RESPONSE OF FORAMINIFERA TO A REVERSE OSMOSIS BRINY DISCHARGE

by

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ABSTRACT

RICHARD EUSTACE AIKEN SMALL. Response of Foraminifera to a reverse osmosis briny discharge. (Under the direction of DR. SCOTT HIPPENSTEEL)

Reverse osmosis water treatment plants are becoming the preferred means of generating potable water for many eastern North Carolina communities. At these facilities, reject brine solutions—sometimes containing up to 10 times the initial concentration of dissolved solids—are created and often discharged into estuarine waters. Several state and federal agencies have expressed concern over the potential ecological impacts this wastewater could have on these sensitive environments. Monitoring of a brine discharge site in Currituck County, North Carolina revealed significantly higher conductivity values within ~50 m of the point source. One group of organisms that have proven useful in other studies for monitoring impact of anthropogenic pollution in estuaries is Foraminifera. Foraminifera are abundant microorganisms that are widespread in most marginal-marine and marine environments; nevertheless, individual taxa are highly selective of their habitat. Nearly all species build shells (tests) that are preserved in coastal sediments, allowing for reconstruction of previous marine conditions. Species abundance data was collected from surface and sub-surface samples taken in the area surrounding the brine point source. Two taxa (*Ammobaculites* spp. and *Ammotium* sp.) accounted for 98.5% of all normalized specimens. Abundance is significantly less in the sub-surface samples (Student's t-test, $p<0.0001$), likely due to taphonomic effects. Abundance does not appear correlated with discharge of the wastewater; instead, natural parameters appear to affect abundance in an assemblage to a greater degree. Species distribution is similar in surface and sub-surface samples. Foraminiferal diversity is significantly less near the discharge based on one sample collected within 5 m of the discharge site; samples at greater distances do not appear affected. Loss of diversity within a few meters of the discharge site is consistent with previous studies, but more data would be needed to confirm the results observed at this site.

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LIST OF ABBREVATIONS

ANOVA	analysis of variance
EPA	Environmental Protection Agency
NPDES	National Pollution Discharge Elimination System
OM	organic matter
PPT	parts per thousand
PTE	potentially toxic elements
RO-WTP	reverse osmosis water treatment plant

CHAPTER 1: INTRODUCTION

1.1 Background

Human population growth and tourism along the North Carolina Atlantic coast have applied anthropogenic stresses to an antiquated infrastructure. Of the eight North Carolina coastal counties, only two (Hyde: -1.5%, and Dare 3.2%) had population growth that did not exceeded the state average (3.3%) from 2010 to 2013 (U.S. Census Bureau, 2014). Historically, potable water in these areas has been derived from shallow surficial aquifers; however, increased anthropogenic removal, saltwater intrusion, sea level rise, and contamination from anthropogenic processes have led to the abandonment of surficial aquifers for deeper, more dependable aquifers (Ward, 2008; Deaton et al., 2010). Local governments have been forced to invest in deeper wells and employ desalination technologies that are quickly becoming the preferred sources for drinking water in the North Carolina Coastal Plain. Area surface waters have not been found suitable for conditioning because the presence of dissolved tannins and lignins at concentrations that often exceed aesthetic standards for drinking water as suggested by the United States Environmental Protection Agency (EPA) (Rulifson et al., 2006b).

The more reliable, deeper aquifers require increased conditioning in the form of desalination to remove salts and metals. The most common method utilized in eastern North Carolina to remove these undesired constituents from the water is reverse osmosis (Kleber, 2010). Although reverse osmosis water treatment plants (RO-WTPs) are able to provide quality drinking water, they have possible negative environmental consequences.

Multiple North Carolina governmental agencies have raised concern regarding impacts to the environment from RO-WTPs (Ward, 2008). The prime concern is the discharge of the concentrated brine solution into North Carolina surface waters.

Reverse osmosis desalination involves the production of potable water from salty or brackish sources by removing salt from water without requiring a change of state (Mahi, 2001). By applying a pressure greater than that of the osmotic pressure of the water, freshwater is allowed to flow through a filtering membrane where it can be captured for further conditioning or public use (Younos and Tulou, 2005). During this process, a concentrated brine solution (wastewater) is also created. Recovery rates for RO-WTPs generally range from 80-90% for “fresh” groundwater sources to as low as 40-60% for seawater sources (Malmrose et al., 2004). The range in treatment efficiency leads to a wide range in salt concentration of effluents; concentrations of source salts in the effluent from brackish sources can range from 1.25 to 10 times that of the source water (Malmrose et al., 2004; Younos, 2005a). The facility that is the focus of this project is designed for a recovery rate of 75% (Carson et al., 2009).

Wastewater from RO-WTPs is influenced by the source water and pre-treatment techniques (Ward, 2008). Salt concentration in the source water determines the efficiency of the desalination process and ultimately the salt concentration of the wastewater. Pre-treatment of incoming groundwater is necessary to protect infrastructure, and generally increases efficiency (Younos, 2005a; Younos and Tulou, 2005). Common pre-treatment chemical additions include sodium hypochlorite, ferric chloride, aluminum chloride, sulfuric acid, hydrochloric acid, sodium hexametaphosphate, and sodium bisulfite (Einav et al., 2002; Sadhwani et al., 2005).

There are also small concentrations of additional chemicals periodically used for membrane cleaning, but these occur in such small concentrations that their potential impact on the environment are considered to be minimal (Sadhwani et al., 2005).

The Federal Clean Water Act of 1972 and subsequent modifications empowers the EPA to regulate point sources of pollution into all U.S. surface waters by permitting facilities through the National Pollution Discharge Elimination System (NPDES) (Johnson and Cholski, 2004). Briny concentrated discharge from desalination plants is considered a point source and is regulated under the NPDES permitting program (Younos, 2005b). Permits are issued by the U.S. EPA, or authorized states, including North Carolina (EPA, 2011). Two types of permits are issued, general and individual. General permits can cover multiple discharging locations using similar industry techniques or discharging similar effluent while individual permits cover only one discharge source (EPA, 2011). As of 2011, North Carolina had only issued individual permits for RO-WTP discharge (EPA, 2011).

The permitting program regulates a wide range of pollutants, with monthly, quarterly, or annual reports on water quality and volume (EPA, 2011). Although brine discharges to U.S. surface waters from RO-WTPs are monitored under the NPDES, there are no national limitations or guidelines for the discharge of residuals into U.S. waters (EPA, 2011). Instead, limitations are decided by states based on applicability of alternative technologies, technology-based best professional judgment limitations, toxicity assessments, and impact to water quality based on the use of the receiving waters (Water Treatment Plant Workgroup, 2003).

Federal and state regulators, through the NPDES program commonly monitor aluminum, copper, dissolved oxygen, iron, lead, pH, temperature, total residual chlorine, total suspended solids, and turbidity in residuals from WTPs (EPA, 2011). Additionally, at RO-WTPs, chloride concentration and total dissolved solids are often monitored (EPA, 2011).

Not only is the type and level of contaminants important, but the method and location of wastewater discharge directly influences the potential impacts to the environment. Disposal sites in the United States, in decreasing order of number are: surface waters, publically owned treatment works, land, deep well, and evaporation ponds (Younos, 2005a). By far the most common disposal method is discharge of wastewater into surface waters (Younos, 2005a).

Discharge into estuaries, as opposed to the ocean, has the potential to have a greater detrimental ecological impact due to decreased wave and tidal mixing of effluent in combination with the sensitive ecological nature of estuaries. Hence, the EPA recommends against discharging desalination wastewater into estuaries and other environmentally sensitive waters (Johnson and Cholski, 2004). However, wastewater from most North Carolina RO-WTPs is discharged into surface waters of the Albemarle-Pamlico estuary system (NCDENR, 2014).

1.2 Justification

Desalination of saline water has been a commonly used method of water treatment in Middle Eastern countries for decades due to the lack of easily accessible fresh water and the abundance of inexpensive energy (Tsiourtis, 2001). Desalination for potable water is gaining popularity globally and the use of these technologies is expected to increase in the future in response to global climate change and increasing global population (Tsiourtis, 2001). The most common method for desalination worldwide is reverse osmosis, largely due to its higher relative energy efficiency and cost efficiency (Mahi, 2001; Einav et al., 2002).

Due to the projected increase in desalination technologies worldwide and statewide, it is important to understand the impacts of these technologies on the environment.

Effluent from these systems often exceeds acute and chronic toxicity levels for aquatic organisms (Water Treatment Plant Workgroup, 2003). Consequently, multiple state and federal agencies have expressed concern over the potential impacts to biotic communities and have stated a desire for more research on the potential impacts (Water Treatment Plant Workgroup, 2003; Ward, 2008). Two recently built RO-WTPs in northeastern North Carolina (Currituck and Pasquotank counties) have forgone a North Carolina Division of Marine Fisheries previously suggested two-year post-construction impact assessment (Deaton et al., 2010).

RO-WTPs are becoming increasingly common in the North Carolina coastal plain as communities look for a reliable source of potable water. As of 2010, 16 RO-WTPs were operating in coastal North Carolina with most operating in the region surrounding the Albemarle-Pamlico estuarine system, despite this being one of the least populated areas along the U.S. eastern seaboard (Deaton et al., 2010).

One group of organisms that would likely respond to the altered surface waters, and provide easily observable evidence of the impacts from a RO-WTP effluent are Foraminifera. Foraminifera are marine and marginal-marine single-celled organisms that are highly selective of their habitat. The sensitivity and abundance of these organisms makes them the ideal candidate for the study of ecological impacts surrounding a point source of brine solution in an estuary.

The majority of previous studies have used macrobiota to investigate impacts from desalination discharges (e.g. Mabrook, 1994; Gacia et al., 2007; Ruso et al., 2007; Kleber, 2010). The use of Foraminifera has several advantages over the use of these larger organisms. Foraminifera are a diverse order of the Protista kingdom, and play an important role in the trophic structure of estuaries linking autotrophs to larger micocarnivores and macrofauna (Lipps and Valentine, 1970). Foraminifera are generally less mobile than macrofauna which limits their ability to enter or leave an environmentally disturbed area (Alve, 1999). Their small size and abundance in most marine and marginal-marine systems can yield statistically significant results in small sediment samples (Phleger, 1960). Additionally, Foraminifera build stout tests that can be preserved in sediment for millions of years providing insight into the environmental conditions prior to the operation of a WTP/modern human influence (Haynes, 1981; Alve et al., 2009; Schönfeld et al., 2012). Foraminifera are particularly selective of their habitat, specifically with respect to salinity (e.g. Culver and Horton, 2005; Kemp et al., 2009).

This project was designed to addresses gaps in research on the impacts of brine solution discharge from desalination WTPs on marginal-marine environments. Previous research has focused on direct physical and chemical measurement of discharge and impacts to macrofauna and flora (e.g. Fernández-Torquemada et al., 2005; Ruso et al., 2007; Ward, 2008; Kleber, 2010).

The goal of this project was to provide quantitative measures of ecological impacts from a RO-WTP, where the results could be used as a basis for future assessment and regulation of concentrated brine solution discharge into estuaries. It was predicted that even minor changes in estuary salinity would be reflected in the lower-trophic, Foraminiferal populations, providing the observer with an early indicator and baseline for later comparison of ecosystem modification.

CHAPTER 2: PREVIOUS LITERATURE

2.1 Foraminifera

Foraminifera have been recognized by humans for thousands of years, but it was not until the 19th century that they began to be classified and used in scientific pursuits (Prothero, 2004). Research using Foraminifera began to grow considerably in the early 20th century due to their usefulness in biostratigraphy, specifically in hydrocarbon exploration (Prothero, 2004). By the mid-to-late 20th century Foraminifera were a global index fossil for biostratigraphy of late Mesozoic and Cenozoic strata (Prothero, 2004). Many of the same characteristics of Foraminifera that have been useful in biostratigraphy are proving equally useful in the more recent endeavors into paleoclimatology and environmental assessment.

Foraminifera, as an Order, are widespread through space and time, yet at the genus or species level, many have small temporal and/or geographic ranges. Foraminifera date from the early Cambrian and are currently distributed globally in nearly all marine and marginal-marine environments (Murray, 1981; Sen Gupta, 2003a). Additionally, a limited number of species are able to survive in freshwater environments (Holzmann et al., 2003). Foraminiferal tests can be preserved in the sediment long after death and the distinct morphology between species allows for identification by researchers (Haynes, 1981; Sen Gupta, 2003b).

Foraminifera range in size from a few micrometers to a few centimeters but most are between one-tenth of a millimeter and one millimeter (Haynes, 1981; Prothero, 2004). Populations can exceed two and a half million individuals per square meter of sea floor (Phleger, 1960). In some areas of the ocean, the majority of the sea floor is made of calcareous tests of Foraminifera (Prothero, 2004). Due to their small size and abundance, small sediment or water samples (cubic centimeters) are able to yield statistically significant distributional data.

It is estimated that there are over 3,600 described genera and 60,000 recognized species with perhaps one sixth of those species extant (Sen Gupta, 2003a; Prothero, 2004). The high taxonomic diversity arises from the short reproductive cycles of Foraminifera, ranging from days to a few years (Armstrong and Brasier, 2009). The short reproductive cycles also encourages rapid response to environmental changes within an assemblage.

Foraminifera have proven useful to researchers because of their high species diversity and abundance in many environments. The Order as a whole is very adapt to thriving in a wide range of marine and marginal-marine habitats, but at the species level, Foraminifera are highly selective of their habitat (Schafer, 2000). These characteristics have contributed to the relative ease of sampling and identification that have provided a financial (e.g. for hydrocarbon exploration) and labor advantage over other methods of environmental analysis and biostratigraphy (e.g. macrofossils).

Observations have shown that Foraminifera exhibit considerable sensitivity to environmental factors including salinity, temperature, dissolved oxygen, nutrients, tides, competition for space, food supply (organic matter), physical disturbances, substrate/grain-size, and controls on anoxia (Murray, 2006; Armynot du Châtelet et al., 2009). Researchers are able to record changes in the environment by observing variations in Foraminiferal assemblages (Murray, 2001). This can be achieved by one of two approaches. One approach relies on evidence of select environmental parameters, such as oxygen isotopes, that are incorporated into the tests of calcareous Foraminifera during growth. Later the researcher can infer abundance of a chemical element in the environment at the time of growth through chemical analysis of the tests (Murray, 2001). A second approach relies on physical changes observed in assemblages due to changes in the habitat. Researchers use relative species distribution and abundance, changes in test morphology, pyritization (fossilization), and biological response of cytoplasm (cell structure) to understand environmental changes (Boltovskoy et al., 1991; Yanko et al., 1994; Yanko et al., 2003).

2.2 Foraminifera as Bioindicators of Pollution

There is a lack of agreement on the use of bioindicators for monitoring environmental change. Some researchers argue that the use of bioindicators is unreasonable in most cases because the narrow niche in which most organisms live makes interpretation over space and time difficult (Cushman et al., 2010; Lindenmayer and Likens, 2011). Others note that direct measurement of chemical and physical parameters is simplistic in nature because it ignores many potential environment-altering factors, and that bioindicators can be useful (Rodrigues and Brooks, 2007; Lindenmayer and Likens, 2011). Though both views have merit, the use of bioindicators provides a more holistic view.

Because of the complex nature of ecosystems, direct measurement of a few select variables is not likely to provide an accurate assessment of the ecological health. Though there are limitations, direct measurement can be beneficial, but repeated sampling is necessary due to spatiotemporal gradients within an estuary. Many natural and anthropogenic contaminants of estuary waters eventually sink to the sediment-water interface where they are buried and stored (Frontalini and Coccioni, 2011). Because pollutants are stored here, benthic bioindicators can be valuable indicators of contamination.

Ambient environmental conditions can act to mask or compound Foraminiferal response to pollutants, complicating observational studies (Yanko et al., 2003). A taxon's preferred habitat with respect to any one pollutant likely includes a range of acceptance (Alve, 1995). Therefore, the proximity of the ambient levels of a particular pollutant to either end of a species tolerance-spectrum plays an important role in the observed species tolerance to a pollutant (Alve, 1995). Additionally, near-shore environments are characterized by large spatiotemporal gradients in environmental parameters (e.g. salinity, temperature) as marine and continental waters meet (Yanko et al., 2003). Separating changes due to pollution from those changes due to natural variability can be a difficult task. In addition to varied natural conditions, pollution within a study area is often from a conglomeration of sources with multiple pollutants from individual points, making isolation of the impacts from a single parameter difficult (Yanko et al., 2003). Also, response to pollutants is likely non-linear, especially when considering multiple pollutants (Alve, 1995).

Over half a century has passed since the first published evidence of Foraminiferal sensitivity to anthropogenic pollution and the subsequent use of Foraminifera as proxies for environmental disturbances (Zalesny, 1959; Watkins, 1961). Since that time, the volume of literature pertaining to the use of Foraminifera for environmental monitoring has steadily increased. Research has described Foraminiferal response to a variety of environmental changes both natural and anthropogenic. Most literature has focused on population dynamics surrounding point sources of organic waste (e.g. from sewage, pulp and paper mills, aquaculture, and fertilizers). Other research has examined impacts from oil and other hydrocarbons, thermal pollution, coal and fuel ash, radioactive waste and various industrial chemicals (including potentially toxic elements) (Alve, 1995; Yanko et al., 2003; Martínez-Colón et al., 2009). The majority of studies have been conducted in near-shore and marginal-marine environments—the area of overlap of major anthropogenic impacts and Foraminiferal habitats (Yanko et al., 2003).

Organic pollutants from municipal sewage, agriculture, aquaculture, and pulp and paper mill point sources have repeatedly been shown to trigger a simple pattern in Foraminiferal populations (Figure 2.1). Even with the varied sources and discharge constituents, studies have consistently shown that Foraminifera respond positively in the form of increased population density and species richness to a limited amount of nutrient-enriched organic effluents (Alve, 1995; Yanko et al., 2003). In a study in the eastern Mediterranean along the coast of Israel, Yanko et al. (1994) found increased species diversity, increased populations and the largest test sizes at a site of domestic sewage discharge. The hypertrophic zone was likely due to the influx of biologically available nutrients from organic waste, however food in the form of bacteria associated with abundant fecal matter from large populations of macrofauna in this area could also have been a contributing factor (Alve, 1995).

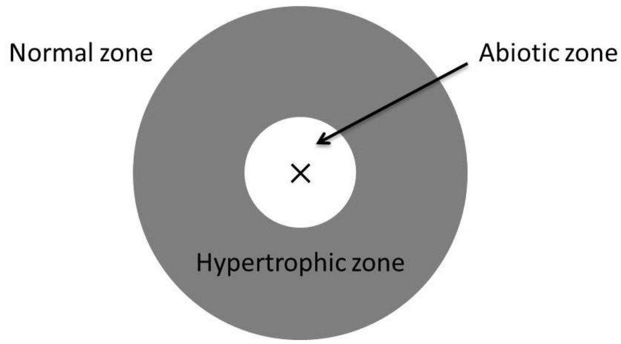

Figure 2.1: Idealized diagram of an organic point source (×) and surrounding biotic zones as described in Alve (1995).

There are limits to the increased biotic activity, however, in the most extreme pollution cases the enrichment can lead to an abiotic zone surrounding the discharge (Alve, 1995; Yanko et al., 2003). In a study in Winyah Bay, SC, Collins et al. (1995) found that Foraminifera disappeared from the bay near the site of high organic pollution, but their population increased downstream towards the mouth of the bay. The abiotic zone of the water in the immediate vicinity of a point source arises from extreme oxygen depletion from eutrophication (Yanko et al., 2003). In addition, predator population changes could also be an important factor in controlling Foraminiferal populations surrounding organic pollution outfalls (Alve, 1995).

Research into the effect of oil and other hydrocarbons on Foraminiferal populations has yielded conflicting results. Findings range from minimal impact to dramatic changes in diversity and abundance and significant increases in morphological deformities (Yanko et al., 2003; Brunner et al., 2013). In laboratory experiments, Morvan et al. (2004) noted abnormalities in reproduction, cell structure, and morphology of Foraminifera exposed to oil. In a field study following the Erika oil spill off the coast of France, Morvan et al. (2004) did not find significant deformities, but instead found low population density and diversity. The researchers attributed the low populations and diversity to either the pollution or the cleanup efforts. In a study along the U.S. Gulf of Mexico coast after the Macondo (British Petroleum) well blow out, Brunner et al. (2013) found a similar population abundance pattern to that found at organic outfalls. They found increased populations (3-5 times) over the control at a slightly polluted site, but a significant decrease in population at a highly polluted site. Alve (1995) suggests that hydrocarbons likely influence Foraminiferal food supply and predator populations. Hydrocarbon pollutions is associated with the release of hydrogen sulfide, methane and ammonia, and increased acidity (Alve, 1995).

Impacts from potentially toxic elements (PTEs) are difficult to assess because they rarely occur in isolation. PTE pollution is generally associated with decreased species diversity, decreased population, and increase in percentage of deformed tests (Yanko et al., 1994; Martínez-Colón et al., 2009). In an assessment of sediment cores in Baltimore Harbor, Maryland, Ellison et al. (1986) found that Foraminifera were almost entirely eliminated in the harbor as copper, zinc and vanadium pollution increased. Population density and diversity dynamics (most significantly, *Ammobaculites* spp. density) indicated decreasing impacts from pollution downstream of the harbor (Ellison et al., 1986). As pollution in the harbor was reduced, a return towards normal populations were observed (Ellison et al., 1986). Though understanding of the impacts from PTEs is growing, the bioavailability of PTEs varies depending on natural environmental conditions (e.g. salinity, pH, temperature), and this complicates the understanding of ecological impacts (Martínez-Colón et al., 2009).

Published research on additional pollution sources is limited (Yanko et al., 2003). Pesticides and other chemical pollutants have been observed to have similar impacts on populations as that of PTE pollution (Yanko et al., 2003; Frontalini and Coccioni, 2011). Thermal pollution has been shown to increase population density while decreasing diversity (Alve, 1995; Yanko et al., 2003). Effects of radioactive waste on assemblages have been inconclusive, and fuel ash pollution likely decreases food supply, limiting reproduction and diversity (Yanko et al., 2003).

2.3 Desalination Impacts on the Environment

A limited number of studies have investigated the impacts of wastewater from desalination WTPs on the environment (e.g. Chesher, 1975; Abdul-Wahab, 2007; Safrai and Zask, 2008; Kleber, 2010). Some of these studies include analysis of perceptible intensity and extension of mixing of wastewater in receiving waters (e.g. Chesher, 1975; Talavera and Quesada Ruiz, 2001; Gacia et al., 2007; Kleber, 2010). Studies found the extension of increased salinity due to discharge of brine solution into marine environments ranged from less than 10 m from the outfall to as far as 4 km (Fernández-Torquemada et al., 2005; Raventós et al., 2006). Plume extension and intensity appears to be strongly controlled by discharge rate and local winds, waves and water currents. Other likely important factors include presence of human engineered marine structures, coastline topography, bathymetry, and degree of disparity between the chemical and physical properties of the effluent and the receiving waters (Table 2.1). At a site in Camden County, North Carolina, Kleber (2010) found no evidence of plume extension beyond 25-50 m from the discharge of a RO-WTP, but noted plume extension and direction was highly dependent on local winds. She also found that the plume was most often apparent ~1.5 m below the water's surface (Kleber, 2010).

In what is perhaps the most comprehensive study to date, Chesher (1975) found an increased concentration of salts contained in the bottom waters of an artificial harbor that received discharge from a desalination WTP. With continued discharge, the concentrated brine would overflow out of the lower harbor bowl into the navigation channel that led to the ocean. Evidence of brine concentrate was observed in the channel ~1,500 m from the discharge.

In addition to the study of physical or chemical parameters surrounding outfall sites of desalination WTPs, a few studies have attempted empirical assessment of ecological impacts at affected sites. Ecological assessments at outfalls of concentrated brine solution have characterized a variety of non-Foraminifera aquatic biota: vegetation, benthic and planktonic fauna, fish, and coral reef systems (e.g. Chesher, 1975; Fernández-Torquemada et al., 2005; Raventós et al., 2006; Ruso et al., 2007; Ruso et al., 2008). Results from these studies range from no significant impact to complete loss of species from the disturbed environment (e.g. Fernández-Torquemada et al., 2005; Raventós et al., 2006). The majority of cases found moderate reduction in vegetative health, or reduction in populations of benthic or planktic organisms (e.g. Chesher, 1975; Mabrook, 1994; Sánchez-Lizaso et al., 2008).

The degree of impact to the ecology in the region is intrinsically linked with the plume extension and intensity. In a review of available literature, Roberts et al. (2010) found that discharge location was the single most critical factor determining ecological impact to a marine environment. A common result of impact assessment studies is differing sensitivity of organisms to the brine solution resulting in changes in population dynamics (e.g. Crockett, 1997; Ruso et al., 2007; Ruso et al., 2008). In a study along the southeast coast of Spain, Ruso et al. (2007) found a substitution of dominant organisms near the outfall of a desalination WTP. The control site was dominated by a relatively even distribution of Polychaetes, Crustaceans, and Molluscans, but near the discharge site, Nematodes were the dominant organisms (up to 95%). In a later study at the same site, Ruso et al. (2008) found Polychaete species had differing sensitivity to pollution that created significant variations in species diversity between the control and disturbed sites. In the study conducted in Camden County, North Carolina, only minor changes in two species of macroinvertebrates within 5 m of the RO-WTP discharge pipe were observed (Kleber, 2010). In that study, the researcher investigated the impacts of the discharge on macroinvertebrates, macrozooplankton, and nekton (Kleber, 2010).

Because of the limited number of published results from empirical studies, concerns remain regarding environmental impacts from concentrated brine solutions discharged from desalination WTPs (e.g. What are the impacts to lower trophic populations?). A critical review of environmental impacts from desalination plant discharge by Roberts et al. (2010) found that most of the literature on the subject is discussion orientated with few papers based on empirical results. Of the literature based on monitoring studies, many lack descriptive methods and/or clear results making the validity of the conclusions difficult to assess. Roberts et al. (2010) suggests that more before-and-after impact assessments, as well as manipulative field experiments, are needed to increase the understanding of the potential environmental consequences of concentrated brine from desalination WTPs discharged into marine and marginal-marine environments.

2.4 Foraminiferal Response to Briny Solutions

After an extensive literature search, only two published studies were found to have used Foraminifera in their ecological assessment of discharges from desalination facilities (Chesher, 1975; Hammond et al., 1998). In a report for the Southwest Florida Water Management District at a RO-WTP discharge site in Antigua, Hammond et al. (1998) found higher concentration of salts within tens of meters of the outfall, but no changes in the Foraminiferal population were attributed to the increase in salinity. Instead, Hammond et al. (1998) attributed minor changes in Foraminiferal assemblages to reduced oxygen content of sediment and water. The report did not include an assessment of the site prior to the WTP operation nor did it include a control site (Hammond et al., 1998).

In another study where Foraminiferal populations were investigated, Chesher (1975) found significant impacts to the microfossils in an artificial harbor in Key West, Florida. Foraminiferal density decreased in close proximity to the discharge from a desalination WTP, but greater densities than at the control site were observed in the remainder of the harbor. However, the desalination WTP that was the focus of the study had additional pollution issues, including copper discharge that created an environment where copper concentrations where often 5-10 times higher than the control concentrations (Chesher, 1975). Much of the adverse impact to the biota surrounding this discharge was likely caused by copper pollution.

Table 2.1: Description of plume extent and ecological impacts from desalination plant discharges.

Reference	**Location**	**Discharge Rate (m^3/D)**	**Salinity of Environment (ppt)**	**Salinity of Discharge (ppt)**	**Plume Extension**	**Biological Impact**
Kleber (2010); Rulifson et al. (2006a)	Camden County, NC	757	1.0-5.0	10.2-15.2	Not detectable beyond 25-50 m. Extension and location highly dependent on local wind	Decreased density of two species of macroinvertebrates within 5m of outfall and increased density outside of 5 m. No effect on macrozooplankton or nekton
Ward (2008)	Tyrrell County, NC	NR	.06-.1	39.6	NR	Tree stress and mortality in wetland downstream from discharge, but no salt accumulation
Altayaran and Madany (1992)	Sitra Island, Bahrain	288,000	45	51	160 m; mixing may have been hampered by jetties	NR
Talavera and Quesada Ruiz (2001)	Canary Island	17,000	37-38	75	Close to background within 20 m; later study (in Einav et al., 2002) found a "stream" of higher salinity water along bottom of ocean up to 60ppt, 100m away	NR
Fernández-Torquemada et al. (2005)[a]	Alicante, Spain	75,000	38	68	<0.5 ppt increase up to 4km away	Echinoderms disappear from outfall location. *Decrease in Posidana oceanica* shoot-division rate
Ruso et al. (2007)[1]	Alicante, Spain	65,000	38	68	NR	Distribution of Polychaetes Nematodes and Bivalves changed from evenly populated to heavily weighted to Nematodes (<98%)
Ruso et al. (2008)[1]	Alicante, Spain	65,000	38-39	68	NR	Decrease diversity of Polychaeta based on Family sensitivity to effluent.
Raventós et al. (2006)	Blanes, Spain	32,877	NR	60 g/L	Not detectable le at 10m	No significant impacts found; high dilution rate and high natural variability
Safrai and Zask (2008)	Ashkelon, Israel	183,561	39.7	NR	3% above at 500m and 1% at ~3km	NR
Gacia et al. (2007)	Spain (western Mediterranean)	2,000	37	41-60 (~50)	~50m	Decreased shoot density of *Posidana oceanica* in patchy meadows, but in continuous meadows shoot density was similar at all sites. Necrosis significantly more prevalent in all meadows (2% at control, 17% at discharge)

Table 2.1: (continued)

Reference	Location	Discharge Rate (m^3/D)	Salinity of Environment (ppt)	Salinity of Discharge (ppt)	Plume Extension	Biological Impact
Abdul-Wahab (2007)	Oman (Gulf of Oman)	max design: 122,100 (current NR)	37	$\geq$ 40ppt	~100 m	NR
Crockett (1997)	McMurdo Station, Antarctica	1,500 at max population	NR	1.2 X environment	NR	Relative abundance of diatom genus *Fragilariopsis* increased to >97.5 near outfall, while only 30%-82.3% at control sites. Significant decrease in Chlorophyll-*a* near discharge
Sánchez-Lizaso et al. (2008)	Near Murcia, Spain	NR	37-38	NR	NR	Significant impact to *Posidana oceanica* only 1-2ppt above ambient. Increase spatial variability and mortality and general decrease in health around outfall
Mabrook (1994)	Eastern Egypt	NR	NR	NR	NR	Most coral disappeared from coastal areas. Many planktonic organisms disappeared. Decreased fish population; some species disappeared
Chesher (1975)	State harbor, Key West, FL	21,803	34.6-38.0	50 (40-55)	Detectable up to 600m from harbor turning basin (~1500 m from outfall) Copper 5-10 times higher than ambient (absorbed into sediment).	Most organisms adversely affected (sea squirts various algae, bryozoans, sabellids, lamellibranches, echinoids). Shells of oysters and clams found but no living specimens found. Foraminifera increase in density in harbor, but decreased in close proximity of outfall (Similar number of species)
Hammond et al. (1998)	Antigua, Antigua and Barbuda	Max: 5,564 (current NR)		57	~16m, dependent on winds and tides	No Foraminifera changes attributed to salt discharge. Some Foraminifera population changes attributed to reduced oxygen. No significant impact to chlorophyll, seagrass, macro-fauna, or benthic microalga communities

[1] – Denotes same desalination plant; NR = Not Reported

CHAPTER 3: SITE DESCRIPTION

The RO-WTP that is the focus of this project is located in Currituck County, North Carolina. The RO-WTP is located on the county mainland, separated from the Atlantic Ocean by Currituck Sound and a narrow barrier island. The discharge site is located near the mouth of the North River off the western shore of the mainland peninsula that extends southward into Albemarle Sound (Figure 3.1). Below is a description of the hydrology and geology of the area with respect to ecological controls and RO-WTP operations.

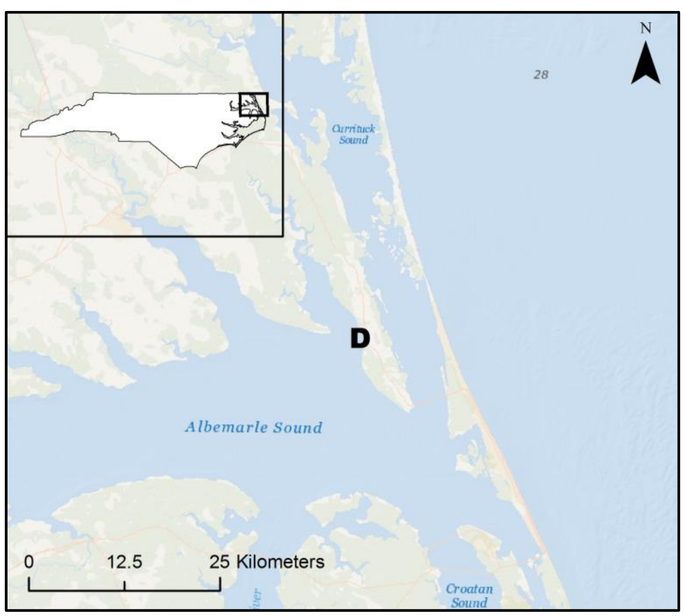

Figure 3.1: Location of Currituck County RO-WTP wastewater discharge site (D) near the mouth of the North River at Albemarle Sound.

Albemarle Sound is a brackish estuary system in northeastern North Carolina that separates the mainland from the Outer Banks. Albemarle Sound drainage basin is the largest in North Carolina, draining ~47,500 km^2 of northeastern North Carolina and southern Virginia. The sound in an extensive area of shallow water covering ~1,243 km^2 (Giese et al., 1985). The maximum depth of the sound is just over 9 m, however average depth is less than 5.5 m (Giese et al., 1985; Wells and Kim, 1989).

Albemarle Sound is protected from the Atlantic Ocean by a continuous strip of land (the Outer Banks) with no direct water exchange between Albemarle Sound and Atlantic Ocean. All exchange of surface water between the Atlantic Ocean and Albemarle Sound is through inlets located in Pamlico Sound. Oregon Inlet is the closest inlet, located south of Roanoke Island at the northeastern boundary of Pamlico Sound. Astronomical tides are negligible in most interior areas of the sound (Giese et al., 1985). Instead, winds are a significant controlling mechanism of local water level and circulation patterns (Giese et al., 1985). Freshwater input from the tributaries is a minor controlling mechanism of local water level, but contributes to the long-term water budget of the estuary (Giese et al., 1985).

The restricted flow of water to the ocean constricts conventional estuary circulation, limiting ocean water intrusion and salinity in Albemarle Sound. Western portions of the sound are nearly fresh (<0.5 part per thousand), whereas salinity in the east increases to more than 10 parts per thousand (ppt) (Giese et al., 1985). Salinity in the sound is inversely proportional to stream discharge and it is historically lowest in March due to increased freshwater runoff from spring rains and highest in December after the minimal autumn precipitation (Bowden and Hobbie, 1977; Giese et al., 1985).

Ambient salinity was ~2.0 ppt in the required monthly assessment at the Currituck RO-WTP discharge site conducted in March of 2014 (Table 3.1). In a 6 month pre-assessment (July-December) report at the Currituck County discharge site, Rulifson et al. (2006a) found average monthly salinities ranged from 2.13 ppt in July to 7.10 ppt in October.

Table 3.1: Currituck County Mainland RO-WTP March 2014 North River outfall samples, measured 3/20/2014 (Pat Irwin, Currituck County Public Utilities, personal communication, 2014).

	Temperature (°C)	pH	Dissolved Oxygen (mg/L)	Conductivity (µS/cm)	Salinity (ppt)
Upstream	11.0	6.82	10.9	3,570	1.9
Downstream	9.5	7.32	11.10	3,810	2.0

Water temperature of the sound closely follows air temperature, with a minimum in January (3-4°C) and a maximum in July (~28°C) (Bowden and Hobbie, 1977). Rulifson et al. (2006a) found average monthly water temperatures at the discharge site ranged from 29°C in July to 8°C in December. Dissolved oxygen is abundant, often 80-90% of saturation (Bowden and Hobbie, 1977). Because of sufficiently robust wave and wind-driven tidal mixing, and the shallow nature of the estuary, only minor, infrequent stratifications are observed in temperature, salinity or dissolved oxygen (Bowden and Hobbie, 1977; Giese et al., 1985; Riggs, 1996; Rulifson et al., 2006a).

Albemarle Sound is underlain by unconsolidated Pliocene and Pleistocene sediments that were deposited as ocean waters advanced and retreated across the landscape during oscillating periods of glaciation and deglaciation (Riggs, 1996). The thickness and depth of these deposits increases seaward. Thickness ranges from a few meters below the western sound to 70 m in the east toward the barrier islands (Riggs, 1996). Underlying these materials is Cretaceous and older Tertiary sediment and sedimentary rock that increase in thickness and dips down to the east in a similar fashion as the overlying sediment (Lautier, 2012).

The bathymetry of the sound and its tributaries has been described as a "shallow, flat-bottom dish" (Riggs, 1996). The shallow shelves along the edges of the sound are covered with recently eroded quartz-rich sand from the Pleistocene shore (Riggs, 1996). The interior basin is generally covered with Holocene material. Due to the weak flow rates in the low gradient tributaries, sediment load is small and consists of predominately organic-rich mud derived from the extensive deciduous swamp forests of the region (Copeland et al., 1983). Likely after multiple iterations of deposition and re-suspension, the mud settles creating a substrate dominated by organic rich muds in the interior regions of the sound (Riggs, 1996). The accumulation of these sediments is controlled by the depth of the wave energy in the sound and ultimately sea-level (Wells and Kim, 1989; Riggs, 1996).

The two processes of (a) recent shoreline erosion and (b) weak stream discharge have created a basin dominated by two dissimilar sediments, (a) a chemically inactive, quartz-rich sand, and (b) a chemically reactive clay-dominated, organic-rich sediment. Secondary contributing sources of sediment to the sound include *in situ* biogenic production and wind-blown coarse sand in the areas directly west of the barrier islands (Wells and Kim, 1989).

The Currituck RO-WTP is located in Maple, NC and draws water from the Yorktown aquifer from two wells that penetrate ~75 m into the ground (Pat Irwin, Currituck County Public Utilities, personal communication, 2014). This aquifer is the uppermost aquifer below the surficial aquifer and is contained in the upper Pliocene

Yorktown Formation and the Miocene Pungo River Formation (Riggs, 1996; Lautier, 2009). As with the other confined aquifers of the North Carolina coastal plain, (Pungo River, Castle Hayne, Beaufort, Peedee, Black Creek, Upper Cape Fear, and Lower Cape Fear) the Yorktown aquifer is most narrow, closest to the surface, and least saline at the western boundary (Riggs, 1996; Lautier, 2009). Towards the Atlantic Ocean the aquifer increases in thickness, depth, and salinity (Riggs, 1996; Lautier, 2009). The aquifer is narrower than 6 m in many places west of the study site, but increases to over 91 m just east of the site in Dare County (Winner and Coble, 1996). The top of the aquifer dips to the east at a rate up to ~3.4 m/km (Winner and Coble, 1996).

Salinity of the source groundwater at the Maple facility is ~1.8 ppt (derived from conductivity) (Table 3.2) (Pat Irwin, Currituck County Public Utilities, personal communication, 2014).

Table 3.2: Samples of raw well water and “clean,” permeate water that has passed through the membranes. Values recorded in May 2014 (Pat Irwin, Currituck County Public Utilities, personal communication, 2014).

Pollutant	Well Water	Permeate Water
Chlorine (free Cl2), mg/L	0.01	0.01
Conductivity as µS/cm	3,350	84.4
Iron, Fe, mg/L	0.333	0
Manganese, Mn, mg/L	0.012	0
Chloride as Cl_2, mg/L	965	25
Nitrate as NO_3, mg/L	0	0.3
Zinc as Zn, mg/L	0	0
pH	7.91	5.82
Total Dissolved Solids, mg/L	1,957	44.4
Sulfate as SO_4, mg/L	5.8	0
Sodium as Na, mg/L (est.)	389.3	22.8
Temperature as C°, Celsius	20.5	20.0

The wastewater from the Maple RO-WTP is pumped ~34 km south along the mainland peninsula to the outfall site (Carson et al., 2009). The plant currently operates and discharges ~12 hours per day (Pat Irwin, Currituck County Public Utilities, personal communication, 2014). The outfall is located ~560 m into the North River from the western shore of Currituck County, ~2.6 m below the surface of the water (Figure 3.2) (Pat Irwin, Currituck County Public Utilities, personal communication, 2014). Immediately shoreward of the discharge, water depth decreases rapidly to ~1 m. A 16-m long and 36-cm diameter pipe with discharge ports acts as a diffuser for the effluent (Carson et al., 2009).

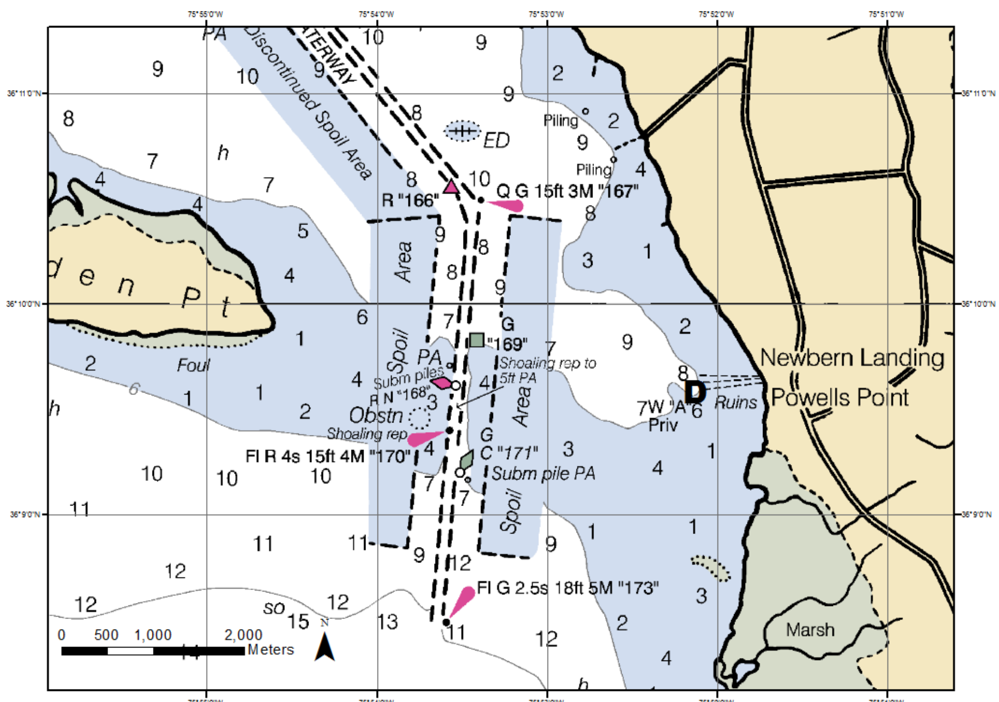

Figure 3.2: Discharge location (D) in the North River, ~560 m from the Currituck County shore. Depths are given in units of feet (divide by 3.281 for meters). (U.S. National Oceanic and Atmospheric administration; www.charts.noaa.gov)

Measurement of water depth taken in a 200 m by 200 m grid surrounding the discharge site indicate that water depth increases from east to west and northwest across the sampling site (Figure 4.1). Depth ranges from a minimum of ~2.3 m at the NE point to a maximum of ~2.9 m at the NW point. Average water depth in the grid is ~2.6 m.

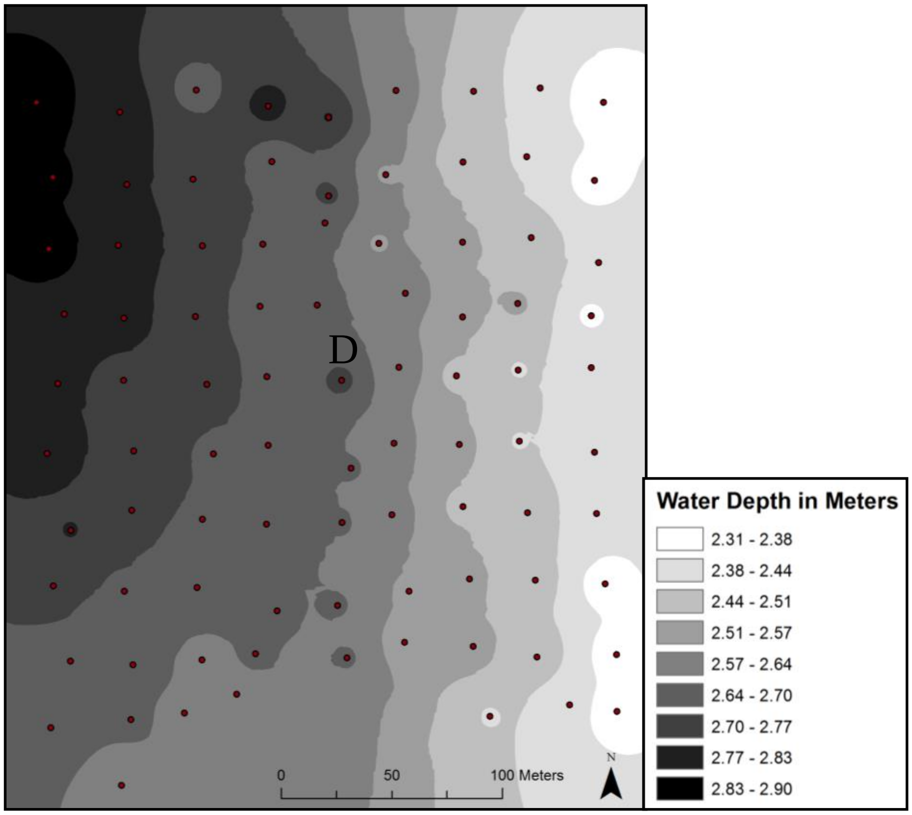

Figure 3.3: Locations of measurements of water properties are marked by red circles. Water depth (m) is contoured surrounding the discharge site (D).

Currently the WTP only operates at partial capacity (757 m^3/D effluent) (Pat Irwin, Currituck County Public Utilities, personal communication, 2014). As required, the capacity will be expanded to the existing infrastructure limits. At plant build-out (~6,321 m^3/D discharge), 12 (~5cm diameter) diffuser ports will act to encourage mixing of effluent; currently only 4 ports are in use, the remaining 8 are capped (Carson et al., 2009).

Major differences between the effluent and the receiving waters are evident with respect to dissolved oxygen and conductivity (Tables 3.1 and 3.3). Dissolved oxygen is much lower in the effluent, ~1/5th of dissolved oxygen in the receiving waters. Salinity of the effluent is nearly five times that of the receiving waters. The effluent is more acidic than the receiving waters (6.82-7.87). Temperature was much higher in the effluent in the March 2014 sample, but as previously described, surface water temperature fluctuates considerably with the seasons. Ammonia concentration is drastically higher (over an order of magnitude) in the effluent than in surface waters based on samples taken 12 km upstream of the discharge site from March to September of 2013, (U.S. Geological Survey, 2014). Ammonia at the upstream USGS site was less than 0.01 mg/L (as N) for four of seven dates of sampling the other three were 0.022, 0.011 and 0.012 mg/L (as N) (U.S. Geological Survey, 2014).

Table 3.3: Currituck County Mainland RO-WTP March 2014 effluent sample. Average daily discharge is an average daily rate for the month of March; all other values are a onetime sample taken 3/20/2014 (Pat Irwin, Currituck County Public Utilities, personal communication, 2014).

Average Daily Discharge (m3/D)	757
Cl_2 Residual (µg/L)	<1
Dissolved Oxygen (ml/L)	2.22
pH	7.87
TDS (mg/L)	9,500
Chloride (mg/L)	4,950
Manganese (mg/L)	<0.010
Iron (mg/L)	1.33
Zinc (mg/L)	0.02
Temperature (°C)	17.7
Salinity (ppt)	9.4
Conductivity (µS/cm)	16,300
NH_3-N (mg/L)	17

CHAPTER 4: METHODOLOGY

4.1 Field Methodology

This project involved the collection of two types of field data: (1) measurements of physical water properties near the outfall of the briny solution and (2) sediment samples for Foraminiferal counts, grain-size analysis and analysis of percent organic matter (OM).

Measurements of water temperature and conductivity were collected within 10 cm of the sediment-water interface using a YSI model 63 handheld pH and conductivity meter. Water measurements were taken in a grid surrounding the discharge on eight days from August to November of 2014. Water measurements were located within a 200 m by 200 m grid centered over the outfall. The number of sampling points in the grid varied by workload and impact of meteorological conditions on sampling rate (i.e. fewer were taken on days when sediment cores were taken or waves slowed progress) and ranged from 20 to 95 (mean=49). The model grid contained 91 sampling points (Figure 4.1). Water depth was recorded during every measurement of water conditions. Supplementary water conductivity measurements of in-plant discharge water were obtained from the RO-WTP operator and were used in some analyses.

Collection and analysis of Foraminiferal samples followed as closely as practicable the guidelines proposed by the FOBIMO (FOraminiferal BIo-MOnitoring) initiative (Schönfeld et al., 2012). The FOBIMO initiative held an inaugural workshop in

2011 in Switzerland that included 37 scientists from 13 countries (Schönfeld et al., 2012). The central goal of the initiative was to define standardized methods for the collection and assessment of Foraminifera for use in bio-monitoring studies that are scientifically sound and allow for practical application (Schönfeld et al., 2012).

A total of 29 sediment cores were taken using a Wildco core sampler (~5-cm diameter). Seventeen cores were taken for primary Foraminiferal analysis: one at the outfall, 16 at distances of 25, 50, 75 and 100 m from the outfall along the 4 cardinal directions (Figure 4.2). Ten (5 X 2) cores were taken and used for micro-variability analysis at two locations approximately 75 m from the discharge site. Each of the five cores at one micro-variability site was within 10 m of the other four cores. Two cores were taken at a control site located >4 km from the outfall. Cores were collected on two separate occasions: the cores used for micro-variability analysis were collected August 27, 2014; the remaining 19 cores were collected November 30, 2014. Two samples were removed from each core immediately following collection. A 0-1 cm depth sample and a 10-11 cm depth sample were placed in plastic bags and sealed to retain moisture. The surface samples were used for modern impact assessment, and the 10-11 cm depth samples were used for pre-impact assessment. Corbett et al. (2007) collected down-core radiometric data at a site 3.65 km northwest of the outfall used in this study. That site was at a similar water depth (2.7 m) and distance from the eastern shoreline (~800 m) as the discharge location in this study. They reported sediment found at 10-11 cm was deposited ~90-100 years before present.

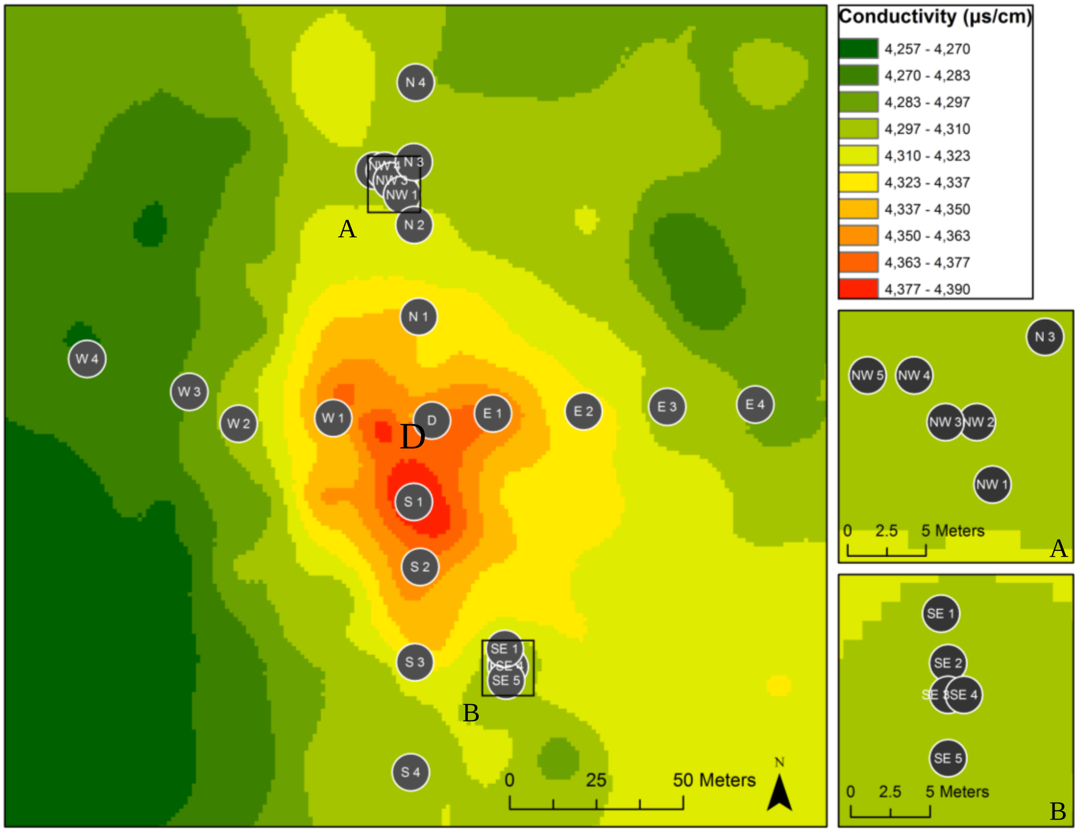

Figure 4.1: Circles represent locations where sediment cores were collected. Contours represent average temperature corrected conductivity values surrounding the discharge site (D) on the six days from 9/15-9/17 and 9/29-10/1.

Samples are labeled with a set of unique alphanumeric characters indicating direction and distance from the discharge followed by a decimal and either a “0” or a “1.” Samples ending in “0” are surface samples; samples ending in “1” are down-core samples.

4.2 Laboratory Methodology

Microfossil analysis involved placing a portion of unaltered, wet sediment on a tray and systematically scanning the entire tray with a binocular microscope. A minimum of 100 specimens were identified to the genus level. Foraminiferal taxonomy was determined based on comparison with figures in published literature (e.g. Loeblich and Tappan, 1988; Vance et al., 2006). For each sample, representative specimens of each taxon identified were archived on a slide for reference. After 100 specimens were identified, the inspected portion of the sample was air-dried and weighed. Weights of inspected portions were then used to convert counts to a standard sample size of one cubic centimeter of wet volume. This was done by comparing weights of inspected portions to air-dried weights of larger portions of the original samples where volumes could more easily be determined.

Organic matter content was determined by mass of sample before and after incineration. Portions of air-dried sediment samples varying from 0.5-0.75 g were crushed to clay-size grains with a mortar and pestle and oven-dried at 110°C for a minimum of 24 hours. The samples were then placed in dried crucibles of known weight and initial weight was recorded. Crucibles and samples were placed in a furnace at 550°C for 1.5 hours then removed and placed in a desiccator to cool. After ~15 minutes, a final weight was recorded to calculate %OM. A duplicate sample was analyzed every ninth sample for error assessment.

Prior to grain-size analysis, organics were removed by digestion with 2 ml of 30% H_2O_2; 10 ml of 30% H_2O_2 was used for two highly organic samples. The procedures used are based on commonly accepted guidelines for organic digestion (Schumacher, 2002). Organic digestion was performed on each sample using the following procedure with the exception of two samples that contained high percentages of organics: approximately 1 g of sediment was hydrated with deionized water in a 250 ml beaker on a shaker table. After 15+ minutes, 0.25 ml of 30% H_2O_2 was added to the mixture and allowed to mix on the shaker table. Fifteen minutes after the initial addition of H_2O_2, the beakers were moved to a ~90°C hot bath and an additional 0.25 ml of 30% H_2O_2 was added. Hydrogen peroxide was added two more times with 15 minutes between additions. At 75 minutes from initial H_2O_2 addition, 1 ml of H_2O_2 was added to the beakers, in four 0.25 ml additions, with ~1 minute between additions. Beakers were monitored for excessive evaporation or boil-over and replenished with deionized water when needed. Beakers remained in a hot bath until reaction had significantly diminished (~120 min in total). Upon observing diminished reaction, beakers were moved to an oven for drying. After 24+ hours at ~110°C samples were weighed, hydrated with 10 ml of deflocculant (10% $Na_4P_2O_7$) and placed in sealed bottles for a minimum of 24 hours. Organic digestion of two highly organic (>5% OM) samples followed a similar procedure as used with the less organic samples. Ten ml of 30% H_2O_2 added at 1 ml additions were used as opposed to a total of 2 ml added at 0.25 ml additions. Grain-size analysis was performed using a Beckman Coulter LS 13 320 Laser Diffraction Particle Size Analyzer. The analyzer is capable of distinguishing particle sizes ranging from 0.04 μm to 2 mm. Inspection of samples prior to analysis with the Beckman Coulter analyzer, revealed no grains larger than 2 mm.

Local meteorological conditions (temperature and precipitation) were collected for the preceding 180, 30 and 7 days for each of the 8 days when water measurements were collected. These data were used as proxies for surface water influx and salinity regime. Average daily values of temperature and precipitation for days of sampling in 2014 were compared to the 30-year normal (1981-2010). Data were collected from the nearby Elizabeth City Coast Guard Air Station Automated Surface Observing System.

Approximate wind speed and direction were documented at the time of sampling and data that are more precise were collected from records at the Duck Pier Field Research Facility located in Duck, North Carolina. Wind data were collected from the Field Research Facility because it is closer to the sampling site than the Elizabeth City station. Historical data at the Field Research Facility is less complete and not suitable for analysis of the 30-year record, however.

4.3 Statistical Analysis

In order to use conductivity values in statistical correlations, average values were calculated using spatial analysis tools in ArcGIS software packages. Using water measurement data from the six days (9/15 to 9/17 and 9/29 to 10/1) with the most complete datasets, an inverse distance weighted function was applied to each day's data to create a continuous raster of conductivity values. The raster data from the 6 days were averaged and a value at each core location was extracted. All conductivity values refer to temperature corrected to 25°C conductivity.

After collection of all pertinent data, statistical analysis involved simple linear regression, calculation of Pearson product-moment correlation coefficients and Spearman's rank-order coefficients (when Pearson was inappropriate), Analysis of Variance (ANOVA), and Student's T-test using water and sediment environmental variables, distance from discharge, and Foraminiferal populations. Variables investigated include Foraminiferal total count, a diversity index (Hill's N_1), distance from discharge, 6-day average conductivity, water depth, %OM, longitudinal location, percent clay, percent silt, percent sand, grain-size standard deviation and mean, and median grain-size. Pearson and Spearman's coefficients are generally similar except when extreme outliers exist and when data are not normally distributed (e.g. distance from the outfall and %OM), in these cases, Spearman's correlation coefficient was used. More complex analysis using similarity between samples (e.g. Bray-Curtis similarity) was unreasonable due to the disproportionate number of specimens identified as one of two very prolific genera. One diversity measure was used in the analysis of Foraminiferal populations: Hill's N_1—a transformed Shannon-Wiener diversity index. Hill's N_1 is calculated using the following equation:

$$N_1 = \exp[-\Sigma_i (p_i \times \ln p_i)]$$

where p_i is the proportion of the total count of one sample arising from the i^{th} genera. Only those genera accounting for 5% of the assemblage or greater were included in the calculation of Hill's N_1. Genera that occur in smaller proportions are more variable and their inclusion in a sample may be by chance when only 100 specimens are counted (Fatela and Taborda, 2002).

CHAPTER 5: RESULTS

5.1 Environmental Variables

Six-day average conductivity is highest nearest the discharge and generally diminishes with increasing distance from the discharge (Figure 4.2). Recorded values of conductivity during the six days of sampling ranged from 3,988 to 5,220 μS/cm. One-way ANOVA performed using the conductivity values from the six days across nine distances confirmed significant difference between conductivity at varying distance to the discharge site (F ratio=3.780, $p<0.0003$).

Conductivity recorded in-plant prior to discharge from August to November ranged from 14,060 to 16,910 μS/cm. Results of two-tailed Student's t-test ($\alpha=0.05$) indicate that conductivity values measured at the WTP before discharge are significantly different from all locations within the river (Table 5.1).

Table 5.1: Table of Student's t-test two-tailed p-values of temperature corrected conductivity at discrete distances (m) from the discharge site. The in-plant values were recorded in the WTP prior to discharge. Group D-12.5 indicates those water measurement locations taken within 12.5 m of the discharge site. Conductivity values were collected from 9-15-2014 to 9-17-2014 and from 9-29-2014 to 10-1-2014. Those values marked with an asterisk indicate statistical significance (α=0.05).

	In-Plant	D-12.5	12.5-37.5	37.5-62.5	62.5-87.5	87.5-112.5	112.5-137.5	137.5-400	400-4000
D-12.5	*<0.001								
12.5-37.5	*<0.001	0.462							
37.5-62.5	*<0.001	0.203	0.277						
62.5-87.5	*<0.001	0.081	*0.023	0.171					
87.5-112.5	*<0.001	0.055	*0.005	*0.043	0.493				
112.5-137.5	*<0.001	*0.045	*0.005	*0.036	0.351	0.715			
137.5-400	*<0.001	*0.013	*0.000	*0.000	*0.009	*0.028	0.094		
400-4,000	*<0.001	0.572	0.890	0.373	0.104	0.055	0.043	0.005	
>4,000	*<0.001	0.467	0.883	0.055	*<0.001	*<0.001	*<0.001	*<0.001	0.943

Percent OM, determined by loss on ignition ranges from 0.78% to 23.76%. Most values are less than 2% (Figure 5.1), and only two samples were found to have OM content greater than 3% (NW1.1=23.76%, and N2.1=7.14%). Percent OM and water depth are positively correlated in down-core samples (Spearman's rho=0.429, p=0.0202), but are not significantly correlated when only examining the surface samples (p>0.05). Combined, down-core and surface samples indicate a significantly positive relationship between water depth and %OM (Spearman's rho=0.3212, p=0.0140).

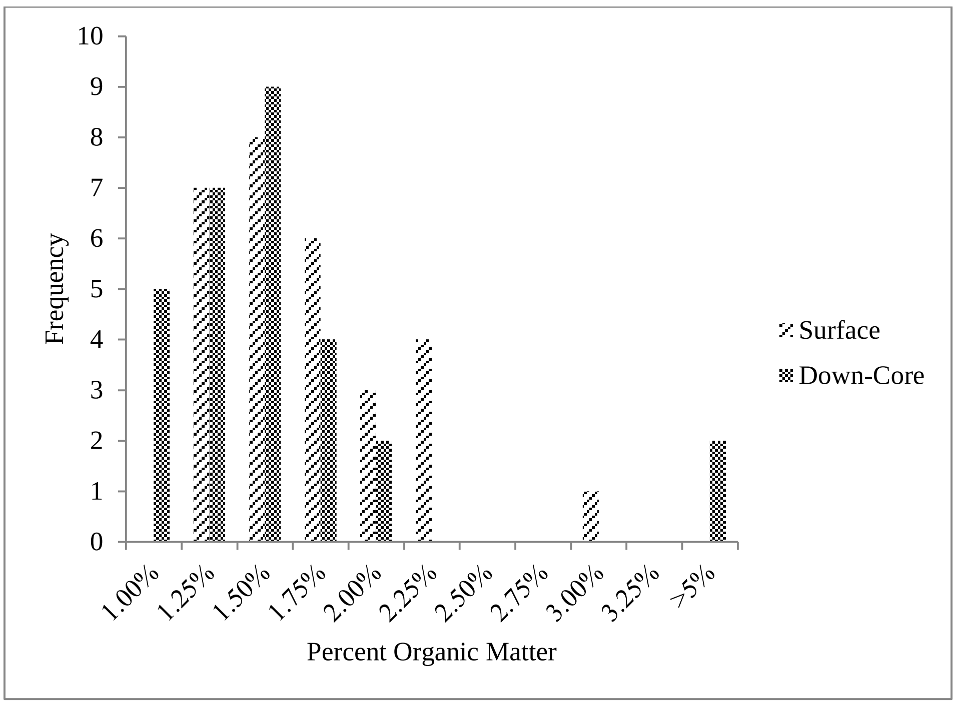

Figure 5.1: Histogram of samples by percentage of organic matter. Surface samples: n=29. Down-core samples: n=29.

Mean grain-size ranges from 10 μm to 177 μm with an average of 91 μm. The average textural composition of all analyzed sediment samples is 87.1% ± 10.5% sand, 12.3% ± 10.0% silt, and 0.6% ± 0.5% clay. The majority of sediments are fine and very fine sand (19% and 64% respectively).

Mean grain-size of the surface samples generally increase with water depth (r=0.376, p=0.044). Percent very fine sand is negatively correlated with water depth (r=-0.716, p<0.0001), but percent fine sand is positively correlated with water depth (r=0.532, p=0.003). Alternatively, mean gain-size in the down-core samples is not significantly correlated with water depth (p=0.29). Percent very fine sand continues to be negatively correlated with water depth (r=-0.533, p=0.0029), but percent fine sand does not show significant correlation with water depth (p=0.0677).

Mean grain-size and %OM are not correlated in either surface or down-core samples. However, percent silt is positively correlated and percent sand is negatively correlated with %OM in both the down-core (Spearman's rho=0.6212, p=0.0003; Spearman's rho=-0.6069, p=0.0005) and surface samples (Spearman's rho=0.6749, p<0.0001; Spearman's rho=-0.6601, p<0.0001). Percent clay and %OM are positively correlated in only the down-core samples (Spearman's rho=0.5108, p=0.0050).

5.2 Meteorology

Based on the precipitation and temperature data collected, 2014 was a relatively normal year from August to December. For all eight days of sampling, 2014 precipitation values for the preceding 30-days and 180-days were in the middle 50th percentile of the 30-year, 1981-2010 record (Figure 5.2). The preceding 7-day precipitation was within the middle 50th percentile for half of the eight days. For October 15th through 17th, and November 30th, the preceding 7-day precipitation was in the highest 25th percentile of the 30-year record (as well as above one standard deviation).

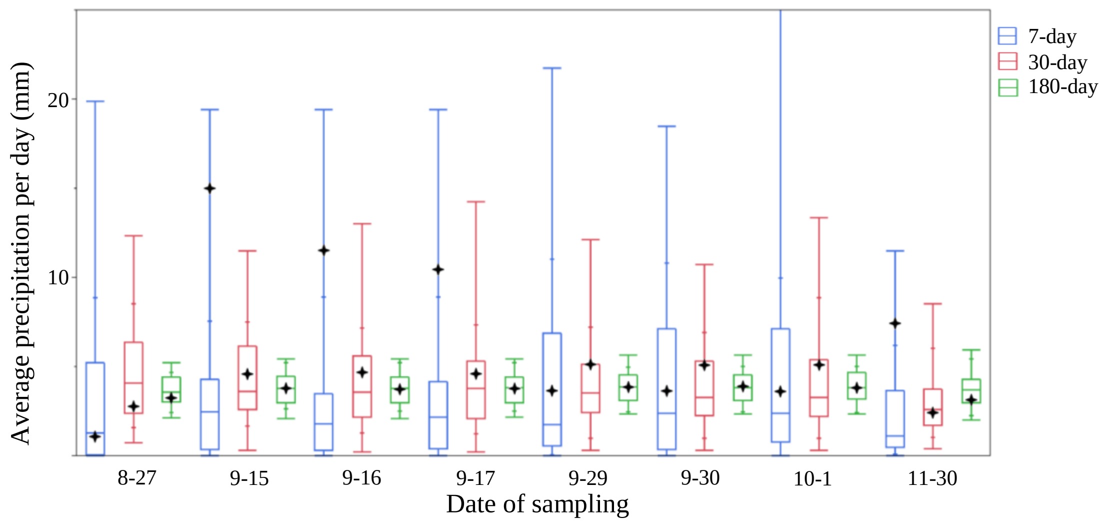

Figure 5.2: Box and whisker plot showing 30-year precipitation record and 2014 values noted by stars. Three colors indicate number of days used in the calculation: preceding 7, 30, or 180 days.

Temperature for the days preceding sampling in 2014 was more variable than precipitation with respect to the 30-year record. Examination of the preceding 180-days subset indicate only the three days in mid-September were within the 50th percentile of the 30-year record, but all except the August date were within one standard deviation. The 30-day preceding temperature average of the August date was lower than any value in the 30-year record (-2.05 standard deviations). November 30th was also below average, in the lowest 25th percentile, but within one standard deviation of the average (Figure 5.3).

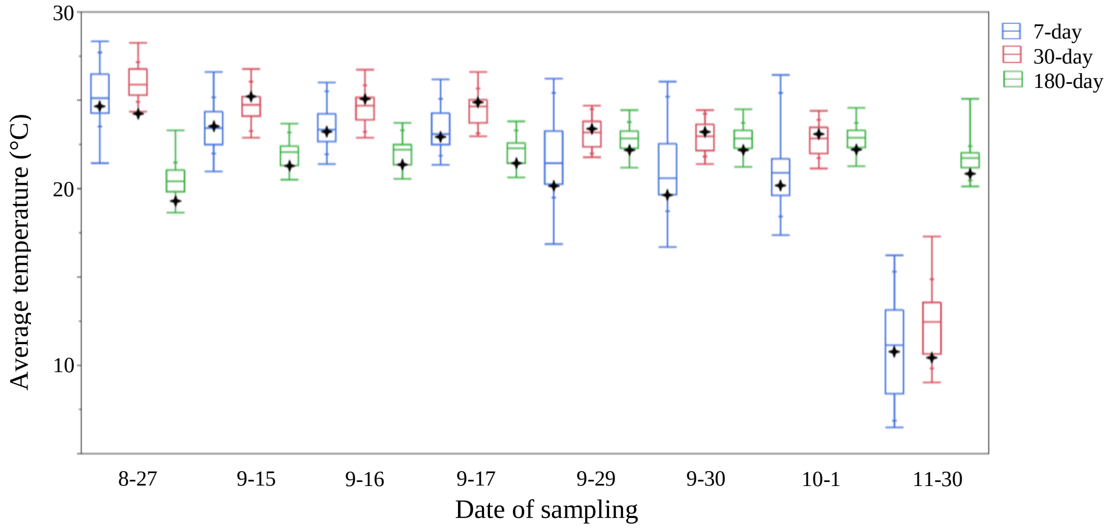

Figure 5.3: Box and whisker plot showing 30-year temperature record and 2014 values noted by stars. Three colors indicate number of days used in the calculation: preceding 7, 30, or 180 days.

Winds during most days of data collection were from the north or east—an unavoidable control applied to the data. To access the discharge site, the wind had to be very calm or blowing from the near shore (east shore). Mean wind direction recorded at the Field Research Facility for the six days used in plume analysis was 25.5° with a circular standard deviation of 28.3°. Average wind speed was 5.34 m/s (2.0 m/s standard deviation) during the hours of data collection for the six days. The collection site is more protected from the wind than Duck Field Research Facility located on the Atlantic Ocean

Winds during the first day of sediment collection (August) were from the northeast (18°, standard deviation of 7.5°) at 8.8 m/s. During sediment collection in November, wind direction averaged 211° (7.5° standard deviation) with and average wind speed of 5.5 m/s.

5.3 Micropaleontology

In 58 (29 surface, 29 down-core) samples, 7,575 Foraminiferal specimens were identified, for an average of 130.6 per sample. Eleven genera were identified, nine with agglutinated tests and two genera with calcareous tests. Distribution of assemblages is heavily skewed towards two coarsely-agglutinated taxa: *Ammotium* sp. and *Ammobaculites* spp. Together these two genera account for over 98.5% of all normalized counts, and both genera are found in every sample. In only two samples (NW1.1 and N2.1; both 10cm) did taxa other than the two coarsely-agglutinated genera comprise greater than 2% of the assemblage. These two samples are also unique in their sediment composition—having the greatest percentage of OM of all samples. Samples NW1.1 and N2.1 contain larger proportions of two calcareous genera: *Ammonia* spp. and *Elphidium* spp. *Ammonia* spp. and *Elphidium* spp. account for 92.4% of specimens in sample NW1.1, the remainder of the assemblage is comprised of *Ammotium* sp. and *Ammobaculites* spp. In sample N2.1, 6.6% of specimens are *Ammonia* spp. or *Elphidium* spp., 85.8% are *Ammotium* sp. or *Ammobaculites* spp., 3.8% are *Miliammina* spp.; the remainder of the taxa are found in smaller proportions (<2%).

Total Foraminiferal specimens/cm^3 averages 1,074 and ranges from 168 to 2,895. Mean surface abundance is significantly higher than the mean down-core abundance (t stat=7.958, $p<.0001$). Surface abundance ranges from 317 to 2,895 and averages 1,486. Down-core abundance ranges from 168 to 1,988 and averages 662. Down-core abundance was found to be higher than surface abundance in only one core (core C2).

5.4 Foraminiferal Relationship to RO-WTP Discharge

Foraminiferal abundance in surface samples is not significantly correlated with either 6-day average conductivity or distance from discharge ($p>0.27$). As would be expected, these relationships are not significant when using all samples nor when using only down-core samples. Assemblage diversity in the form of Hill's N_1 is not significantly correlated with 6-day average conductivity or distance from discharge either ($p>0.345$).

One-way ANOVA using the dependent variable total count of the surface samples between six distances (Discharge, 25 m, 50 m, 75 m, 100 m, Control) and the four directions (N, E, S, W) revealed no significant differences between distance or direction (Figures 5.4 and 5.5). Additionally, one-way ANOVA using count of *Ammotium* sp. and count *Ammobaculites* spp. independently yielded no significant disparities between distances or directions ($p>0.1$). Surprisingly, a one-way ANOVA of total counts of down-core samples revealed significant differences between the six distances (F ratio= 4.045, $p=0.0196$); no significant differences were observed between direction in the down-core samples ($p>0.05$).

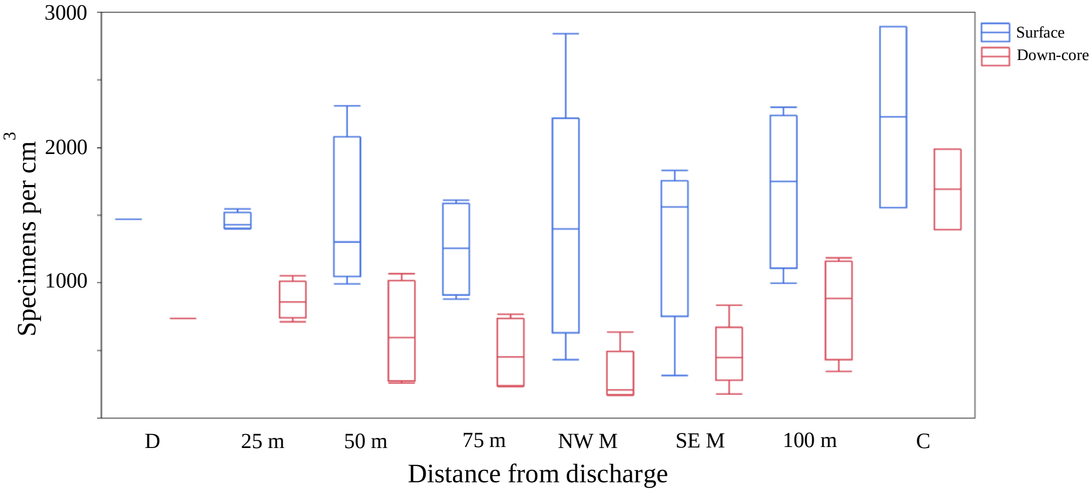

Figure 5.4: Box and whisker plot of total count of Foraminifera per cubic centimeter at varying distance (m) from discharge site. "D" denotes the samples taken at the discharge point. "SE M" and "NW M" denote the microanalysis samples southeast and northwest of the discharge point. "C" represents the control samples.

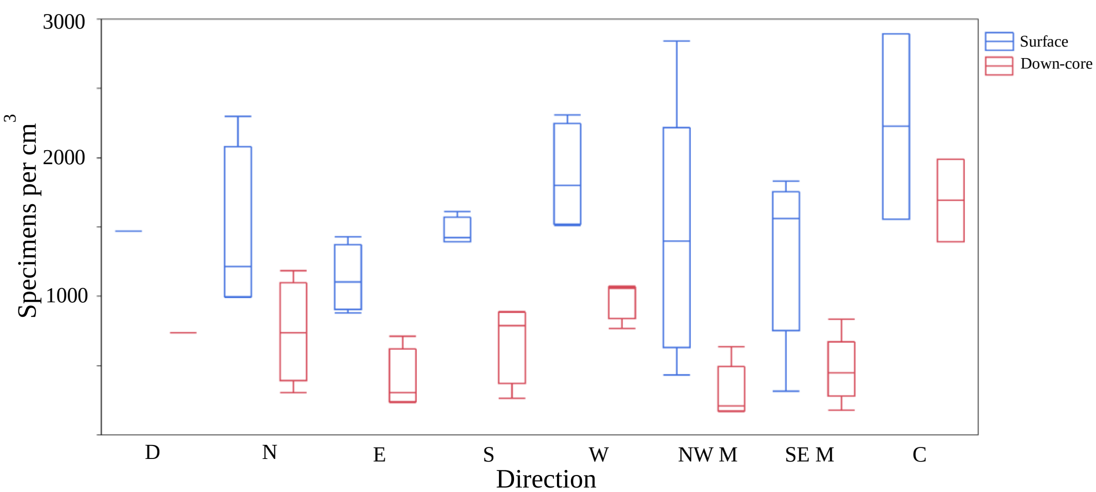

Figure 5.5: Box and whisker plot of total count of Foraminifera per cubic centimeter along transects in the four cardinal directions. "D" denotes the samples taken at the discharge site. "SE M" and "NW M" denote the microanalysis samples southeast and northwest of the discharge point. "C" represents the control samples.

One-way ANOVA using Hill's N_1 in the surface samples revealed significant differences between the six distances (F ratio=4.625, p=0.0121), and between the four directions (F ratio=4.692, p=0.0217). An array of Student's t-tests between the six distances revealed the only significant differences was between the discharge site and each of the ensuing four distances (not the control). Only one sample was taken at the discharge site, considerably affecting confidence (in this case the t-test relied on the assumption of equal variance) (Table 5.2). Student's t-test between directions revealed that the southern transect was significantly different from the east and west transects (Table 5.3). ANOVA of Hill's N_1 in down-core samples did not reveal significant differences between varying distance or direction.

Table 5.2: Table of Student's t-test two-tailed p-values of Hill's N_1 across varying distances (m) from the discharge site. Those values marked with an asterisk indicate statistical significance (α=0.05).

	D	25	50	75	100
25	*0.008				
50	*0.004	0.997			
75	*0.005	0.835	0.819		
100	*0.037	0.484	0.469	0.559	
C	0.427	0.616	0.617	0.643	0.760

Table 5.3: Table of Student's t-test two-tailed p-values of Hill's N_1 across transects in the four cardinal directions extending from the discharge site. Those values marked with an asterisk indicate statistical significance (α=0.05).

	N	S	E
S	0.803		
E	0.090	*0.011	
W	0.140	*0.019	0.212

5.5 Foraminiferal Relationship to Non-Anthropogenic Parameters

Correlation coefficients between Foraminiferal assemblages and natural environmental variables are stronger than the observed relationships with the variables associated with the discharge. Control samples are not included in the following relationships because of the concern that the environment could be influenced by other factors at that location.

Total count and %OM are positively correlated when both down-core and surface samples are included in the analysis (Spearman's rho=0.3422, p=0.0113) and when using only surface samples (Pearson's r=0.5183, p=0.0056). This relationship is not significant in only the down-core samples (p>0.45). Hill's N_1 and %OM are not significantly correlated when examining both surface and down-core samples (p>0.05), or when examining only the surface samples (p>0.30). A significant positive correlation does exist between Hill's N_1 and %OM in the down-core samples (Spearman's rho=0.4842, p=0.0105).

If the analysis of Foraminiferal assemblages is limited to those samples that contain less than 3% OM (i.e. exclude samples NW1.1 and N2.1), a strong correlation exist when using surface and down-core samples. In this case, total count in down-core and surface samples is significantly correlated with %OM (r=0.5614, p<0.0001). Total count and %OM in down-core samples are not significantly correlated (p>0.05), and surface samples correlations are unchanged from the above described relationship. Diversity does not appear significantly correlated with %OM in this case of removing the two outlying samples (p>0.05).

Total count and water depth are not significantly correlated (p>0.05) when using only surface samples, only down-core samples, or when using both surface and down-core samples. Significant correlations do not exist between Hill's N_1 and water depth either (p>0.10).

Abundance appears to increases across the study site from east to west. Using surface samples, down-core samples, and both surface and down-core samples, total count is significantly negatively correlated with easting value (r=0.5546, p=0.0027;

r=0.5183, p=0.0056; r=0.5983, p<0.0001). Diversity based on Hill's N_1 does not appear correlated with longitudinal location across the study site (p>0.30).

Abundance is not significantly correlated with mean or median grain-size or standard deviation of grain-size for any of the three previously mentioned subsets of data (only surface, only down-core, and down-core and surface samples combined). The only significant total count and grain-size correlation observed was found using the surface samples. A positive correlation between total count and percent silt was identified and a negative correlation with percent sand was identified (r=0.4118, p=0.0328; r=-0.4100, p=0.0337). No significant correlations between Hill's N_1 and grain-size were found in only the surface samples. When both down-core and surface samples are included in the analysis, however, mean grain-size and Hill's N_1 show a significant negative correlation (Spearman's rho=-0.2812, p=0.0394). In only the down-core samples, percent clay, and percent silt are positively correlated with Hill's N_1 (Spearman's rho=0.4631, p=0.015; Spearman's rho=0.3907, p=0.0439). Alternatively, mean and median grain-size are negatively correlated with Hill's N_1 in the down-core samples (Spearman's rho=0.4454, p=0.0199; Spearman's rho=0.4356, p=0.0231).

CHAPTER 6: DISCUSSION

The RO-WTP effluent plume is easily detectable with a YSI conductivity probe within ~50 m of the discharge. Conductivity is significantly different at the discharge site than it is at distance of 120.5-400 m from the discharge site. Oddly, conductivity values at the sites further than 400 m from the discharge site are more similar to the values observed in direct vicinity of the discharge than to the sites 50-400 m from the discharge. This is likely an indication of the high natural variability in the region. Based on Student's t-test (Table 5.1) results and the 6-day average conductivity map, the detectable level of higher conductivity appears to extend to ~75 m from the discharge. This value is in agreement with evidence from previous literature. The median value of detectable plume extension of the nine studies listed in Table 2.1 is 100 m.

Effluent dispersal does not appear to be controlled by water depth according to the map (Figures 4.1 and 4.2) of conductivity values surrounding the discharge site. A sufficiently strong density gradient to cause the wastewater to flow westward and downhill does not exist. There appears to be a north-south trend of higher values over the discharge site; highest values are recorded south and southeast of the discharge. The higher values south of the discharge could possibly be explained by either the direction of the prevailing wind or natural basin drainage. The wind during most days of data collection was from the northeast. Wind or natural drainage does not explain the southeastwardly (towards shore) extension, however. Stratification of the effluent intersecting the sediment-water interface could explain the apparent southeastward extension, but confirming such a hypothesis would require extensive data collection during a longer timescale.

Foraminiferal assemblages at this study site are similar to other studies conducted in the area with respect to species identified, and density of specimens (Grossman and Benson, 1967; Abbene et al., 2006; Vance et al., 2006). The number of specimens per volume of sediment varies widely in the area, but diversity is relatively consistent due to the overwhelming number of the two coarsely-agglutinated taxa.

The down-core samples have considerably fewer numbers of specimens, likely the result of taphonomic destruction. This assumption is supported by the increased number of broken tests observed in the down-core samples. It is possible that the increased numbers in the surface samples are a result of changes in the broader environment (e.g. increased fertilizer runoff), but it is unlikely that it is a result of the RO-WTP. At other sites in Albemarle Sound, large changes in abundance have been observed from surface to down-core samples. At three sites in central Albemarle Sound, Vance et al. (2006) found increases of 2300%, and 470% and a decrease of 67% between surface and 10-13 cm samples. Because of the reduced abundance in down-core samples, use of the down-core samples in the analysis of the discharge was limited to changes in relative distribution of taxa in the assemblages. No significant correlations were observed between parameters associated with the effluent and parameters associated with changes in diversity or composition of the assemblages between the surface and down-core samples, however.

The lack of significant correlation between abundance in the surface samples and distance from discharge or 6-day average conductivity is the strongest argument against the discharge having an impact on Foraminiferal populations. ANOVA results confirm that there are no significant differences between the total count in the samples at varying distances or directions. At the control sites, normalized counts of 1,555 and 2,894.5 specimens were recorded. These samples are the 11^{th} (of 29) and first in terms of the number of specimens per cm^3 in the surface samples. Additionally, at a site located between the control and the discharge sites used in this study, Vance et al. (2006) identified 1,152 specimens per cm^3 of surface sediment. These results indicate that Foraminiferal abundance is not impacted by the brine solution at the geographic scale studied, but instead abundance is naturally widely variable.

Diversity in the form of Hill's N_1 does change with distance and direction, but the confidence in the calculated diversity indices values is less than that of the total count. In each surface sample, *Ammotium* sp. and *Ammobaculites* spp. were the only genera present in proportions greater than 5%. Therefore, Hill's N_1 relies exclusively on the distribution of these two genera. These two genera are coarsely-agglutinated and have a similar morphology and appearance—particularly when only fragments of their test remain. While differentiation of the taxa was completed as carefully and consistently as possible, differentiation between these two groups is significantly more difficult than between other taxa.

A Student's t-test using Hill's N_1 and six distances (discharge, 25 m, 50 m, 75 m, 100 m, and control) revealed that diversity of the sample at the discharge is significantly less than all other distances except the control samples (Table 5.2). Loss of select species leading to a decrease in diversity is likely surrounding the outfall, based on response of macrofauna and flora monitored in previous studies (e.g. Crockett, 1997; Ruso et al., 2007; Ruso et al., 2008; Kleber, 2010). Unfortunately, only one core was collected within 10 m of the discharge site in this study. The surface sample taken at the discharge site has the highest percentage of *Ammotium* sp. and the smallest percentage of *Ammobaculites* spp. of all surface samples. Perhaps, the effluent only imparts a control on the assemblages within a few meters of the discharge as Kleber (2010) identified in the macroinvertebrate communities at the Camden County RO-WTP wastewater discharge site. If that is the case, the grid used in this project was too large to resolve the changes in assemblages.

A Student's t-test using Hill's N_1 and four cardinal directions shows that the south transect is significantly different from the east and west transects (Figure 5.3). The north and south transects are more similar to each other than to the east or west transects, and the east and west are more similar to each other than to the north or south transects. Hill's N_1 is generally lower in the north and south transects. This is curious because of the pattern seen in the map of 6-day average conductivity (Figure 4.2). However, no significant correlations were identified between Hill's N_1 and the 6-day average conductivity values.

Correlation among environmental variables limits their use as covariates in determining Foraminiferal response (i.e. %OM is correlated with water depth, longitudinal location and grain-size). However, previous literature suggests variations in %OM is the best factor available for explaining Foraminiferal abundance. When analysis is limited to those samples containing less than 3% OM, %OM explains ~36% of the variability. This increase in abundance with increase of OM fits the model seen in previous studies (e.g. Alve, 1995). The two samples containing high percentages of OM also fit the model of increased OM to the point of toxicity, causing a decrease in Foraminiferal abundance. This is not thought to be the case here; instead, assemblages in those two samples were likely controlled by other factors. Based on taxa found, it is postulated that the specimens were transported to the site on grass mats from back-barrier marshes.

The lack of response by the assemblages to the briny discharge can be explained by the tolerant nature of the two most prevalent genera. In a study in Pamlico Sound, Grossman and Benson (1967) found *Ammobaculites* spp. in salinities ranging from 1-23 ppt, preferring ~10-15 ppt, and in water depths of 2-15 ft. (0.6-4.6 m). *Ammotium* sp. was found in salinities ranging from 1-25 ppt, preferring ~10 ppt, and in water depths of 1-10 ft. (0.3-3.0 m). In northern North Carolina, Vance et al. (2006) found *Ammotium* sp. and *Ammobaculites* spp. at concentrations greater than 90% of the assemblage at sites ranging from the narrows of the Pasquotank river to within a few kilometers of Oregon Inlet in Pamlico Sound. The inland brackish waters of the Albemarle-Pamlico estuary system appear to be dominated by these two prolific and cosmopolitan genera. Other taxa, including some of those identified in minor quantities in this study, prefer higher salinity and/or marsh environments (Grossman and Benson, 1967; Abbene et al., 2006; Vance et al., 2006; Kemp et al., 2012; Hippensteel and Garcia, 2014).

The Foraminiferal taxa found in the inland areas of the sounds of North Carolina do not appear useful for monitoring briny discharges, except perhaps, in cases where the effluent from the point source is drastically different from the receiving waters. An extremely saline effluent at a site further inland might elicit changes in assemblages due to the greater discrepancy between the effluent and the receiving water. It is also possible that variability across the discharge site could be observed surrounding the outfall at the Currituck facility during extreme climatic variability, or following the landfall of tropical system.

CHAPTER 7: CONCLUSION

A RO-WTP in Currituck County, North Carolina creates a briny wastewater solution that is discharged in the North River. The wastewater contains significantly higher concentration of dissolved solids than the receiving waters. Although minimal, significantly higher values of conductivity are observed surrounding the discharge site. The highest conductivity values, those recorded in the WTP, cannot be observed in the river with the tools used in this study, as the wastewater efficiently mixes with the receiving water in a very short distance.

It was presumed that the briny solution would have an impact on the Foraminiferal assemblages surrounding the discharge site, but high variability and natural controls on abundance, along with the tolerant nature of the two most numerous genera likely limited the response of Foraminifera. Assemblage abundance did not appear to be impacted by the effluent. The single sample taken within 5 m of the discharge site did show a possible response to the effluent, however. This sample had significantly lower diversity than the surrounding area, but questions remain concerning the applicability of the diversity index and the single data point limits confidence in the finding. More evidence would be needed to confirm this relationship, but decreased diversity within a few meters of the discharge is consistent with some previous studies (e.g. Hammond et al., 1998; Kleber, 2010).

Previous research indicates that abundance is chiefly controlled by %OM (Alve, 1995; Collins et al., 1995; Yanko et al., 2003). Percent OM is highly correlated with other environmental variables, but it provides the strongest correlation coefficients with Foraminiferal abundance (when excluding the two highly organic samples) in this study. Natural variability elicits a stronger control on Foraminiferal abundance across the entire site than any impact from the wastewater, but the wastewater may be causing a decrease in diversity in close proximity to the discharge site.

The understanding of impacts to the environment from wastewater from RO-WTPs will become increasingly important due to population increases and needs for potable water. This study will contribute to a foundation of literature to help form a basis for comparisons.

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APPENDIX A: COLLECTED DATA

Counts of Foraminifera genera identified in surface samples normalized to 1 cm^3

Location	*Ammotium* sp.	*Ammobaculites* spp.	*Miliammina* sp.	*Trochammina* sp.	*Ammoastuta* sp.	*Jadammina* sp.	*Ammonia* spp.	*Elphidium* spp.	*Textularia* spp.	*Arenporella* sp.	*Pseudothurammina* sp.	Total Count
D.0	1149.7	287.4			23.0				11.5			1471.7
N 1.0	491.1	942.5										1433.6
E 1.0	762.7	663.8										1426.5
S 1.0	569.0	829.8										1398.8
W 1.0	809.4	726.1	11.9									1547.4
N 2.0	404.6	588.5										993.1
E 2.0	668.0	532.1							11.3			1211.4
S 2.0	902.3	491.0										1393.3
W 2.0	1256.6	1054.4										2310.9
N 3.0	408.0	582.8							9.7			1000.5
E 3.0	430.1	438.8	8.8									877.7
S 3.0	563.3	1050.8										1614.2
W 3.0	913.5	595.1										1508.6
N 4.0	1561.8	661.5	18.4						36.7		18.4	2296.8
E 4.0	517.5	469.6	9.6									996.7
S 4.0	522.2	928.4										1450.6
W 4.0	1177.0	856.0					17.8					2050.9
SE 1.0	1010.5	535.0			14.9							1560.3
SE 2.0	1198.3	611.7	5.0	5.0	10.0							1830.0
SE 3.0	950.4	712.8	4.8	4.8				4.8	4.8			1682.6
SE 4.0	698.5	468.4			4.1				16.4			1187.4
SE 5.0	227.4	84.1	6.2									317.7
NW 1.0	1073.3	315.2					7.5					1396.0
NW 2.0	333.3	90.9	7.6									431.7
NW 3.0	1220.1	370.8										1591.0
NW 4.0	1481.1	1361.0										2842.0
NW 5.0	560.2	272.8										833.0
C 1.0	761.7	2132.8										2894.5
C 2.0	783.5	747.4	12.1		12.1							1555.0

Counts of Foraminifera genera identified in down-core samples normalized to 1 cm^3

Location	*Ammotium* sp.	*Ammobaculites* spp.	*Miliammina* sp.	*Trochammina* sp.	*Ammoastuta* sp.	*Jadammina* sp.	*Ammonia* spp.	*Elphidium* spp.	*Textularia* spp.	*Arenporella* sp.	*Pseudothurammina* sp.	Total Count
D.1	575.2	163.4										738.6
N 1.1	318.1	496.6			15.5							830.3
E 1.1	487.1	219.8	5.9									712.8
S 1.1	400.5	487.5										888.0
W 1.1	474.7	577.9										1052.5
N 2.1	194.5	69.7	11.6		5.8		20.3	2.9		2.9		307.6
E 2.1	150.8	103.1	2.5	2.5							2.5	261.4
S 2.1	325.1	553.5										878.6
W 2.1	532.7	522.5		10.2								1065.4
N 3.1	209.1	412.1	6.2		6.2				6.2			639.7
E 3.1	117.8	111.1	2.3					2.3				233.4
S 3.1	139.3	124.1										263.4
W 3.1	392.4	377.0										769.3
N 4.1	598.1	579.1	9.5									1186.6
E 4.1	231.3	100.5	6.7		3.4		3.4					345.2
S 4.1	242.2	453.4										695.6
W 4.1	603.4	468.1										1071.5
SE 1.1	625.4	200.1		8.3								833.8
SE 2.1	355.0	87.7				4.2						446.9
SE 3.1	445.4	58.1		4.8								508.3
SE 4.1	292.8	84.2		3.7								380.7
SE 5.1	132.1	47.4										179.5
NW 1.1	6.6	19.8					95.8	224.5				346.7
NW 2.1	144.9	27.0	1.7									173.6
NW 3.1	175.5	28.2	2.0								2.0	207.8
NW 4.1	150.3	16.2			1.6							168.1
NW 5.1	487.8	137.7		5.7	5.7							637.0
C 1.1	518.0	836.7	19.9		19.9							1394.5
C 2.1	1158.3	777.9			17.3	17.3	17.3					1988.1

Environmental variables from sample locations. Coordinates are based on WGS 1984, UTM zone 18 N.

Location	Distance From Discharge (m)	Water Depth (m)	6-Day avg. Temperature corrected conductivity (25°C) (µS/cm)	Northing	Easting
D	3.5	2.6	4371.8	4002006	421859
N 1	31.4	2.5	4336.7	4002036	421855
E 1	21.1	2.6	4364.4	4002008	421876
S 1	22.1	2.5	4390.2	4001983	421853
W 1	25.4	2.6	4353.0	4002007	421830
N 2	57.9	2.5	4313.4	4002062	421853
E 2	47.1	2.4	4328.1	4002009	421902
S 2	40.7	2.5	4360.7	4001964	421855
W 2	52.5	2.7	4302.0	4002005	421803
N 3	76.0	2.5	4302.4	4002080	421853
E 3	71.2	2.3	4304.1	4002010	421926
S 3	68.3	2.5	4327.3	4001936	421854
W 3	67.5	2.7	4285.8	4002014	421789
N 4	98.9	2.6	4298.4	4002103	421854
E 4	96.6	2.2	4294.7	4002011	421952
S 4	100.0	2.5	4309.0	4001905	421853
W 4	98.1	2.8	4270.9	4002024	421759
SE 1	68.8	2.4	4305.3	4001940	421880
SE 2	71.8	2.4	4301.1	4001937	421880
SE 3	73.7	2.4	4300.6	4001935	421880
SE 4	74.1	2.4	4300.5	4001935	421881
SE 5	77.5	2.4	4302.1	4001931	421880
NW 1	66.7	2.8	4307.8	4002071	421850
NW 2	70.8	2.8	4305.9	4002075	421849
NW 3	71.0	2.8	4306.4	4002075	421847
NW 4	74.2	2.8	4305.5	4002078	421845
NW 5	74.7	2.8	4306.7	4002078	421842
C 1	4222.9	2.9	NA	4005504	419492
C 2	4281.7	3.0	NA	4005527	419421

Environmental data from surface samples

Location	LOI (%OM by weight)	Mean Grain Size (µm)	Median Grain Size (µm)	Grain Size Standard Deviation (µm)	Percent Clay	Percent Silt	Percent Sand
D.0	1.5%	86.4	101.3	2.5	0.4%	11.9%	87.6%
N 1.0	1.8%	85.0	100.7	2.4	0.4%	10.5%	89.1%
E 1.0	1.3%	94.8	106.1	2.4	0.4%	9.5%	90.1%
S 1.0	1.4%	80.7	99.2	2.6	0.5%	13.1%	86.4%
W 1.0	1.2%	84.8	101.5	2.5	0.4%	10.9%	88.7%
N 2.0	2.0%	82.5	100.9	2.7	0.4%	13.0%	86.6%
E 2.0	1.3%	99.2	106.6	2.6	0.4%	9.2%	90.4%
S 2.0	1.6%	78.9	98.6	2.7	0.4%	14.3%	85.2%
W 2.0	1.2%	83.9	100.9	2.5	0.5%	11.7%	87.9%
N 3.0	1.3%	97.8	107.1	2.6	0.4%	9.4%	90.2%
E 3.0	1.2%	88.6	103.8	2.4	0.4%	8.8%	90.8%
S 3.0	1.4%	84.5	101.1	2.5	0.4%	11.2%	88.4%
W 3.0	1.5%	82.7	99.9	2.5	0.5%	12.8%	86.8%
N 4.0	1.6%	99.3	108.4	2.8	0.4%	10.5%	89.1%
E 4.0	1.0%	93.4	106.1	2.2	0.4%	6.7%	92.9%
S 4.0	1.6%	78.9	99.7	2.7	0.5%	14.2%	85.3%
W 4.0	1.9%	76.4	98.8	2.8	0.4%	16.5%	83.0%
SE 1.0	1.6%	84.0	100.2	2.5	0.4%	11.5%	88.1%
SE 2.0	2.1%	73.2	97.7	2.9	0.5%	17.5%	82.0%
SE 3.0	1.2%	82.4	100.1	2.5	0.4%	13.0%	86.5%
SE 4.0	2.0%	75.1	98.8	2.9	0.5%	16.6%	82.9%
SE 5.0	1.2%	85.1	101.1	2.4	0.5%	10.1%	89.5%
NW 1.0	1.4%	97.7	105.0	2.9	0.4%	11.7%	87.9%
NW 2.0	1.5%	120.4	112.8	3.1	0.4%	10.2%	89.4%
NW 3.0	1.3%	115.7	112.7	2.9	0.3%	9.7%	90.0%
NW 4.0	2.8%	96.7	107.0	3.2	0.4%	13.7%	85.9%
NW 5.0	1.0%	90.3	105.1	2.5	0.4%	9.4%	90.2%
C 2.0	2.1%	92.7	116.6	2.7	0.4%	11.4%	88.1%
C 1.0	1.9%	92.1	114.0	2.7	0.4%	11.7%	87.9%

Environmental data from down-core samples

Location	LOI (%OM by weight)	Mean Grain Size (μm)	Median Grain Size (μm)	Grain Size Standard Deviation (μm)	Percent Clay	Percent Silt	Percent Sand
D.1	1.4%	92.3	104.0	2.5	0.5%	8.6%	90.8%
N 1.1	0.9%	91.1	102.5	2.5	0.5%	8.4%	91.1%
E 1.1	1.3%	82.6	95.9	2.4	0.5%	12.1%	87.5%
S 1.1	1.4%	83.2	101.1	2.6	0.5%	10.7%	88.7%
W 1.1	1.7%	84.5	101.0	2.5	0.6%	10.4%	89.1%
N 2.1	7.1%	37.0	75.8	5.4	1.9%	40.4%	57.7%
E 2.1	1.3%	84.9	100.1	2.5	0.5%	10.5%	89.0%
S 2.1	1.0%	88.6	103.8	2.5	0.6%	8.6%	90.8%
W 2.1	1.2%	85.1	101.0	2.5	0.6%	10.5%	88.9%
N 3.1	1.5%	83.2	98.5	2.5	0.6%	10.4%	89.0%
E 3.1	1.2%	99.0	108.4	2.5	0.5%	7.1%	92.4%
S 3.1	1.5%	88.3	103.2	2.5	0.5%	9.3%	90.2%
W 3.1	1.4%	85.6	102.3	2.6	0.6%	10.8%	88.7%
N 4.1	1.9%	115.1	114.8	2.8	0.4%	8.3%	91.3%
E 4.1	1.2%	88.8	103.4	2.4	0.5%	8.2%	91.3%
S 4.1	1.2%	85.9	102.9	2.6	0.6%	9.7%	89.7%
W 4.1	1.5%	92.7	105.0	2.7	0.5%	10.1%	89.4%
SE 1.1	1.5%	82.1	98.7	2.5	0.6%	11.0%	88.4%
SE 2.1	0.9%	89.4	103.4	2.4	0.5%	8.6%	90.9%
SE 3.1	1.1%	87.5	102.6	2.5	0.5%	9.6%	89.9%
SE 4.1	1.1%	88.6	102.5	2.4	0.5%	8.8%	90.6%
SE 5.1	1.0%	89.1	103.8	2.5	0.5%	8.9%	90.6%
NW 1.1	23.8%	9.8	10.5	5.9	3.7%	79.5%	16.8%
NW 2.1	1.0%	177.2	208.1	3.5	0.5%	7.9%	91.6%
NW 3.1	1.4%	134.0	132.7	3.1	0.5%	9.6%	90.0%
NW 4.1	1.5%	92.1	104.9	2.7	0.6%	10.1%	89.3%
NW 5.1	0.8%	168.0	148.7	2.9	0.3%	6.3%	93.5%
C 1.1	1.6%	94.3	113.6	2.6	0.5%	9.3%	90.2%
C 2.1	1.9%	89.3	116.9	3.0	0.7%	11.7%	87.7%

Raw counts of Foraminifera genera identified in surface samples

Location	*Ammotium* sp.	*Ammobaculites* spp.	*Miliammina* sp.	*Trochammina* sp.	*Ammoastuta* sp.	*Jadammina* sp.	*Ammonia* spp.	*Elphidium* spp.	*Textularia* spp.	*Arenporella* sp.	*Pseudothurammina* sp.	Total Count	Volume of sediment (cm^3)
D.0	100	25			2				1			128	0.0870
N 1.0	37	71										108	0.0753
E 1.0	54	47										101	0.0708
S 1.0	48	70										118	0.0844
W 1.0	68	61	1									130	0.0840
N 2.0	44	64										108	0.1088
E 2.0	59	47							1			107	0.0883
S 2.0	68	37										105	0.0754
W 2.0	87	73										160	0.0692
N 3.0	85	36	1						2		1	125	0.0544
E 3.0	49	50	1									100	0.1139
S 3.0	52	97										149	0.0923
W 3.0	66	43										109	0.0723
N 4.0	42	60							1			103	0.1029
E 4.0	54	49	1									104	0.1043
S 4.0	36	64										100	0.0689
W 4.0	66	48					1					115	0.0561
SE 1.0	68	36			1							105	0.0673
SE 2.0	239	122	1	1	2							365	0.1995
SE 3.0	196	147	1	1				1	1			347	0.2062
SE 4.0	170	114			1				4			289	0.2434
SE 5.0	73	27	2									102	0.3211
NW 1.0	143	42					1					186	0.1332
NW 2.0	88	24	2									114	0.2640
NW 3.0	102	31										133	0.0836
NW 4.0	148	136										284	0.0999
NW 5.0	191	93										284	0.3409
C 1.0	40	112										152	0.0525
C 2.0	65	62	1		1							129	0.0830

Raw counts of Foraminifera genera identified in down-core samples

Location	*Ammotium* sp.	*Ammobaculites* spp.	*Miliammina* sp.	*Trochammina* sp.	*Ammoastuta* sp.	*Jadammina* sp.	*Ammonia* spp.	*Elphidium* spp.	*Textularia* spp.	*Arenporella* sp.	*Pseudothurammina* sp.	Total Count	Volume of sediment (cm^3)
D.1	88	25										113	0.1530
N 1.1	41	64			2							107	0.1289
E 1.1	82	37	1									120	0.1684
S 1.1	46	56										102	0.1149
W 1.1	46	56										102	0.0969
N 2.1	67	24	4		2		7	1		1		106	0.3445
E 2.1	60	41	1	1							1	104	0.3979
S 2.1	37	63										100	0.1138
W 2.1	52	51		1								104	0.0976
N 3.1	34	67	1		1				1			104	0.1626
E 3.1	52	49	1					1				103	0.4412
S 3.1	55	49										104	0.3949
W 3.1	51	49										100	0.1300
N 4.1	63	61	1									125	0.1053
E 4.1	69	30	2		1		1					103	0.2984
S 4.1	39	73										112	0.1610
W 4.1	58	45										103	0.0961
SE 1.1	75	24		1								100	0.1199
SE 2.1	85	21				1						107	0.2394
SE 3.1	92	12		1								105	0.2066
SE 4.1	80	23		1								104	0.2732
SE 5.1	78	28										106	0.5906
NW 1.1	2	6					29	68				105	0.3029
NW 2.1	86	16	1									103	0.5934
NW 3.1	87	14	1								1	103	0.4956
NW 4.1	93	10			1							104	0.6189
NW 5.1	85	24		1	1							111	0.1743
C 1.1	52	84	2		2							140	0.1004
C 2.1	67	45			1	1	1					115	0.0578

APPENDIX B: CORRELATION COEFFICIENTS

Correlation coefficients of variables associated with Foraminifera assemblages in surface and down-core samples combined. Relationships do not include control samples.

Variable X	Variable Y	n	Pearson's r	p-Value	Spearman's ρ	p-Value
Total Count/cm^3	Distance from Discharge	54	-0.0405	0.7711	-0.0964	0.488
Total Count/cm^3	6-day Average Conductivity	54	0.0285	0.8379	-0.0284	0.8386
Total Count/cm^3	Water Depth	54	0.1943	0.1593	0.1032	0.4579
Total Count/cm^3	% OM	54	-0.1225	0.3774	0.3422	*0.0113
Total Count/cm^3	% OM (without two outliers)	52	0.5614	*<.0001	0.4577	*0.0006
Total Count/cm^3	Longitudinal Location	54	-0.3114	*0.0219	-0.2003	0.1465
Total Count/cm^3	Mean Grain Size	54	-0.0981	0.4805	-0.2372	0.0842
Total Count/cm^3	Median Grain Size	54	-0.077	0.5802	-0.1895	0.17
Total Count/cm^3	Grain Size Standard Deviation	54	-0.1484	0.2843	0.1049	0.4501
Total Count/cm^3	% Sand	54	0.076	0.5849	-0.4739	*0.0003
Total Count/cm^3	% Silt	54	-0.0673	0.6287	0.4987	*0.0001
Total Count/cm^3	% Clay	54	-0.2592	0.0584	-0.5004	*0.0001
Hill's N_1	Distance from Discharge	54	-0.0554	0.6907	-0.1365	0.325
Hill's N_1	6-day Average Conductivity	54	0.063	0.6509	-0.024	0.8635
Hill's N_1	Water Depth	54	-0.1672	0.2268	-0.0844	0.5438
Hill's N_1	% OM	54	0.3597	*0.0075	0.232	0.0913
Hill's N_1	% OM (without two outliers)	52	0.2495	0.0745	0.1399	0.3225
Hill's N_1	Longitudinal Location	54	-0.0794	0.5683	-0.0994	0.4745
Hill's N_1	Mean Grain Size	54	-0.5531	*<.0001	-0.2812	*0.0394
Hill's N_1	Median Grain Size	54	-0.5096	*<.0001	-0.2046	0.1378
Hill's N_1	Grain Size Standard Deviation	54	0.2487	0.0698	0.0135	0.923
Hill's N_1	% Sand	54	-0.3997	*0.0028	-0.213	0.122
Hill's N_1	% Silt	54	0.401	*0.0027	0.2096	0.1283
Hill's N_1	% Clay	54	0.3579	*0.0079	0.0869	0.5319

Correlation coefficients of variables associated with Foraminifera assemblages in surface samples only. Relationships do not include control samples.

Variable X	Variable Y	n	Pearson's r	p-Value	Spearman's ρ	p-Value
Total Count/cm^3	Distance from Discharge	27	0.0347	0.8636	0.0061	0.9759
Total Count/cm^3	6-day Average Conductivity	27	-0.064	0.7511	-0.1886	0.346
Total Count/cm^3	Water Depth	27	0.3222	0.1012	0.2358	0.2364
Total Count/cm^3	% OM	27	0.5183	*0.0056	0.3645	0.0616
Total Count/cm^3	Longitudinal Location	27	-0.4094	*0.034	-0.3317	0.091
Total Count/cm^3	Mean Grain Size	27	-0.1744	0.3844	-0.21	0.2931
Total Count/cm^3	Median Grain Size	27	-0.1304	0.5169	-0.1905	0.3413
Total Count/cm^3	Grain Size Standard Deviation	27	0.3225	0.1009	0.2637	0.1838
Total Count/cm^3	% Sand	27	-0.41	*0.0337	-0.464	*0.0148
Total Count/cm^3	% Silt	27	0.4118	*0.0328	0.4573	*0.0165
Total Count/cm^3	% Clay	27	0.1223	0.5432	0.1801	0.3687
Hill's N_1	Distance from Discharge	27	0.074	0.7136	0.0006	0.9976
Hill's N_1	6-day Average Conductivity	27	-0.0673	0.7387	-0.1246	0.5359
Hill's N_1	Water Depth	27	-0.404	*0.0366	-0.3102	0.1153
Hill's N_1	% OM	27	0.0916	0.6496	-0.2052	0.3046
Hill's N_1	Longitudinal Location	27	0.066	0.7435	0.1011	0.6158
Hill's N_1	Mean Grain Size	27	-0.4702	*0.0133	-0.1453	0.4695
Hill's N_1	Median Grain Size	27	-0.3645	0.0616	-0.033	0.8703
Hill's N_1	Grain Size Standard Deviation	27	-0.3115	0.1138	-0.279	0.1587
Hill's N_1	% Sand	27	-0.1018	0.6133	0.084	0.6772
Hill's N_1	% Silt	27	0.0988	0.6238	-0.0888	0.6595
Hill's N_1	% Clay	27	0.2587	0.1925	0.0085	0.9662

Correlation coefficients of variables associated with Foraminifera assemblages in down-core samples only. Relationships do not include control samples.

Variable X	Variable Y	n	Pearson's r	p-Value	Spearman's ρ	p-Value
Total Count/cm^3	Distance from Discharge	27	-0.2163	0.2786	-0.2045	0.3062
Total Count/cm^3	6-day Average Conductivity	27	0.2197	0.2709	0.0263	0.8966
Total Count/cm^3	Water Depth	27	0.2034	0.3089	0.1058	0.5996
Total Count/cm^3	% OM	27	-0.1684	0.4011	0.1587	0.4291
Total Count/cm^3	% OM (without two outliers)	25	0.3739	0.0656	0.3038	0.1398
Total Count/cm^3	Longitudinal Location	27	-0.5087	*0.0067	-0.2902	0.1421
Total Count/cm^3	Mean Grain Size	27	-0.0387	0.8478	-0.2204	0.2693
Total Count/cm^3	Median Grain Size	27	-0.0842	0.6761	-0.2051	0.3047
Total Count/cm^3	Grain Size Standard Deviation	27	-0.2676	0.1772	0.0244	0.9038
Total Count/cm^3	% Sand	27	0.1856	0.354	-0.1728	0.3888
Total Count/cm^3	% Silt	27	-0.1848	0.356	0.2021	0.3121
Total Count/cm^3	% Clay	27	-0.2013	0.3139	0.1777	0.3753
Hill's N_1	Distance from Discharge	27	-0.1247	0.5353	-0.2459	0.2163
Hill's N_1	6-day Average Conductivity	27	0.1336	0.5066	0.069	0.7322
Hill's N_1	Water Depth	27	-0.07	0.7285	0.0472	0.8151
Hill's N_1	% OM	27	0.4339	*0.0237	0.4842	*0.0105
Hill's N_1	% OM (without two outliers)	25	0.2802	0.1749	0.35	0.0864
Hill's N_1	Longitudinal Location	27	-0.1592	0.4278	-0.2017	0.3129
Hill's N_1	Mean Grain Size	27	-0.5688	*0.002	-0.4454	*0.0199
Hill's N_1	Median Grain Size	27	-0.5474	*0.0031	-0.4356	*0.0231
Hill's N_1	Grain Size Standard Deviation	27	0.3691	0.0582	0.2111	0.2905
Hill's N_1	% Sand	27	-0.4664	*0.0142	-0.3733	0.0551
Hill's N_1	% Silt	27	0.4661	*0.0143	0.3907	*0.0439
Hill's N_1	% Clay	27	0.4724	*0.0128	0.4631	*0.015

Correlation coefficients of variables associated with water depth and sediment composition in surface and down-core samples combined. Relationships include control samples.

Variable X	Variable Y	n	Pearson's r	p-Value	Spearman's ρ	p-Value
LOI (%OM by weight)	Water Depth	58	0.1571	0.2388	0.3212	*0.014
LOI (%OM by weight)	Mean Grain Size	58	-0.5653	*<.0001	-0.3346	*0.0103
LOI (%OM by weight)	Median Grain Size	58	-0.6524	*<.0001	-0.2505	0.0579
LOI (%OM by weight)	Grain Size Standard Deviation	58	0.8351	*<.0001	0.5484	*<.0001
LOI (%OM by weight)	% Sand	58	-0.9742	*<.0001	-0.6814	*<.0001
LOI (%OM by weight)	% Silt	58	0.9729	*<.0001	0.6835	*<.0001
LOI (%OM by weight)	% Clay	58	0.9646	*<.0001	0.0209	0.876
LOI (%OM by weight)	% Fine Sand	58	-0.3574	*0.0059	-0.0819	0.5413
LOI (%OM by weight)	% Very Fine Sand	58	-0.6685	*<.0001	-0.3682	*0.0045
Water Depth	Mean Grain Size	58	0.2254	0.0889	0.3214	*0.0139
Water Depth	Median Grain Size	58	0.2022	0.1279	0.3951	*0.0021
Water Depth	Grain Size Standard Deviation	58	0.2711	*0.0395	0.5557	*<.0001
Water Depth	% Sand	58	-0.1238	0.3545	-0.1831	0.1689
Water Depth	% Silt	58	0.1247	0.3512	0.1827	0.1698
Water Depth	% Clay	58	0.1009	0.451	-0.0362	0.7873
Water Depth	% Fine Sand	58	0.4332	*0.0007	0.1586	0.2344
Water Depth	% Very Fine Sand	58	-0.5322	*<.0001	-0.6568	*<.0001
% OM (without two outliers)	Water Depth	56	0.3234	*0.0151	0.3119	*0.0193
% OM (without two outliers)	Mean Grain Size	56	-0.2871	*0.0319	-0.2607	0.0523
% OM (without two outliers)	Median Grain Size	56	-0.1991	0.1412	-0.1673	0.2179
% OM (without two outliers)	Grain Size Standard Deviation	56	0.3953	*0.0026	0.4983	*<.0001
% OM (without two outliers)	% Sand	56	-0.67	*<.0001	-0.6461	*<.0001
% OM (without two outliers)	% Silt	56	0.6718	*<.0001	0.6484	*<.0001
% OM (without two outliers)	% Clay	56	-0.0523	0.7019	-0.0878	0.5201
% OM (without two outliers)	% Fine Sand	56	0.2347	0.0816	0.0201	0.8831
% OM (without two outliers)	% Very Fine Sand	56	-0.1264	0.3531	-0.3077	*0.0211

Correlation coefficients of variables associated with water depth and sediment composition in surface samples only. Relationships include control samples.

Variable X	Variable Y	n	Pearson's r	p-Value	Spearman's ρ	p-Value
LOI (%OM by weight)	Water Depth	29	0.2893	0.1279	0.2036	0.2894
LOI (%OM by weight)	Mean Grain Size	29	-0.1865	0.3328	-0.2764	0.1467
LOI (%OM by weight)	Median Grain Size	29	0.0408	0.8335	-0.1852	0.3361
LOI (%OM by weight)	Grain Size Standard Deviation	29	0.6788	*<.0001	0.6493	*0.0001
LOI (%OM by weight)	% Sand	29	-0.646	*0.0002	-0.6601	*<.0001
LOI (%OM by weight)	% Silt	29	0.6496	*0.0001	0.6749	*<.0001
LOI (%OM by weight)	% Clay	29	0.1412	0.4651	0.1315	0.4964
LOI (%OM by weight)	% Fine Sand	29	0.1351	0.4849	-0.1719	0.3725
LOI (%OM by weight)	% Very Fine Sand	29	-0.4386	*0.0173	-0.4438	*0.0159
Water Depth	Mean Grain Size	29	0.3762	*0.0443	0.3884	*0.0373
Water Depth	Median Grain Size	29	0.5803	*0.001	0.484	*0.0078
Water Depth	Grain Size Standard Deviation	29	0.4285	*0.0204	0.3949	*0.034
Water Depth	% Sand	29	-0.053	0.7847	-0.0394	0.8393
Water Depth	% Silt	29	0.0557	0.7741	0.0394	0.8393
Water Depth	% Clay	29	-0.1453	0.4519	-0.2101	0.2741
Water Depth	% Fine Sand	29	0.532	*0.003	0.2098	0.2747
Water Depth	% Very Fine Sand	29	-0.7164	*<.0001	-0.6287	*0.0003

Correlation coefficients of variables associated with water depth and sediment composition in down-core samples only. Relationships include control samples.

Variable X	Variable Y	n	Pearson's r	p-Value	Spearman's ρ	p-Value
LOI (%OM by weight)	Water Depth	29	0.198	0.3031	0.429	*0.0202
LOI (%OM by weight)	Mean Grain Size	29	-0.6117	*0.0004	-0.3399	0.0712
LOI (%OM by weight)	Median Grain Size	29	-0.6781	*<.0001	-0.2468	0.1968
LOI (%OM by weight)	Grain Size Standard Deviation	29	0.8494	*<.0001	0.4704	*0.01
LOI (%OM by weight)	% Sand	29	-0.9835	*<.0001	-0.6069	*0.0005
LOI (%OM by weight)	% Silt	29	0.9834	*<.0001	0.6212	*0.0003
LOI (%OM by weight)	% Clay	29	0.9817	*<.0001	0.5108	*0.0046
LOI (%OM by weight)	% Fine Sand	29	-0.4985	*0.0059	0.0946	0.6255
LOI (%OM by weight)	% Very Fine Sand	29	-0.7004	*<.0001	-0.2394	0.211
Water Depth	Mean Grain Size	29	0.2045	0.2873	0.2561	0.1799
Water Depth	Median Grain Size	29	0.1882	0.3283	0.3406	0.0706
Water Depth	Grain Size Standard Deviation	29	0.285	0.134	0.7253	*<.0001
Water Depth	% Sand	29	-0.169	0.3808	-0.2351	0.2196
Water Depth	% Silt	29	0.1695	0.3794	0.247	0.1965
Water Depth	% Clay	29	0.1575	0.4144	0.2685	0.159
Water Depth	% Fine Sand	29	0.3444	0.0674	0.1412	0.465
Water Depth	% Very Fine Sand	29	-0.5328	*0.0029	-0.652	*0.0001
% OM (without two outliers)	Water Depth	27	-0.3283	0.0945	0.4186	*0.0298
% OM (without two outliers)	Mean Grain Size	27	-0.2697	0.1736	-0.1819	0.3638
% OM (without two outliers)	Median Grain Size	27	0.0687	0.7333	-0.0665	0.7416
% OM (without two outliers)	Grain Size Standard Deviation	27	-0.5617	*0.0023	0.3437	0.0792
% OM (without two outliers)	% Sand	27	0.5616	*0.0023	-0.5128	*0.0062
% OM (without two outliers)	% Silt	27	0.3712	0.0566	0.5305	*0.0044
% OM (without two outliers)	% Clay	27	0.4873	*0.0099	0.3938	*0.0421
% OM (without two outliers)	% Fine Sand	27	0.0621	0.7584	0.3565	0.0679
% OM (without two outliers)	% Very Fine Sand	27	0.4212	*0.0287	-0.0965	0.6322