year or season (Appendix 4-D), indicating that each of these effects are dependent on each other. In 2014, there was a decrease in female crab capture between the planting season and the growth season, followed by an increase in capture from the growth to harvesting seasons. In 2015, there was an increase in capture between the planting and growth season, then a steady capture rate into the harvesting season (Figure 4-18).

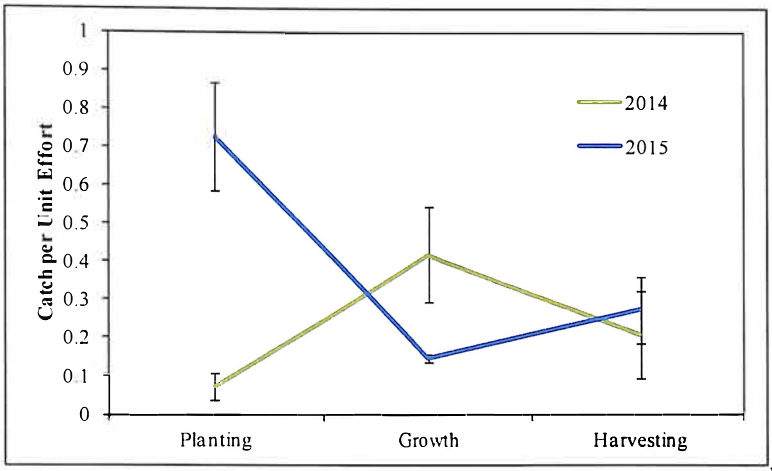

Figure 4-18. Mean catch per unit effort of female blue crabs (crabs trawl^{-1} m^{-2}) across years and seasons in the 2014-2015 experimental design ± 1 standard error.

4.2.3. *2014-2015 Experimental Design: Fish Trawls*

There were no significant differences in capture of primary consumers or opportunistic omnivores across years, seasons, agricultural land cover, or any interaction of these factors (Appendix 4-E). There was a significant decrease in capture per unit effort of secondary consumers from 2014 (30.6 fish trawl^{-1} m^{-2} ± 9.22 SE) to 2015 (8.9 fish trawl^{-1} m^{-2} ± 1.29 SE) ($F(1,14) = 6.550$; $p = 0.023$) but no significant changes according to agricultural season or treatment (Appendix 4-E).

There was a significant increase in total fish capture from 2014 (144.7 fish trawl^{-1} m^{-2} ± 18.04 SE) to 2015 (615.4 fish trawl^{-1} m^{-2} ± 156.05 SE). There was a significant difference in the species richness between seasons ($F(2,14) = 6.936$; $p = 0.008$)

(Appendix 4-F). Richness was significantly greater in the harvesting season than the planting season ($U = 1.00$, $p < 0.001$). Margalef's Index was significantly greater in the growth season than the planting season ($U = 7.00$, $p = 0.011$), in the growth season than the planting season ($U = 18.00$, $p = 0.015$), and in the harvesting season than the planting season ($U = 1.00$, $p < 0.001$) (Figure 4-11). There was a significant difference in the Shannon-Weiner Index across seasons as well ($F(2,14) = 4.416$; $p = 0.033$), with a significantly greater value occurring during the harvesting season than the growth season ($U = 22.00$, $p = 0.035$) (Figure 4-19). There were no other significant differences of other biodiversity indices based on year, season, agriculture land cover, or any interaction of these factors (Appendix 4-F).

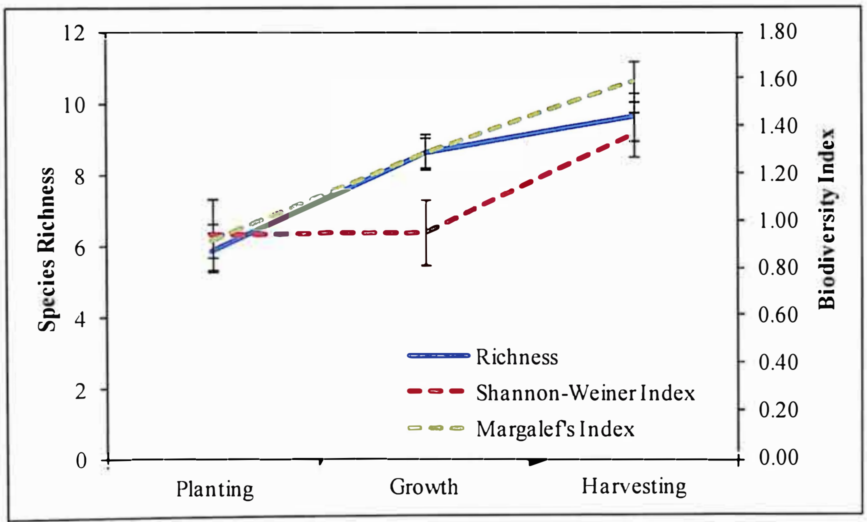

Figure 4-19. Response of Species Richness (primary y-axis) and the Shannon-Weiner and Margalef's Indices (secondary y-axis) across agriculture seasons in the 2014-2015 experimental design ± 1 standard error.

4.2.4. *2015 Experimental Design: Blue Crab Pots*

There were no significant differences in catch per unit effort of recruit blue crabs (< 30 mm carapace width), molting crabs, or either gender in pots in the 2015 experimental design (Appendix 4-G). Non-molting ($F(2,20) = 3.681$; $p = 0.044$) and adult crab ($F(2,20) = 8.998$; $p = 0.002$) captures were significantly different according to the GLM between agriculture seasons (Figure 4-20; Appendix 4-G). There was a lower capture of non-molting crabs during the planting season than the other two seasons, where capture rates of non-molting crabs were not statistically different from each other. No adult crabs were captured in the planting season in 2015, so capture of adults in the growth and harvesting seasons were greater. There was also a difference in capture between the growth season (0.36 crabs pot^{-1} ± 0.086 SE) and the harvesting season (0.17 crabs pot^{-1} ± 0.073 SE) (Figure 4-20).

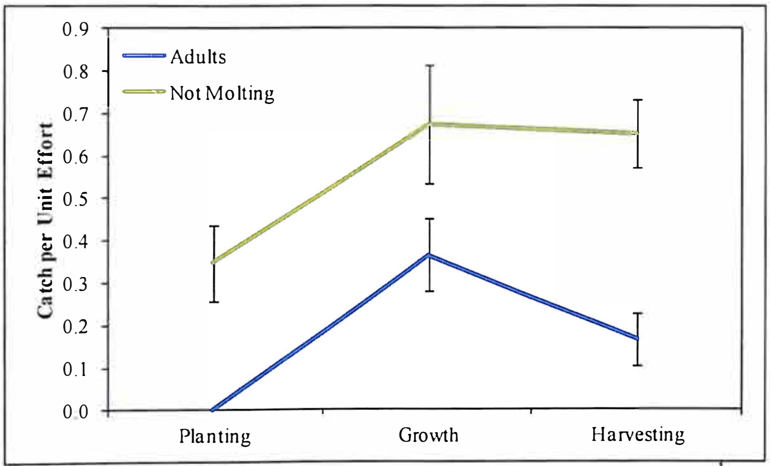

Figure 4-20. Mean catch per unit effort of adult blue crabs (crabs pot^{-1}) and non-molting crabs in pots in the 2015 experimental design ± 1 standard error.

There was a significant difference in total catch per unit effort of all crabs between the three agriculture seasons ($F(2,20) = 7.500$; $p = 0.004$), but not between agricultural treatments (Appendix 4-G). However, there was also statistical significance in the season × treatment interaction term ($F(6,20) = 3.419$; $p = 0.017$), indicating that the seasonal differences are dependent on the agricultural treatment. There was a greater catch per unit effort during the growth season than the planting season for crabs captured in both the 20%-30% agriculture block and the 30%-40% agriculture block (Figure 4-21). There was a lower capture rate during the harvesting season than the growth season and a greater capture rate than during the planting season in the 30%-40% agriculture block. There was a greater reduction in total catch in the 30%-40% block (1.3 crabs pot^{-1} ± 0.29 SE to 0.8 crabs pot^{-1} ± 0.25 SE) than the 20%-30% block (1.1 crabs pot^{-1} ± 0.08 SE to 0.9 crabs pot^{-1} ± 0.38 SE) between the growth and harvesting seasons. In the 10%-20% agriculture block, capture per unit effort was not different across the planting and growth seasons, but was greater in the harvesting season.

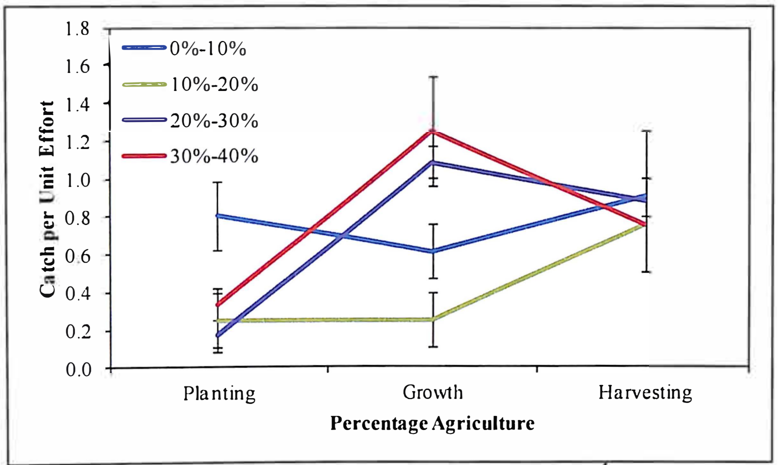

Figure 4-21. Mean total crab catch per unit effort (crabs pot^{-1}) across agriculture seasons and adjacent cropland percentage in the 2015 experimental design ± 1 standard error.

There were significant differences in juvenile capture between both seasons ($F(2,20) = 8.615$; $p = 0.002$) and treatments ($F(3,20) = 4.532$; $p = 0.014$), and also in the interaction term ($F(6,20) = 5.276$; $p = 0.002$), indicating that the main effects are dependent on each other (Appendix 4-G). In the 0%-10% agriculture block, there was a decrease in capture from the crop planting season to the growth season, followed by an increase in capture during the harvesting season (Figure 4-21). There was an increase in capture from the planting to growth season in the 30%-40% agriculture block, followed by a reduction in capture of juveniles in the harvesting season. For both the 10%-20% and 20%-30% blocks, there was a significant increase in capture per unit effort from the growth season to the harvesting season (Figure 4-22).

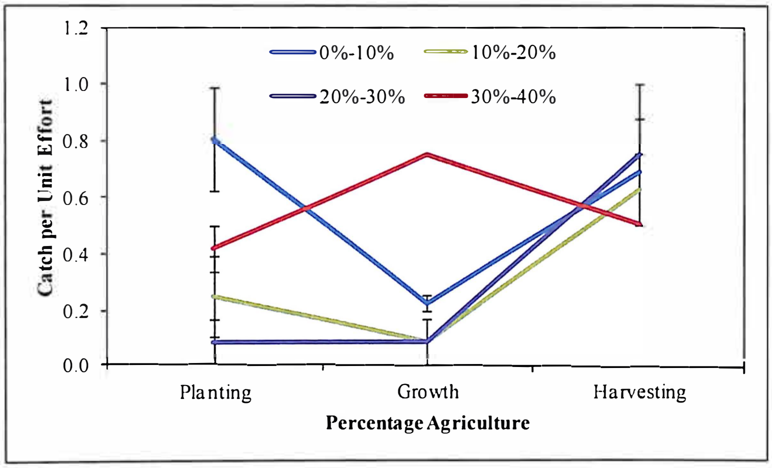

Figure 4-22. Mean catch per unit effort of juvenile blue crabs in pots (crabs pot^{-1}) in the 2015 experimental design ± 1 standard error.

The Spearman Rank correlations revealed a significant increase in total adults captured per unit effort with increased percentage of agriculture (Table 4-3). There were no other significant correlations.

Table 4-3. Spearman Rank correlations between percentage agriculture and blue crab pot capture per unit effort. Emboldened p-values represent significant correlations.

Variable (per unit effort)	Correlation Coefficient	Significance
Total catch	**0.204**	**0.066**
Total recruits	-0.084	0.455
Total juveniles	**0.082**	**0.463**
Total adults	0.251	**0.023**
Total molting crabs	**0.003**	**0.978**
Total non-molting crabs	0.209	0.060
Total males	**0.200**	**0.071**
Total females	0.167	0.133

4.2.5. *2015 Experimental Design: Blue Crab Trawls*

There were no significant differences in trawl catch per unit effort of recruit crabs (< 30 mm carapace width) or molting crabs across seasons or treatments, or within the interaction term in the 2015 experimental design (Appendix 4-H). There were seasonal differences in capture per unit effort of adults ($F(2,20) = 4.023$; $p = 0.034$) and male crabs ($F(2,20) = 6.727$; $p = 0.006$) (Figure 4-23). A greater number of males were captured in the planting season than the other two seasons, and there was no difference in catch of males across the growth and harvesting seasons. There was a greater catch per unit effort of adults during the growth season than the other two seasons, and no difference in adult capture between the planting and harvesting seasons.

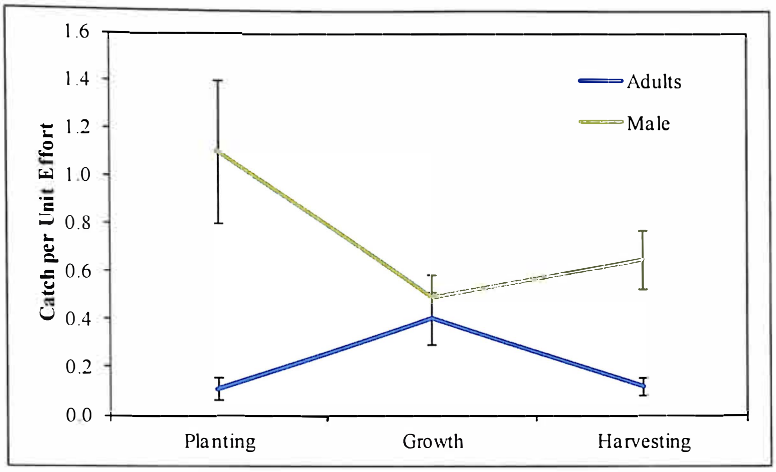

Figure 4-23. Mean capture per unit effort of adult crabs and male crabs in trawls (crabs trawl^{-1} m^{-2}) across agriculture seasons in the 2015 experimental design ± 1 standard error.

There were significant differences across seasons and treatments, and in the interaction term, in total catch ($F(6,20) = 4.910$; $p = 0.003$), juvenile capture ($F(6,20) = 4.333$; $p = 0.006$), capture of non-molting crabs ($F(6,20) = 5.179$; $p = 0.002$), and female catch ($F(6,20) = 9.562$; $p < 0.001$) (Appendix 4-H). Because there was significance in the interactions, the capture rates by season and treatment are dependent upon each other. For total catch, there was an increase in the 10%-20% agriculture from the planting to the growth seasons (Figure 4-24). There was a reduction in total catch from the planting to growth seasons in both the 20%-30% and 30%-40% agriculture blocks (Figure 4-24). For total capture of females, there was an increase in the 10%-20% agriculture from the planting to the growth seasons. There was a reduction in total female catch from the planting to growth seasons in both the 20%-30% and 30%-40% agriculture blocks (Figure 4-24). For capture of non-molting crabs, there was an increase in the 10%-20% agriculture from the planting to the growth seasons. There was a reduction in total catch of non-molting crabs from the planting to growth seasons in both the 20%-30% and 30%-40% agriculture blocks (Figure 4-24). For juvenile catch per unit effort, there was a reduction from the planting to growth seasons in both the 20%-30% and 30%-40% agriculture blocks, and the reduction was greater in the 30%-40% agriculture treatment (Figure 4-24).

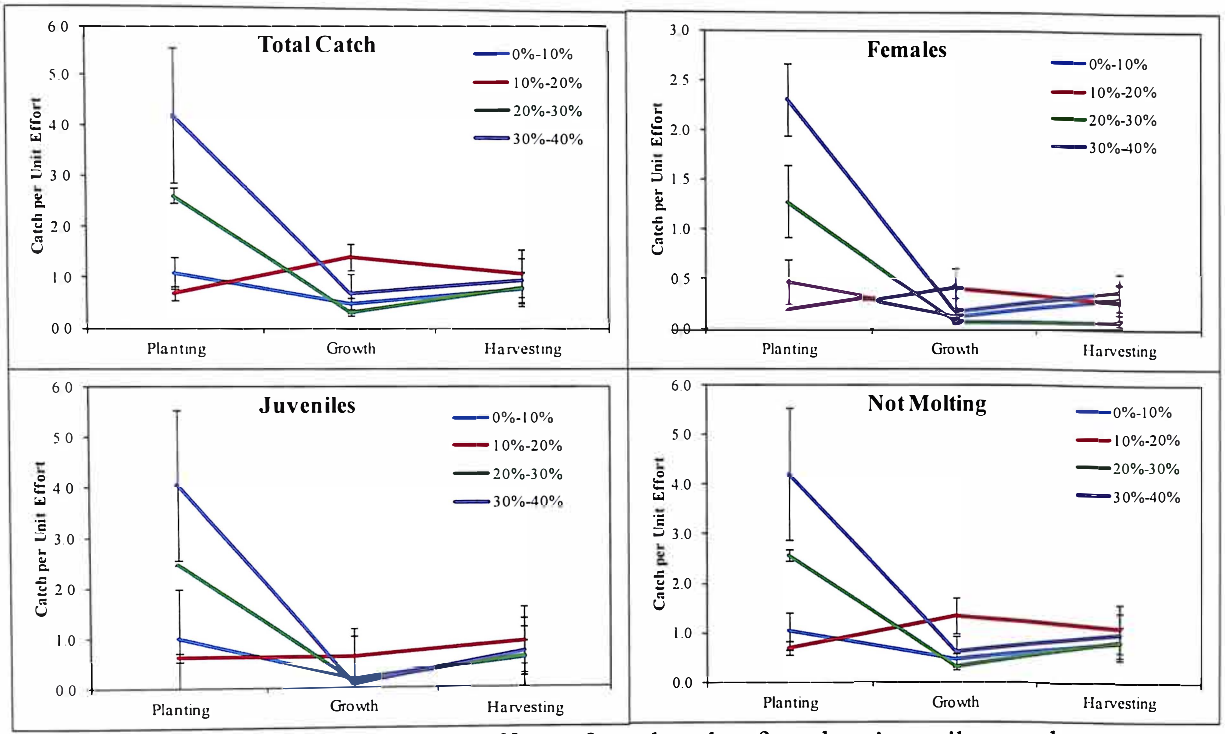

Figure 4-24. Mean catch per unit effort of total crabs, females, juveniles, and non-molting crabs in trawls (crabs trawl^{-1} m^{-2}) across seasons and percent agriculture treatments ± 1 standard error.

There were no significant correlations between percentage of adjacent agricultural land and crab catch per unit effort in trawls in 2015 (Table 4-4).

Table 4-4. Spearman Rank correlations between percentage agriculture and blue crab capture per unit effort. Emboldened p-values represent significant correlations.

Variable (per unit effort)	Correlation Coefficient	Significance
Total catch	**-0.006**	**0.959**
Total recruits	0.022	0.848
Total juveniles	**-0.020**	**0.862**
Total adults	0.038	0.738
Total molting crabs	**0.100**	**0.379**
Total non-molting crabs	-0.058	0.610
Total males	**-0.058**	**0.610**
Total females	-0.058	0.610

4.2.6. *2015 Experimental Design: Fish Trawls*

There were no significant differences between agriculture seasons or adjacent land cover treatment for primary consumer or opportunistic omnivore capture per unit effort (Appendix 4-I). There was a significant difference between treatments in primary consumer capture per unit effort according to the GLM ($F(3,20) = 5.354$; $p = 0.007$) (Figure 4-24), with the greatest catch occurring in the 0%-10% agriculture blocks. The Mann-Whitney U tests showed significant differences between the 0%-10% block and the 10%-20% block ($U = 12.00, p = 0.038$), the 20%-30% ($U = 6.00, p = 0.005$), and the 30%-40% block ($U = 1.00, p < 0.001$). There was also a significant difference in primary consumer capture between the 10%-20% block and the 20%-30% block ($U = 12.00, p = 0.038$), with a general reduction in capture (Figure 4-25).

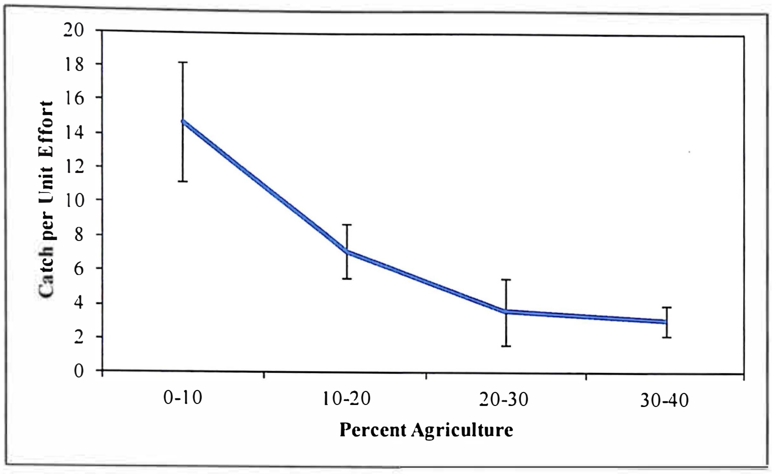

Figure 4-25. Response of catch per unit effort of primary consumers (fish trawl^{-1} m^{-2}) across agricultural treatments in the 2015 experimental design ± 1 standard error.

According to the GLM, there were no significant differences in total capture per unit effort, the Simpson Inverse, or the Shannon-Weiner Index across agriculture seasons and land cover treatments (Appendix 4-J). Species richness (F(2,20) = 4.583; p = 0.023) and Margalef's Index (F(2,20) = 8.426; p = 0.002) were significantly different across seasons. Both measures were also significant across treatments: richness F(2,20) = 3.406; p = 0.038 and Margalef's Indix F(2,20) = 3.335; p = 0.040, but not for the interaction term (Figure 4-26; Appendix 4-J).

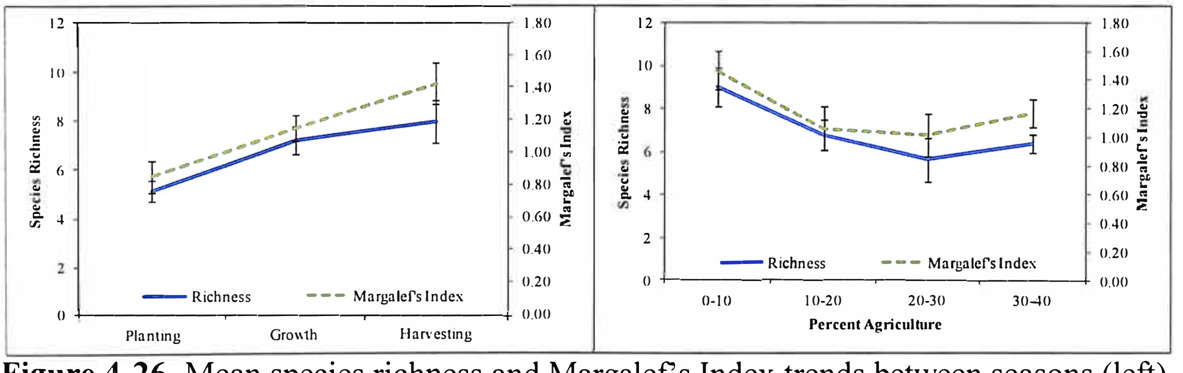

Figure 4-26. Mean species richness and Margalef's Index trends between seasons (left)

and percent adjacent agricultural land covers (right) in the 2015 experimental design ± 1 standard error.

The Spearman Rank correlations for 2015 data revealed significant inverse relationships between percent agriculture and primary consumers (p = 0.038), catch of Bay anchovy (*p* = 0.015), and capture of Striped bass (p = 0.033). There were significant direct correlations between percent agriculture and Margalef's Index (p = 0.048) and capture of Brown bullhead (p = 0.009) (Table 4-5).

Table 4-5. Spearman Rank correlations across agriculture percentage throughout Blackbird Creek in 2015. Emboldened values represent significance ($p < 0.05$).

Variable	Correlation Coefficient	Significance
Primary consumers	**-0.232**	**0.038**
Secondary consumers	0.023	0.841
Opportunistic omnivores	**0.214**	**0.056**
American eel	.0109	0.336
Channel catfish	**0.141**	**0.213**
Common carp	0.067	0.558
Hogchoker	**-0.127**	**0.262**
White perch	0.187	0.096
Weakfish	**-0.167**	**0.139**
Atlantic croaker	-0.046	0.687
White catfish	**0.141**	**0.211**
Bay anchovy	-0.271	**0.015**
Striped bass	**-0.238**	**0.033**
Black drum	-0.110	0.329
Brown bullhead	**0.288**	**0.009**
Spot	-0.175	0.119
Naked goby	**0.154**	**0.173**
Alewife	-0.114	0.313
Silver perch	**-0.134**	**0.235**
Total catch (N)	0.156	0.166
Species richness (R)	**-0.095**	**0.401**
Simpson Inverse (*1/λ*)	-0.175	0.118
Shannon-Weiner Index (*H'*)	**-0.143**	**0.205**
Margalef's Index (*M*)	-0.221	**0.048**

CHAPTER 5

GENERAL CONCLUSIONS AND FUTURE RESEARCH

5.1. Part I – Water quality relationship to land use and land cover in Blackbird Creek, Delaware

Generally, there were no differences between treatments for any of the water quality parameters that were studied in Blackbird Creek. Variability in temperature followed predictable seasonal patterns, and was not any different throughout the creek on any given date. This was expected because the portion of Blackbird Creek that was investigated was not in an area with significant amounts of shade from trees or buildings. Thus, any temperature changes were accountable to environmental changes.

Salinity throughout Blackbird Creek showed predictable trends over time. Again, there was no difference between sites, but there were seasonal changes similar to previous work. Summer months have characteristically low rain and high temperatures in the Mid-Atlantic States. The tidal regime was also important. If the tide was on its way in from the Delaware Bay, there was a greater likelihood that salinity would be greater due to the source of the water, especially if the tide had been incoming for several hours.

Turbidity throughout the waterway was extremely high, with extremes in 2014 on 19 September (8 NTU) and 06 October (167 NTU) and 04 November (32 NTU) and 14 May (222 NTU) in 2015. While there are no standards set forth by regulatory agencies to maintain the turbidity levels, studies have shown that values such as these can be detrimental to the ecosystem (Cloern, 1987). For example, fish that rely on sight and speed for prey capture may flee highly turbid water in order to find a food source. Also, high turbidity can reduce light penetration into the water to the point that aquatic primary producers cannot photosynthesize for lack of light energy. These high turbidity values were likely a function of the nature of a tidal creek and not reflective of adjacent agricultural land. Modeling the turbidity in these ecosystems is challenging due to the complicated hydrodynamics and rheological behavior of the soft mud which can dissociate easily under varying conditions (Uncles *et al.*, 2006; Dyer *et al.*, 2004). High rainfall, for example, can cause mud flats to break up and move into the waterway, especially during low tide. The changing environmental parameters of air temperature, rainfall, and wind may account for the significant differences in turbidity between seasons. However, it cannot be ignored that the turbidity was higher during the planting season – that in which there may have been considerable tilling of agriculture fields which loosened soil particles and increased the likelihood of the soil to runoff into the waterway.

The dissolved oxygen concentration in the creek was at appropriate levels according to the minimum acceptable standard to minimize growth effects of fish based on 5.0 mg L^{-1} established by the state of Delaware (DNREC, 2006) or 4.8 mg L^{-1} by the Environmental Protection Agency (USEPA, 2000). There were a few instances when the levels were indeed too low based on this standard. It is likely that this is primarily based on the tidal structure at the time of collection. Nearly all of the measurements of lower dissolved oxygen were recorded at or near low slack tide, when movement of water was minimal. The extreme lows were recorded when readings were taken at low tide and mid-day during the peak summer months. Higher temperature water is not capable of holding as much dissolved oxygen as cooler water and standing water does not afford the opportunity of direct injection of oxygen from the air-water interface via wave action (Falkowski and Raven, 2007). However, most readings were taken when the tide was in flux, thus providing for sufficient ecological DO levels.

The buffering capacity of Blackbird Creek was high. The pH fluctuated little throughout the entire study period, and was an appropriate brackish water level, with a mean of 7.18 ± 0.3 SD. The high alkalinity values throughout the watershed were responsible for this, with an average of 87.76 mg L^{-1} $CaCO_3$ ± 23.71 SD, suitable to sustain aquatic life and retain the pH levels between agricultural seasons and treatments.

While it was likely that most of the contribution to alkalinity levels is from the carbonate and bicarbonate levels, some of the contribution may be from phosphates (Millero, 1996). Blackbird Creek had elevated levels of orthophosphate (DNREC, 2006), where the maximum total phosphorous concentration in the system should not exceed 0.20 mg L^{-1}. The data collected in this project revealed orthophosphate levels between 0.16 mg L^{-1} and 1.02 mg L^{-1}. Orthophosphate (or simply phosphate) is the simplest form of phosphorous and is most readily available for biological use and typically enters the waterway by way of runoff of eroded soil from land. While the values were elevated, there was no suggestion in the data that phosphate concentrations were influenced by agricultural practices. There were only significant seasonal differences, which represent typical phosphorous cycling, with a reduction during the growth season due to plant uptake followed by an increase in aquatic phosphate in the harvesting season when plants die and phosphate is remineralized into the soil (Correll, 1998). The elevated concentrations were possibly evident due to the nature of tidal systems in general, whereby incoming and outgoing tides continually erode the creek channel, which could be responsible for both increased turbidity and, by extension, increased phosphorous from upstream shorelines.

The surface water nitrogen parameters that were included in this project were nitrate, nitrite, and ammonia. Nitrite concentrations proved to be extremely low for most of the sampling season in both years and across all treatments and never measured greater than 0.08 mg L^{-1}. Given that the acceptable level from the EPA is 1.0 mg NO_2 L^{-1}, there was no concern with the nitrite concentrations in Blackbird Creek.

Nitrate concentrations were also well below the EPA-designated maximum contaminant level of 10 mg L^{-1} (USEPA, 2013). The highest recorded level in Blackbird Creek was 7.53 mg L^{-1}, and occurred on the second sampling date in 2014. At this particular site, there was no riparian buffer in place to prevent runoff of contaminants – the only known part of Blackbird Creek where this was so. Previous work in the creek by Roeske (2014) suggested that a crop field near the location where this sample was extracted grew corn in 2013. In 2014, the field ran fallow. It may be suggested that legacy nutrients which were not used by crops in 2013 were then able to run into the waterway early in the season. Never again after this sampling date were nitrate values greater than 1.41 mg L^{-1}. There was no effect of percent agriculture in the 2015 experimental design, but there was a significant difference between the two treatments (agriculture versus non-agriculture) for the 2014-2015 experimental design. There was also an interaction between the treatments and years. It is possible that this is due to the single aforementioned elevated data point at the corn plot in early 2014. Nevertheless, it cannot be ignored that there were increased nitrate levels which proved to be significant between cropland and forested wetland.

The final parameter of interest was ammonia. The levels were low (0.09 mg L^{-1} ± 0.043 SD). There is no EPA-designated standard contaminant level for ammonia, but it was recommended by the agency in 2013 that 1.9 mg L^{-1} should be the maximum level to sustain aquatic life (USEPA, 2013). There was a significant difference in ammonia levels between seasons for both data sets, suggesting that there may have been some impact on the parameter by agricultural use. However, because there is no difference between agriculture treatments, this was unlikely.

The seasonal variability of the nitrogen parameters in general may be more likely attributable to tidal creek geo-hydrodynamics and microbial nitrification and denitrification processes. Concomitant research in Blackbird Creek identified several nitrogen-fixing bacteria that influence nitrogen species concentrations (Ozbay *et al.*, 2014). Several studies have shown that microbial concentrations and associated nitrogen-fixation processes fluctuate seasonally (i.e., Biesboer, 1984; Ravikumar *et al.*, 2012). Bacteria tend to be greatest in number at the height of the growing season. Because the population is increased, there is greater potential for mediating the nitrification and denitrification processes and reducing concentrations of ammonia and nitrate in surface waters, especially at the soil-water interface. Thus, even if there were some excess runoff from cropland (which was not proven in statistical analysis), much of that would be taken up by these bacteria for their consumptive and regulatory processes, a phenomenon would might explain the seasonal variation in nitrogen variables in Blackbird Creek without any significance between treatments.

Based on this research, Blackbird Creek does not appear to be influenced by the agriculture land cover. In only one instance was there a water quality parameter that approached the EPA maximum recommended level (nitrate), but that signal faded quickly. Orthophosphate values were greater than the EPA-recommended maximum, but it is unlikely that this is due to agricultural land use because there were no differences between the treatments in either of the experimental designs. It is more likely that the phosphate was high due to tidal scouring, a natural component of a tidal creek.

Turbidity values were elevated in the system, but that was to be expected for a Mid-Atlantic tidal creek and was probably not attributable to erosion of soil from modified land. Indeed, this reflects the results from the National Estuarine Research Reserve data synthesis report, where it was shown that in areas of intermediate salinity, turbidity was generally elevated (Brush *et al.*, 2007).

All data suggest that the riparian buffer system in Blackbird Creek appeared to be working efficiently as an appropriate nutrient management practice by disallowing runoff of contaminants into the waterway.

5.2. Land use and blue crab population dynamics, fish biodiversity, and fish trophic dynamics in Blackbird Creek, Delaware

There were seasonal differences in catch per unit effort of total crab catch, blue crab recruits, blue crab adults, non-molting crabs, and male blue crabs, without assumptions of gear type. While it may be tempting to suggest that these differences do indeed represent the planting, growth, and harvest seasons of agricultural practice, this is more likely a simple function of life history of the crabs, which is dependent on temperature-driven seasons (spring, summer, autumn) rather than nutrient-driven seasons (planting, growth, harvesting). Blue crabs generally mate between May and October (Williams, 1984) in low-salinity lower regions of upper estuaries (Hill *et al.*, 1989). Female blue crabs then migrate to high-salinity waters and spawn. The larvae grow through their first several zoeal stages offshore then return to the mouth of the estuary as megalopae where there is abundant food and predator avoidance opportunity and where they can transition to the first stage crabs while becoming benthic. Growth and maturation proceed through a series of molts which become less frequent with increasing size (Van Engel, 1958). Therefore, it is most likely that more juveniles are captured throughout the beginning of the sampling season (i.e., planting season), followed by adults and, indeed, this was what was recorded (Figures 4-11; 4-15; 4-20; 4-22). At the end of the season, young-of-year recruits then become prominent due to blue crab life history and not necessarily due to increased availability of nutrients at any particular part of the season. This is showcased further by the fact that there were no significant changes in nutrient concentrations over time, as evidenced in Chapter 3.

Seasonal changes in fish biodiversity were evident as well. There were increases in Richness, the Margalef's Index, and the Shannon-Weiner Index. Blackbird Creek is a nursery ecosystem with several species of anadromous fish entering the system throughout the spring and summer in order to spawn. This is almost certainly the explanation for these increases through the season and is not dependent on nutrient runoff from adjacent croplands. Indeed, within the watershed, there were no differences between treatments for any of the biodiversity indices that were calculated. Similar seasonal changes in the Simpson Inverse were not recorded because that index is weighted more to dominant species than to general evenness. Therefore, with a handful of dominant species that did not change between seasons (i.e., American eel, white perch, hogchoker), the Simpson Inverse was not changed.

There were some interesting trends between 2014 and 2015 with these data, especially in the interaction term coupled with seasonal variation. There is no evidence available to suggest that fertilizer application increased between years or that there were increases in the nutrient signals between years (Appendix 4-A). There were significance differences in the model for total catch, total juvenile capture, capture of non-molting crabs, and female catch, where in 2014, there was an increase in catch per unit effort of each of these categories from the planting to growth seasons, followed by a reduction in catch per unit effort into the harvesting season. There was an opposite trend in 2015 (Figures 4-12; 4-14; 4-17; 4-18). This is most likely just blue crab population inter-annual variability. Unfortunately, with only two years of data, there is not enough evidence to make assumptions of long-term trends. Similarly, it is difficult to ascertain why there were differences in secondary consumers between years, and is most likely attributable to typical variability of temperature and salinity (i.e., Kimmerer *et al.*, 2001).

There was a significantly greater catch per unit effort of female blue crabs between treatments in the 2014-2015 experimental design. Again, it is tempting to suppose that these are truly a function of the defined treatments of non-agriculture and agriculture. However, it is important to note the life history of the animals and couple that information with where the actual sites were identified. The non-agriculture sites were located primarily in the lower part of the tidal creek near the mouth of the Bay while the agriculture sites were upstream, beyond the Stave Landing boat launch. Because of the nature of GLMs, data for these two treatments are pooled to include everything through all seasons and both years, which would include variation in the total numbers of crabs caught, number of females caught, and capture of non-molting crabs. To consider the treatment without factoring in seasonal changes is not sufficient for analysis. Indeed, in the 2015 experimental design, there was significance in the season × treatment interaction terms for total catch, non-molting animals, females, and juveniles, all with records of drops in capture from the planting season to the growth seasons in the two treatments with the greatest percentage of adjacent agricultural land cover, which may indicate that there was a more suitable habitat for crabs in these categories under these treatments. However, further data collection and analysis is necessary.

Based on the above study, it was concluded that, while there were variations in blue crab ecological parameters throughout time and space, it is unlikely that these variations are the effects of agricultural use land use in Blackbird Creek. The creek represents constant tidal changes that impact temperature, salinity, turbidity, and some water chemistry, and those changes are represented in blue crab abundance, life history stage, molting stage, and gender, and fish biodiversity. The daily fluctuation of the tide is partially responsible for the effects seen in the analysis, as are natural fluctuations in various parameters over seasons. Blackbird Creek is lined with marsh grasses and forested areas, which act as filters that intercept nutrients that may be running off from croplands. It is unknown how efficient these riparian buffers are because there is no information on how much fertilizer is placed onto the fields. Nonetheless, there is no evidence in this study to suggest that blue crab and fish population dynamics are being influenced by agriculture in this watershed.

5.3. General conclusions and future research

Rivers are increasingly being studied at landscape scales, both as ecosystems in their own rights (i.e., Wiens, 1989), and as ecosystems that are impacted by their adjacent land uses within the watershed (Townsend *et al.*, 2003). It has been apparent for the past few decades that human activity at the landscape scale is a principal threat to the ecological integrity of aquatic ecosystems, influencing food web dynamics, water quality, and habitat availability (Townsend *et al.*, 2003). Blackbird Creek represents a watershed with a range of land uses, including agricultural cover throughout much of the catchment.

Based on the percentage of agricultural cover and other pressures afforded by human impact, there should be a reduction in the overall biological integrity of the ecosystem (Allan, 2004; Norris and Thomas, 1999; Figure 5-1). These relationships are typically based on the sensitivity of the response variables (in the case of this study, these variables include water quality, blue crab population dynamics, and fish species biodiversity). However, based on the data collected and analyzed for this project, there does not appear to have been any significant impact by humans on the aquatic structure of Blackbird Creek.

These results are perhaps attributable to a complex riparian buffer network throughout the watershed and the nature of the tidal creek ecosystem. Unfortunately, there is no data available to identify how well farmers throughout the watershed are managing their fields. However, it may be suggested that the amounts of fertilizer being used (most likely manure and/or poultry litter) are in the appropriate concentrations and are applied at the correct times of the year in order to minimize loss of phosphorous and nitrogen from the crops to runoff due to heavy rainfall.

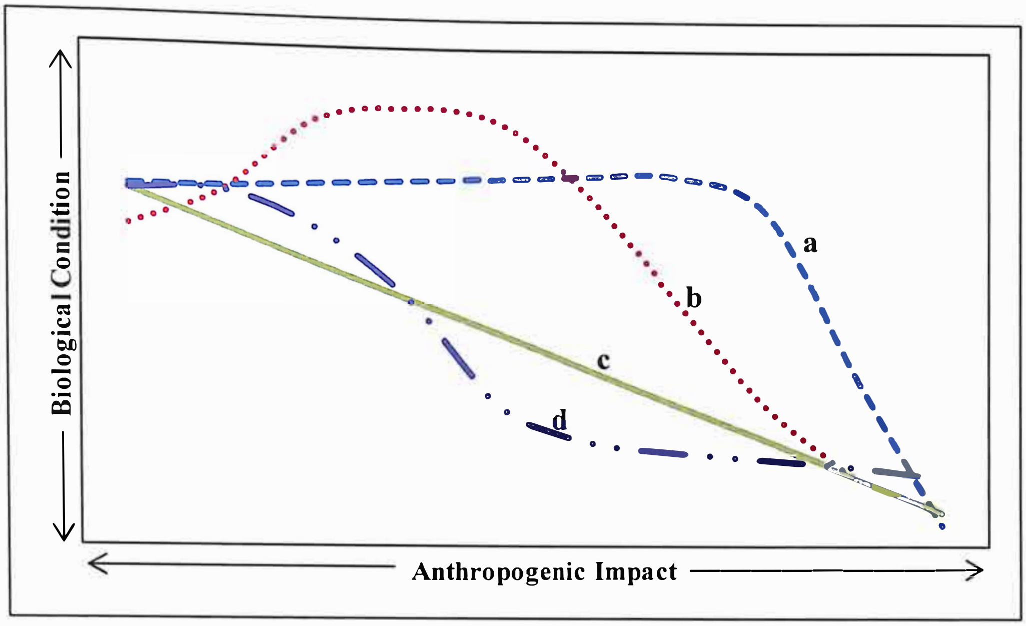

Figure 5-1. Possible hypothetical relationships of human impacts on aquatic biological condition (i.e., index of biotic integrity, species diversity and richness, biotic abundance). Possible responses include: (a) a nonlinear reduction in biological condition in the high range of impact, (b) the subsidy-stress response, (c) a linear response, or (d) the threshold response. Curves (a) and (d) are contingent on community sensitivity to the stressor. Adapted from Allan (2004).

In addition to practical and appropriate application of fertilizer onto crop fields, it must be noted that nearly the entire watershed was protected by some sort of riparian buffer (Figure 5-2). In the downstream portion of the creek, the buffer was in the form of marsh grasses. Upstream, the waterway was protected by both marsh grasses and stands of sweet gum, red maple, and green ash trees. These riparian buffers can intercept nutrients and other contaminants from running off into the water (Schilling and Jacobson, 2014; Bingham *et al*., 1980; Randhir and Ekness, 2013). The natural and man-made buffers that exist in Blackbird Creek appear to be suitable to play this role.

Figure 5-2. Aerial imagery of Blackbird Creek, Delaware. Top: Downstream; mostly marsh wetland. Photo courtesy: Laurieann Phalen. Bottom: Upstream; agriculture. Photo courtesy: Kristopher Roeske. Note the riparian buffers between the cropland and the waterway.

Tidal creeks are extremely dynamic ecosystems that have spatial and temporal heterogeneity, even in small systems such as Blackbird Creek. If there is a significant loss of nutrients to such a system, it is likely that the aquatic signature of this loss would be removed quickly due to the 1-meter semi-daily tidal fluctuations. Hence, the flushing rate is extremely high as water moves into and out of the creek. It seems more likely that any opportunity to see how solubilized nutrients run into the water would be an analysis of pore water. Soil geochemistry could be studied to identify how water flows through subsurface interstitial spaces and into the creek channel. This would also provide information to the researcher about how much fertilizer is applied to a field and how well the riparian buffer is performing its duty to minimize runoff.

The working hypothesis for this thesis provided that blue crab populations and fish biodiversity and population dynamics would be impacted by poor water quality and, by extension, human land use. Based on the data, however, this was not the case. The only water quality parameter that was of any major concern was turbidity. However, the high values of this parameter could not necessarily be attributed to adjacent land use and land cover. While there is, indeed, clear land modification throughout the watershed to accommodate crop growth, the extreme turbidity is likely just a function of the nature of the tidal structure of a Mid-Atlantic creek. With water moving in and out of the channel so quickly and so often, the energy from this movement was carving away the sides of the channel, thus increasing total suspended solids and reducing clarity. Perhaps there was an impact from land modification, but with no other watershed to study as a benchmark comparison, it cannot be concluded that management practices were unsuccessful and have allowed eroded surface soil into the waterway.

There were a few concerns with the overall project design that should be addressed. These include the use of random site selection, the nature of otter trawling, and the choices for sampling protocol in general. In 2015, fourteen sites were chosen for study based on a random grid established in GIS software. Had that grid been placed a few hundred meters in any direction, the percentages of land cover would have been different. Since there were no significant differences in water chemistry through space, however, it may be supposed that changes in the grid layout would not have made any real difference in the hallmark of the study (that nutrient chemistry is different across percentages of agriculture). Also, of those fourteen sites, ten were chosen randomly for sampling on any given date. There were some dates where, for example, no sites were sampled that were between 20% and 30% agriculture. Unfortunately, this is the nature of random sampling and the repercussions could not be avoided for lack of time on a sampling day. But it must be noted that some information was not gathered due to this limitation.

A second concern was the use of otter trawling as the sole means to sample fish. The net was a given size and, based on the scope ratio, was designed (especially in 2015) to trawl along the bottom of the channel. Therefore, any fish that were "pelagic" (not bottom-dwelling or resting low in the water column at the time of sampling) were not sampled. This was also true of any animals that reside at the edges of the creek. For example, mummichogs (*Fundulus heteroclitus*) were captured a number of times in the crab pots, but were not a part of the representative sample in the otter trawls. This is because *F. heteroclitus* tends to inhabit sheltered shorelines (Kneib, 1984), which were missed due to the necessity to trawl through the center of the creek.

Because of this, there has to be some way to incorporate more than one or two gear types in an overall sampling protocol. This was done to a small degree in this project, whereby two different sized trawl nets were used and adjusted to a catch per unit effort, then coupled with collections from crab pots. There was no way, however, to claim that the effort from setting a pot (passive sampling) was comparable to the active otter trawling. Therefore, they were not compared. Perhaps there is a way to model the selectivity of pots versus trawls or other sampling procedures (i.e., weirs, seines, electrofishers in non-saline water). This is an opportunity for future research and analysis of previously gathered data.

Over the past decade or so, reports have been published (DNREC, 2006; Biohabitats Inc, 2007) that have outlined the maximum contaminant values of various nutrients in Blackbird Creek. Based on the data collected for this research, not only is the overall water quality in the ecosystem high, but any possible fluctuations in chemical parameters do not appear to be influenced by human activity. Therefore, the hypotheses outlined in this thesis are rejected. Water quality, blue crab population dynamics, fish biodiversity, and fish trophic assemblages are *not* impacted by adjacent agricultural land use.

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APPENDICES

Appendix 4-A. Results of GLM run on water quality parameters across years, agricultural seasons, agricultural treatments, and all included interactions for the 2014-2015 experimental design. Emboldened p-values reflect significant differences.

Source	Dependent variable (DV)	DV df	Error df	F-statistic	p-value
Year	NH_3	1	96	0.306	**0.582**
	NO_3	1	96	15.534	**<0.001**
	NO_2	1	96	5.531	**0.021**
	PO_4^{3-}	1	96	1.071	0.303
	Alkalinity	1	96	3.071	**0.083**
	Turbidity	1	96	4.072	0.380
	Temperature	1	96	0.300	**0.585**
	Salinity	1	96	42.992	**<0.001**
	pH	1	96	3.679	**0.058**
	DO	1	96	42.159	**<0.001**
Season	NH_3	2	96	16.685	**<0.001**
	NO_3	2	96	6.794	**0.002**
	NO_2	2	96	20.778	**<0.001**
	PO_4^{3-}	2	96	12.266	**<0.001**
	Alkalinity	2	96	10.183	**<0.001**
	Turbidity	2	96	8.995	**<0.001**
	Temperature	2	96	24.789	**<0.001**
	Salinity	2	96	126.272	**<0.001**
	pH	2	96	1.341	**0.267**
	DO	2	96	0.622	0.539
Treatment	NH_3	1	96	1.208	**0.311**
	NO_3	1	96	3.130	**0.029**
	NO_2	1	96	1.059	**0.370**
	PO_4^{3-}	1	96	0.156	0.925
	Alkalinity	1	96	0.202	**0.894**
	Turbidity	1	96	1.021	0.387
	Temperature	1	96	1.393	**0.250**
	Salinity	1	96	1.598	0.195
	pH	1	96	2.146	**0.099**
	Dissolved Oxygen	1	96	0.799	0.497

Season × Treatment Interaction	NH_3	2	96	0.949	0.464
	NO_3	2	96	0.316	0.927
	NO_2	2	96	0.516	0.795
	PO_4^{3-}	2	96	0.459	0.837
	Alkalinity	2	96	0.314	0.928
	Turbidity	2	96	0.944	0.468
	Temperature	2	96	0.542	0.775
	Salinity	2	96	0.453	0.841
	pH	2	96	2.290	0.041
	Dissolved Oxygen	2	96	1.164	0.332
Year × Treatment Interaction	NH_3	1	96	0.361	0.549
	NO_3	1	96	4.912	0.029
	NO_2	1	96	1.676	0.199
	PO_4^{3-}	1	96	0.143	0.706
	Alkalinity	1	96	2.064	0.154
	Turbidity	1	96	2.390	0.125
	Temperature	1	96	0.416	0.520
	Salinity	1	96	0.149	0.700
	pH	1	96	0.414	0.522
	Dissolved Oxygen	1	96	3.789	0.540
Year × Season Interaction	NH_3	2	96	0.169	0.682
	NO_3	2	96	7.358	0.008
	NO_2	2	96	9.929	0.002
	PO_4^{3-}	2	96	0.193	0.662
	Alkalinity	2	96	6.672	0.011
	Turbidity	2	96	2.756	0.100
	Temperature	2	96	0.024	0.878
	Salinity	2	96	14.682	<0.001
	pH	2	96	2.615	0.109
	Dissolved Oxygen	2	96	10.980	0.001
Year × Season × Treatment Interaction	NH_3	2	96	0.287	0.593
	NO_3	2	96	0.707	0.403
	NO_2	2	96	0.355	0.553
	PO_4^{3-}	2	96	0.153	0.553
	Alkalinity	2	96	0.619	0.433
	Turbidity	2	96	4.712	0.138
	Temperature	2	96	0.436	0.510
	Salinity	2	96	0.306	0.581
	pH	2	96	0.013	0.908
	Dissolved Oxygen	2	96	0.271	0.604

Appendix 4-B. Results of GLM run on water quality parameters across agricultural seasons and treatments, and season × treatment interaction for the 2015 experimental design. Emboldened p-values reflect significant differences.

Source	Dependent Variable (DV)	df DV	df Error	F-statistic	p-value
Agriculture Treatment	NH_3	3	71	1.153	0.334
	NO_3	3	71	0.357	0.784
	NO_2	3	71	0.126	0.945
	PO_4^{3-}	3	71	0.033	0.992
	Alkalinity	3	71	0.855	0.469
	Turbidity	3	71	1.163	0.330
	Temperature	3	71	1.718	0.171
	Salinity	3	71	2.112	0.106
	pH	3	71	1.039	0.381
	Dissolved Oxygen	3	71	0.398	0.755
Agriculture Season	NH_3	2	71	15.545	<0.001
	NO_3	2	71	0.680	0.510
	NO_2	2	71	30.971	<0.001
	PO_4^{3-}	2	71	5.843	0.004
	Alkalinity	2	71	7.943	0.001
	Turbidity	2	71	13.355	<0.001
	Temperature	2	71	21.342	<0.001
	Salinity	2	71	162.016	<0.001
	pH	2	71	1.734	0.184
	Dissolved Oxygen	2	71	7.703	0.001
Season × Treatment	NH_3	6	71	0.752	0.610
	NO_3	6	71	0.290	0.940
	NO_2	6	71	0.476	0.824
	PO_4^{3-}	6	71	0.405	0.873
	Alkalinity	6	71	0.352	0.907
	Turbidity	6	71	2.965	0.012
	Temperature	6	71	0.658	0.683
	Salinity	6	71	0.517	0.794
	pH	6	71	1.743	0.124
	Dissolved Oxygen	6	71	1.079	0.383

Appendix 4-C. Results of GLM run on pot-capture per unit effort experimental design for 2014 and 2015. Independent sources and interaction terms are included with degrees of freedom (df) of both the dependent variables and the error terms, and the F-statistic. Emboldened p-values (< 0.05) represent significant variation of dependent variables dependent upon the source.

Source	Dependent Variable (DV) per Unit Effort	df DV	df Error	F-statistic	p-value
Year	Total catch	1	28	0.686	0.414
	Total recruits	1	28	1.046	0.315
	Total juveniles	1	28	1.000	0.326
	Total adults	1	28	4.024	0.055

	Total molting crabs	1	28	0.015	0.903
	Total non-molting crabs	1	28	0.765	0.389
	Total males	1	28	0.469	0.499
	Total females	1	28	0.750	0.394
Agriculture Season	Total catch	2	28	5.752	0.008
	Total recruits	2	28	2.144	0.136
	Total juveniles	2	28	2.619	0.091
	Total adults	2	28	8.790	**0.001**
	Total molting crabs	2	28	1.725	0.197
	Total non-molting crabs	2	28	4.497	0.071
	Total males	2	28	5.296	0.011
	Total females	2	28	2.913	**0.020**
Agriculture Treatment	Total catch	1	28	0.006	0.939
	Total recruits	1	28	1.632	0.212
	Total juveniles	1	28	0.035	0.853
	Total adults	1	28	0.826	0.371
	Total molting crabs	1	28	0.320	0.576
	Total non-molting crabs	1	28	0.096	0.758
	Total males	1	28	0.689	0.413
	Total females	1	28	2.288	0.142
Year × Treatment Interaction	Total catch	1	28	0.147	0.704
	Total recruits	1	28	1.351	0.255
	Total juveniles	1	28	1.991	0.169
	Total adults	1	28	0.032	0.860
	Total molting crabs	1	28	0.338	0.566
	Total non-molting crabs	1	28	0.415	0.525
	Total males	1	28	0.798	0.379
	Total females	1	28	0.213	0.648
Year × Season Interaction	Total catch	2	28	1.975	0.158
	Total recruits	2	28	2.885	0.073
	Total juveniles	2	28	0.699	0.505
	Total adults	2	28	1.230	0.308
	Total molting crabs	2	28	0.888	0.423
	Total non-molting crabs	2	28	1.462	0.249
	Total males	2	28	1.337	0.279
	Total females	2	28	2.821	0.077
Season × Treatment Interaction	Total catch	1	28	0.843	0.441
	Total recruits	1	28	1.781	0.187
	Total juveniles	1	28	0.807	0.456
	Total adults	1	28	0.341	0.714
	Total molting crabs	1	28	0.420	0.661
	Total non-molting crabs	1	28	1.009	0.377
	Total males	1	28	1.387	0.266
	Total females	1	28	0.068	0.935

Year × Season × Treatment Interaction	**Total catch**	**2**	**28**	**0.795**	**0.462**
	Total recruits	2	28	1.129	0.337
	Total juveniles	**2**	**28**	**1.035**	**0.368**
	Total adults	2	28	0.292	0.749
	Total molting crabs	**2**	**28**	**0.346**	**0.711**
	Total non-molting crabs	2	28	0.557	0.579
	Total males	**2**	**28**	**0.163**	**0.850**
	Total females	2	28	1.788	0.186

Appendix 4-D: Results of GLM run on trawl-capture per unit effort experimental design for 2014 and 2015. Independent sources and interaction terms are included with degrees of freedom (df) of both the dependent variables and the error terms, and the F-statistic. Emboldened p-values (< 0.05) represent significant variation of dependent variables dependent upon the source.

Source	**Dependent Variable (DV) per Unit Effort**	**df DV**	**df Error**	**F-statistic**	**p-value**
Year	**Total catch**	**1**	**14**	**4.437**	**0.054**
	Total recruits	1	14	0.115	0.739
	Total juveniles	**1**	**14**	**4.925**	**0.044**
	Total adults	1	14	0.423	0.526
	Total molting crabs	**1**	**14**	**1.168**	**0.298**
	Total non-molting crabs	1	14	5.207	**0.039**
	Total males	**1**	**14**	**5.868**	**0.030**
	Total females	1	14	3.904	0.068
Agriculture Season	**Total catch**	**2**	**14**	**0.281**	**0.759**
	Total recruits	2	14	5.773	**0.015**
	Total juveniles	**2**	**14**	**2.115**	**0.158**
	Total adults	2	14	4.176	**0.038**
	Total molting crabs	**2**	**14**	**4.019**	**0.042**
	Total non-molting crabs	2	14	0.478	0.630
	Total males	**2**	**14**	**0.135**	**0.875**
	Total females	2	14	1.332	0.295
Agriculture Treatment	**Total catch**	**1**	**14**	**5.049**	**0.041**
	Total recruits	1	14	0.285	0.602
	Total juveniles	**1**	**14**	**3.507**	**0.082**
	Total adults	1	14	1.597	0.227
	Total molting crabs	**1**	**14**	**0.000**	**0.990**
	Total non-molting crabs	1	14	5.272	**0.038**
	Total males	**1**	**14**	**3.352**	**0.088**
	Total females	1	14	5.808	**0.030**
Year × Season Interaction	**Total catch**	**2**	**14**	**4.880**	**0.025**
	Total recruits	2	14	1.947	0.179
	Total juveniles	**2**	**14**	**6.388**	**0.011**
	Total adults	2	14	0.565	0.581

	Total molting crabs	2	14	3.381	0.063
	Total non-molting crabs	2	14	4.237	**0.036**
	Total males	2	14	2.228	0.145
	Total females	2	14	11.413	**0.001**
Treatment × Season Interaction	**Total catch**	2	14	0.360	0.704
	Total recruits	2	14	1.123	0.353
	Total juveniles	2	14	0.233	0.795
	Total adults	2	14	0.372	0.696
	Total molting crabs	2	14	0.307	0.740
	Total non-molting crabs	2	14	0.390	0.684
	Total males	2	14	0.953	0.409
	Total females	2	14	0.281	0.759
Year × Treatment Interaction	**Total catch**	2	14	0.021	0.887
	Total recruits	2	14	2.228	0.158
	Total juveniles	2	14	0.086	0.774
	Total adults	2	14	0.034	0.856
	Total molting crabs	2	14	0.652	0.433
	Total non-molting crabs	2	14	0.003	0.959
	Total males	2	14	0.147	0.707
	Total females	2	14	0.336	0.572
Year × Treatment × Season Interaction	**Total catch**	2	14	0.776	0.479
	Total recruits	2	14	1.351	0.291
	Total juveniles	2	14	0.503	0.615
	Total adults	2	14	0.324	0.729
	Total molting crabs	2	14	0.033	0.968
	Total non-molting crabs	2	14	0.847	0.450
	Total males	2	14	0.330	0.725
	Total females	2	14	1.689	0.220

Appendix 4-E. Results from GLM run on fish species according to their trophic levels in the 2014-2015 experimental design. Emboldened p-values represent significance within the source variance.

Source	**Dependent Variable (DV) per Unit Effort**	**df DV**	**df Error**	**F-statistic**	**p-value**
Year	**Primary consumers**	1	14	1.544	0.234
	Secondary consumers	1	14	6.550	**0.023**
	Opportunistic omnivores	1	14	0.108	0.747
Agriculture Season	Primary consumers	2	14	0.875	0.439
	Secondary consumers	2	14	2.649	0.106
	Opportunistic omnivores	2	14	1.708	0.217
Agriculture Treatment	**Primary consumers**	1	14	0.252	0.604
	Secondary consumers	1	14	1.420	0.253
	Opportunistic omnivores	1	14	1.662	0.218
Year ×	Primary consumers	2	14	0.016	0.984

Season Interaction	**Secondary consumers**	**2**	**14**	**1.684**	**0.189**
	Opportunistic omnivores	2	14	1.557	0.246
Year × Treatment Interaction	**Primary consumers**	**1**	**14**	**1.377**	**0.260**
	Secondary consumers	1	14	2.020	0.177
	Opportunistic omnivores	**1**	**14**	**1.575**	**0.230**
Season × Treatment Interaction	Primary consumers	2	14	0.264	0.772
	Secondary consumers	**2**	**14**	**0.716**	**0.506**
	Opportunistic omnivores	2	14	0.563	0.582
Year × Season × Treatment Interaction	**Primary consumers**	**2**	**14**	**0.448**	**0.648**
	Secondary consumers	2	14	0.550	0.580
	Opportunistic omnivores	**2**	**14**	**1.135**	**0.349**

Appendix 4-F: Results from GLM run on fish biodiversity indices in the 2014-2015 experimental design. Emboldened p-values represent significance within the source variance.

Source	Dependent Variable (DV) per Unit Effort	df DV	df Error	F-statistic	p-value
Year	**Total capture**	**1**	**14**	**5.505**	**0.034**
	Species Richness	1	14	2.713	0.122
	Simpson Inverse	**1**	**14**	**0.334**	**0.572**
	Shannon-Weiner Index	1	14	0.383	0.546
	Margalef's Index	**1**	**14**	**0.075**	**0.788**
Agriculture Season	Total capture	2	14	1.638	0.229
	Species Richness	**2**	**14**	**6.936**	**0.008**
	Simpson Inverse	2	14	2.179	0.150
	Shannon-Weiner Index	**2**	**14**	**4.416**	**0.033**
	Margalef's Index	2	14	12.353	**0.001**
Agricultural Treatment	**Total capture**	**1**	**14**	**1.488**	**0.243**
	Species Richness	1	14	0.301	0.592
	Simpson Inverse	**1**	**14**	**1.352**	**0.263**
	Shannon-Weiner Index	1	14	0.686	0.421
	Margalef's Index	**1**	**14**	**2.224**	**0.158**
Year × Season Interaction	Total capture	2	14	1.363	0.288
	Species Richness	**2**	**14**	**0.997**	**0.394**
	Simpson Inverse	2	14	1.549	0.247
	Shannon-Weiner Index	**2**	**14**	**2.090**	**0.161**
	Margalef's Index	2	14	1.241	0.319
Year × Treatment Interaction	**Total capture**	**1**	**14**	**1.589**	**0.228**
	Species Richness	1	14	0.012	0.914
	Simpson Inverse	**1**	**14**	**0.175**	**0.682**
	Shannon-Weiner Index	1	14	1.391	0.258
	Margalef's Index	**1**	**14**	**0.115**	**0.740**
Season ×	Total capture	2	14	0.605	0.560

Treatment Interaction	**Species Richness**	**2**	**14**	**0.004**	**0.996**
	Simpson Inverse	2	14	0.257	0.777
	Shannon-Weiner Index	**2**	**14**	**0.973**	**0.402**
	Margalef's Index	2	14	0.050	0.951
Year × Season × Treatment Interaction	**Total capture**	**2**	**14**	**0.731**	**0.499**
	Species Richness	2	14	0.094	0.911
	Simpson Inverse	**2**	**14**	**0.659**	**0.633**
	Shannon-Weiner Index	2	14	0.611	0.557
	Margalef's Index	**2**	**14**	**0.082**	**0.922**

Appendix 4-G: Results of GLM run on blue crab pot-capture per unit effort experimental design for 2015. Independent sources and interaction terms are included with degrees of freedom (df) of both the dependent variables and the error terms, and the F-statistic. Emboldened p-values (< 0.05) represent significant variation of dependent variables dependent upon the source.

Source	Dependent Variable (DV) per Unit Effort	df DV	df Error	F-statistic	p-value
Agriculture Treatment	**Total catch**	**3**	**20**	**2.580**	**0.082**
	Total recruits	3	20	0.720	0.552
	Total juveniles	**3**	**20**	**4.532**	**0.014**
	Total adults	3	20	0.727	0.548
	Total molting crabs	**3**	**20**	**0.667**	**0.582**
	Total non-molting crabs	3	20	1.855	0.170
	Total males	**3**	**20**	**1.628**	**0.215**
	Total females	3	20	0.726	0.549
Agriculture Season	**Total catch**	**2**	**20**	**7.500**	**0.004**
	Total recruits	2	20	1.281	0.300
	Total juveniles	**2**	**20**	**8.615**	**0.002**
	Total adults	2	20	8.998	**0.002**
	Total molting crabs	**2**	**20**	**1.580**	**0.231**
	Total non-molting crabs	2	20	3.681	**0.044**
	Total males	**2**	**20**	**2.387**	**0.118**
	Total females	2	20	2.089	0.150
Treatment × Season Interaction	**Total catch**	**6**	**20**	**3.419**	**0.017**
	Total recruits	6	20	0.343	0.906
	Total juveniles	**6**	**20**	**5.276**	**0.002**
	Total adults	6	20	0.706	0.648
	Total molting crabs	**6**	**20**	**0.485**	**0.812**
	Total non-molting crabs	6	20	2.132	0.094
	Total males	**6**	**20**	**0.988**	**0.460**
	Total females	6	20	2.489	0.058

Appendix 4-H: Results of GLM run on blue crab trawl-capture per unit effort experimental design for 2015. Independent sources and interaction terms are included with degrees of freedom (df) of both the dependent variables and the error terms, and the F-statistic. Emboldened p-values (< 0.05) represent significant variation of dependent variables dependent upon the source.

Source	Dependent Variable (DV) per Unit Effort	df DV	df Error	F-statistic	p-value
Agriculture Treatment	Total catch	[illegible]	20	3.925	0.024
	Total recruits	[illegible]	20	1.108	0.369
	Total juveniles	[illegible]	20	3.242	0.044
	Total adults	3	20	1.711	0.197
	Total molting crabs	3	20	0.018	0.997
	Total non-molting crabs	3	20	4.15	0.019
	Total males	3	20	1.573	0.227
	Total females	3	20	9.607	<0.001
Agriculture Season	Total catch	2	20	11.153	0.001
	Total recruits	2	20	2.514	0.106
	Total juveniles	2	20	15.877	<0.001
	Total adults	2	20	6.727	0.006
	Total molting crabs	2	20	1.334	0.286
	Total non-molting crabs	2	20	11.801	<0.001
	Total males	2	20	4.023	0.034
	Total females	2	20	28.083	<0.001
Treatment × Season Interaction	Total catch	6	20	4.910	0.003
	Total recruits	6	20	0.793	0.586
	Total juveniles	6	20	4.333	0.006
	Total adults	6	20	1.984	0.116
	Total molting crabs	6	20	0.757	0.611
	Total non-molting crabs	6	20	5.179	0.002
	Total males	6	20	2.136	0.094
	Total females	6	20	9.562	<0.001

Appendix 4-I: Results from GLM run on fish species broken into their trophic levels in the 2015 experimental design. Emboldened p-values represent significance within the source variance.

Source	Dependent Variable (DV) per Unit Effort	df DV	df Error	F-statistic	p-value
Season	Primary consumers	2	20	1.916	0.173
	Secondary consumers	2	20	1.870	0.180
	Opportunistic omnivores	2	20	3.039	0.070
Treatment	Primary consumers	3	20	5.354	0.007
	Secondary consumers	3	20	2.589	0.081

	Opportunistic omnivores	3	20	0.642	0.597
Season × Treatment Interaction	**Primary consumers**	**6**	**20**	**0.995**	**0.455**
	Secondary consumers	6	20	1.120	0.385
	Opportunistic omnivores	**6**	**20**	**0.544**	**0.768**

Appendix 4-J: Results from GLM run on fish biodiversity indices in the 2015 experimental design. Emboldened p-values represent significance within the source variance.

Source	Dependent Variable (DV) per Unit Effort	df DV	df Error	F-statistic	p-value
Season	**Total capture**	**2**	**20**	**2.907**	**0.078**
	Species Richness	2	20	4.583	**0.023**
	Simpson Inverse	**2**	**20**	**1.285**	**0.298**
	Shannon-Weiner Index	2	20	2.244	0.132
	Margalef's Index	**2**	**20**	**8.426**	**0.002**
Treatment	Treatment Total capture	3	20	0.675	0.577
	Species Richness	**3**	**20**	**3.406**	**0.038**
	Simpson Inverse	3	20	1.274	0.310
	Shannon-Weiner Index	**3**	**20**	**1.834**	**0.173**
	Margalef's Index	3	20	3.335	**0.040**
Season × Treatment Interaction	**Total capture**	**6**	**20**	**0.619**	**0.713**
	Species Richness	6	20	0.676	0.671
	Simpson Inverse	**6**	**20**	**0.621**	**0.711**
	Shannon-Weiner Index	6	20	0.246	0.955
	Margalef's Index	**6**	**20**	**1.068**	**0.414**

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Department of Agriculture and Natural Resources
Delaware State University
College of Agriculture and Related Sciences

EDUCATION

2016 M.S. Natural Resources (projected graduation May 2016)
Delaware State University, Dover, DE
Thesis: Land use practice and water quality, blue crab population dynamics, and fish biodiversity in Blackbird Creek, Delaware

2011 M.S. Marine Science
University of Southern Mississippi, Stennis Space Center, MS
Thesis: Effects of fluctuating light on two strains of marine phytoplankton

2005 B.S. Biology
Anderson University, Anderson, SC

EMPLOYMENT

2016-present **Intern**
Chester County Conservation District, Kennett Square, PA

- Wrote report on PA Act 167 municipality stormwater management ordinances
- Completed farm visit data entry for Chesapeake basin
- Participated in site visits for NPDES and ENS inspections, compliance, and residential complaints

2014-present **Graduate Field and Laboratory Research Assistant**
Department of Natural Resources, Delaware State University, Dover, DE

- Collected field samples (soil, water, fish, crabs, grasses) in a tidal marsh
- Processed samples for analysis of water quality, soil chemistry, and stable isotopes
- Analyzed data using SPSS statistical software package
- Assisted with oyster remote set project
- Presented data products at local and international conferences

2011-2012 **Adjunct Instructor**
Department of Biology, Anderson University, Anderson, SC

- Created 56 lectures for two courses: Integrated Science and Environmental Science
- Presented lectures and answered questions twice weekly per course
- Developed and graded homework assignments and exams for 90 student

2006-2011 **Graduate Field and Laboratory Research Assistant**
Department of Marine Science, University of Southern Mississippi, Stennis Space Center, MS

- Monitored and maintained ~100 phytoplankton cultures
- Measured plankton strains for carbon, nitrogen, particle counts, fluorescence, production, and quantum yield
- Assisted with field collection of water for nutrient analysis and algal photobiology

2006 **Seasonal Naturalist**
Audubon South Carolina at Francis Beidler Forest, Harleysville, SC

- Led canoe trips through the swamp trail
- Led boardwalk environmental tours to visitors and school groups
- Performed basic maintenance of the sanctuary

RESEARCH EXPERIENCE

2014-present **Research Assistant**
Delaware State University, Dover, DE
Blue Crab and Fish Population Dynamics as Related to Adjacent Land Use Practices in Blackbird Creek, Delaware

- Identified land use practice using ArcMAP GIS software
- Conducted weekly surveys of blue crab and fish populations in collaboration with the Delaware Department of Natural Resources and Environmental Control using pot and otter trawl collection practices
- Collected environmental data to investigate site treatments and yearly and seasonal changes

2014-present **Research Assistant**
Delaware State University, Dover, DE
Easter Oyster Remote Set Efficiency of Different Gear Types

- Set 3 gear types into a remote set system and allowed oyster larvae to settle
- Wrote manuscript to identify which gear type was most effective

2014-2015 **Research Assistant**
Delaware State University, Dover, DE
Native and Invasive Marsh Grasses and Their Ecosystem Roles

- Collected marsh grass root, leaf, and stem samples for genetic analysis

2006-2011 **Research Assistant**
The University of Southern Mississippi, Stennis Space Center, MS
Phytoplankton Biofuels

- Ran analysis of plankton samples for C:N, primary production (^{14}C method), quantum yield, particle counts based on nutrient consumption, and lipid content

2006-2008 **Research Assistant**
The University of Southern Mississippi, Stennis Space Center, MS
Water Quality in the Bay of St. Louis, MS

- Conducted field collection of water for nutrient, phytoplankton, and bacterial analysis

MANUSCRIPTS IN PREPARATION/SUBMISSION

Stone, M., K. Roeske, L. K. Chintapenta, L. Phalen, and G. Ozbay. (in prep). Water quality evidence of successful management of Blackbird Creek, Delaware, a marginally-impacted watershed. Journal of the American Water Resources Association.

Stone, M., L. K. Chintapenta, V. Kalavacharla, L. Phalen, K. Roeske, and G. Ozbay (in prep). An ecosystem assessment of the Blackbird Creek watershed, Delaware.Ecosystems.

Roeske, K., **M. Stone**, and G. Ozbay (in prep). Relationship between native and invasive marsh grasses to blue crab population dynamics.

AWARDS

2016 Atlantic Estuarine Research Society: Ann C. Powell Student Travel Scholarship
2014 NOAA Educational Partnership Program: Student Travel Award
2014 Atlantic Estuarine Research Society: Ann C. Powell Student Travel Scholarship

INVITED TALKS

2015 Department of Agriculture and Natural Resources, Delaware State University
- *Land use impacts on blue crabs in Blackbird Creek, Delaware*

2010 Department of Marine Science, University of Southern Mississippi
- *Effects of fluctuating light on two strains of marine phytoplankton*

2009 Department of Biology, Anderson College
- *How phytoplankton can end America's dependence on fossil fuels: A small part in a large project*

PRESENTATIONS

Stone, M., L. Phalen, K. Chintapenta, V. Kalavacharla, and G. Ozbay. 2016. Evidence of successful management in an agriculturally-impacted Delaware estuary. Oral presentation. Atlatnic Estuarine Research Society. Virginia Beach, VA. 10-12 March.

Ozbay, G., J. Faye, K. Hannum, and L. K. Chintapenta (Presented by **M. Stone**). 2015. Investigating total bacteria, total Vibrionaceae, and *Vibrio paraheaemolyticus* in Eastern Oysters (*C. virginica*) from the Delaware Inland Bays, USA. Oral presentation. 6th Annual Oyster Symposium. Cape Cod, MA. 20-23 October. http://oystersymposium.org/about-ios6/schedule/

Ozbay, G., B. Reckenbeil, F. Marenghi, and L, Phalen (Presented by **M. Stone**). 2015. Eastern oyster larvae *C. virginica* remote set practices in Delaware, USA. Oral Presentation. 6th

International Oyster Symposium and Conference. Cape Cod, MA. 20-23 October. http://oystersymposium.org/about-ios6/schedule/

Roeske, K. P., P. Jivoff, and G. Ozbay. Blue crab (*Callinectes sapidus*) population dynamics relative to salt marsh grasses in Blackbird Creek, Delaware. 2014. Oral presentation by **M. Stone** and G. Ozbay. Mid-Atlantic Chapter – American Fisheries Society. Cape Henlopen, DE. 06-07 November.

Stone, M. L., L. K. Chintapenta, G. Ozbay, K. Coyne, and C. Dixon. 2014. Evaluation of benthic diatoms as water quality indicators in the Blackbird Creek watershed, Delaware. Poster presentation. Atlantic Estuarine Research Society. Ocean City, MD. 12-16 March.

Stone, M. L., K. Ommanney, L. Phalen, K. Chintapenta, V. Kalavacharla, and G. Ozbay. 2014. Nekton abundance and diversity relative to marsh grasses in Blackbird Creek, Delaware. Oral presentation. Mid-Atlantic Chapter of the American Fisheries Society Annual Meeting. Cape Henlopen State Park – Lewes, DE. 06-07 November.

Stone, M. L., K. Ommanney, L. Phalen, K. Chintapenta, V. Kalavacharla, and G. Ozbay. 2014. Nekton abundance and diversity relative to water quality in Blackbird Creek, Delaware. Oral presentation. National Oceanic and Atmospheric Administration – Educational Partnership Program. Princess Anne, MD. 26-29 October.

Stone, M., and G. Ozbay. 2015. Tidal tale: Bi-directional waterway reveals nutrient runoff from cropland. Poster Presentation. Graduate Student Symposium: Delaware Environmental Institute. Newark, DE. October 8.

Stone, M., G. Ozbay, and V. Kalavacharla. 2015. Environmental factors influencing population dynamics of the blue crab (*Callinectes sapidus*) in a large tidal creek near the Delaware Bay and possible implications to stable isotope analysis. Poster Presentation. 107th Annual Meeting: National Shellfisheries Association. Monterey, CA. March 22-26.

Stone, M., G. Ozbay, and V. Kalavacharla. 2015. Stable isotope analysis of blue crabs relative to water quality in Blackbird Creek, Delaware. Oral Presentation. 107th Annual Meeting: National Shellfisheries Association. Monterey, CA. March 22-26.

Stone, M. L., D. Redalje, and A. Mojzis. 2008. Comparison of environmental parameters of the Bay of St. Louis, Mississippi before and after Hurricane Katrina. 72nd Annual Meeting: Mississippi Academy of Sciences. Olive Branch, MS. 20-22 February.

Stone, M. L., A. Wiltbank, L. K. Chintapenta, V. Kalavacharla, and G. Ozbay. 2014. Studying the macrophyte-microbe interactions in the salt tolerance mechanism of the Common reed *Phragmites australis* in Blackbird Creek, Delaware. Poster presentation. Delaware National Estuarine Research Reserve annual symposium. Dover, DE. 21 March.

MENTORED PROJECTS

Chintapenta, L. K., D. L. Carter, **M. Stone**, V. Kalavacharla, and G. Ozbay. 2014. Assessment of ecosystem health in Blackbird Creek, Delaware using diatoms as biological indicators. Oral presentation. Mid-Atlantic Chapter – American Fisheries Society. Cape Henlopen, DE. 06-07 November.

Chintapenta, L. K., **M. Stone**, A. Pappas, K. Lee, K. Coyne, and G. Ozbay. 2014. Assessment of benthic diatoms as water quality Indicators in the Blackbird Creek Watershed, Delaware. Delaware National Estuarine Research Reserve Research Symposium. Dover, DE. Poster presentation. 21 March.

Johnson, E. R., **M. L. Stone**, L. R. Phalen,G. Ozbay, L. K. Chintapenta, and V. Kalavacharla. 2014. Blue crab (*Callinectes sapidus*) population dynamics across the salinity gradient in Blackbird Creek, Delaware. Poster presentation. Center for Integrated Biological and Environmental Research forum. 29 June.

Ommanney, K., **M. L. Stone**, L. Phalen, K. Chintapenta, V. Kalavacharla, and G. Ozbay. 2014. Relationship between aquatic species richness and salt marsh grasses in Blackbird Creek, Delaware. National Oceanic and Atmospheric Administration – Educational Partnership Program. Princess Anne, MD. Poster presentation. 26-29 October.

Ozbay, G., L. K. Chintapenta, K. P. Roeske, **M. L. Stone**, and L. Phalen. 2014. Blackbird Creek monitoring program to study the impacts of climate change and land use. American Geophysical Union. San Francisco, CA. Poster presentation. 15-19 December. https://agu.confex.com/agu/fm14/meetingapp.cgi/Paper/31975

Ozbay, G., L. K. Chintapenta, **M. L. Stone**, K. P. Roeske, and L. Phalen. 2016. Saltmarsh habitat of Blackbird Creek and impact of climate change and land uses. Delaware Wetlands Conference. Wilmington, DE. Poster presentation. 03-04 February.

Wiltbank, A., **M. L. Stone**, L. K. Chintapenta, V. Kalavacharla, and G. Ozbay. 2014. Studying the macrophyte-microbe interactions in the salt tolerance mechanism of the native marsh grass *Spartina alterniflora* in Blackbird Creek, Delaware. Poster presentation. AAAS EPSCoR Review Meeting. Dover, DE. 6 February.

PROFESSIONAL AFFILIATION

Atlantic Estuarine Research Society, American Fisheries Society – Mid-Atlantic Chapter, National Shellfish Society, World Aquaculture Society

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