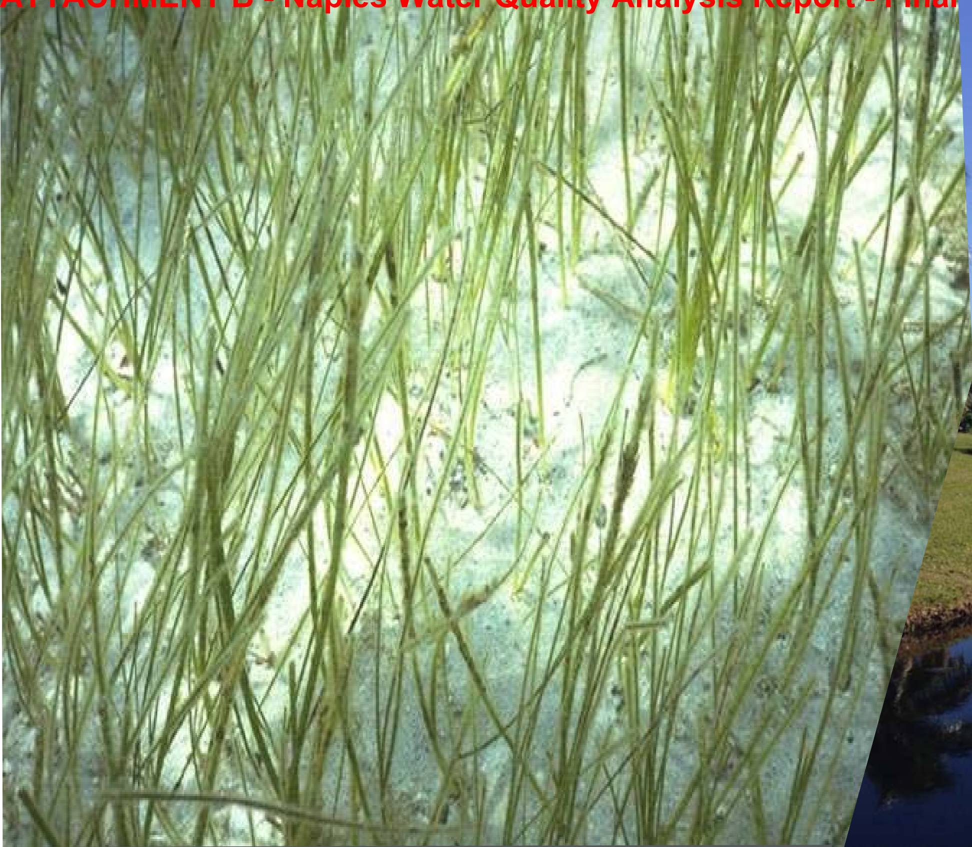

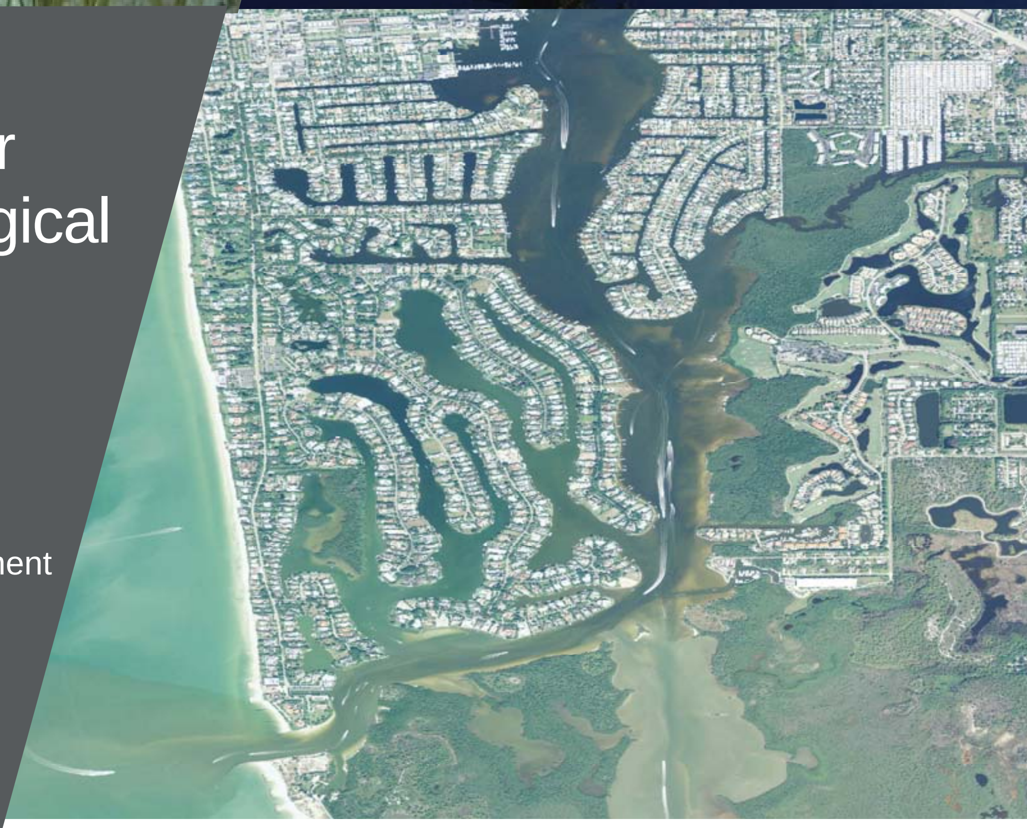

# Naples Bay Water Quality and Biological Analysis Project

Prepared for:
The City of Naples
Streets and Stormwater Department
Natural Resources Division

# Final Report

August 2015

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# Document Information

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# Executive Summary

This study provides a comprehensive update on the current status of water quality and biology in Naples Bay. The analysis and information here will be valuable for making informed decisions regarding the management and restoration of Naples Bay. This effort focuses on Naples Bay water quality and biological communities (fish and seagrass), stormwater inputs to the Bay (stormwater lakes and pump stations), and effects of the Golden Gate Canal system, along with a comparison of the current status with a previous similar study conducted in the late 1970s (Simpson *et al.* 1979).

Data for this project were compiled from publicly available sources, focusing primarily on the water quality and biological monitoring conducted by the City of Naples Natural Resource Division. Additional data were compiled from Collier County, the Florida Department of Environmental Protection (FDEP), the South Florida Water Management District (SFWMD), and the National Oceanic and Atmospheric Administration (NOAA). Statistical analyses were conducted to identify significant trends in water quality, water quantity, and biology; identify links between water quality and effects on biology; determine the potential effect of stormwater inputs on Bay water quality; and attempt to quantify the effects of ongoing management activities.

A summary of the major findings of this effort are:

- Naples Bay exhibits a statistically significant decreasing trend in nutrients;
- Increasing trends in copper, chlorophyll *a*, turbidity, and bacteria (fecal coliform and enterococci) are observed;
- Naples Bay water quality is significantly improved relative to available data from the late 1970s;
- Freshwater contributions along with nutrient and solids loadings from the Golden Gate Canal significantly affect water quality in Naples Bay;
- Seagrass in southern Naples Bay exhibits a decreasing trend over time;
- Nutrient and solids loading to Naples Bay likely contribute to the decreasing trend in seagrass;
- Univariate metrics (abundance, richness, and diversity) of the fish community in Naples Bay show similarity over time, across zones, and seasons;
- Fish community structure in Naples Bay is similar to that of other southwest Florida estuaries with less urban development and less impact from flow diversions;

This effort identifies focus areas for further investigation that will inform ongoing management and restoration efforts. Statistically significant trends in water quality and biology in Naples Bay were identified that will be useful to resource managers. The characterization of the current biological community provides a baseline for future management actions to measure progress and achieve restoration goals.

# 1 Introduction

The Naples Bay estuary is a focal point in southwestern Florida providing both a commercial working waterfront and a beautiful residential setting with abundant recreational and sporting opportunities for the residents and guests of the City of Naples. Naples Bay has a long history of transformation and development since the first European settlers arrived in the 1860s (Schmid *et al.* 2005). Urbanization and dredge-and-fill activities since the 1950s and 1960s have affected the function of the shallow-water Naples Bay estuary (Schmid *et al.* 2005). Significant canal drainage, dredged channels, and urban development have altered the Bay's water quality with the most notable change being the freshwater inflows from the Golden Gate Canal (GGC) system, which was constructed in the 1960s (SFWMD 2007). The Naples Bay watershed historically drained approximately ten square miles, but now drains approximately 120 square miles as a result of the GGC drainage system (SFWMD 2007, Schmid *et al.* 2005, FDEP 2010).

Studies have shown that the human-induced changes to the Naples Bay watershed have had a significant effect on the biological character of the estuarine system. Schmid *et al.* (2005) reported that Naples Bay has lost 90 percent of its seagrass beds, 80 percent of the oyster reefs, and 70 percent of the mangrove fringe since the 1950s. Salinity stresses from unnatural freshwater inflows have affected plankton, benthic, and fish communities (FDEP 2010). Reports dating back to the 1950s have documented impacts to the Naples Bay aquatic system as a result of the hydrologic alterations (Baum 1973, Simpson *et al.* 1979, SFWMD 2007).

In addition to the effect of freshwater on the biological community in Naples Bay, stormwater directed into Naples Bay carries pollutants, such as heavy metals, bacteria, sediment, fertilizers, herbicides and pesticides (Simpson *et al.* 1979, City of Naples 2010). Copper has been identified as a major pollutant in Naples Bay. The Florida Department of Environmental Protection (FDEP) listed Naples Bay as impaired for copper and iron during their Cycle 2, Group 1 assessment in 2009. Copper sulfate has been used for decades in Naples (and throughout Florida and the country) as an algaecide in stormwater retention lakes. Over time, copper can accumulate in stormwater lakes and be released into receiving waters (in this case Naples Bay) where it can become toxic to estuarine life. Evidence of copper accumulation in Naples Bay was described in a report by the National Oceanic and Atmospheric Administration (NOAA) which stated oysters in Naples Bay had some of the highest copper concentrations observed anywhere in the nation (Kimbrough *et al.* 2008).

The story of Naples Bay is not unique and mimics that of other estuaries that experience a rapid rate of development and urbanization. Residents and guests alike are drawn to the natural appeal of the estuary and the recreational, sporting, and commercial opportunities it offers, but all too often the urbanization that follows creates adverse environmental effects that diminish that very appeal. However, the story of Naples Bay is far from complete and the City of Naples and other stakeholders are pro-actively engaging in identifying sources of the adverse effects and creating restoration plans to mitigate for them. The City of Naples Twenty Year Restoration Plan (City of Naples 2010) outlines several efforts currently underway to identify and reverse the adverse environmental effects described above including diversion of freshwater inflows from the GGC system, installation of floating wetlands in stormwater lakes, adoption of a fertilizer ordinance, adoption of a city ordinance banning the use of copper sulfate, creation of filter marshes, public outreach and educational programs, along with oyster, mangrove, and seagrass restoration efforts.

A critical component of the process of restoring Naples Bay is a water quality and biological monitoring program directed at identifying environmental issues and their sources in addition to tracking progress and improvements associated with the restoration activities. In 2006, the City's Natural Resources Division implemented a monitoring program in Naples and Moorings Bays that includes a wide range of water quality constituents of interest paired with seagrass monitoring and trawling efforts to characterize the fish communities of the Bays. Some changes to the program were made in 2010, but the monitoring continues today.

Cardno, Inc. (Cardno) was retained by the City of Naples, Streets and Stormwater Department to complete the Naples Bay Water Quality Analysis Project aimed at characterizing the current status of water quality and biological communities in Naples Bay along with the effects of ongoing management and restoration activities. This project uses the data that have been collected by the City since 2006 as well as other publicly available sources (Section 2) to identify statistically and ecologically significant trends and inter-connected relationships between the water quality and biological variables. The goal is to tell a scientifically sound and statistically defensible story of Naples Bay and the management efforts dedicated to restoring it. In addition, recommendations for changes to the current water quality and biology monitoring programs are made based on what was learned during the analysis of existing data.

At the outset, a series of focused questions were developed as guiding principles for the project. These questions centered on identifying quantifiable relationships in the data that can be relied upon to inform current and future management activities. These questions are listed below:

1. Are statistically significant trends in Naples Bay water quality data observed spatially and temporally?
2. Are statistically significant trends in Naples Bay biological data (fish and seagrass) observed spatially and temporally?
3. Are statistically and ecologically significant changes in the Naples Bay biological community observed as a result of changes in water quality?
4. What measurable effect, if any, do contributions from the City's stormwater lakes and pump stations have on water quality and biology in Naples Bay?
5. What quantifiable effect are the City's management activities having on water quality and biology in Naples Bay?

This document details the effort to complete the Naples Bay Water Quality Analysis Project and the results of the investigation. We also present the implications for management and restoration of Naples Bay.

# 2 Data Sources

Data from the City of Naples monitoring programs along with publicly available data were used as the basis for the analysis presented in this report. Water quality and biological data from Naples Bay, along with relevant data from other nearby estuaries such as Rookery Bay, Estero Bay, Fakahatchee Bay, Pumpkin Bay, and Faka Union Bay were compiled. Each sampling agency maintains their own monitoring program with differing sampling frequencies and constituents, and this section briefly describes the data compiled from each sampling entity. All water quality and biological data are maintained in a Microsoft Access database.

## 2.1 Water Quality and Quantity Data

The water quality analytical effort focused on constituents of concern to the City and those that are of regulatory concern to the FDEP with regard to the health of Naples Bay. Particular attention was paid to nutrients and nutrient response variables, heavy metals (copper), bacteria counts, and freshwater inputs (measured as salinity and/or conductivity). These parameters have been identified in previous studies and discussions with the City as those of the most interest for this effort.

Water quality data for this effort were obtained from the South Florida Water Management District (SFWMD), Collier County, and the Florida **STO**rage and **RET**rieval (STORET) database, although the primary source of data was the City of Naples Natural Resources Division. Data from each source are briefly described below (Table 2-1). A depiction of all water quantity and quality data locations is provided in Figures 2-1 and 2-2, respectively.

Data were summarized by calendar year or water year (WY) for certain analyses throughout the report and are noted as such where applicable. Some analyses were also conducted on seasonal data, with December through May designated as the dry season and June through November designated as the wet season. The division of months into each season was based on the designations in the biological data provided by the City of Naples and supported by an analysis of flow data from the Golden Gate Canal. In order to make links between water quality and biological data, the same seasonal divisions were used for seasonal analysis of the water quality dataset.

**Table 2-1. Water quality and quantity data sources, *Naples Bay Water Quality Analysis Project.***

| Data Source | Location | Data type | Number of Stations | Date Range* | Number of Records |
|---|---|---|---|---|---|
| City of Naples | Naples Bay | Grab | 16 | 2005–2010 | 480 |
| | Naples Bay | Grab | 8 | 2011–Present | 384 |
| | Stormwater Lakes | Grab | 15 | 2010–Present | 81 |
| | Pump Stations | Grab | 3 | 2010–Present | 32 |
| | Pump Stations | Flow | 3 | 2011–2014 | Annual Totals |
| Collier County | Naples Bay and Tributaries | Grab | 7 | 1995–2014 | 1900 |
| | Collier County Facilities Management | Rainfall | 1 | 2008–2014 | Daily Records |
| USGS | Naples Bay | Continuous Recorder | 4 | 2011–2014 | 440,420 |
| SFWMD | Naples Bay | Grab | 14 | 2000–2014 | 49,260 |
| | Golden Gate Canal | Flow | 1 | 2008–2014 | Daily Records |
| NOAA–NERRS | Henderson Ck | Continuous Recorder | 1 | 2011–2014 | 118,000 |
| NOAA | Golden Gate Canal | Rainfall | 1 | 1977–2014 | Daily Records |
| FDEP STORET | Naples Bay | Grab | 62 | 1998–2014 | 770 |
| FDEP | Estero Bay | Continuous Recorders | 3 | 2011–2014 | 143,140 |

* Represents longest range for the data source; individual station ranges may differ from time frame listed

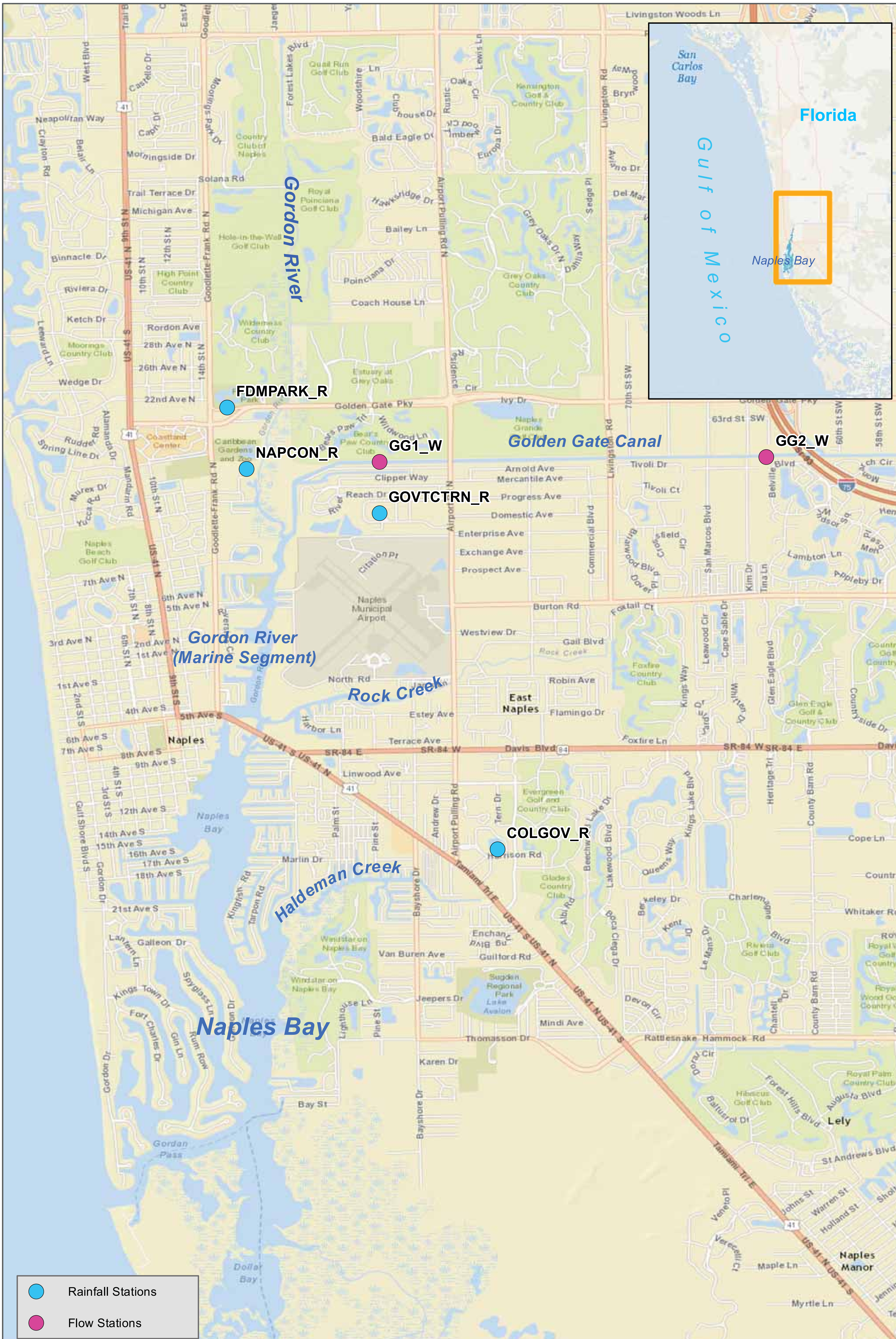


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**Figure 2-1. Water Quantity Data Locations, Naples Bay**

City of Naples, Natural Resources Division
Collier County, Florida

0 1 Miles
0 1 2 Kilometers

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GIS Analyst: adam.bausch

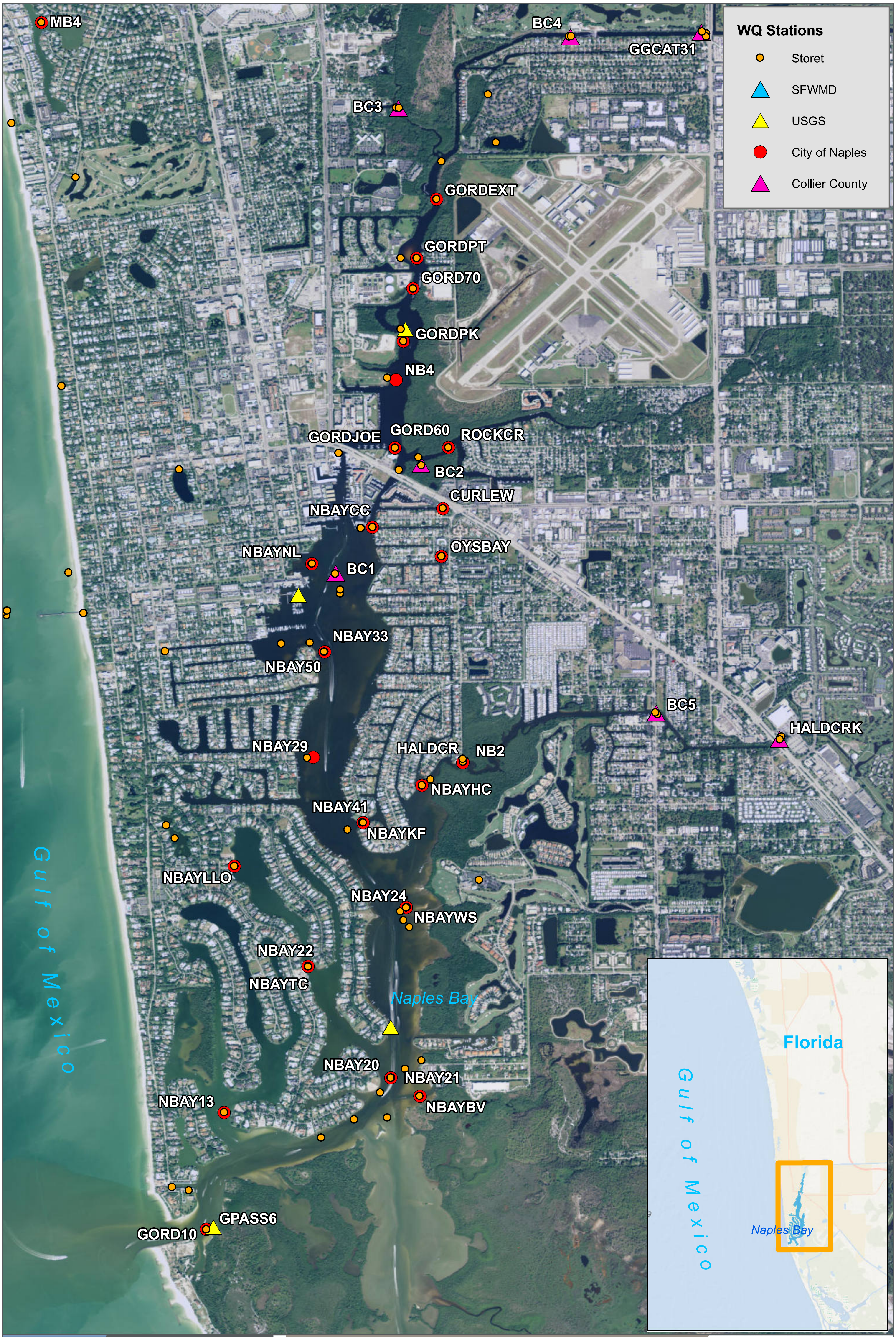


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**Figure 2-2. Water Quality Data Locations, Naples Bay**
City of Naples, Natural Resources Dvision
Collier County, Florida

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GIS Analyst: Lauren.Federsel

## 2.2 Biological Data

The primary source of biological data was the City of Naples Natural Resources Division ongoing monitoring efforts. Additional data, provided by the City to Cardno, were compiled from other southwest Florida estuary monitoring programs to serve as comparison to Naples Bay data. Monitoring programs are very similar between the southwest Florida estuaries. A brief description of the biological data used in this effort is provided below.

**Table 2-2. Biological data sources, *Naples Bay Water Quality Analysis Project*, 2006-2014.**

| Organization | Sample Type | Location | Approximate Date Range | Description |
|---|---|---|---|---|
| City of Naples | Seagrass | Southern Naples Bay | 2006–2014 | Five transects sampled once or twice per year between April and October. Quadrats placed at fixed points along transect: species composition, cover (Braun-Blanquet scale), shoot count, blade length, qualitative sediment type, water depth, and relative epiphyte coverage recorded. |
| | Fish - Trawling | Naples and Moorings Bays | 2009–2011 | Otter trawls pulled for specific lengths and times at four fixed locations in each Bay. Naples Bay was trawled approximately six times per year; Moorings Bay was trawled four times per year. Species identity and abundance recorded. Length of first 20 individuals of each species recorded. Bycatch and environmental conditions recorded. |
| | | Naples and Moorings Bays | 2011–2014 | Otter trawls pulled for specific length and time. Four grid zones established in each Bay. A random grid box is selected within each zone for sampling in each Bay during each event. Naples Bay is trawled six times per year; Moorings Bay is trawled four times per year. Species were identified and abundance recorded. Length of first 20 individuals of each species recorded. Bycatch and environmental conditions recorded. |
| Rookery Bay National Estuarine Research Reserve | Fish - trawling | Rookery Bay | 2009–2011 | Otter trawls pulled for specific length and time. A random grid box was selected for sampling at each event. Sampling approximately every other month from Apr 2009 to Apr 2011. Species identity and abundance recorded. Length of first 20 individuals of each species recorded. Bycatch and environmental conditions recorded. |
| | | Fakahatchee Bay | 2009–2013 | Otter trawls pulled for specific length and time. A random grid box was selected for sampling within each bay during each event. All bays trawled six times per year. Species identity and abundance recorded. Length of first 20 individuals of each species recorded. Bycatch and environmental conditions recorded. |
| | | Faka Union Bay | | |
| | | Pumpkin Bay | | |

# 3 Naples Bay Water Quality and Quantity

Changes in water quality and biology in Naples Bay as a result of rapid urbanization and hydrologic changes from the GGC are a long-standing concern. The first step in solving any issue is identifying the problem and its sources then developing scientifically defensible and economically feasible solutions. Several public entities have been collecting water quality in Naples Bay, its tributaries, and the GGC dating back to the late 1990s. However, it wasn't until 2006 when the City of Naples instituted a more robust water quality and biological monitoring program that a more comprehensive characterization of Naples Bay was possible. Combining all of the available data from Naples Bay and its contributing sources provides the opportunity to not only characterize the current status, but also to identify statistically and ecologically significant trends over time and the sources of those trends. This effort will assist in determining if Naples Bay water quality is in compliance with applicable water quality criteria and whether water quality and biological conditions are trending toward improvement or degradation, as well as provide the basis for determining if ongoing management programs are improving Naples Bay.

This section provides a characterization of the water sources to the Bay, quantification of volumes and loadings to the Bay, and a statistical analysis of the significant trends in Naples Bay water quality.

## 3.1 Sources of Water to Naples Bay

Naples Bay is a shallow, narrow estuary, oriented north to south along Florida's southwest coast with several freshwater inputs (FDEP 2010). The Bay has a single pass (Gordon Pass) at the southern end of the Bay providing water exchange with the Gulf of Mexico. Naples Bay south of Gordon Pass is connected to Rookery Bay which connects to the Marco River further south by a shallow dredged channel (FDEP 2010). The major sources of freshwater to Naples Bay include the GGC, the Gordon River, Rock Creek, Haldeman Creek, and urban stormwater runoff from the surrounding areas. During the economic boom of the early 2000s, the City of Naples and Collier County were among the fastest growing areas in Florida (FDEP 2010).

A characterization of the water quality and quantity of freshwater sources to Naples Bay is provided here. This discussion focuses on the GGC and the stormwater inputs to Naples Bay as a lack of information exists on the contributions of the other sources to the Bay. The SFWMD measures gauge height at the weirs from the Gordon River and Haldeman Creek into Naples Bay, but lacks flow measurements necessary to determine pollutant loads to Naples Bay from these sources. Some water quality data are available for Haldeman Creek, which is included in the discussion in the next section (Section 3.2). Paired flow and water quality measurements from the Gordon River, Rock Creek, and Haldeman Creek would be valuable in establishing a more robust characterization of the sources of water to Naples Bay.

### 3.1.1 Golden Gate Main Canal

The Golden Gate Canal system is widely recognized as the major source of freshwater to Naples Bay (Laakkonen 2014, Schmid *et al.* 2005, SFWMD 2007, FDEP 2010, Simpson *et al.* 1979). The canal system was built in the 1960s to drain wetland systems to the northeast of Naples Bay and facilitate residential development (SFWMD 2006). Historically, Naples Bay had a drainage area of approximately 10 square miles; however, following the construction of the GGC, the area draining to Naples Bay grew to approximately 120 square miles (Laakkonen 2014 and City of Naples 2010) (Figure 3-1). The SFWMD operates three weirs along the Golden Gate Main Canal system which have upgraded through the 2000s to improve flood control and better manage freshwater flows into Naples Bay (SFWMD 2006).

Current flow contributions to the Gordon River (Marine Segment) and Naples Bay are available from flow data recorded at the downstream most weir (GGC1). Daily flow data from this gauge are available from September 21, 2008 through December 31, 2014 (Figure 3-2). Average daily flow over this time period was approximately 77 mgd, including times of no flow. When the GGC1 weir is flowing the average daily discharge is 123 mgd. As expected, flow from the canal system is rainfall driven and, therefore, the highest magnitude flows are concentrated during the wet season (approximately June through November).

The period of record flow dataset was not complete because the flow gauge at GGC1 was not operating during some time periods. A regression model estimating flow at station GGC1 and the next upstream gauge in the Golden Gate Main Canal system, GGC2, was used to estimate flow over the GGC1 weir into the Gordon River (Marine Segment) and Naples Bay where data were missing (Figure 3-3). A strong correlation ($R^2$ = 0.9) between flow at GGC1 and GGC2 provided the opportunity to predict flow at GGC1 during times when the flow gauge was not operating. Including the time periods of estimated flow allows for a more robust characterization of the flow regime from the GGC system into receiving waters. The estimated flows are shown in green in Figure 3-2.

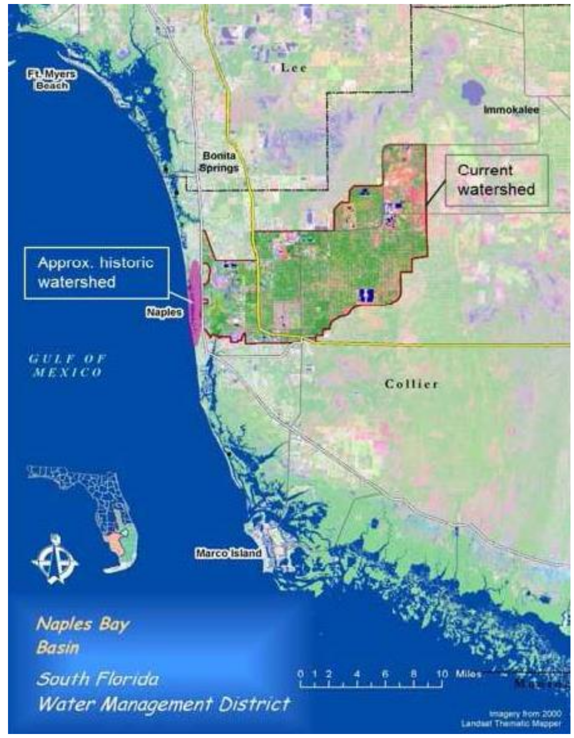


**Figure 3-1. Historical and current Naples Bay watershed resulting from construction of Golden Gate Canal. (Reprinted from Laakkonen 2014.)**

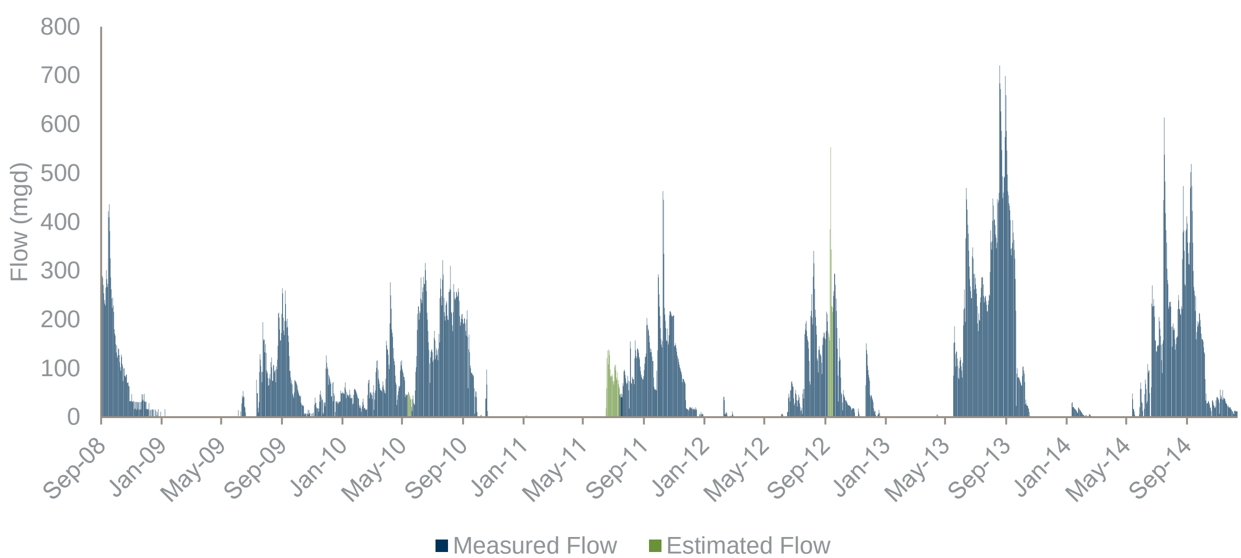


**Figure 3-2. Golden Gate Main Canal daily flow into Gordon River (Marine Segment), September 21, 2008–December 31, 2014. SFWMD.**

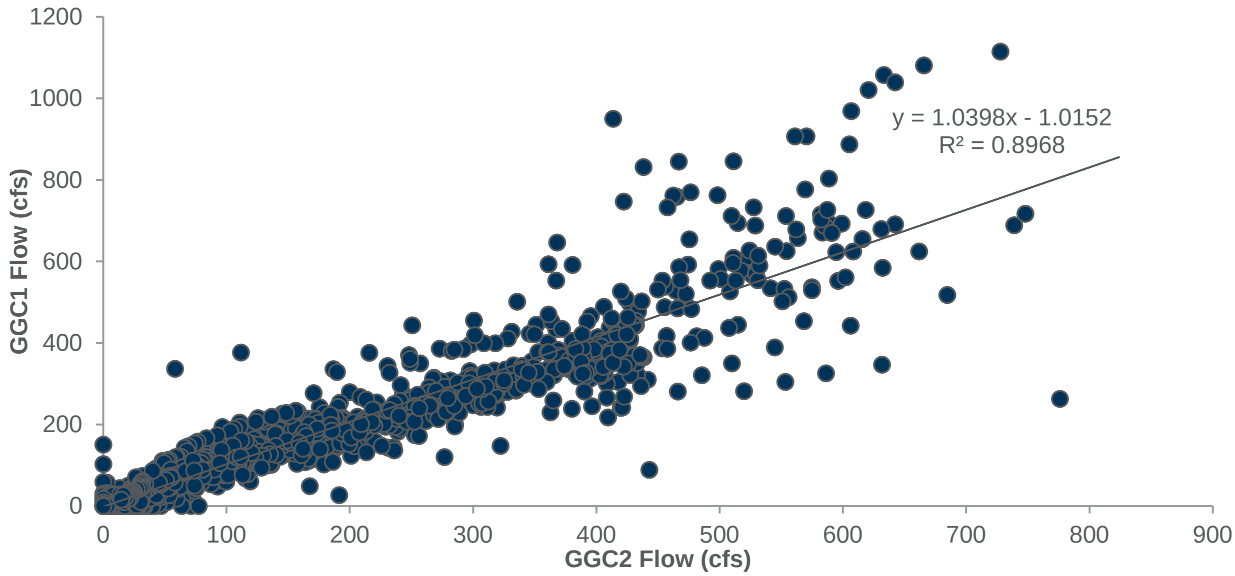


**Figure 3-3.** **Flow relationship between Golden Gate Canal Weir 1 and Golden Gate Canal Weir 2, September 21, 2008–December 31, 2014.**

Total freshwater flow during the six months of the wet season (June–November) ranges from approximately 10 to over 40 billion gallons, typically constituting over 90 percent of the annual freshwater flow delivered from the GGC to Naples Bay (Figure 3-4). An exception was when higher dry season rainfall amounts during 2010 led to higher than normal flow from the GGC during that time period. The highest flows were observed during 2013, when canal flow during the six months of the wet season was larger than the total annual flow for any other year for which data are available.

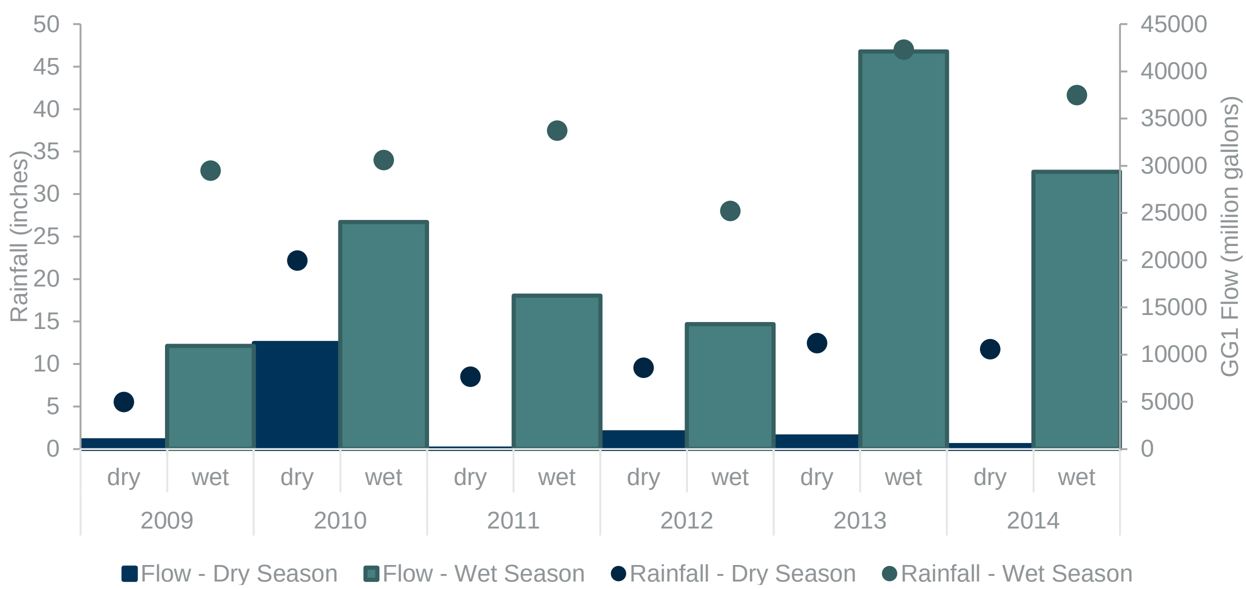


**Figure 3-4.** **Total wet and dry season rainfall and flows from the Golden Gate Main Canal into Gordon River (Marine Segment), December 2008–November 2014.**

Along with the large volume of freshwater, the GGC also delivers significant loadings of potential pollutants to Naples Bay. Collier County monitors water quality in the GGC upstream of the GGC1 weir (station GGCAT31) that allows for loading calculations of the canal contributions to the Gordon River (Marine Segment) and Naples Bay (see Figure 2-2). Although several water quality constituents are monitored at this location, this analysis will focus on nutrients, copper, and suspended solids as the constituents of concern that represent potential impacts to the Bay. Loadings were calculated for 2009–2014 using water quality measurements from the GGCAT31 sampling location. Monthly (or quarterly, in the case of copper) values were assumed to be representative of the contribution for that calendar month (or quarter).

As expected, the time periods with the highest loadings (2010 and 2013) were observed during years with the greatest flow from GGC (Figure 3-5). Copper was the only constituent that didn't follow this pattern with increasing loads from 2009 through 2012 and greatly reduced loading in 2013 when flow was the highest. Over the 2009-2014 time period the average daily loadings from the GGC were approximately 0.45 lbs/day copper; 495 lbs/day nitrogen; 16.5 lbs/day phosphorus; and 1,945 lbs/day suspended solids.

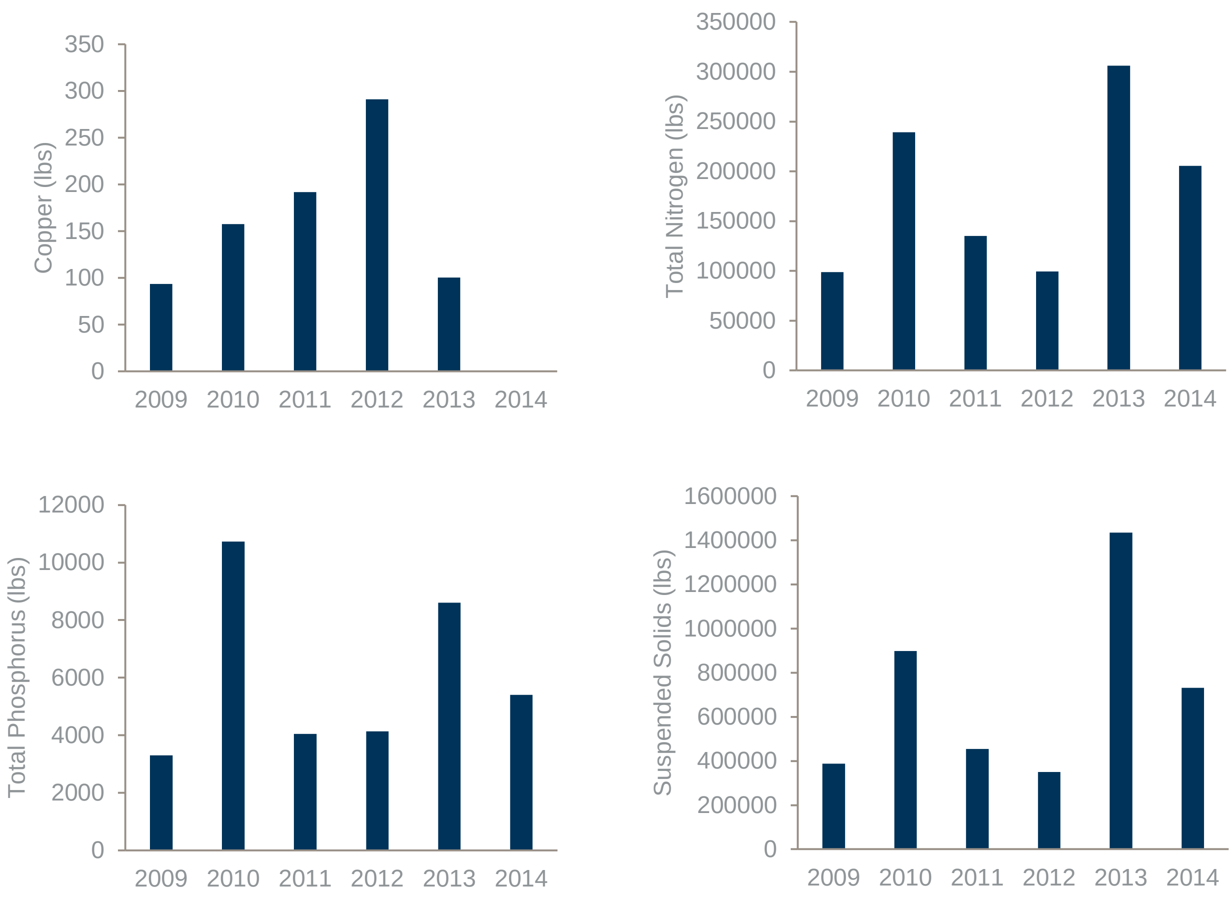


**Figure 3-5. Total annual loads from the Golden Gate Canal System into the Gordon River (Marine Segment) and Naples Bay, 2009–2014.**

To give perspective to the estimated loads from the GGC, a comparison is made between the GGC loadings to Naples Bay and the estimated loads to Tampa Bay. This comparison is drawn not to establish Tampa Bay as a reference for Naples Bay, but to illustrate the magnitude of the loads into Naples Bay relative to loads into another bay that has significant ongoing restoration activities and has experienced well documented seagrass resource recovery.

The differences between the two bays in terms of surface area, volume, drainage area, and land use make a direct comparison of loadings into each bay impractical. However, a relative comparison of the loadings into each based on bay volume can provide useful information regarding the magnitude and potential impact of the loading on Naples Bay. Relative loading comparisons between bays were available for total nitrogen, total phosphorus, and total suspended solids. Relative to bay volume, Naples Bay receives significantly more nitrogen, phosphorus, and solids loading than Tampa Bay (Table 3-1). Average phosphorus loading is approximately double, suspended solids loading is approximately six times greater, and nitrogen loading is approximately 17 times greater to Naples Bay than loading into Tampa Bay by volume. It is also significant to note that the Tampa Bay loading calculation includes all sources to the Bay (domestic wastewater and industrial point sources, non-point sources, groundwater, and atmospheric deposition), while the Naples Bay loading is only from the Golden Gate Canal. As described earlier in this section, the vast majority of the flow from the GGC occurs during the six months of the wet season (June – November) indicating these significant loads are delivered during a relatively short time period each year.

**Table 3-1. Relative loading comparison between Naples Bay and Tampa Bay. Naples Bay loads calculated from Golden Gate Canal only, Tampa Bay loads calculated from all sources.**

| Water Body | Bay Volume (million cubic yards) | Parameter | Average Annual Load (tons) | Relative Loading by Volume (tons/million cubic yards) |
|---|---|---|---|---|
| Naples Bay | 10.8* | TN | 90.3 | 8.3 |
| | | TP | 3.2 | 0.3 |
| | | TSS | 354.9 | 32.6 |
| Tampa Bay | 5321.8** | TN | 2,437.4 | 0.5 |
| | | TP | 747.2 | 0.14 |
| | | TSS | 29,457 | 5.6 |

Naples Bay loads - 2009-2014
Tampa Bay loads - 2007-2011 (Janicki 2013)
* Volume derived using bathymetry data from City of Naples GIS
** Weisberg and Zheng 2006

The Naples Bay watershed is approximately 120 sq. miles, 110 sq. miles of which is the result of area drained by the Golden Gate Canal system. Relative to volume, Naples Bay receives drainage from approximately 11 sq. miles for every million cubic yards of volume. In contrast, Tampa Bay, with a watershed of approximately 2,200 sq. miles (Yates *et al.* 2011) has a watershed to bay volume ratio of 0.4. This means that Naples Bay receives drainage from a much larger area per unit of bay volume than Tampa Bay and is an important consideration when assessing the potential impact of the loadings into Naples Bay. The GGC loads to Naples Bay should be considered an important factor during planning and implementation of bay wide management activities and restoration efforts.

This comparison demonstrates the magnitude of nutrient and solids loading to Naples Bay from the GGC. The relative loading from this single source to Naples Bay is many times larger than total loadings to Tampa Bay from all sources, which has exhibited significant documented resource recovery. While the GGC is known to be the largest source of freshwater to Naples Bay, the magnitude and timing of nutrient and solids loading to the Bay is also a critical consideration for management and restoration planning. For example, ongoing seagrass and oyster restoration efforts by the City will need to consider the GGC loadings. With the vast majority of loadings delivered to the Bay during the wet season, which is also the seagrass growing season, restoration activities will likely show limited success unless simultaneous efforts to address nutrient and solids loadings from the GGC are implemented. Subsequent sections of this report provide additional evidence supporting this assertion.

### 3.1.2 Urban Stormwater Runoff

Rapid urbanization in the City of Naples inevitably brought with it changes in land use, an increase in impervious cover, and increased urban runoff of stormwater into receiving waters. Stormwater within the City limits is routed either directly into the receiving waters or to one of 28 stormwater lakes, and/or through one of the City's three pump stations prior to entering receiving waters. In December 2010, the City began water quality monitoring of discharge from the stormwater lakes and pump stations. Characterization of water quality and quantity that has a direct runoff to receiving waters from the City's urban areas was not possible at the time of this report, therefore, the characterization of stormwater lake and pump station quality is used here to represent stormwater runoff to Moorings Bay, Naples Bay and the Gulf of Mexico.

Currently, 15 of the 28 stormwater lakes within the City and all three pump stations are included in the water quality monitoring program. Of the stormwater lakes in the monitoring program, four discharge to Moorings Bay, six discharge to the Gordon River (Marine Segment) above the SR 41 bridge, one discharges to northern Naples Bay, two discharge to the Port Royal canal area, and two discharge to the Gulf of Mexico (Figure 3-6). All three of the pump stations discharge into Naples Bay or the Gordon River (Marine Segment). The routes of discharge from the stormwater lakes are not direct: flow travels either through swales, ditches, and/or pipes to one of the three pump stations or roadside swales and culverts prior to entering the receiving waters. The stormwater lakes have generally been monitored twice per year, once in the wet season and once in the dry season, and the pump stations are generally monitored quarterly. This section is devoted to describing the water quality and quantity of the stormwater lake and pump station contributions to the receiving waters. At the time of this report, estimates of flow volume from the stormwater lakes were not available so the discussion of these lakes will focus on water quality.

Characterization of water quality in stormwater contributions was focused on the major parameters of concern for the City and potential pollutants in Naples Bay: copper, nutrients, suspended solids, and bacteria (fecal coliform and enterococci) (Figure 3-7). For the purposes of representing the water quality that enters receiving waterbodies, only data collected at the discharge point of each stormwater lake (characterized with a "B" after each lake number) were included here. Between five and ten individual data points have been collected in each stormwater lake since December 2010 (Appendix A, Table A-1). The small sample size and inconsistent sampling frequency from each individual lake precluded the use of formal time series analyses within each lake, so only descriptive summary statistics are shown here (Figure 3-7). Visual examination of the stormwater lake water quality data do not show any changes over time for any parameter, with the exception of total nitrogen in lake 20B, Forest Lake, where total nitrogen appears to be increasing over the three most recent sampling events (November 2013, December 2014, and February 2015) (Figure 3-8).

When grouped by receiving waterbody (Moorings Bay, Naples Bay, or Gulf of Mexico), cumulative distribution plots show some differences in nutrients and copper concentrations among the waterbody groups (Figure 3-9). However, these differences may simply be the result of the large variation in number of samples collected within each lake. Recommendations for enhancement to the survey design which would increase the ability to detect spatial and temporal trends in the stormwater lakes are provided in the accompanying *Naples Bay Monitoring Design* report.

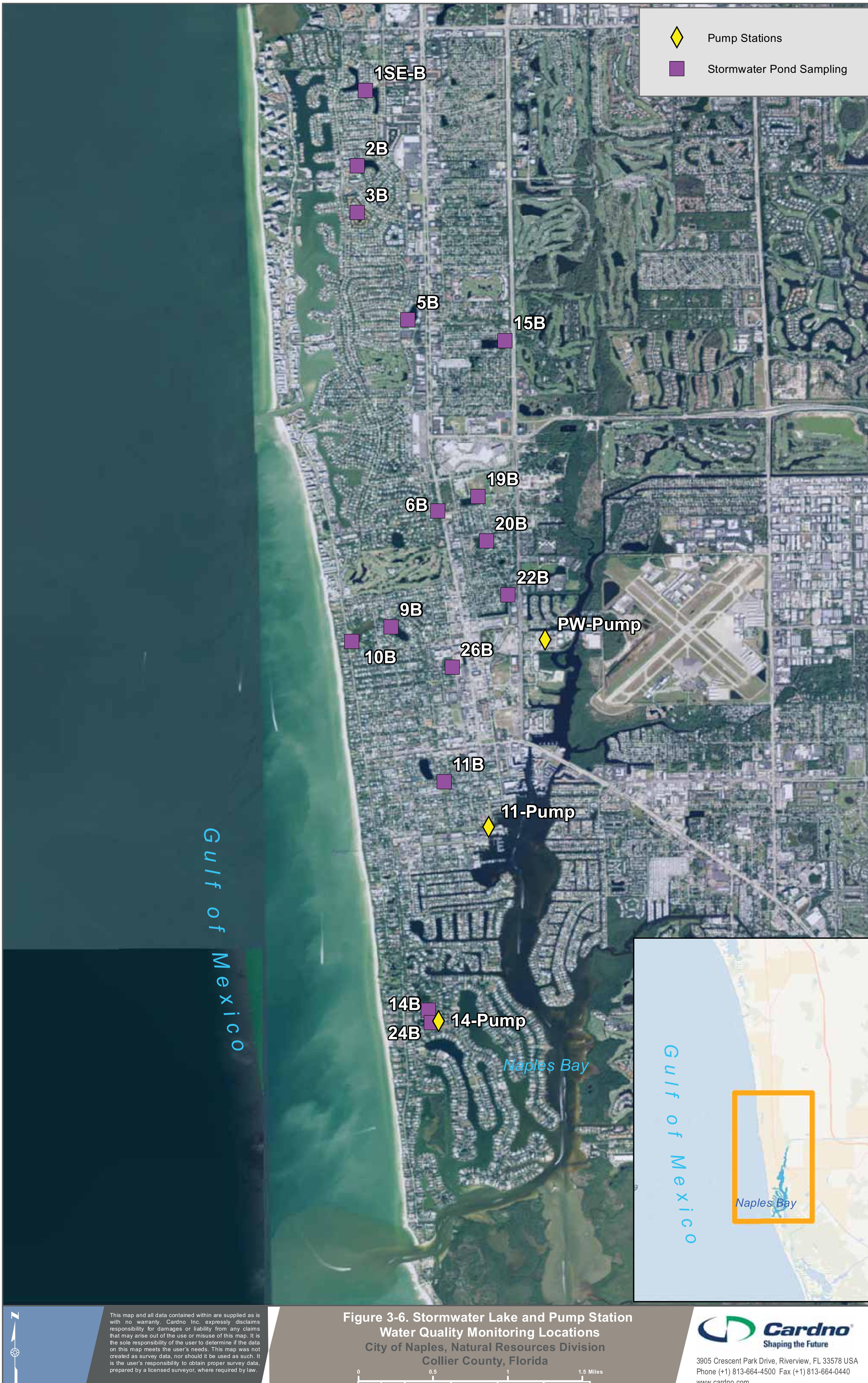


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**Figure 3-6. Stormwater Lake and Pump Station Water Quality Monitoring Locations**

City of Naples, Natural Resources Division
Collier County, Florida

Cardno
Shaping the Future
3905 Crescent Park Drive, Riverview, FL 33578 USA
Phone (+1) 813-664-4500 Fax (+1) 813-664-0440
www.cardno.com

Date Created: 5/8/2015 Date Revised: 5/8/2015 File Path: Q:\UnitedStates\Florida\Tampa\City_Of_Naples\Water Quality Analysis Project\working\arcmap\Fig3_6_CON_Stmwtr_Pond_Locations_20150505.mxd
GIS Analyst: adam.bausch

Figure 3-7. Water quality summary for stormwater lakes within the City of Naples, December 2010–February 2015.

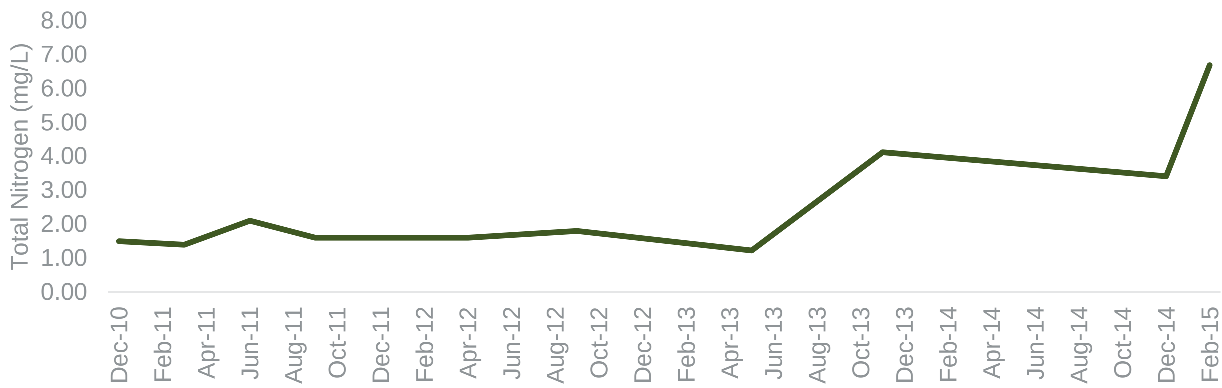


**Figure 3-8.** **Forest Lake (Lake 20) total nitrogen concentrations, December 2010–February 2015.**

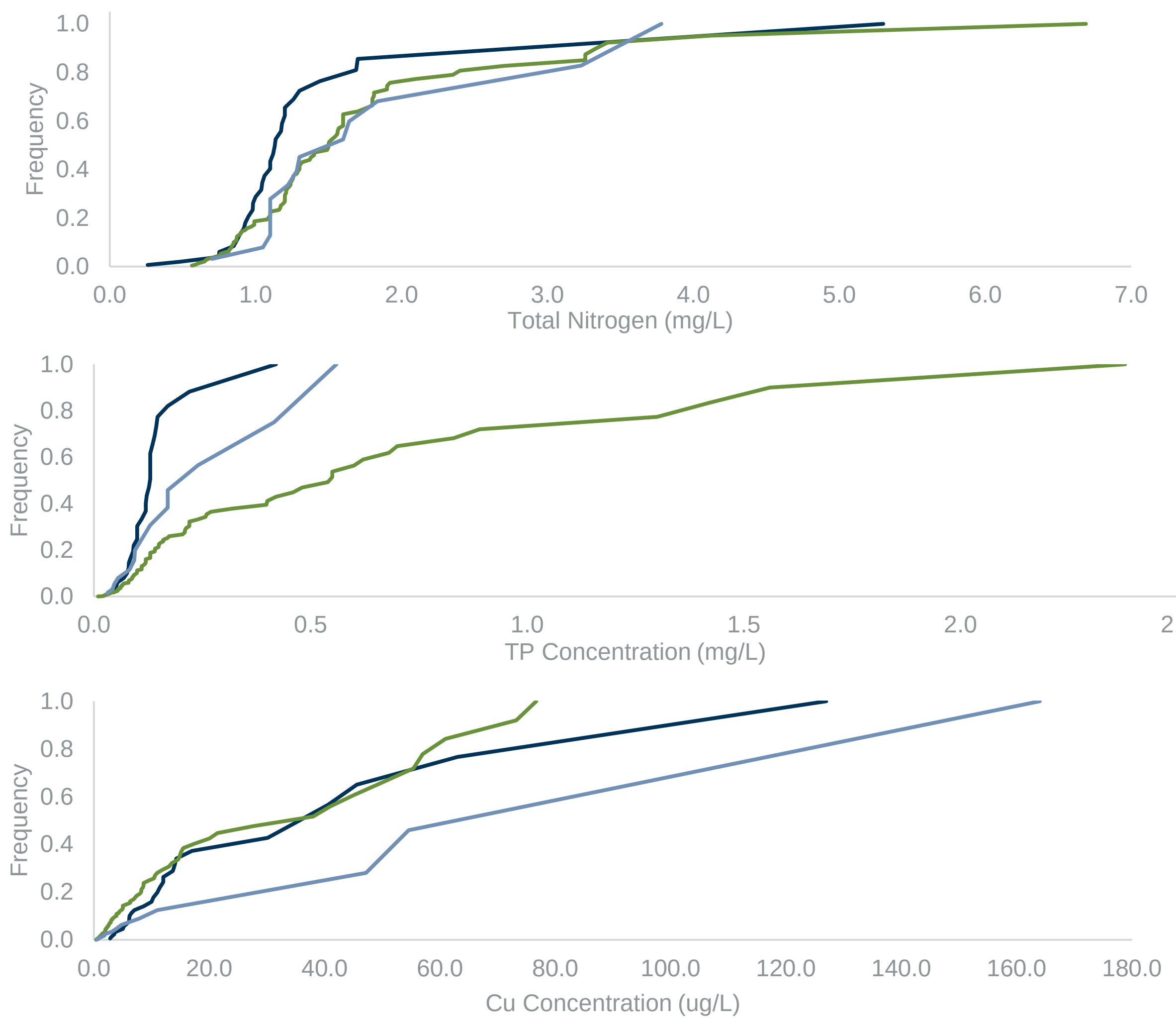


**Figure 3-9.** **Cumulative distribution plots of nitrogen (top), phosphorus (middle), and copper (bottom) in stormwater lakes discharging to Moorings Bay (navy), Naples Bay (green), and Gulf of Mexico (blue).**

The City's stormwater system is designed to conduct some stormwater directly to receiving waters and some to one of three pump stations through swales, ditches, and pipes. The Public Works Pump Station (PW-Pump) directs stormwater to the Gordon River (Marine Segment), the Cove Pump Station (11-Pump) discharges into northern Naples Bay, and the Port Royal Pump Station (14-Pump) discharges into the canals of the Port Royal area in the southern portion of Naples Bay (see Figure 3-6). Water quality in water discharged from the pump stations into Naples Bay (Figure 3-10, Table A-2) is very similar to the stormwater lakes, with the exception of bacteria (fecal coliform and *enterococci*), which was significantly higher in the pump station discharge than in the stormwater lake discharge (one-way ANOVA, $p < 0.01$).

The predominant land use across the areas contributing stormwater through the pump stations and into Naples Bay is residential. A comparison of the water quality through the Naples pump stations to typical runoff concentrations indicates the stormwater quality in Naples is within the range of that observed from other residential land uses in Florida (Table 3-2). In fact, the average concentrations of copper and suspended solids observed in water traveling through the pump stations are somewhat lower than those observed by FDEP in their review of residential land uses (FDEP 2007).

**Table 3-2.** **Mean Naples pump station water quality (December 2010–February 2015) and typical Florida residential runoff concentrations.**

| Parameter | Residential Land Uses* | Naples Pump Stations |
|---|---|---|
| | Range | Mean |
| Copper (µg/L) | 8–16 | 6.7 |
| Total Nitrogen (mg/L) | 1.61–2.32 | 1.92 |
| Total Phosphorus (mg/L) | 0.19–0.52 | 0.23 |
| Suspended Solids (mg/L) | 23–78 | 8 |

* Source: FDEP 2007

In addition to the concentration of certain pollutants in stormwater, an estimate of loading to receiving waters is a valuable management tool. Calculations of loads to Naples Bay from the pump stations is possible for three distinct time periods: water year 2012 (October–September 2012); calendar year 2013, and calendar year 2014. During previous upgrades to the City's supervisory control and data acquisition (SCADA) system, a significant amount of volume data for the pump stations were lost limiting the available time frame for loadings calculations. Data needed for the loading calculations associated with the PW-Pump station were only available for water year 2012. Loads were calculated using the total volume for the available time period and the mean concentration recorded during the time period.

The loadings calculated from the available time periods from the pump stations were highly variable between pump stations and over time for most parameters, with the 14-Pump discharging higher loadings of suspended solids than the other pump stations during all timeframes (Table 3-3). The average daily loadings to Naples Bay from the three pump stations over the available time period were approximately 0.032 lbs/day copper; 12.9 lbs/day total nitrogen; 1.9 lbs/day total phosphorus; and 37 lbs/day total suspended solids.

**Figure 3-10. City of Naples pump station water quality summary, December 2010–February 2015.**

**Table 3-3. Total annual loads delivered to Naples Bay through City of Naples pump stations.**

| Pump Station | Parameter | Total Annual Loads (lbs) | | |
|---|---|---|---|---|
| | | WY2012 | 2013 | 2014 |
| PW-Pump | Copper | 33.1 | -- | -- |
| | Total Nitrogen | 2566.9 | -- | -- |
| | Total Phosphorus | 162.6 | -- | -- |
| | Suspended Solids | 10425.3 | -- | -- |
| 11-Pump | Copper | 8.3 | 11.3 | 5.4 |
| | Total Nitrogen | 5730.3 | 8755.5 | 1978.6 |
| | Total Phosphorus | 930.5 | 829.5 | 346.5 |
| | Suspended Solids | 14371.9 | 11514.7 | 5376.1 |
| 14-Pump | Copper | 5.9 | 8.4 | 9.9 |
| | Total Nitrogen | 356.1 | 9393.5 | 4290.4 |
| | Total Phosphorus | 130.7 | 982.0 | 1427.8 |
| | Suspended Solids | 15219.8 | 24549.4 | 13526.9 |

The lack of consistent data over time hinders the ability to provide a more robust characterization of stormwater discharge to Naples Bay. More frequent monitoring of stormwater lakes along with discharge volume estimates from the lakes and pump stations would allow for additional statistical analyses of trends in water quality and quantity of stormwater delivered to Naples Bay (see *Naples Bay Monitoring Design* report).

## 3.2 Naples Bay Water Quality

This section covers the statistical methods and analysis of current water quality and trends within Naples Bay using the available data from the City's monitoring program as well as other publicly available data sources (see Section 2). The purpose is to provide a robust characterization of the current status of water quality in the Bay that can and will be used as a management tool for regulatory decisions concerning water quality compliance as well as providing a complete understanding of the factors that can and do impact biological communities. The analysis provided here focuses the constituents that both affect water quality in Naples Bay and have regulatory significance: salinity, nutrients, chlorophyll *a*, copper, turbidity, dissolved oxygen, and bacteria (fecal coliform and enterococci).

### 3.2.1 Statistical Methods Summary

In addition to a graphical and tabular interpretation of the current conditions of water quality in Naples Bay, several types of statistical analyses were performed for each constituent of concern at long-term data stations throughout the Bay: autoregressive error time-series models, predictive models between salinity and flow, Inverse Distance Weighting spatial interpolation, regression analysis, and parametric and nonparametric correlation analyses.

#### *3.2.1.1 Autoregressive Error Models for Time Series*

In order to identify trends in the water quality data from Naples Bay over time, we chose to use an Autoregressive Error Model (AEM). For many water quality variables, observations over time are temporally correlated. For example, the value of salinity at any given time (*t*) is correlated with the salinity value at an earlier time (*t-1*). Fitting a simple regression model through this data violates many of the statistical assumptions that are required for a proper trend detection. AEM is a simple model that reduces the chance of an incorrectly specified time series model that does not take temporal correlation into account.

Mathematically, the model can be written as:

$$Y_t = \beta_0 + \beta_1 * X_t + \gamma_t$$
$$\gamma_t = \varepsilon_t - \theta_{t-1}\gamma_{t-1} - .. - ..\theta_m\gamma_{t-m}$$
$$\varepsilon_t \sim N(0, \sigma^2)$$

Where *y* are dependent values, *t* represents a time step, *x* are covariates (in this case, simply the time that *y* is observed, e.g., month = 4), *m* is a lag function of *1 …. n*, σ is standard deviation, *θ* is a measure of temporal correlation at lag *m*, and ε is the model error which is normally distributed (*N*).

Effectively, the model predicts *y* at time *t* as a function of time, where the error term in the model accounts for any temporal correlation that exists in the time series. Therefore, the errors from the model are normal, thus meeting the statistical assumptions for trend detection. Using this form, a test of $H_0$: $\beta_1 = 0$, is used to detect trend.

For time series analysis, the frequency of sampling must be consistent; because sampling prior to 2011 was conducted only bimonthly, the monthly sampling data available after 2011 for Naples Bay were subset to only include samples from every other month. Using only a bimonthly subset of the data allows data from a longer time period to be included in the models. In addition, the time series analysis was limited to years where flow data from GGC were available for use as a covariate, 2008–2014. For parameters with suitable datasets, time series AEM were applied to data for four locations in the Gordon River (Marine Segment) and Naples Bay. Stations GPASS6 (Gordon Pass), NBAYWS (mid estuary), and NBAYNL (northern Naples Bay) were selected because of their long-term continuous data set dating back to the beginning of the City's monitoring program and, collectively, they represent upper, middle, and lower Naples Bay (Figure 3-11). A single long-term data station within the area influenced by GGC was not available for the Gordon River (Marine Segment) so stations GORDEXT and GORDPT were combined based on their proximity to each other to represent a single long-term dataset and the marine section of the Gordon River above the SR 41 bridge. Two potential covariates, natural log-transformed daily flow from the GGC and monthly total rainfall, were considered for each model. The best fit models, using total model $r^2$ and corrected AIC (Akaike Information Criterion), were ones that included flow and rainfall for almost all parameters, with the exception of TN (flow was not included in the best fit model) and TP (flow was not included in a time series model that extended back to 2005, when flow data were not available). Water quality data, with the exception of dissolved oxygen and salinity, were also natural log-transformed.

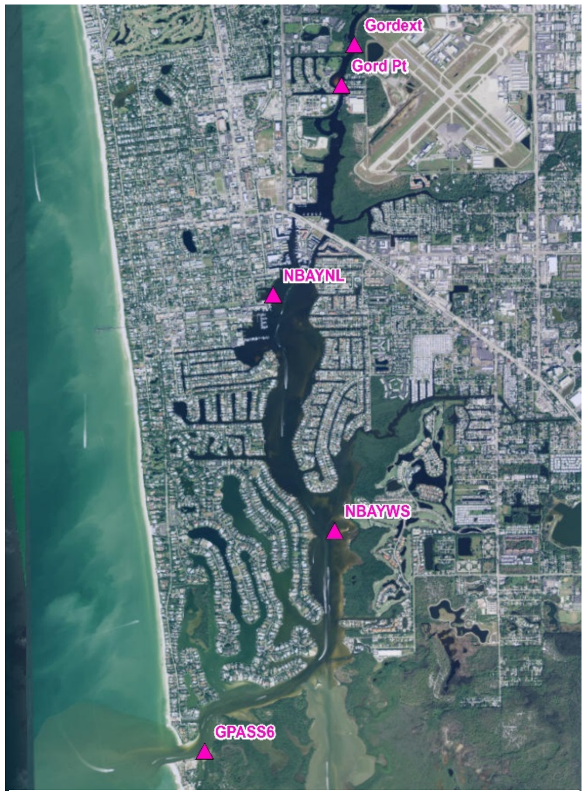


**Figure 3-11. Naples Bay long term water quality monitoring locations, 2005-2014.**

The model results shown in the following sections for each water quality parameter are the best fit models from 2008–2014. Models were also run with just rainfall as a covariate or with no covariates from 2008–2014 and 2005–2014, but results were only reported if trends appeared that were not in in the full model scenario. In many cases, extending the model back to 2005 and without covariates eliminated significant trends seen in the best fit models.

#### *3.2.1.2 Predictive Models*

Cardno developed a model designed to evaluate the effect of freshwater flow from GGC on salinity concentrations at downstream locations (see Section 3.2.2). Sensitivity testing of several time series model forms was implemented, including the use of autoregressive integrated moving average (ARIMA) models, autoregressive error models, and general linear models with trigonometric functions. We also evaluated the use of daily, weekly, and monthly data. The final model was fit to three years of monthly data (August 2011–July 2014, n = 36) for each of four downstream USGS continuous monitoring locations (Gordon River, City Dock, Mid-estuary, and Gordon Pass). The use of monthly data effectively smoothed the model development dataset and provided a reasonable model fit for all areas. The model equation was as follows:

$$Salinity_t = B_0 + B_1 * \ln(flow_t) + E_t$$

Effectively, the model predicts salinity at any month (*t*) as a function of the natural log of flow in that month. The degree of response that salinity has to flow decreases as the distance from GGC increases. The model was developed to estimate the change in salinity in Naples Bay as a result of potential GGC flow reduction scenarios. Three scenarios were chosen to represent a 30, 50, and 70 percent reduction in GGC flow. Additional details and graphics are provided in Section 3.2.2 and Appendix B.

#### *3.2.1.3 Inverse Distance Weighting*

In order to assess the spatial distribution of water quality concentrations within Naples Bay, a method of interpolation called Inverse Distance Weighted was used in ArcGIS to develop a relative raster surface of concentrations where samples points did not exist. Inverse Distance Weighting (IDW) is an interpolation method that assumes that points close together are more similar than those points father apart. To make a prediction at any one location, IDW assigns weights to neighboring observed values inversely related to the distance between the prediction and observation. The weights are assigned to each observation based on an inverse power function: $w(d) = 1/d^p$, where $w$ is the assigned weight, $d$ is the distance between points, and $p$ is the exponent of the power function. With lower values of $p$, more weight is given to neighbors that are farther away, resulting in a smoother predicted surface. With higher values of $p$, almost all weight is given to very close neighbors, which increases local attenuation.

To implement IDW in Naples Bay, we had to assume that the low number of samples points was sufficiently distributed across the study area to provide a valid interpretation of values at unknown locations. To do this and accommodate the non-linear nature of the study area, barriers to interpolation were set where a line-of-sight rule from one sample point to another was maintained and a fixed distance rule of 9000 feet was required for inclusion into the interpolation. This means that in order for a neighboring sample point to be included in the interpolation surrounding another sample location it had to be within a line-of-sight of that point and within 9000 feet to be considered a neighbor. With the small number of samples, this required that a minimum of three to five nearest neighbors be used in the interpolation. The significance of surrounding points on the interpolated value, Power or $p$, was also set at a central value of 2. The output is a 20-ft cell size raster surface of interpolated concentrations, which can be color graded to indicate a simplified visualization of what that parameters distribution across the study area.

For most of the water quality parameters discussed below, the spatial distribution of concentrations using IDW are presented using data from two selected years. In general, when sufficient data were available, one figure shows the beginning of the AEM model period of record (usually 2008) and the other shows a year near the end of the period (usually 2012 or 2013).

#### *3.2.1.4 Other Analyses*

Other analyses were used to assess the water quality data, including linear regression and correlation analysis. Exponential linear regression of salinity and GGC flow was used to estimate the relationship between them before the predictive model described above was developed. The equation for the regression is:

$$Salinity = B_0 * e^{(B1 * flow)}$$

Where $B_0$ is the intercept and $B_1$ is the slope.

Parametric (Pearson's product moment correlation coefficient) and non-parametric (Kendall Tau for correlations over time, Spearman's for correlations between variables) analyses were also used throughout the report to evaluate relationships between water quality variables or between water quality and time.

### 3.2.2 Salinity

Hydrologic alterations within the GGC system and their effect on freshwater influx and the salinity regime of Naples Bay is a primary concern for water quality and biological communities in Naples Bay (SFWMD 2007, Schmid *et al.* 2005, FDEP 2010, Simpson *et al.* 1979, City of Naples 2010, Laakkonen 2014, and Baum 1973). This study provides an in-depth look at the current salinity regime of Naples Bay and the effect of the freshwater inflow from the GGC using advanced statistical analysis and more recent data that were not available during the previous studies. A thorough understanding of the current salinity regime and the effect of the GGC freshwater inflow provides the basis for determining what potential effects the freshwater may be having on the biological communities in Naples Bay. In turn, this information is essential for developing appropriate and cost effective management programs and actions to protect, manage, and restore Naples Bay.

#### *3.2.2.1 Salinity-Flow Relationship*

The magnitude and timing of freshwater inflow has a significant effect on salinity in the marine segment of the Gordon River (upstream of the SR 41 bridge) and Naples Bay. Four USGS continuous recorders in the Gordon River (Marine Segment) and Naples Bay (Figure 3-12) operated for approximately three years, mid-2011 or early 2012 through October 2014, collecting a surface and bottom salinity measurement at 15 minute intervals throughout their deployment. The continuous recorder data provide a unique opportunity to characterize how the freshwater inflow from the GGC affects salinity in Naples Bay at four different locations at the same time. This allows for a spatial and temporal analysis of freshwater inflows and salinity that would not have been possible otherwise.

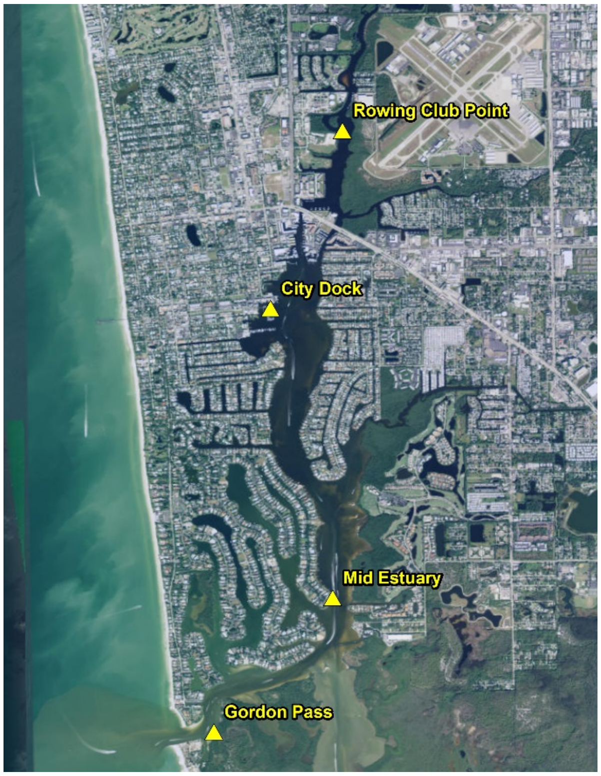


**Figure 3-12. USGS continuous recorder locations, 2011-2014.**

The effect of freshwater flow on salinity is apparent throughout the Bay, with average daily salinity at all four continuous recorders showing a response to GGC flow (Figure 3-13). As expected, the greatest impact is observed at Gordon River at Rowing Club Point, the station closest to the GGC, and the station at Gordon Pass showed the least change in average daily salinity during times of high GGC flow. All four stations show a significant negative relationship between GGC flow and average daily salinity (National Exponential Regression: Rowing Club Point $R^2 = 0.86$, City Dock $R^2 = 0.92$, Mid Estuary $R^2 = 0.90$, Gordon Pass $R^2 = 0.79$; $p < 0.05$), indicating that the entire Bay is affected by the GGC flow (Figure 3-14). When GGC flow is greater than approximately 300 cfs, the average salinity in the Gordon River above the SR 41 bridge drops below the regulatory threshold of 2.7 ppt for marine water (62-302.200(30), F.A.C.), turning this section of the Gordon River into a freshwater system. The vast majority of flow from the GGC occurs during the wet season and, as a result, the average salinity during summer months is much lower than during the winter (Table 3-4). The seasonal differences in flow result in a more dramatic salt gradient during the wet season that pushes into northern Naples Bay from the Gordon River (Figure 3-15).

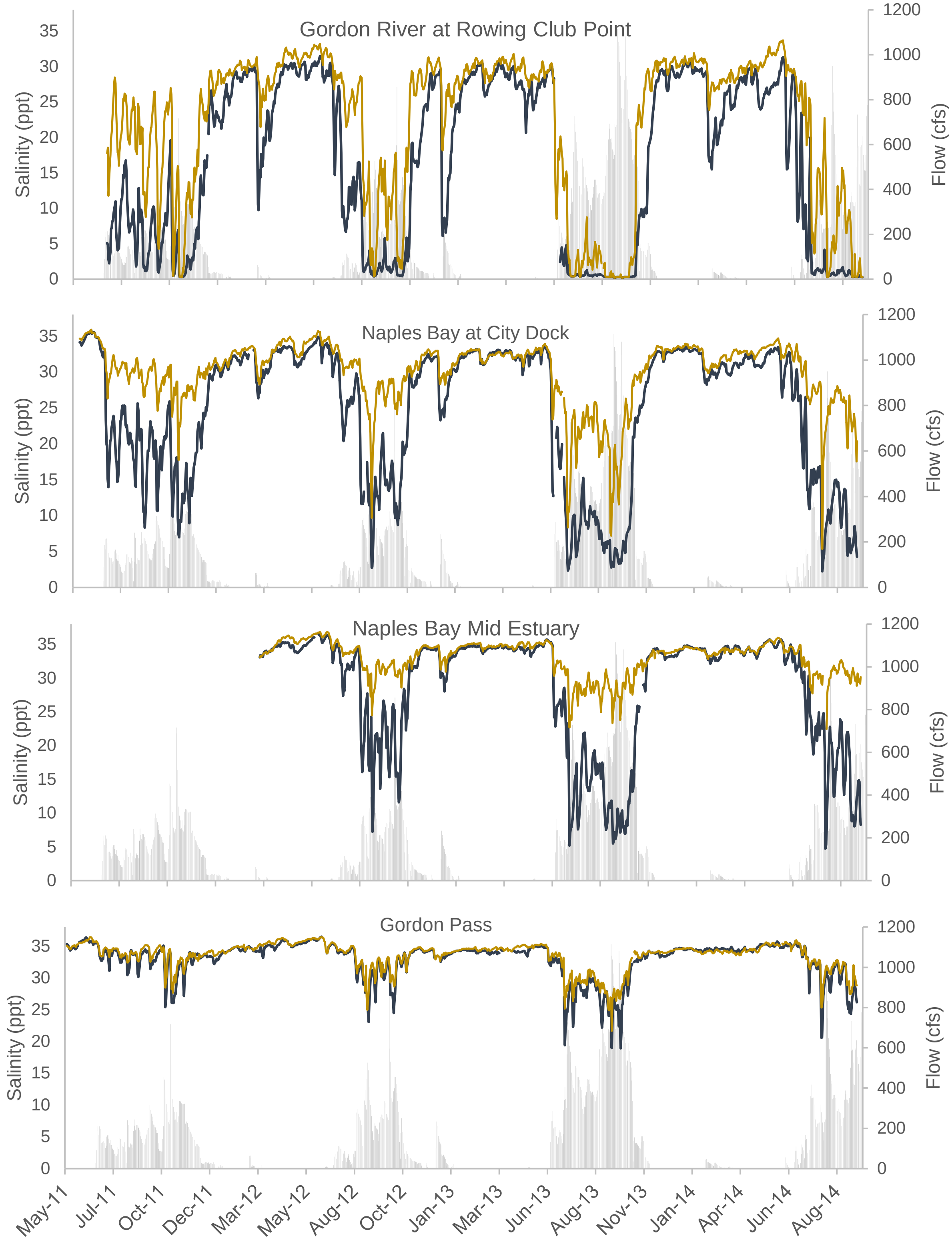


**Figure 3-13.** **Golden Gate Main Canal flow and average daily surface and bottom salinity in the Gordon River (Marine Segment) and Naples Bay, May 2011–September 2014. Data from USGS continuous recorders. Blue is surface salinity; gold is bottom salinity.**

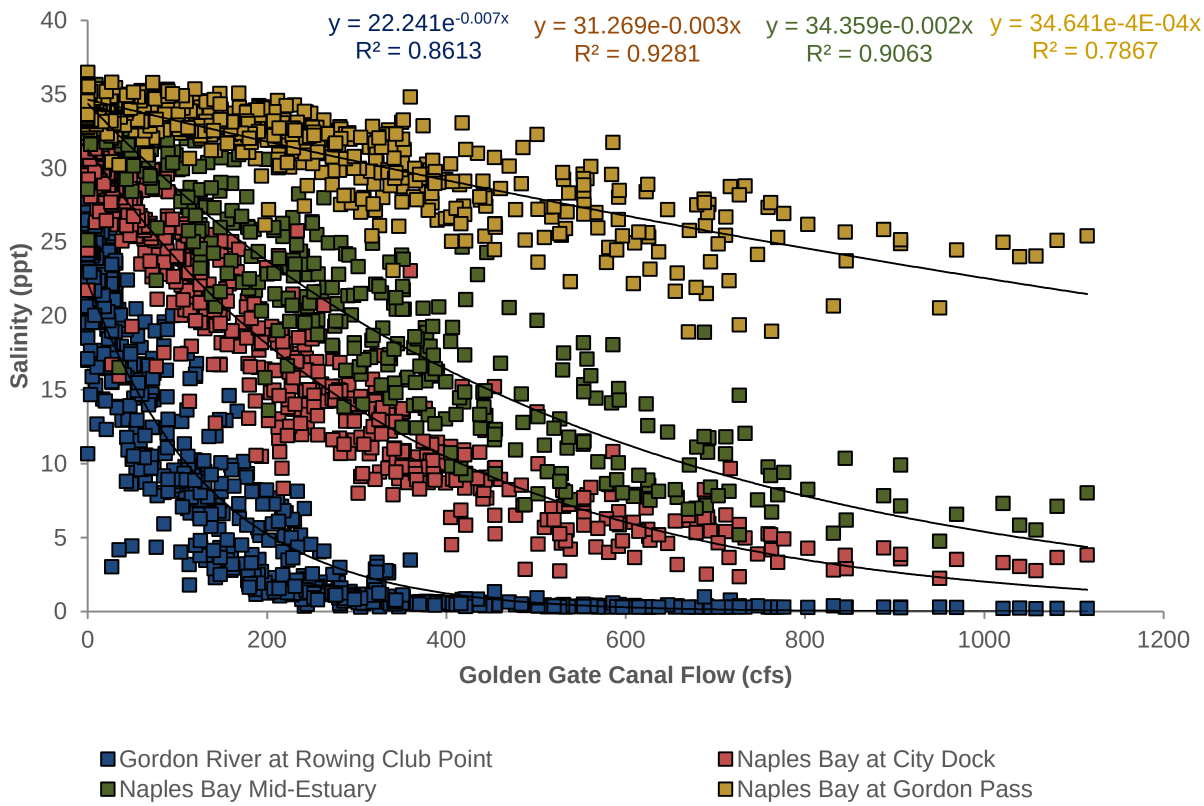


**Figure 3-14.** **Relationship between GGC flow and daily average salinity in the Gordon River (Marine Segment) and Naples Bay. All regressions were statistically significant ($p < 0.05$).**

**Table 3-4.** **Average salinity in Gordon River (Marine Segment) and Naples Bay by season from USGS continuous recorders (mid-2011 or early 2012 through October 2014).**

| Location | Mean Salinity (ppt) | | | |
|---|---|---|---|---|
| | Dry Season (December–May) | | Wet Season (June–November) | |
| | Surface | Bottom | Surface | Bottom |
| Gordon River at Rowing Club Point | 26.3 | 29.4 | 8.4 | 15.9 |
| Naples Bay at City Dock | 31.4 | 32.4 | 19.3 | 27.9 |
| Naples Bay Mid Estuary | 34.2 | 34.6 | 23.5 | 31.6 |
| Naples Bay at Gordon Pass | 34.4 | 34.6 | 31.8 | 32.7 |

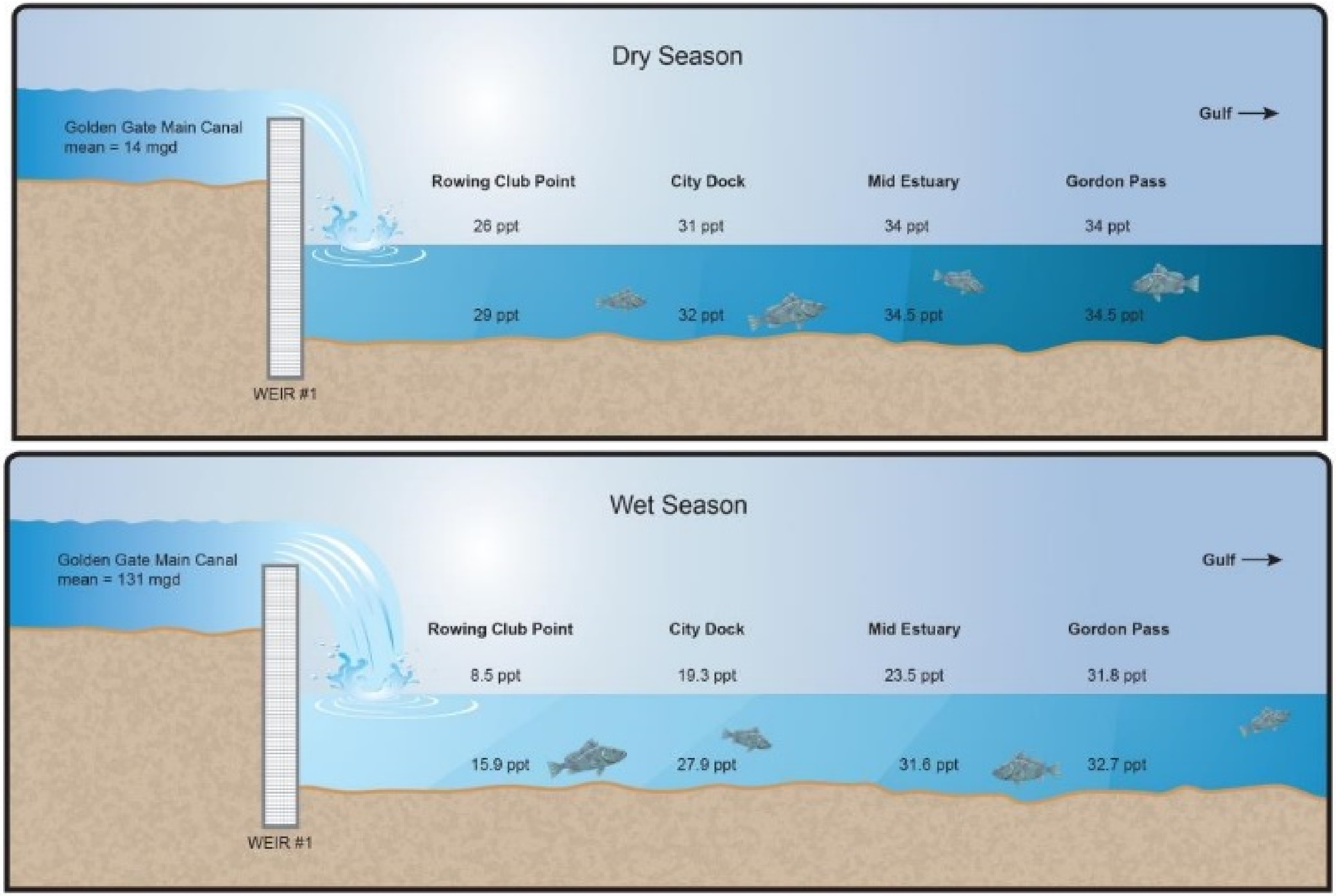


**Figure 3-15.** **Wet and dry season salinity gradient in the Gordon River (Marine Segment) and Naples Bay. Seasonal average salinity concentrations. Data from USGS continuous recorders, 2011–2014.**

Average daily salinity alone does not provide a complete characterization of salinity dynamics in the Naples Bay estuary. The change in salinity over the course of a day as a result of the interchange between freshwater inflow and the tides is also an important measure of the salinity regime in Naples Bay. The daily swing in salinity can be a more appropriate measure of how the estuary is responding to freshwater inputs and the effect of tidal exchange. The onset of the wet season and GGC flow causes larger fluctuations in daily salinity than typically occur during the dry season at all stations throughout the Bay (Figure 3-16). Although the typical daily range across stations in the wet season, when GGC flow is the highest, is between approximately 5 ppt and 15 ppt, some time periods experience daily salinity ranges of well over 20 ppt (Table 3-5) Such large salinity swings within relative short time periods could have an impact on the health of the biological communities in Naples Bay. In general, estuarine organisms are tolerant to a wide range of salinities; however, multiple large changes over the course of a few days could cause stress in some species.

**Table 3-5. Average and Maximum Range of Daily Salinity in Gordon River (Marine Segment) and Naples Bay, Wet and Dry Season (mid-2011 or early 2012 through October 2014).**

| Location | Mean Daily Salinity Range (ppt) | | | | Maximum Daily Salinity Range (ppt) | | | |
|---|---|---|---|---|---|---|---|---|
| | Dry Season (December - May) | | Wet Season (June - November) | | Dry Season (December - May) | | Wet Season (June - November) | |
| | Surface | Bottom | Surface | Bottom | Surface | Bottom | Surface | Bottom |
| Gordon River at Rowing Club Point | 6.2 | 5.9 | 6.5 | 15.1 | 22.1 | 24.1 | 23.1 | 28.5 |
| Naples Bay at City Dock | 1.5 | 1.7 | 4.6 | 8.6 | 8.8 | 10.4 | 12.8 | 25.4 |
| Naples Bay Mid Estuary | 2 | 2.1 | 10.2 | 10.9 | 10.9 | 13.6 | 26.8 | 28.7 |
| Naples Bay at Gordon Pass | 0.9 | 0.98 | 7.7 | 6.9 | 8.1 | 11.9 | 25.3 | 22.8 |

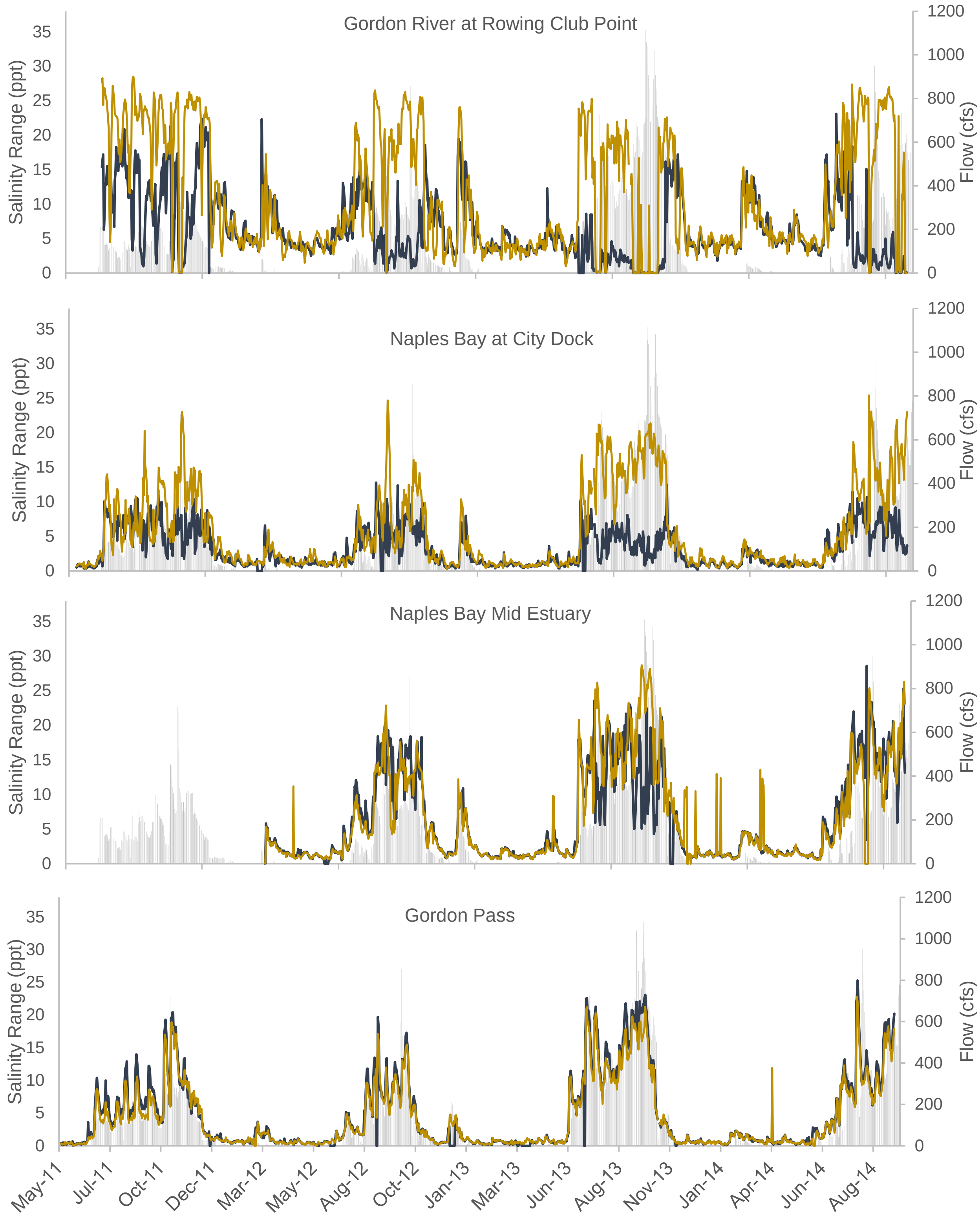


**Figure 3-16.** **Gordon River (Marine Segment) and Naples Bay daily salinity range and Golden Gate Canal Flow, May 2011–September 2014. Data from USGS continuous recorders. Blue is surface range; gold is bottom range.**

### 3.2.2.2 *Salinity Trend Analyses*

Daily and seasonal salinity fluctuations in Naples Bay can be large, with flow from the GGC playing a significant role in determining the salinity regime in the Bay. In order to determine if the salinity regime in the Bay is changing over time, an AEM time-series model was fit to the available salinity data with daily average flow and monthly average rainfall as covariates (see Section 3.2.1). The results indicate salinity in Naples Bay is not changing over time ($p > 0.05$), although the model confirmed that GGC flow and rainfall have a statistically significant negative relationship with salinity in Naples Bay for most stations ($p < 0.05$, Table 3-6).

**Table 3-6 Results of AEM time series models of bimonthly salinity in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.8 | 21.9 | 0.3 | -0.00008 | 0.9 | -3.5 | **0.003** | -1.4 | **0.0001** | None |
| NBAYNL | 0.7 | 52.5 | 0.05 | -0.001 | 0.3 | -2.2 | **0.007** | -0.9 | **0.0001** | None |
| NBAYWS | 0.5 | 26.7 | 0.2 | 0.0002 | 0.8 | -1.9 | **0.02** | -0.6 | **0.0005** | None |
| GPASS6 | 0.2 | 34.5 | 0.04 | -0.00002 | 0.9 | -0.7 | 0.2 | -0.2 | **0.04** | None |

### 3.2.2.3 *Salinity-Flow Management Decision Tools*

As a key component of any overall management and restoration plan for Naples Bay, focus should be given to controlling and reducing flow from the GGC. The Surface Water Improvement and Management Plan (SWIM) for the Naples Bay watershed includes strategies and actions to evaluate the magnitude and timing of freshwater inflow from the GGC to determine how to best manage the freshwater effect and minimize impact to Naples Bay (SFWMD 2007). This effort is supported by elements of the Big Cypress Basin Strategic Plan 2013-2018 (SFWMD 2013) which has suggested plans to improve the quantity, quality, timing, and distribution of water delivered to Naples Bay and Rookery Bay including the Northern Golden Gate Estates Flowway, North Belle Meade Rehydration, and the Henderson Creek Diversion projects. Although these potential projects are part of the Big Cypress Basin Strategic Plan and the Collier County Watershed Management Plan, investigations into the feasibility and potential consequences of each project are ongoing and no definitive water diversion projects are in progress at this time. This section briefly discusses the potential effects of flow reductions to Naples Bay in terms of salinity and offers insights into how flow reduction and alternative uses of GGC water may benefit Naples Bay.

As a management tool designed to provide the City with information necessary to understand the effect of reduced inflow from the GGC on the salinity regime of Naples Bay, we developed a flow and salinity predictive model. This model was developed using the salinity data from the four USGS continuous recorders that were operating in the Bay from 2011 through 2014. A description of the model development and set up can be found in Section 3.2.1 and Appendix B.

The current condition and three GGC flow reduction scenarios were modeled: 30 percent, 50 percent, and 70 percent. The 30 percent flow reduction scenario (scenario 1) was chosen to represent the suggested potential diversion of GGC flow into the Henderson Creek watershed; the 50 percent reduction scenario (scenario 2) represents the Henderson Creek diversion along with the Aquifer Storage and Recovery (ASR) systems included in future planning for the City of Naples as well as the City of Marco Island (City of Naples 2010). Finally, the 70 percent reduction scenario (scenario 3) is meant to represent a potential maximum feasible GGC flow reduction from a combination of potential water diversion projects to provide an estimate of how much change in salinity regime can be expected from the current condition. Note that the 70 percent maximum flow reduction is an arbitrary threshold used for modeling purposes only, and does not represent any analysis that suggests this is the maximum level of flow reduction possible.

Assuming that the observed data from the USGS continuous recorders from 2011 to 2014 are representative of any future salinity and flow concentrations that could occur (given environmental conditions) without any explicit management of the GGC flow, we simulated the expected (or mean) percent increase in salinity as a function of the expected (or mean) percent decrease in flow. The regression model (Appendix B) was used to predict the base case salinity using the observed GGC flow in the stated time frame. A reduction in flow was simulated simply by reducing the observed GGC flow by the selected percentage in each reduction scenario. An example of the model fit and predicted results is provided in Figure 3-17.

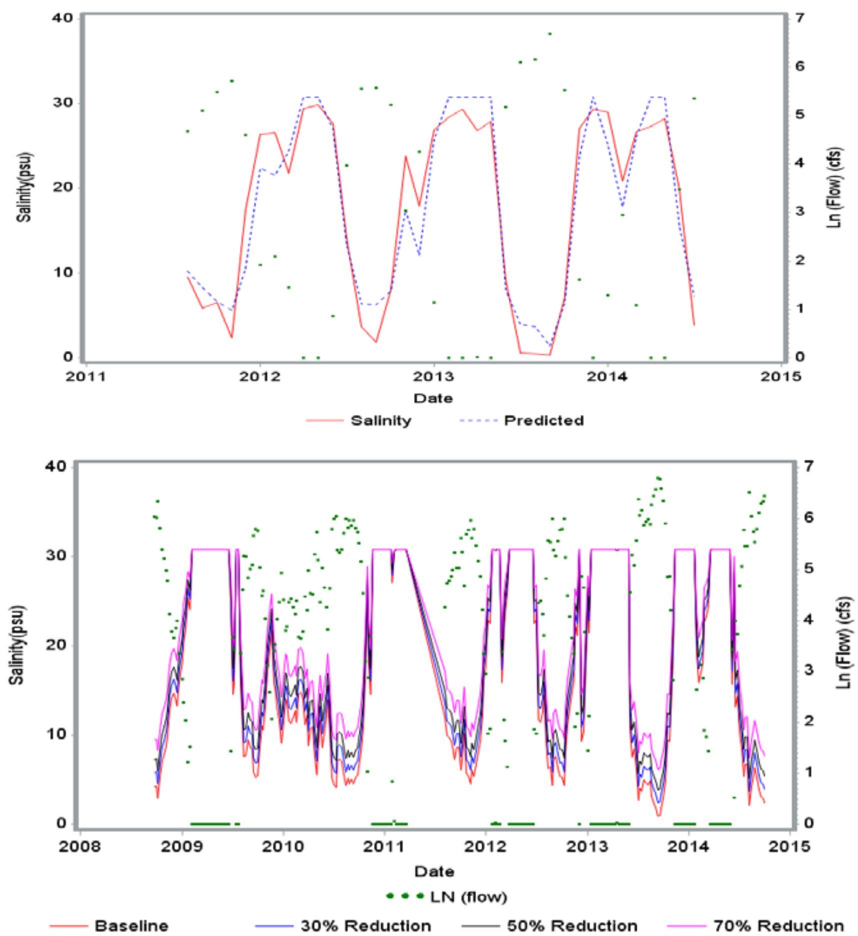


**Figure 3-17. Example model fit (top) and resulting salinity under different flow reduction scenarios (bottom) for Gordon River location based on predictive model output.**

The model was used to predict percent change in salinity and the average wet season salinity at each station in each of the three reduction scenarios (Table 3-7). The majority (over 90 percent) of the GGC flow occurs during the wet season, therefore the change in salinity regime would be expected to be concentrated in only the wet season. Under the 30 percent diversion plan (scenario 1), the model predicts the average salinity would increase by between 0.5 to 14 percent from south to north, respectively, in

Naples Bay. In this scenario, the predicted wet season average surface salinity at Gordon Pass is predicted to increase from 31.8 to 31.96 ppt. Under the same scenario, the model predicts the Gordon River location to exhibit a wet season salinity increase of 14.2 percent resulting in an increase from 8.4 ppt in the current condition to 9.6 ppt on average. Scenario 3 (70 percent reduction in GGC flow) results in the largest predicted percentage change in salinity in Naples Bay, with the Gordon River location predicted to show an increase in wet season average salinity from 8.4 ppt in the current condition to 12.4 ppt on average. Yet, a 70 percent reduction in GGC flow is only predicted to increase average wet season salinity at Gordon Pass from 31.8 ppt to 32.4 ppt on average.

**Table 3-7. Predicted percent change in average salinity and predicted average wet season salinity in Naples Bay under different Golden Gate Canal flow reduction scenarios.**

| Location | Predicted % Increase in Mean Salinity | | | Current and Predicted Average Wet Season Salinity (ppt) | | | |
|---|---|---|---|---|---|---|---|
| | Scenario 1 | Scenario 2 | Scenario 3 | Current Condition | Scenario 1 | Scenario 2 | Scenario 3 |
| Gordon River at Rowing Club Point | 14.18 | 27.49 | 47.52 | 8.4 | 9.59 | 10.71 | 12.39 |
| Naples Bay at City Dock | 3.85 | 7.46 | 12.84 | 19.3 | 20.04 | 20.74 | 21.78 |
| Naples Bay Mid Estuary | 2.45 | 4.75 | 8.18 | 23.5 | 24.08 | 24.62 | 25.42 |
| Naples Bay at Gordon Pass | 0.56 | 1.07 | 1.85 | 31.8 | 31.96 | 32.14 | 32.38 |

The model-predicted changes in the salinity regime in Naples Bay shown here are similar to the predicted changes in salinity developed by Weisberg and Zheng (2007) with a modeled 350 cfs reduction in GGC flow from 2005 conditions. Weisberg and Zheng (2007) used a Finite-Volume Coastal Ocean Model (FVCOM) that could be used to describe the circulation of the Rookery Bay estuarine complex (including Naples Bay) and study the relationship between freshwater inflows and salinity patterns. The results of the modeling effort estimate salinity would increase between 2.2 and 5.1 ppt with a 350 cfs reduction in GGC flow (Weisberg and Zheng 2007). The range of salinity increase is similar to that observed in our predictive modeling effort.

The predicted changes in mean salinity under the three flow reduction scenarios are relatively modest given the large daily and seasonal swings in salinity that Naples Bay currently exhibits. As described in sections 4 and 5 of this document, the fish and seagrass communities of Naples Bay may not show a significant response to these predicted changes in salinity. However, in terms of management of Naples Bay, this result does not suggest that flow diversion projects would not be beneficial. Reduction in freshwater flow from the GGC into Naples Bay can significantly reduce loadings of solids and nutrients to the Bay. Concentrations of solids and nutrients delivered to Naples Bay are relatively low; however, the extremely high volume of water flowing from the GGC results in a large load delivery (see Section 3.1.1). Significant load reductions from a combination of water diversion projects may have a significant positive impact on Naples Bay biology (*i.e.* seagrass) (see Sections 4.1 and 5.1). In addition to the potential water diversion projects already discussed, alternative uses for the GGC water such as water supply and salt water intrusion barriers should also be considered as viable options to significantly reduce the inflow to Naples Bay as part of a holistic water management strategy that benefits not only the ecology of Naples Bay, but the water demands of one of the fastest growing regions in the United States.

### 3.2.3 Copper

The FDEP listed Naples Bay (**W**ater **B**ody **Id**entification Number (WBID) 3278R4) as impaired for copper in 2009 along with Rock Creek (WBID 3278R3), Haldeman Creek (WBID 3278R1), and the Gordon River (Marine Segment) (WBID 3278R5) that contribute to Naples Bay. Therefore, copper is a major water quality issue for the Bay and tributaries. Copper is an essential trace element for many aquatic organisms, but can be toxic at levels slightly above those necessary for growth and reproduction (Hall *et al.* 1988). In estuarine environments, sources of copper include atmospheric deposition, industrial and municipal discharges, urban runoff, and antifouling marine paints (Hall *et al.* 1988). Copper sulfate is also very commonly used as an herbicide in lake management applications to control algae.

We evaluated the spatial and temporal status of copper in Naples Bay relative to the Class II water quality standard of 3.7 µg/L. Over the period of record, higher copper concentrations are typically found in upper Naples Bay, with the majority of locations that exceed the water quality standard found in this area (Figure 3-18). IDW was used to represent copper concentrations in Naples Bay geographically and identify potential “hot spots” of high concentration. Because of the limited number of monitoring locations available in the spatial dataset (especially in later years) and their irregular spacing, the results of the IDW analysis should be used only for illustration purposes; the interpolated values of copper between known sampling locations are sparse, which is typically associated with uncertainty in IDW interpolations. The highest copper concentrations are consistently found in Haldeman Creek, where annual average concentrations are four to eight times higher than the water quality standard at the SR 41 (Tamiami Trail Rd.) monitoring location.

Statistical analysis of copper concentrations over time in Naples Bay was hindered by changes in laboratory method detection limits (MDL) over the period of record. In 2013 and 2014, the laboratory MDL was increased to 3.0 µg/L, masking the ability to detect copper below this concentration. As such, AEM time-series models were not the most appropriate tool to look for trends over time. In order to look at the relationship between copper and time at all of the available stations, a correlation analysis (Pearson product moment correlation) was used to determine if the percentage of samples per year over a certain concentration was increasing or decreasing over time. The concentration chosen for the threshold was 3.7 µg/L, the state marine copper standard. While this analysis does not address trends in the actual copper concentrations over time, it does allow for a determination of whether the frequency of samples with concentrations above the water quality standard is increasing or decreasing over time. Thus, this analysis can show if a station is more frequently exceeding the water quality standard in more recent years, even though it can’t be determined whether the average concentration of copper is going up over time. This is an effective alternative to trend analysis that allows for characterization of copper in Naples Bay over time.

The copper analysis was broadened to include all of the stations where data are still being collected, not just the four stations used in the AEM time series analysis: GGCAT31/3495, BC3, BC2, HALDCRK, and BC5 (2001–2013); GORDEXT/GORDPT and NBAYWS (2005–2014); NBAYNL and GPASS6 (2006–2014); and ROCKCR, CURLEW, OYSBAY, and HALDCR (2011–2014) (see Figure 2-2). All of these stations were chosen (even those with short periods of record) because of the spatial importance of copper in Naples Bay. A potential source of copper to Naples Bay is stormwater inflow from upland applications of copper sulfate to control algae in stormwater lakes. Therefore the ability to identify and describe patterns of where copper may be entering Naples Bay is as important, if not more important, than identifying overall changes within the Bay over time. For the purposes of this analysis, copper in the GGC and other freshwater sources was evaluated against the marine water quality standard because it represents concentrations delivered to the marine portion of the Gordon River that is currently listed as impaired for copper.

Correlation analysis will not show significant changes at stations with data that are always below and/or above the water quality standard. Copper concentrations at GGCAT31/3945 and GPASS6 were always below the 3.7 µg/L criteria and, therefore, were not included in the analysis. This indicates that copper concentrations delivered to Naples Bay from the GGC and at Gordon Pass do not exceed the water quality standard. Two other stations, HALDCRK and BC5, were above the threshold almost 100 percent of the time. The data from HALDCRK and BC5 were analyzed using the annual average concentrations instead of percentage of samples above the water quality standard.

When considering the frequency of results above the threshold of 3.7 µg/L (Figure 3-19), patterns vary from station to station. GORDEXT/GORDPT has a statistically significant increase in the percent of copper measurements above 3.7 µg/L over time ($r = 0.66$, $p < 0.05$). NBAYNL shows a statistically significant decrease over time in the percent of copper measurements above 3.7 µg/L from 2006 to 2012 ($r = -0.80$, $p < 0.05$) and then a statistically significant increase over time from 2012 to 2014 ($r = 0.99$, $p < 0.05$). At NBAYWS, the percent of copper measurements above 3.7 µg/L was relatively consistent from 2005–2009, dropped to 0 percent in 2010, and then showed a strong but not significant increase from 2010 to 2014. BC2 showed a strong, but not statistically significant, increase in percent of copper measurements above 3.7 µg/L. Of the four stations with only four years of data (ROCKCR, CURLEW, OYSBAY, HALDCR), one showed a statistically significant increase (OYSBAY, $r = 0.98$, $p < 0.05$) in percent of copper above 3.7 µg/L over time. The other three showed high correlation coefficients (> 0.77) and visually clear increases in the percent of samples with copper concentrations above 3.7 µg/L over time from 2011 to 2014, but the increases were not statistically significant because of the small sample size ($0.1 < p < 0.25$). These four stations are located where tributaries enter Naples Bay (HALDCR and ROCKCR) or in dead end canals (OYSBAY and CURLEW) where stormwater enters the Bay.

The annual average copper concentrations at BC5 and HALDCRK (the stations where copper concentrations are almost always above 3.7 µg/L) do not show a significant correlation with time for either the arithmetic or geometric mean over the period of record (2001–2013). Although concentrations at HALDCRK appear to be increasing between 2007 and 2010, there is no consistent overall correlation with time (Figure 3-20). It is important to note however that even if no significant increase in concentrations is observed, all of the annual average concentration values in the dataset exceed the marine water quality standard at these two locations.

Although this analysis shows that while copper is spatially variable among the stations in Naples Bay and the tributaries, several stations appear to exhibit copper that is more frequently above the water quality standard in more recent years compared to earlier years in the dataset. This is why copper continues to be an important parameter of concern for Naples Bay.

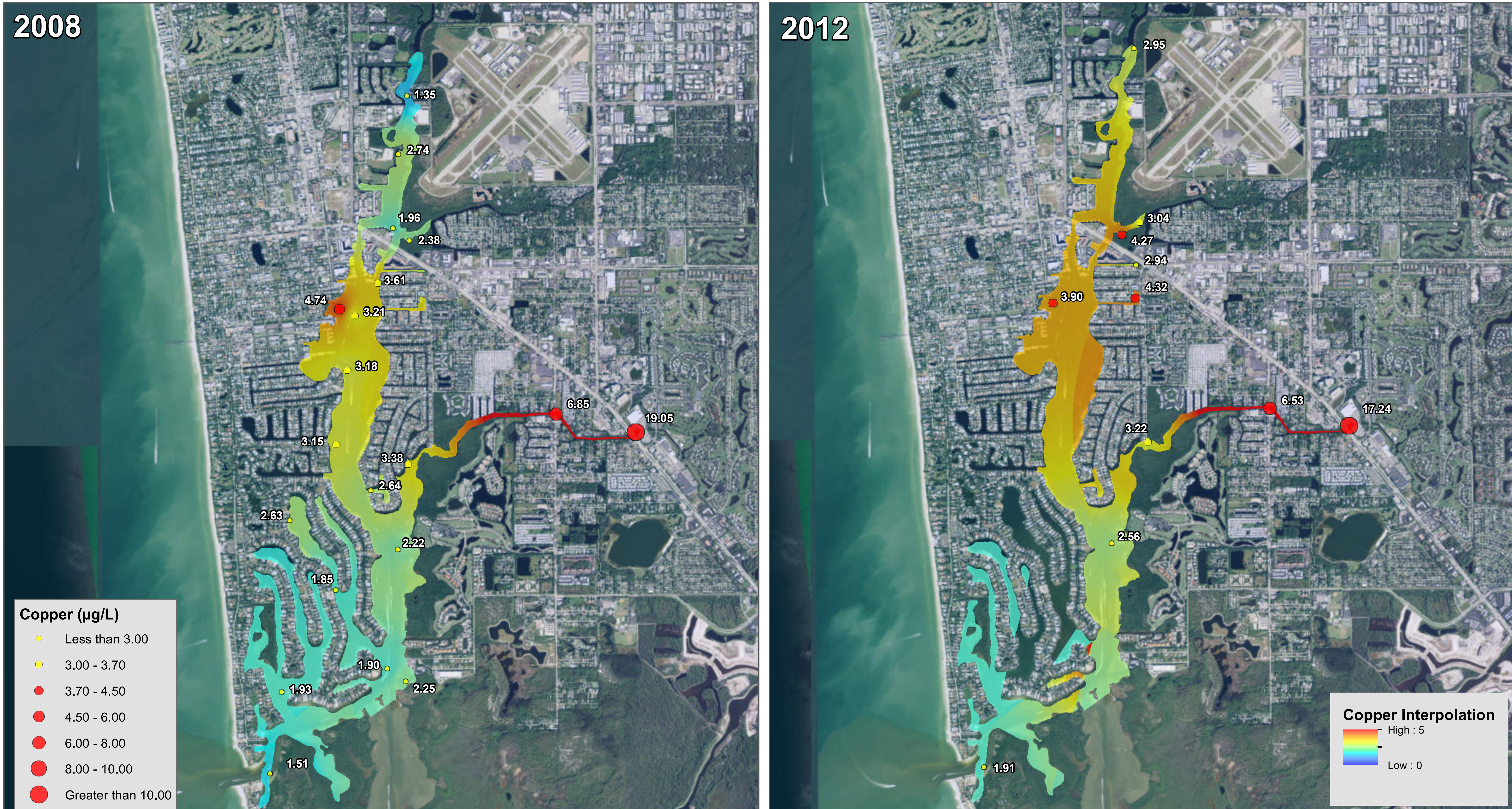


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**Figure 3-18. Annual Average Copper Concentrations in Naples Bay and Gordon River (Marine Segment) with IDW Interpolation, 2008 and 2012**

City of Naples, Natural Resources Division
Collier County, Florida

0 0.5 1 1.5 2 Miles
0 1 2 3 Kilometers

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GIS Analyst: Lauren.Federsel

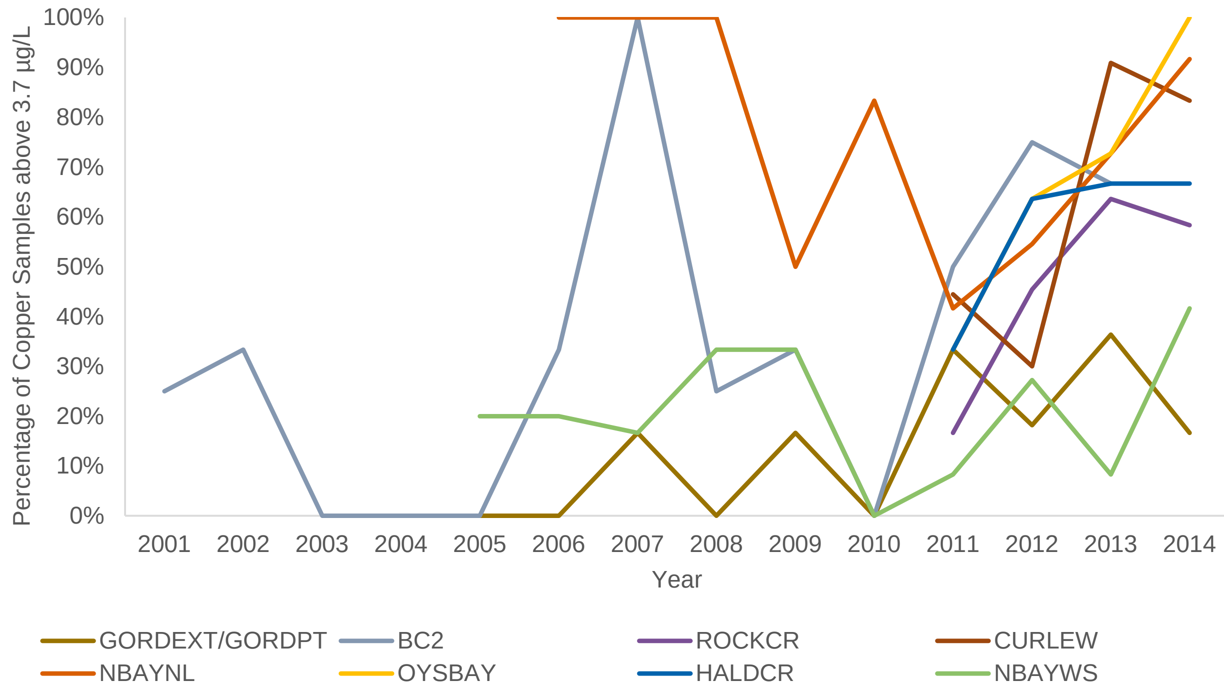


**Figure 3-19. Annual percent of copper concentrations greater than 3.7 µg/L in Naples Bay and Gordon River (Marine Segment).**

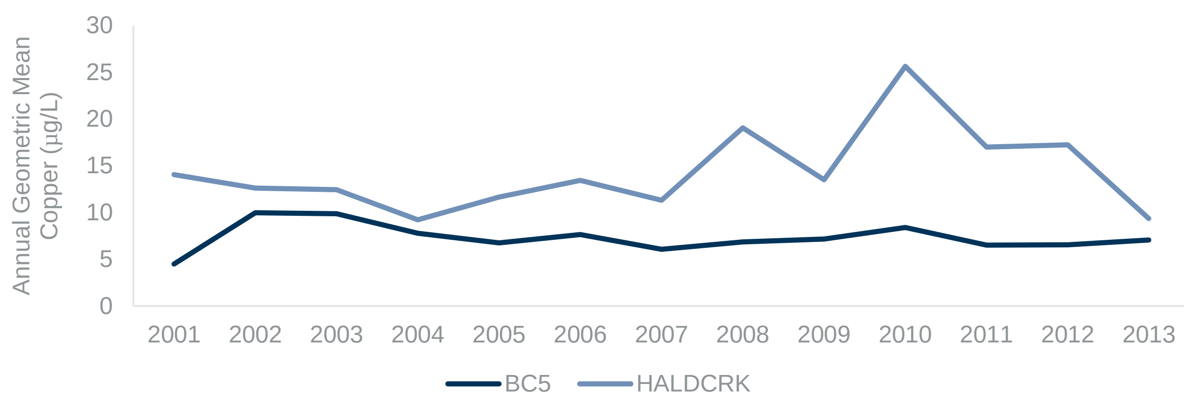


**Figure 3-20. Annual geometric mean copper concentrations over time in Haldeman Creek.**

### 3.2.4 Nutrients: TN and TP

Excess nutrients (nitrogen and phosphorus) are an issue of growing concern in waterbodies throughout the country. The EPA began providing guidance on development of numeric nutrient criteria (NNC) in 2000 and in Florida the process of developing NNC was hastened in 2009 with the EPA's necessity determination that NNC were required under the Clean Water Act (CWA). Following multiple lawsuits and parallel criteria development tracts by the FDEP and EPA, NNC for many waterbodies, including most estuaries were adopted by FDEP in 2012. NNC for the remaining waterbodies became effective in October 2014. NNC for Naples Bay were adopted by the FDEP's Environmental Regulatory Commission (ERC) in 2011 and approved by EPA in 2012. The Naples Bay NNC are expressed as annual geometric mean concentrations that are not to be exceeded more than once in a three year period. The allowable concentrations are as follows:

Total Nitrogen (TN) = 0.57 mg/L;
Total Phosphorus (TP) = 0.045 mg/L, and
Chlorophyll *a* = 4.3 µg/L.

The nutrient discussion and analysis provided here is conducted in light of the newly adopted NNC for Naples Bay to provide context for the observed nutrient conditions.

Prior to 2006 and the initiation of the City of Naples' water quality monitoring program, few monitoring locations existed in the Bay (Figure 3-21) which inhibited comprehensive characterization of the Bay as a whole. The City's program included many portions of the Bay that were not previously monitored, especially the southern portion of the Bay closer to Gordon Pass. Some elements of monitoring program changed in 2011, including the elimination of some locations and the movement of others to more accurately represent inputs to the Bay, but the current program still provides a more robust characterization of the whole Bay than previous monitoring activities.

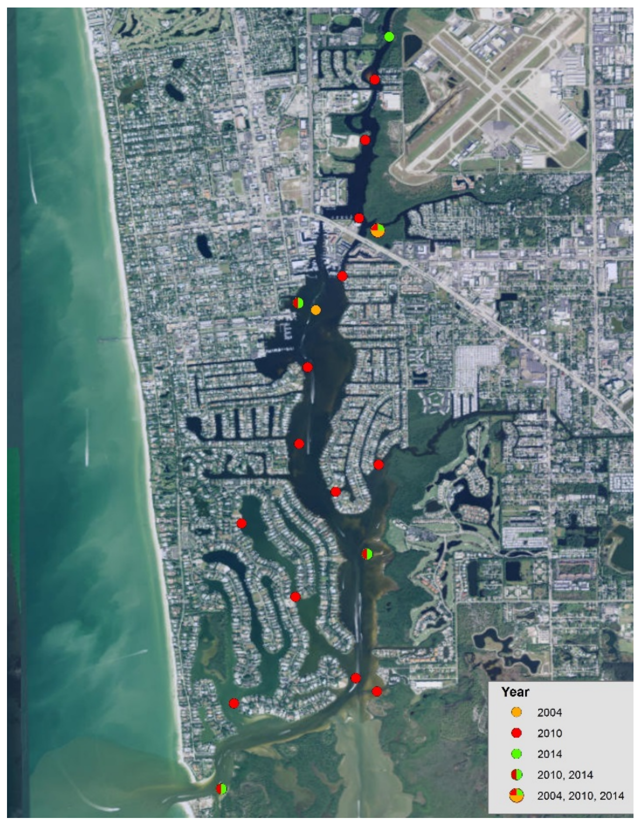


**Figure 3-21. Naples Bay monitoring locations during 2004, 2010, and 2014, illustrating changes over time.**

The water quality monitorng program in Naples Bay is particularly important in the context of nutrient regulations and compliance. Implementation of the NNC requires assessment of the waterbody on a WBID scale, incorporating all available data from all locations within the WBID in the calculation of the annual geometric mean concentration. Therefore, a more robust monitoring program leads to a more accurate representation of the nutrient condition of the WBID as a whole and is not as influenced by localized conditions at individual stations.

Over the period of record (2002–2014) the TN and TP annual geometric mean concentrations in WBID 3278R4 (Naples Bay) appear to have decreased while TN and TP appear to remain relatively stable in WBID 3278R5 (Gordon River Marine Segment) (Figures 3-22 and 3-23). WBID 3278R4 (Naples Bay) has achieved the newly adopted NNC every year since 2006, with TP achieving the criteria every year since 2003, indicating the Bay is in compliance with the NNC. TN in WBID 3278R5 (Gordon River Marine Segment) typically fluctuates above and below the criteria with more than one year in a three year period above the limit, indicating non-compliance with the NNC. Prior to 2011, TP in WBID 3278R5 (Gordon River Marine Segment) exceeded the annual geometric mean NNC limit of 0.045 mg/L at least once in three years, but shows compliance with the NNC since after 2011.


**Figure 3-22. Gordon River (Marine Segment) (WBID 3278R5) and Naples Bay (WBID 3278R4) total nitrogen annual geometric mean concentrations and total nitrogen numeric nutrient criterion, 2001–2014.**

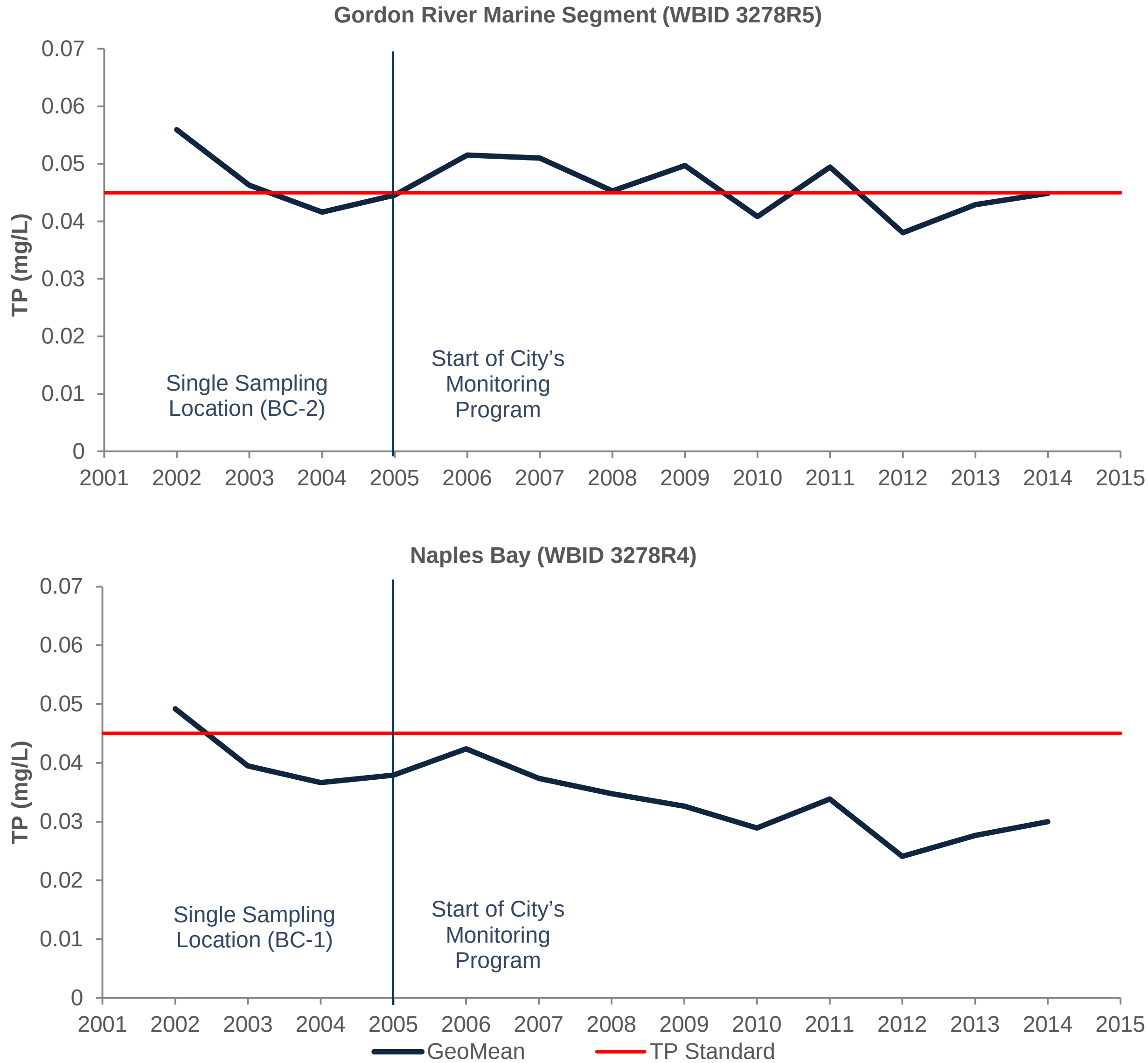


**Figure 3-23. Gordon River (Marine Segment) (WBID 3278R5) and Naples Bay (WBID 3278R4) total phosphorus annual geometric mean concentrations and total phosphorus numeric nutrient criterion, 2002–2014.**

Four stations in WBID 3278R4 (Naples Bay) and WBID 3278R5 (Gordon River Marine Segment) had enough long-term monitoring data to examine trends in nutrient concentrations over time at individual stations, accounting for the effects of flow from the Golden Gate Canal and regional rainfall. AEM time series models indicate a statistically significant decreasing trend in TN over time for the 2008–2014 period at all of the long-term stations (GORDEXT/GORDPT, NBAYNL, NBAYWS ($p < 0.05$); GPASS6 ($p < 0.1$)) (Table 3-8, Figure 3-24). Flow was not a significant covariate at any of the stations (and was not included in the best-fitting model), however rainfall showed a statistically significant positive relationship with TN at the Gordon River (GORDEXT/GORDPT) and mid estuary (NBAYWS) long-term stations. No statistically significant trends over time in TP were observed at any of the long-term stations when assessed over the 2008–2014 time period, and the models had very poor fit and few significant relationships with flow and rainfall (Table 3-9).

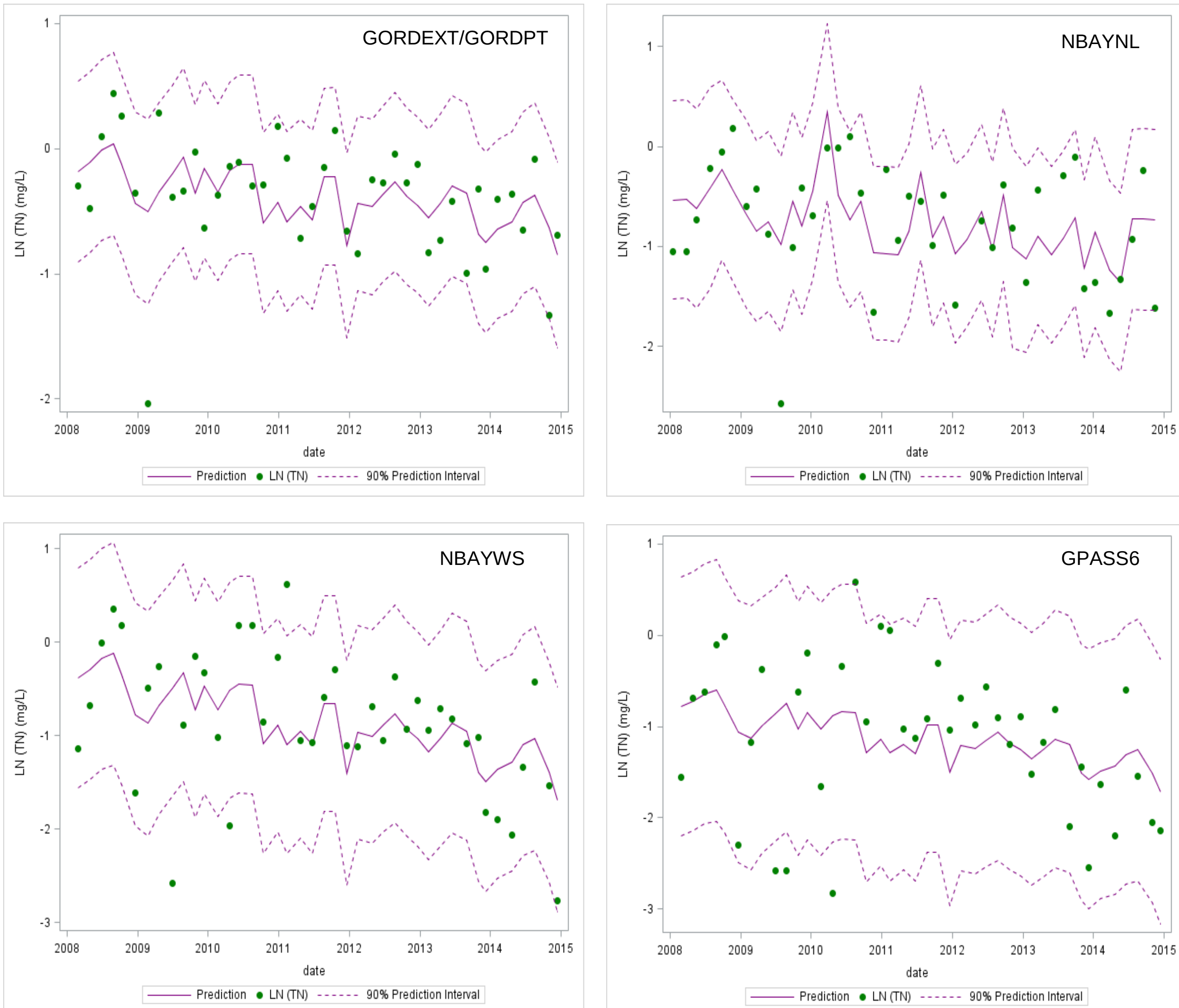


**Figure 3-24. Results of AEM time series models of bimonthly TN in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including observed, predicted, and 90 percent prediction intervals.**

**Table 3-8. Results of AEM time series models of bimonthly TN in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.2 | 2.9 | 0.07 | -0.0001 | **0.03** | 0.14 | **0.01** | n/a | | None |
| NBAYNL | 0.3 | 3.5 | 0.02 | 0.0002 | **0.01** | 0.06 | 0.3 | | | 8 |
| NBAYWS | 0.3 | 6.7 | 0.01 | -0.0004 | **0.005** | 0.18 | **0.04** | | | None |
| GPASS6 | 0.1 | 4.3 | 0.2 | -0.0002 | **0.08** | 0.13 | 0.24 | | | None |

**Table 3-9. Results of AEM time series models of bimonthly TP in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.1 | -1.5 | 0.4 | -0.00008 | 0.4 | -0.03 | 0.6 | -0.01 | 0.2 | None |
| NBAYNL | 0.1 | -2.4 | 0.3 | -0.00005 | 0.7 | 0.11 | 0.2 | -0.006 | 0.7 | None |
| NBAYWS | 0.1 | -1.5 | 0.3 | -0.0001 | 0.2 | 0.02 | 0.6 | 0.01 | 0.2 | None |
| GPASS6 | 0.2 | -4.6 | 0.07 | 0.00004 | 0.7 | 0.03 | 0.6 | 0.04 | **0.05** | None |

Because the annual geometric mean plots (Figure 3-23) showed evidence of a decreasing trend in TP at the WBID scale that was not significant in the AEM time series at individual stations from 2008 to 2014, the time-series model was extended back to 2005 by omitting the GGC flow covariate, which was not available before 2008. In the time-series analysis for 2005–2014, TP did show a statistically significant decreasing trend at the northern bay (NBAYNL) and mid-estuary stations (NBAYWS) in a model that included rainfall as a covariate (Table 3-10, Figure 3-25).

**Table 3-10. Results of time-series models of bimonthly TP in Gordon River (Marine Segment) and Naples Bay, 2005–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.02 | -1.8 | 0.1 | -0.00006 | 0.3 | 0.02 | 0.7 | | | None |
| NBAYNL | 0.14 | -0.02 | 0.9 | -0.0001 | **0.01** | 0.08 | **0.06** | | | None |
| NBAYWS | 0.18 | -1.5 | 0.08 | -0.0001 | **0.03** | 0.11 | **0.01** | | | None |
| GPASS6 | 0.06 | -2.4 | 0.1 | -0.00007 | 0.4 | 0.11 | **0.09** | | | None |

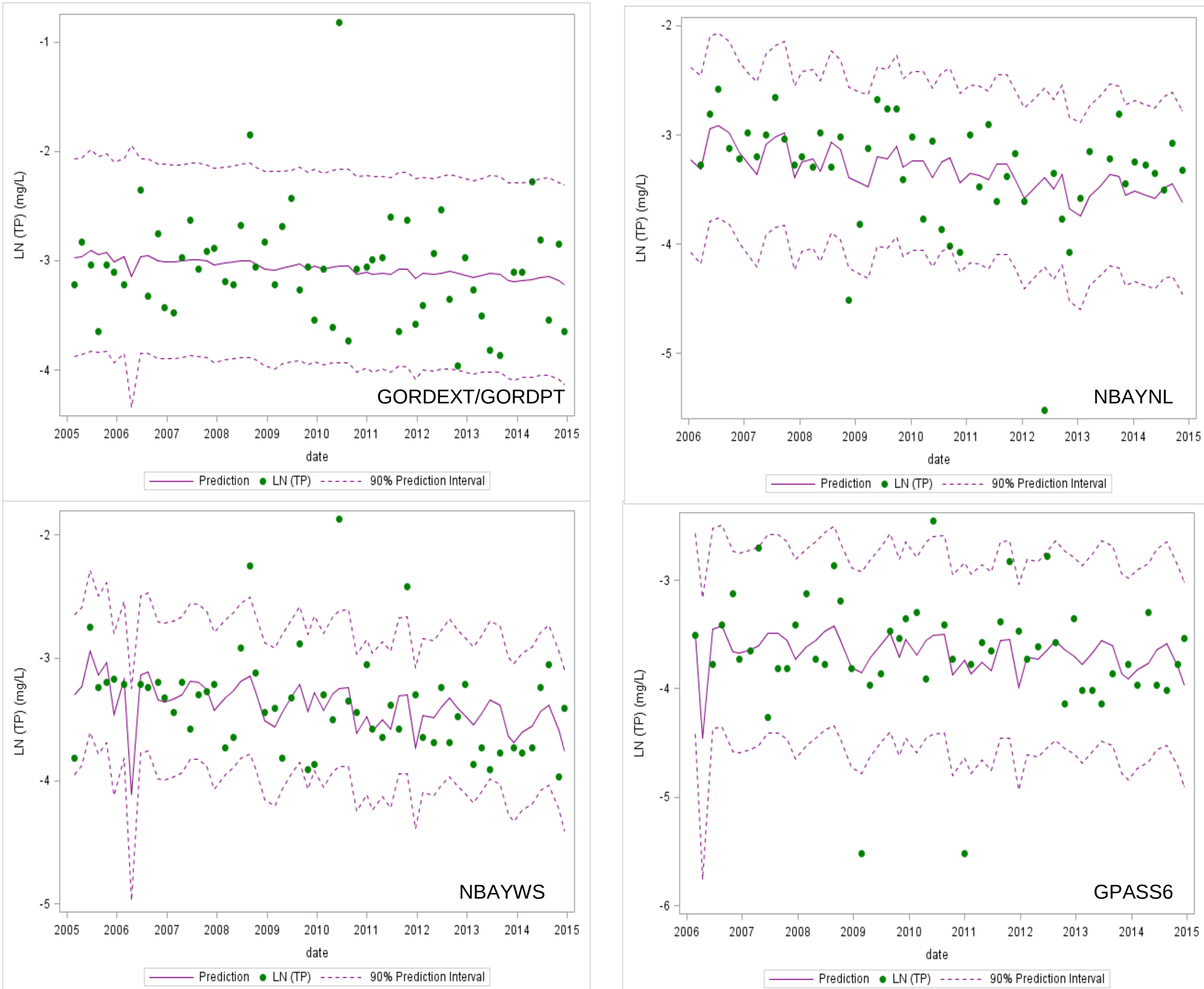


**Figure 3-25.** **Results of AEM time series models of bimonthly TP in Gordon River (Marine Segment) and Naples Bay, 2005–2014, including observed, predicted, and 90 percent prediction intervals.**

Spatial interpolation of annual average TN were created (for illustration purposes) showing slightly higher nitrogen concentrations are typical in the upper portions of Naples Bay and the Gordon River (Marine Segment) compared to the lower portions of the Bay (Figure 3-26). This is not unexpected as the upper portion of the Bay is influenced by urban runoff and costal tributaries with less expected tidal exchange with the relatively low nutrient Gulf water. The decreasing overall nitrogen concentrations are apparent when data from 2008 are compared to 2013 data.

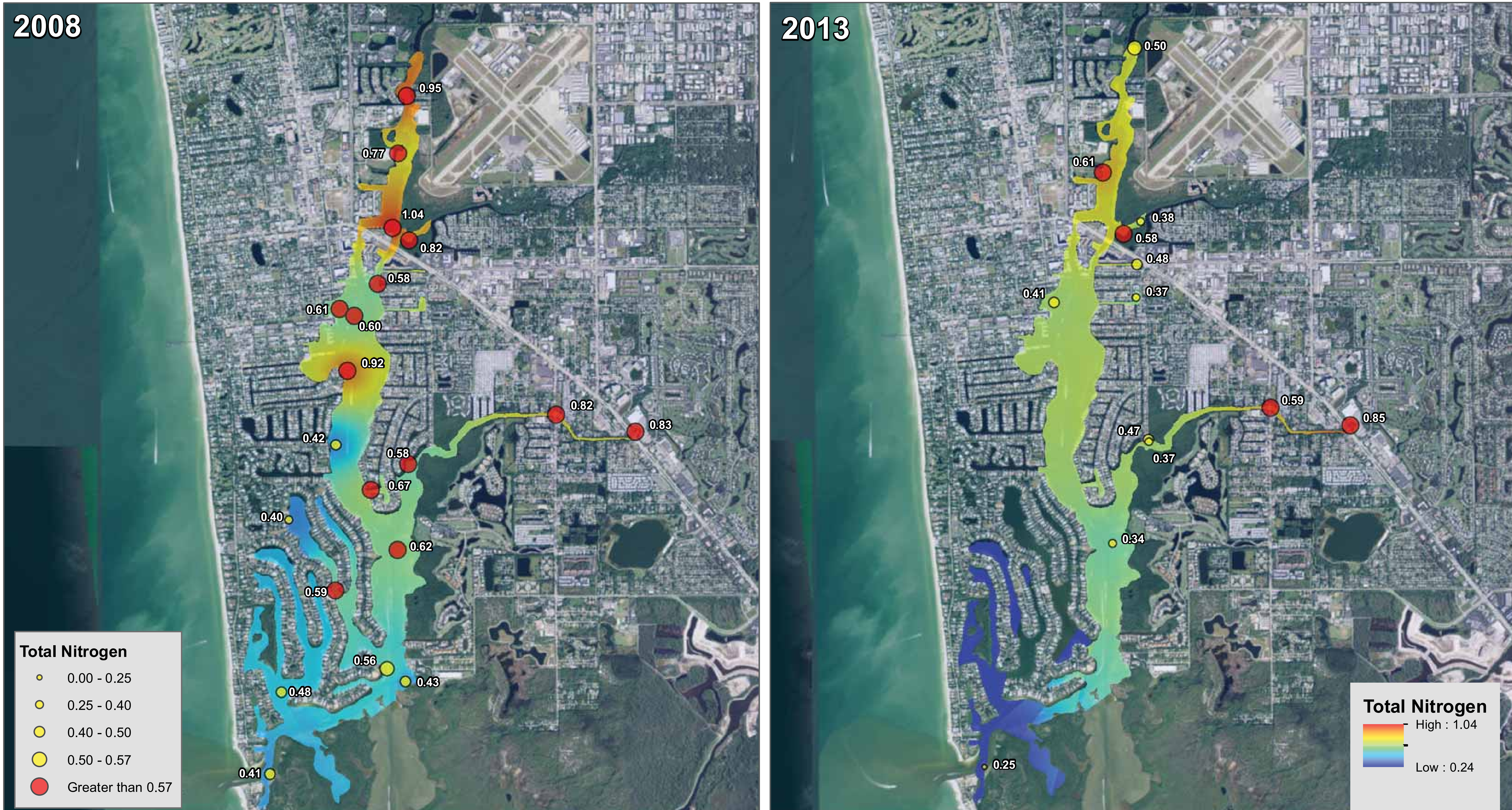


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**Figure 3-26. Annual Average Total Nitrogen Concentrations in Naples Bay and Gordon River (Marine Segment) with IDW Interpolation, 2008 and 2013**

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GIS Analyst: adam.bausch

### 3.2.5 Chlorophyll *a*

The estuarine NNC also includes a limit for chlorophyll *a* in WBID 3278R4 (Naples Bay) and the WBID 3278R5 (Gordon River Marine Segment). The criterion is expressed as an annual geometric mean concentration of 4.3 µg/L not to be exceeded more than once in a three year period. The onset of the City's monitoring program in 2006 allowed for a more robust characterization of chlorophyll *a* concentrations in Naples Bay.

Over the period of record (2000–2014), chlorophyll *a* concentrations in the WBID 3278R5 (Gordon River Marine Segment) and WBID 3278R4 (Naples Bay) have fluctuated around the newly adopted NNC criteria (Figure 3-27). More than one year in each three year period has exceeded the threshold since 2005, indicating chlorophyll *a* is not in compliance with the NNC. A total of 18 individual monitoring locations are included in this assessment for Naples Bay (WBID 3278R4), but only three have sufficient chlorophyll *a* data since 2011. Similarly, six individual locations were used in the assessment of the Gordon River (WBID 3278R5), with only two having data since 2011.

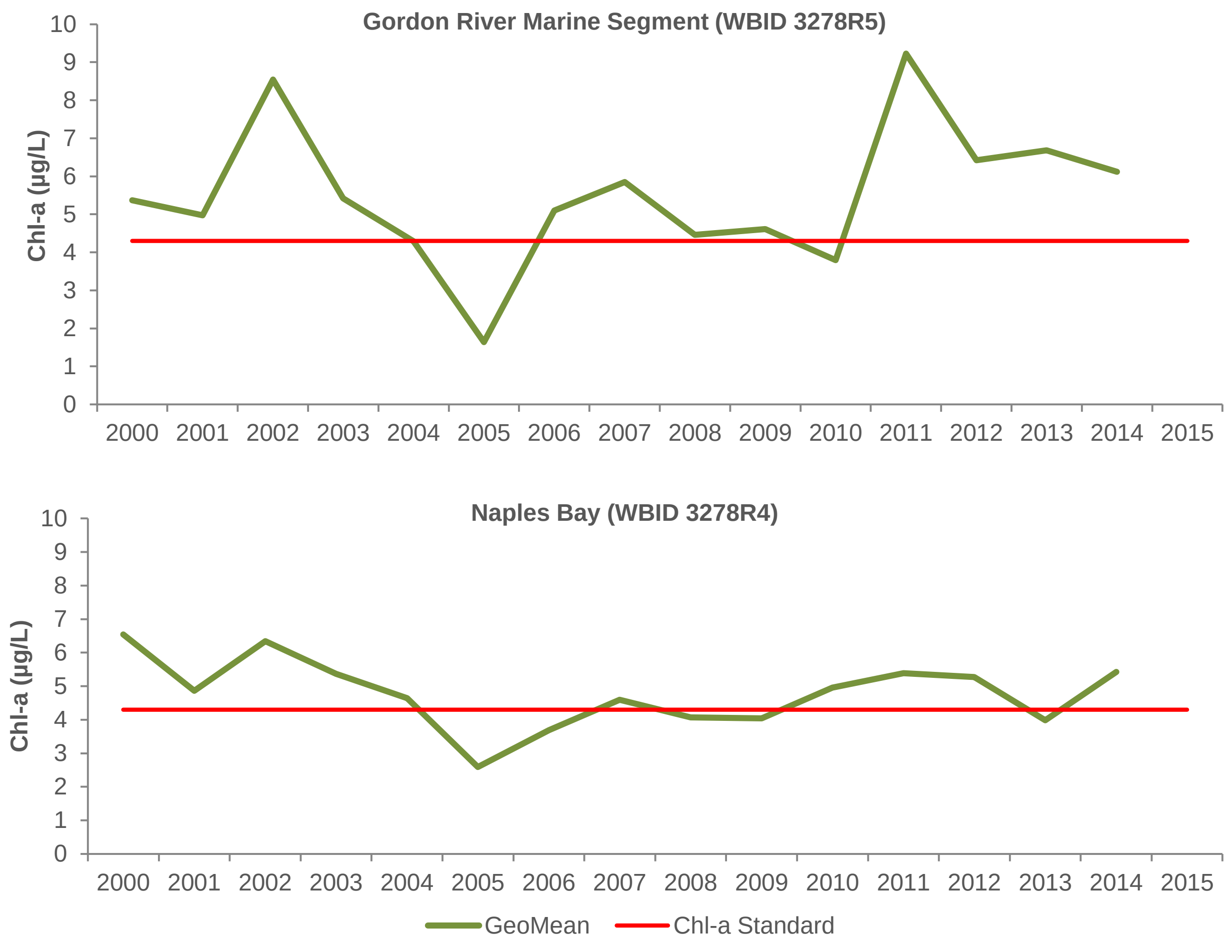


**Figure 3-27. Naples Bay (WBID 3278R4) and Gordon River (Marine Segment) (WBID 3278R5) chlorophyll *a* annual geometric mean concentrations and numeric nutrient criteria, 2000–2014.**

AEM time series models of chlorophyll *a* over time (2008–2014) show a statistically significant increasing trend at all of the long-term sampling locations (GODREXT/GORDPT, NBAYNL, NBAYWS) with the exception of the Gordon Pass location (GPASS6) (Table 3-11, Figure 3-28). However, it is possible that a higher laboratory MDL in the older data from the Gordon Pass location may be impacting this analysis. Flow was a significant covariate at the GORDEXT/GORDPT location with a negative relationship (chlorophyll *a* decreases as flow increases). Flow was not a statistically significant covariate at any other station. Rainfall was a statistically significant covariate at the NBAYWS and GPASS6 locations with a positive relationship indicating that chlorophyll *a* increases when rainfall increases. This may represent the localized effect of tributaries and rainfall on chlorophyll *a* concentrations instead of the GGC freshwater flow.

GORDEXT/GORDPT

NBAYNL

NBAYWS

GPASS6

**Figure 3-28. Results of AEM time series models of bimonthly chlorophyll *a* in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including observed, predicted, and 90 percent prediction intervals.**

**Table 3-11. Results of time-series models of bimonthly chlorophyll *a* in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.4 | -4.3 | 0.007 | 0.0003 | **0.001** | 0.07 | 0.3 | -0.54 | **0.005** | 4 |
| NBAYNL | 0.3 | -4.5 | 0.1 | 0.0003 | **0.03** | 0.09 | 0.3 | 0.03 | 0.1 | None |
| NBAYWS | 0.4 | -3.7 | 0.09 | 0.0002 | **0.02** | 0.20 | **0.01** | 0.01 | 0.4 | None |
| GPASS6 | 0.2 | 1.8 | 0.3 | -0.00004 | 0.7 | 0.19 | **0.01** | -0.01 | 0.4 | None |

As a result of the increasing trend observed in chlorophyll *a* we explored the potential connection with nutrients as the cause. A Spearman's Rank Order Correlation on individual observations of chlorophyll *a* and either TN or TP was conducted. This correlation was used because log transformation did not meet the standards for normality, so parametric correlation was not appropriate.

For Naples Bay (WBID 3278R4), chlorophyll *a* is weakly positively correlated with both TN and TP ($0.1 > [r_s] > 0.12$, $p < 0.05$). For the Gordon River Marine Segment (WBID 3278R5), chlorophyll *a* is weakly negatively correlated with TN ($-0.12 > [r_s]$, $p < 0.05$) and weakly negatively correlated with TP (Spearman's rank correlation, $0.21 > [r_s]$, $p < 0.05$). The weak results indicate that nutrient concentrations are not an accurate predictor of chlorophyll *a* in either waterbody.

IDW interpolation was used to show the spatial distribution of chlorophyll *a* concentrations in 2008 and 2013 (Figure 3-29). Higher chlorophyll *a* concentrations are typically found in the northern bay and Gordon River (Marine Segment), with the highest values observed in Haldeman Creek.

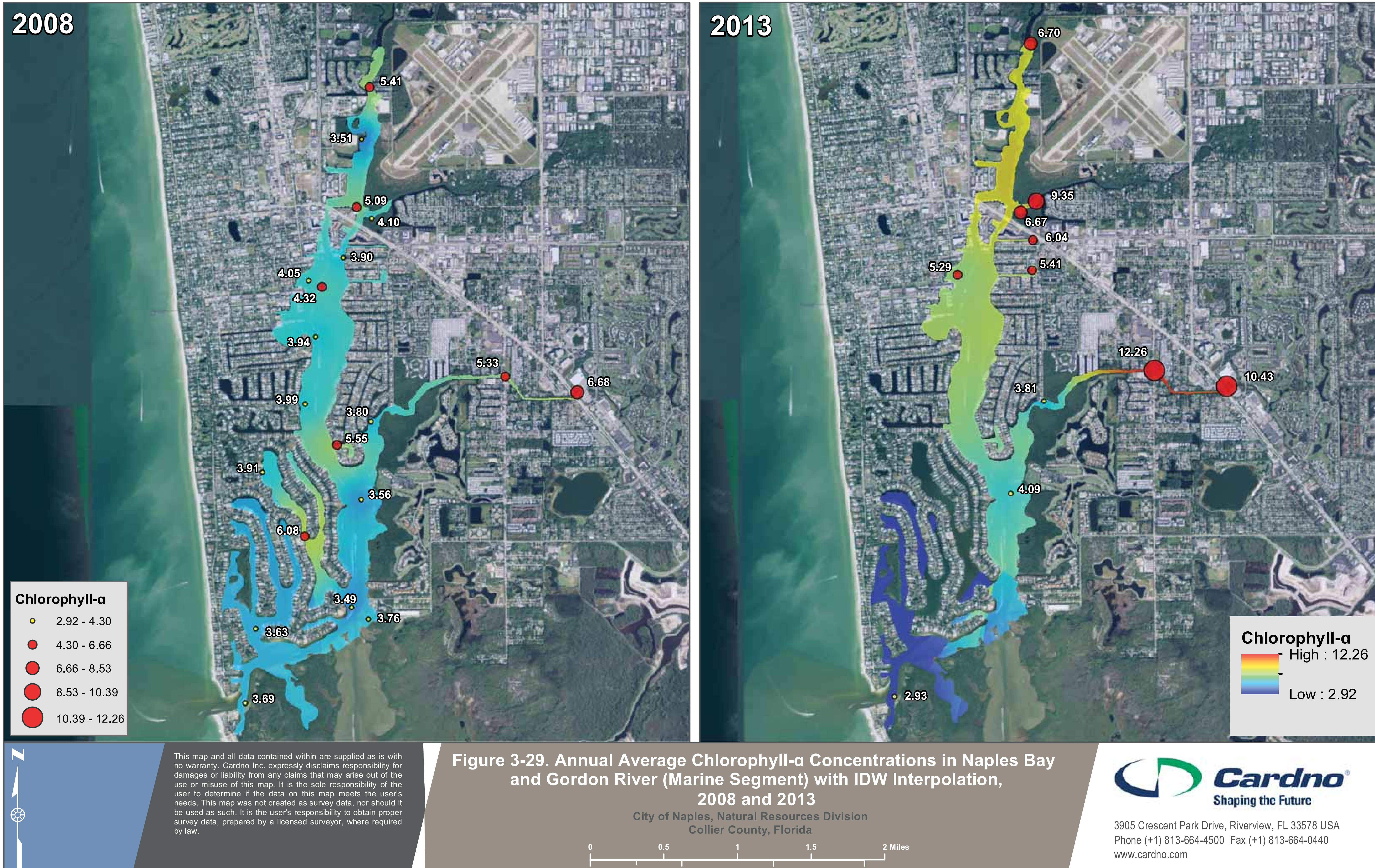


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**Figure 3-29. Annual Average Chlorophyll-a Concentrations in Naples Bay and Gordon River (Marine Segment) with IDW Interpolation, 2008 and 2013**

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GIS Analyst: adam.bausch

### 3.2.6 Dissolved Oxygen

Dissolved oxygen (DO) is viewed as a general indicator of waterbody health because it is essential to aquatic life. Since the 1970s and until recently, the marine water quality standard for DO in Florida required a minimum daily average of 5.0 mg/L, with instantaneous levels not to fall below 4.0 mg/L. However, these levels were derived with little information and were intended to be revised once more Florida specific information and data were available (FDEP 2013). In 2013, FDEP adopted revised DO criteria for fresh and marine waters. The new marine DO criteria is based on percent saturation instead of concentration and requires DO to maintain a daily average of 42 percent saturation (62-302.533, F.A.C.). In addition to the daily average, a seven day average percent saturation of 51 and a 30 day average percent saturation of 56 shall also be maintained. In this report, Naples Bay DO is evaluated against the revised DO criteria for Florida which more appropriately represent necessary aquatic life conditions for Florida estuaries.

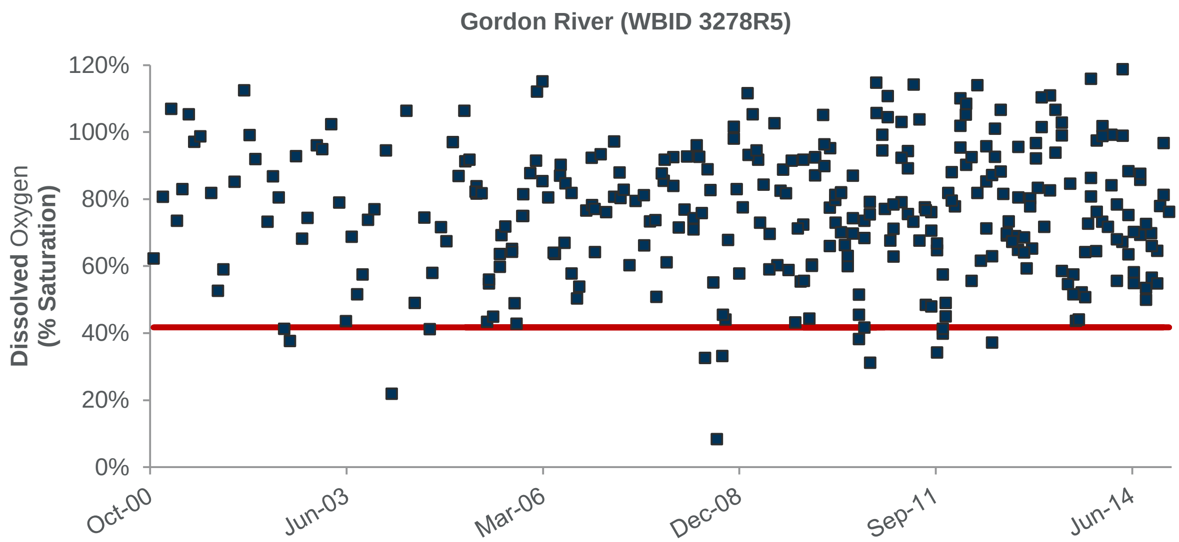
For comparisons to the marine water quality standard, DO is assessed at the WBID scale. Naples Bay had previously been listed by FDEP as impaired for DO, but more recent analysis indicated a low DO condition was natural and the Bay was removed from the impaired list. All available DO measurements, beginning in 2000, were used in this analysis to assess the pattern of DO in Naples Bay with respect to the new marine DO criteria. DO percent saturation data were calculated from the measured DO concentration (mg/L), temperature, and salinity at the time of collection.

WBIDs 3278R5 (Gordon River Marine Segment) and 3278R4 (Naples Bay) both achieve the DO criteria with far less than 10 percent of measurements below the 42 percent saturation benchmark (Figure 3-30). The grab sample data available (typically collected on a monthly or bi-monthly schedule) are insufficient to assess the seven day and 30 day average components of the criteria; however, with the vast majority of measurements above the 51 and 56 percent thresholds, there is no reason to suspect any exceedance of the DO weekly and monthly thresholds in Naples Bay.

Four stations in WBID 3278R4 (Naples Bay) and WBID 3278R5 (Gordon River Marine Segment) had enough long-term monitoring data to examine trends in dissolved oxygen concentrations over time at individual stations, accounting for the effects of flow from the GGC and regional rainfall. AEM time series models show no statistically significant trends in DO over time from 2008 to 2014 at the four long-term monitoring locations ($p > 0.05$, Table 3-12). The models for the northern Naples Bay and mid-estuary stations (NBAYNL and NBAYWS) had rainfall as a significant negative covariate, with flow also a significant negative covariate at NBAYWS.

**Table 3-12. Results of AEM time series models of bimonthly DO in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.2 | 3.7 | 0.5 | 0.00005 | 0.8 | -0.16 | 0.4 | -0.08 | 0.09 | None |
| NBAYNL | 0.2 | 4.6 | 0.4 | 0.00009 | 0.8 | -0.48 | **0.01** | -0.01 | 0.7 | None |
| NBAYWS | 0.3 | 4.5 | 0.4 | 0.0001 | 0.7 | -0.38 | **0.04** | -0.07 | **0.05** | None |
| GPASS6 | 0.6 | 5.2 | 0.01 | 0.00005 | 0.5 | -0.14 | 0.3 | -0.01 | 0.6 | 6, 8 |



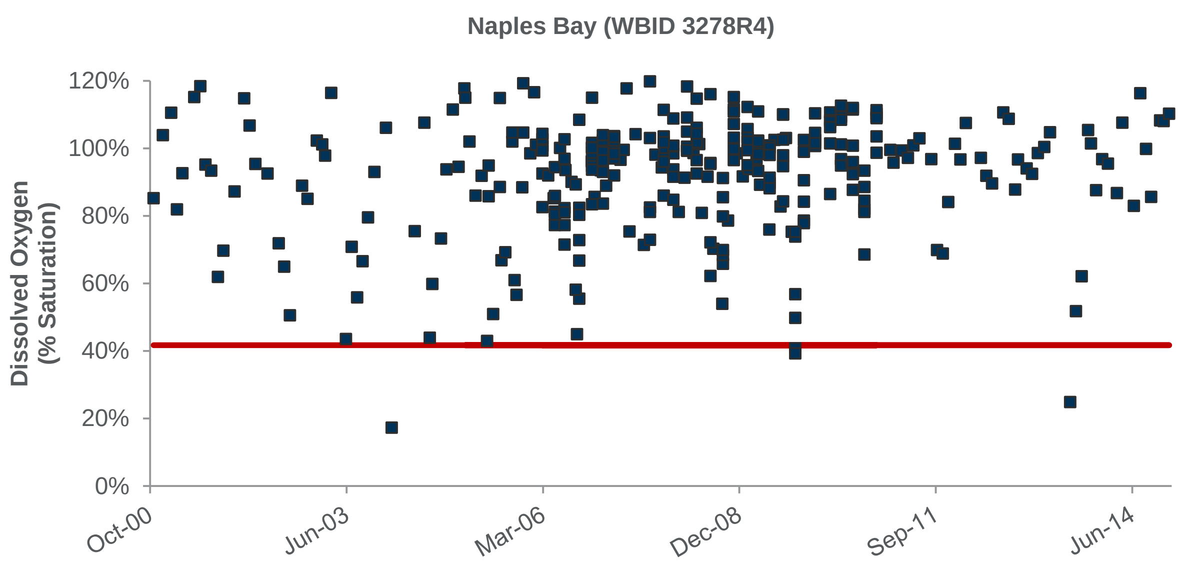




**Figure 3-30. Naples Bay (WBID 3278R4) and Gordon River Marine Segment (WBID 3278R5) dissolved oxygen percent saturation and the revised Class II criteria (42 percent saturation.**

### 3.2.7 Turbidity

Turbidity is an important measure of water clarity in estuarine systems. It measures to what extent the amount of suspended material in the water column decreases the passage of light through the water. Turbidity is measured in nephelometric turbidity units (NTU), where the higher the NTU value, the more suspended materials are hindering light passage in the water. Although there is a marine water quality standard for turbidity, the standard is based on comparisons relative to natural background conditions, which are not defined for Naples Bay. In addition, turbidity values in Naples Bay are low relative to the exceedance values defined in the standard. Thus, since insufficient data exist to compare turbidity in Naples Bay to the water quality standard, turbidity trends were examined by station rather than by WBID.

Four stations in Naples Bay and Gordon River (Marine Segment) had enough long-term monitoring data to examine trends in turbidity over time at individual stations, accounting for the effects of flow from the GGC and regional rainfall with AEM time series models. Three of the four locations show a statistically significant increasing trend in turbidity in the 2008–2014 time period (GODREXT/GORDPT, NBAYNL, and GPASS6) (Figure 3-31 and Table 3-13). NBAYWS was the only station that did not show a significant increasing trend. Flow was a statistically significant covariate at the northern Naples Bay (NBAYNL) and mid estuary (NBAYWS) locations with a negative relationship (as flow increases, turbidity decreases). Rainfall was a significant covariate at the Gordon River location (GORDEXT/GORDPT) and the mid estuary location (NBAYWS) with a positive relationship.

**Table 3-13. Results of AEM time series models of bimonthly turbidity in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.4 | -5.7 | 0.009 | 0.0003 | **0.006** | 0.21 | **0.02** | -0.03 | 0.12 | 2 |
| NBAYNL | 0.3 | -4.6 | 0.01 | 0.0002 | **0.005** | 0.05 | 0.3 | -0.02 | **0.06** | None |
| NBAYWS | 0.4 | -2.2 | 0.3 | 0.0001 | 0.14 | 0.16 | **0.02** | -0.06 | **0.001** | |
| GPASS6 | 0.2 | -12 | 0.01 | 0.0007 | **0.01** | -0.04 | 0.8 | 0.02 | 0.5 | None |

IDW interpolation was used to show the spatial distribution of turbidity in 2008 and 2013 (Figure 3-32). Turbidity appears to be increasing from 2008 to 2013, with slightly higher values observed in the northern portion of the Bay.

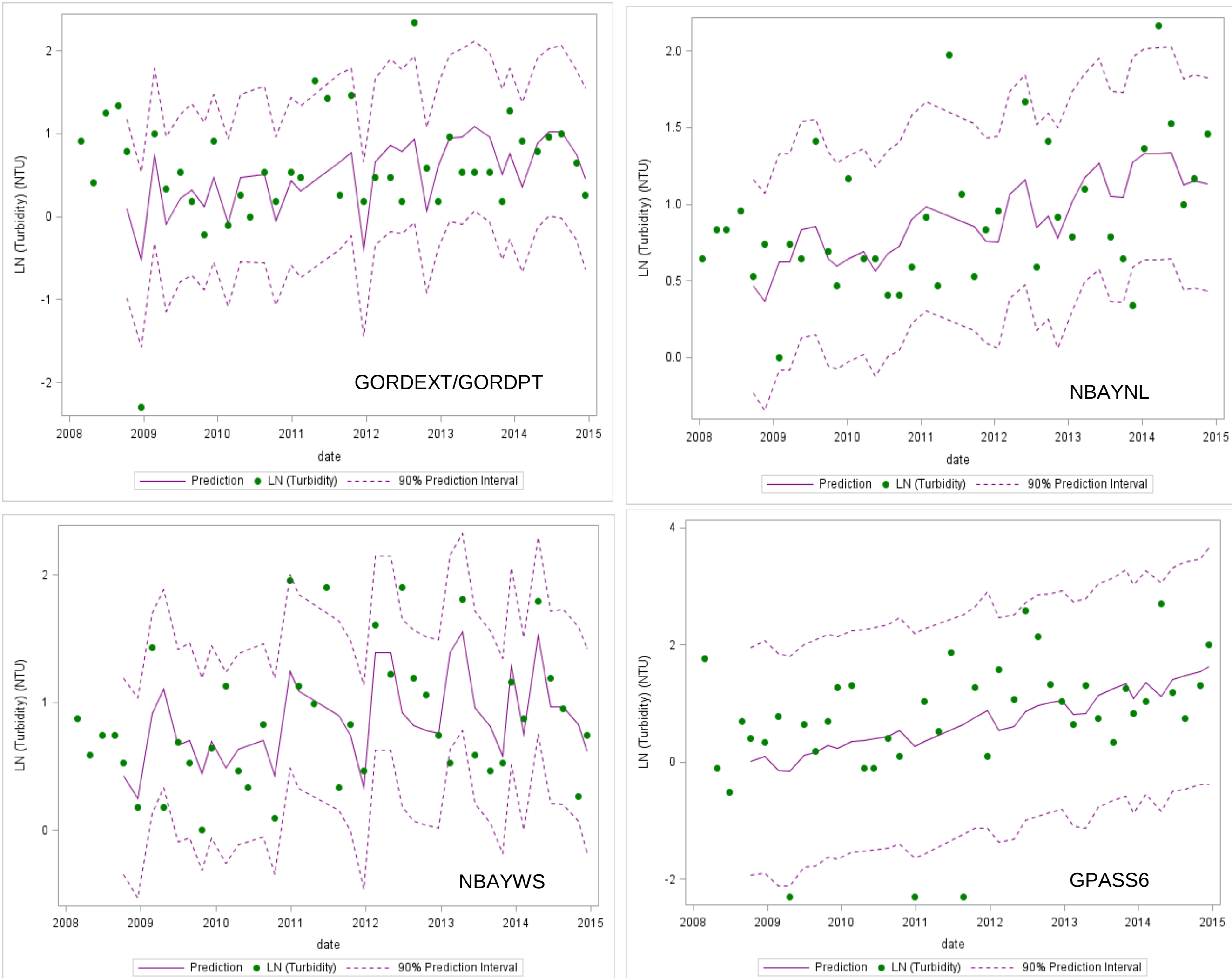



**Figure 3-31.** **Results of AEM time series models of bimonthly turbidity in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including observed, predicted, and 90 percent prediction intervals.**

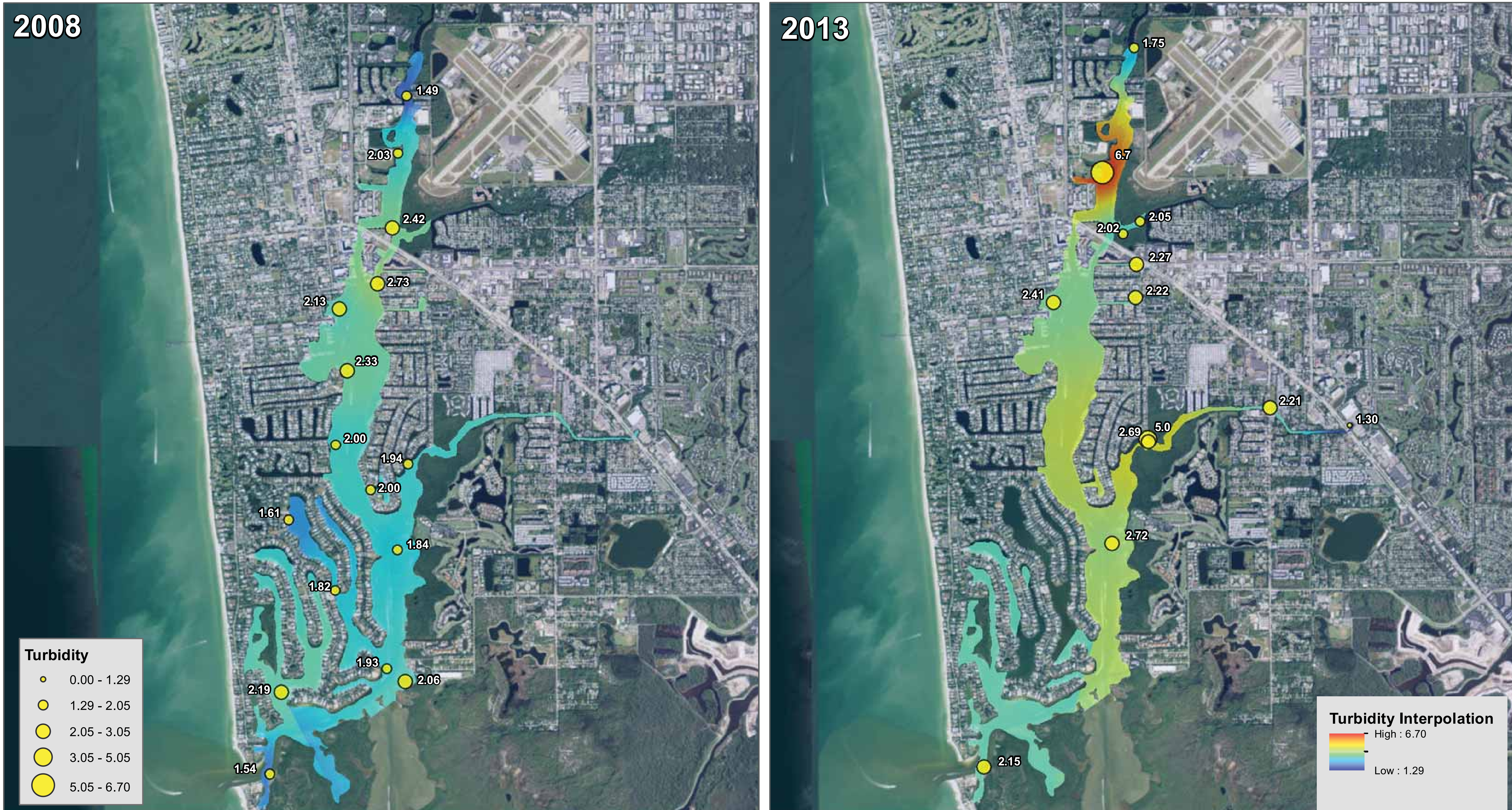



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**Figure 3-32. Annual Average Turbidity Concentrations in Naples Bay and Gordon River (Marine Segment) with IDW Interpolation, 2008 and 2013**

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GIS Analyst: adam.bausch

### 3.2.8 Bacteria (Fecal Coliform and Enterococci)

In estuarine waters, the Class II water quality standard states that fecal coliform counts shall not exceed a median value of 14 cfu/100 ml, with not more than 10 percent of the samples exceeding 43 cfu/100 ml, and may not exceed 800 cfu/100 ml on any one day (62-302.530(6), F.A.C.). It is worth noting that the Class II water quality standard lists fecal coliform units as most probable number (MPN), and units reported here are cfu/100 ml, which are an identical translation of the MPN criteria. The FDEP and EPA are in the process of reviewing the bacteriological water quality standard and soon may revise the marine criteria to be based on enterococci instead of fecal coliform. In 2012 EPA released recreational water quality guidance recommendations for enterococci in marine waters (EPA 2012). The recommended limit of enterococci in marine waters to protect human health is a geometric mean of 35 cfu/100 mL with no more than 10 percent of values to exceed 130 cfu/100 mL (EPA 2012).

Four stations in Naples Bay and Gordon River (Marine Segment) had enough long-term monitoring data to examine trends in bacteria concentrations over time at individual stations, accounting for the effects of flow from the GGC and regional rainfall with AEM time series models. Models for fecal coliform bacteria show a statistically significant increasing trend over time (2008–2014) at the northern most stations (GORDEXT/GORDPT ($p < 0.05$) and NBAYNL ($p < 0.1$)) (Table 3-14 and Figure 3-33). Flow and rainfall covariates were seldom statistically significant for the best-fitting fecal coliform auto-regressive time series models.

Enterococci bacteria show a statistically significant increasing trend over time at all long-term stations with the exception of the Gordon River location (Table 3-15 and Figure 3-34). GGC flow was not a significant covariate at any station, however rainfall was a significant covariate with a positive relationship at all three Naples Bay long-term locations (NBAYNL, NBAYWS, and GPASS6). Enterococci bacteria persist in marine water much longer than fecal coliform bacteria and therefore may explain the identification of enterococci trends in the more consistently marine locations.

IDW interpolation graphics were created to show the spatial distribution of enterococci counts in 2008 and 2013 (Figure 3-35). Enterococci levels appear to be higher in 2013 than 2008.

**Table 3-14. Results of AEM time series models of bimonthly fecal coliform in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.3 | -13 | 0.03 | 0.0009 | **0.005** | 0.12 | 0.5 | -0.06 | 0.2 | None |
| NBAYNL | 0.2 | -8.6 | 0.2 | 0.0006 | **0.06** | 0.3 | 0.1 | -0.006 | 0.8 | None |
| NBAYWS | 0.4 | -7.4 | 0.2 | 0.0004 | 0.2 | 0.41 | **0.08** | 0.12 | **0.01** | None |
| GPASS6 | 0.2 | -0.68 | 0.9 | 0.00008 | 0.8 | 0.52 | **0.02** | 0.03 | 0.4 | None |

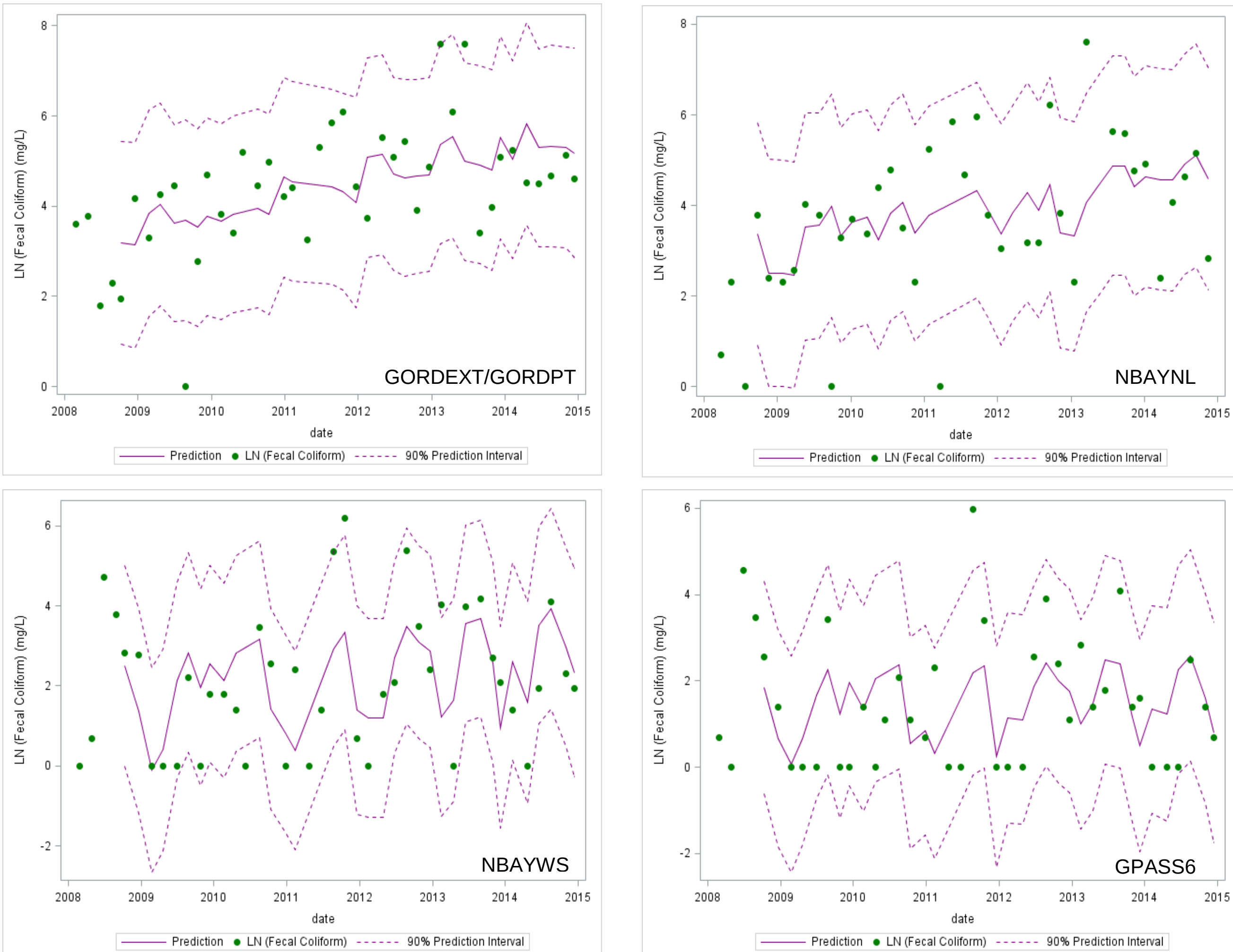


**Figure 3-33. Results of AEM Time Series Models of Bimonthly Fecal Coliform in Gordon River (Marine Segment) and Naples Bay, 2008–2014, Including Observed, Predicted, and 90 percent Prediction Intervals.**

**Table 3-15. Results of AEM time series models of bimonthly Enterococci in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including total model fit ($r^2$); intercept, time, and covariate parameter estimates and p values; and statistically significant autoregression frequency.**

| Station | Total Model $r^2$ | Intercept | | Time (Date) | | LN Rain | | LN Flow | | Auto-regression (Months) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $B_0$ | p | $B_1$ | p | $X_1$ | p | $X_2$ | p | |
| GORDEXT/ GORDPT | 0.1 | -2.5 | 0.6 | 0.0003 | 0.2 | 0.12 | 0.5 | -0.04 | 0.2 | None |
| NBAYNL | 0.3 | -13 | 0.03 | 0.0008 | **0.01** | 0.48 | **0.03** | -0.08 | 0.1 | 10 |
| NBAYWS | 0.4 | -11 | 0.06 | 0.0006 | **0.03** | 0.57 | **0.008** | 0.007 | 0.8 | None |
| GPASS6 | 0.5 | -12 | 0.01 | 0.0007 | **0.005** | 0.47 | **0.01** | 0.03 | 0.32 | 12 |

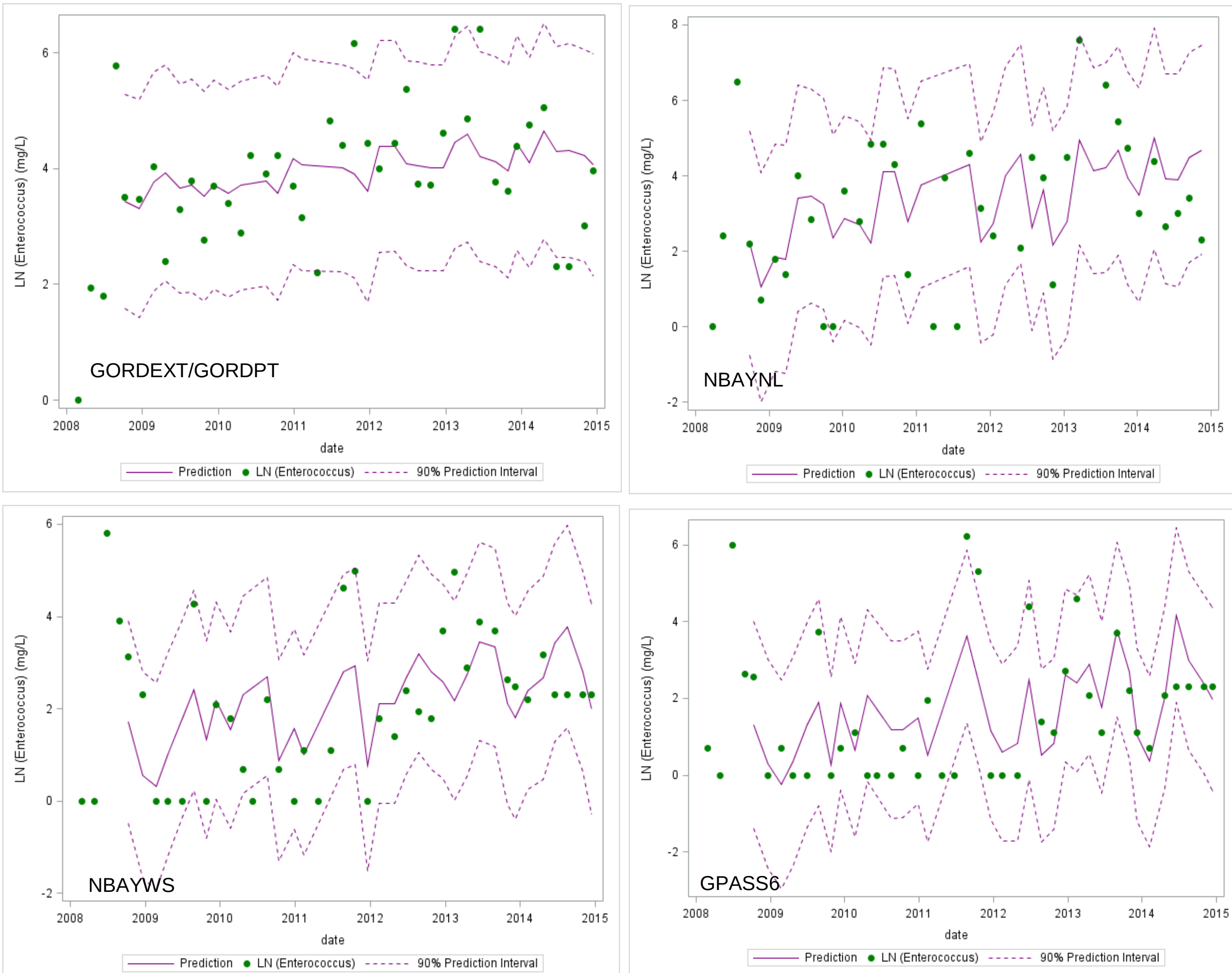


**Figure 3-34. Results of AEM time series models of bimonthly Enterococci in Gordon River (Marine Segment) and Naples Bay, 2008–2014, including observed, predicted, and 90 percent prediction intervals.**

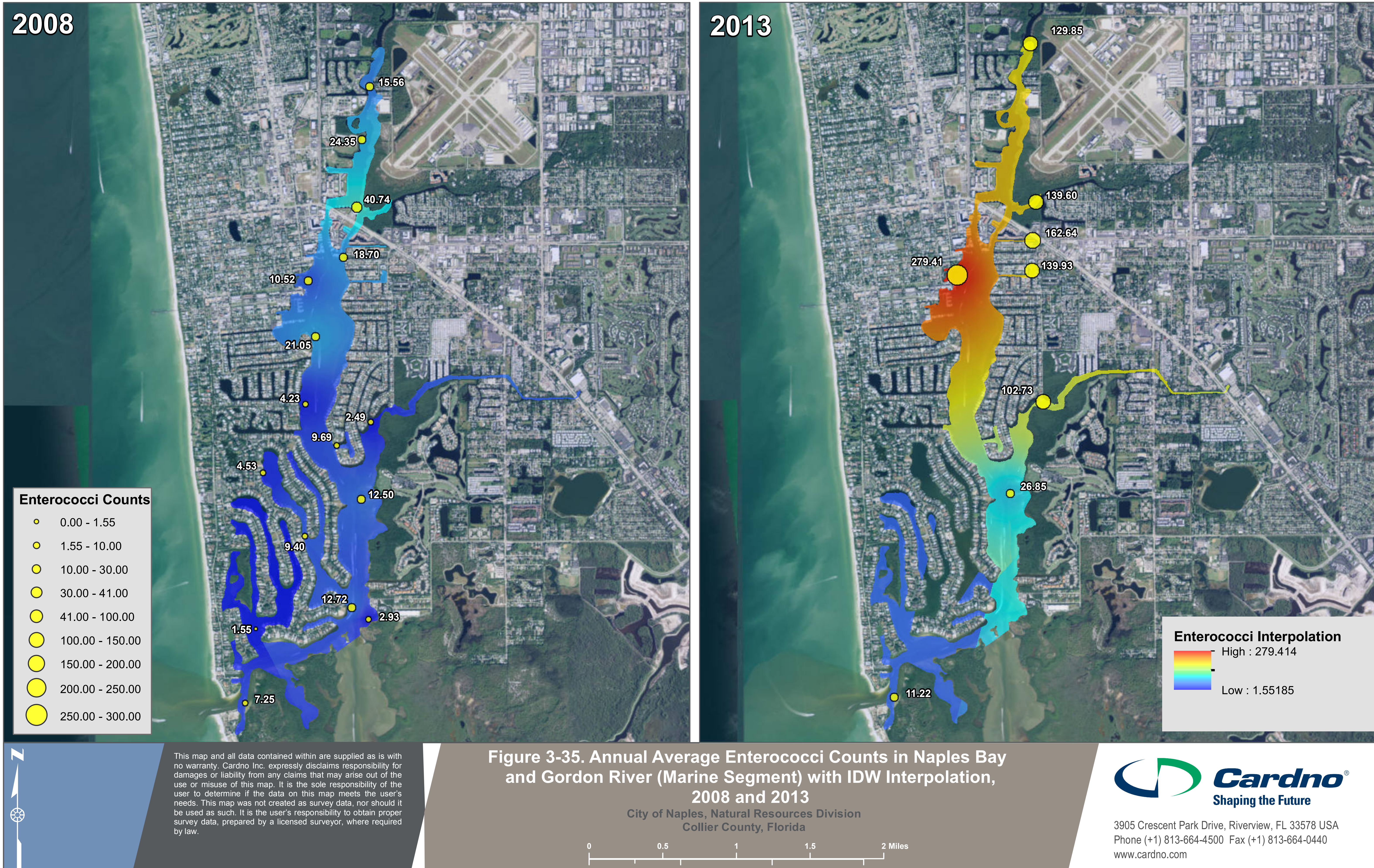




**Figure 3-35. Annual Average Enterococci Counts in Naples Bay and Gordon River (Marine Segment) with IDW Interpolation, 2008 and 2013**

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GIS Analyst: Lauren.Federsel

# 4 Naples Bay Biological Community

This section is devoted to the identification of statistically significant trends in biological community data in Naples Bay. The potential for changes in biological community over time or between different zones within Naples Bay are explored. The analysis presented here focuses on the seagrass and fish community monitoring programs conducted by the City of Naples. The City has been monitoring seagrass since 2006 and fish since 2009. Analysis of the current status of these communities along with quantifying any significant changes over time is an important tool in terms of resource management.

## 4.1 Seagrass Community

The City of Naples monitors five fixed transects located in three separate seagrass areas (designated BV, NChannel, and SPortRoyal) located in the southernmost portion of the Bay (Figure 4-1). These beds represent the majority of seagrass known in the Bay. The following indicators were used to evaluate and identify general patterns in the seagrass systems of Naples Bay over time:

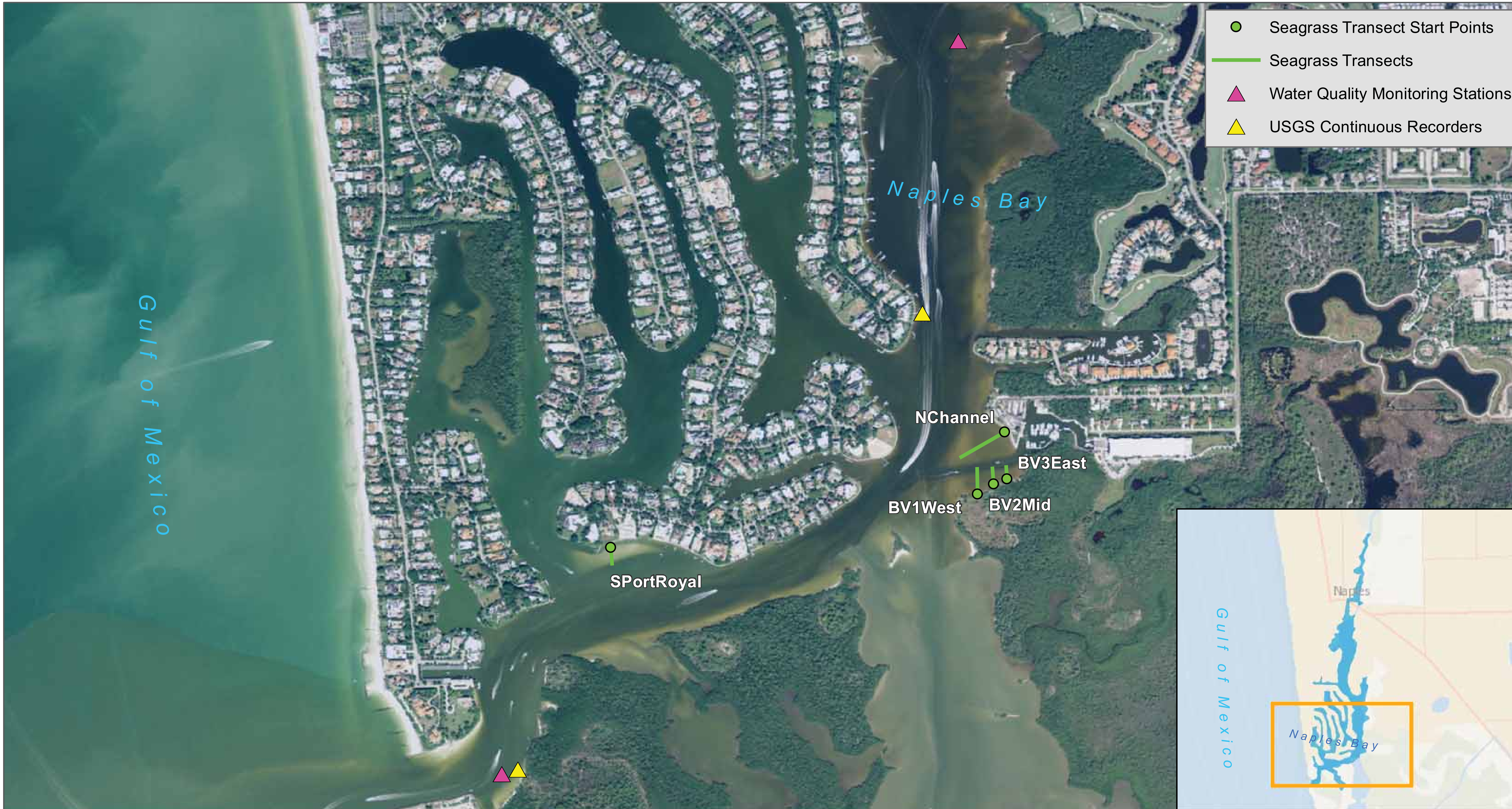
- Seagrass composition: Number of species present
- Seagrass cover: Categories of percent cover
- Seagrass density: Number of seagrass short shoots per square meter
- Seagrass depth distribution: Maximum water depth and depth range

On any sampling day, the variance among the location-specific measurements was relatively small, indicting little spatial variance at the time of the sample. Therefore, for the purpose of analysis (unless otherwise noted), data for each metric were pooled, resulting in a single value per sampling day.

Seagrass transects were monitored once or twice per year during the growing season from 2006 to 2014. In early years, surveys were generally once in the early part of the season and once later in the season. In the last four years, surveys were only conducted once, and for the last three years, only in the later part of the season (Table 4-1). This shift in methodology may complicate identification of temporal patterns difficult in the dataset as a whole.

**Table 4-1. Timing of seagrass surveys for each transect in Naples Bay, 2006–2014**

| Year | Transect ID | | | | |
|---|---|---|---|---|---|
| | BV1West | BV2Mid | BV3East | NChannel | SPortRoyal |
| 2006 | July | July | August | September | October |
| 2007 | April, September | April, September | September | September | June, November |
| 2008 | May, October | May, October | May, October | May, October | May, October |
| 2009 | May, October | May, October | May, October | May, October | June, October |
| 2010 | June, September | June, September | June, September | June, September | June, September |
| 2011 | June | June | June | June | June |
| 2012 | August | August | August | August | August |
| 2013 | September | September | September | September | September |
| 2014 | August | August | August | August | September |

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# Figure 4-1. Naples Bay Seagrass Monitoring Transects and Nearby Water Quality Monitoring Stations

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0 500 1,000 1,500 2,000 Feet
0 100 200 300 400 500 600 Meters

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GIS Analyst: adam.bausch

### 4.1.1 Seagrass Species Composition

Three species of seagrass were observed in the survey area: *Halodue wrightii*, *Halophila decipiens*, and *Halophila englemannii*. Two species of rhizophytic, bed-forming macroalgae were also occasionally present: *Caulerpa prolifera* and *Caulerpa mexicana*. *H. wrightii*, generally growing in monospecific beds, was by far the most common seagrass, occurring in 88 percent of the quadrats surveyed along the transects (Table 4-2). *Halophila decipiens* and *Halophila engelmannii* were much less common, occurring in less than 5 percent of quadrats surveyed for the whole survey period from 2006 to 2014.

*H. wrightii* is one of the most commonly occurring species of seagrass in Florida (Dawes 2004). It can tolerate a wide range of salinity, nutrient, and physical environments (Zieman 1982, van Tussenbroek *et al.* 2010) and can be found in intertidal and subtidal areas (Zieman and Zieman 1989). Subtidally, *H. wrightii* can grow in both monospecific beds and mixed with other seagrasses (Yarbro & Carlson 2013). *Halophila engelmanni* and *Halophila decipiens* are generally considered to be low-light species and can grow in much deeper depths than many other Florida species; however, both species can be found at shallower depths where water is more turbid (van Tussenbroek, *et al.*, 2010). Within Florida, *Halophila englemanni* is most commonly found along the southwest coast (Yabro and Carlson 2013) and generally only grows as an understory to other species (van Tussenbroek, *et al.* 2010). *Halophila decipiens* is limited to areas with near-marine salinities (Zieman 1982, van Tussenbroek *et al.* 2010).

**Table 4-2. Percentage occurrence of seagrasses and rhizophytic algae by species at fixed monitoring stations Naples Bay, 2006–2014.**

| Species | Year | | | | | | | | | Total |
|---|---|---|---|---|---|---|---|---|---|---|
| | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | |
| *Halodule wrightii* | 88.2 | 85.2 | 91.2 | 97.0 | 91.4 | 93.8 | 88.2 | 93.3 | 62.1 | 87.8 |
| *Halophila decipiens* | -- | 3.7 | 8.8 | 15.2 | -- | -- | 5.9 | -- | -- | 4.2 |
| *Halophila engelmannii* | 5.9 | -- | -- | -- | -- | -- | -- | 6.7 | -- | 1.3 |
| *Caulerpa prolifera* | -- | -- | -- | -- | 2.9 | -- | 5.9 | -- | -- | 0.8 |
| *Caulerpa mexicana* | -- | 3.7 | -- | -- | -- | -- | -- | -- | -- | 0.4 |
| none | 5.9 | 11.1 | 8.8 | 3.0 | 5.7 | 6.3 | 11.8 | 6.7 | 34.5 | 10.5 |

### 4.1.2 Seagrass Cover

Seagrass cover was assessed using a modified Braun-Blanquet scale where a categorical score is assigned to a range of percent bottom cover. Total seagrass cover was generally low across all transects over the entire survey period (Figure 4-1); the highest Braun-Blanquet cover score recorded from 2006 to 2014 was a 2 which corresponds to 5–25 percent cover. The most frequently recorded score was 1, which indicates less than 5 percent seagrass cover. The qualitative Braun-Blanquet cover score method does not allow detection of small changes in seagrass cover because the range of percentages covered by one score is quite large (Bell *et al.* 2008). For low density systems like Naples Bay, where small gains would be worth documenting, more quantitative methods like actual percent cover or biomass measurements would allow for a more in depth statistical analysis of seagrass patterns.

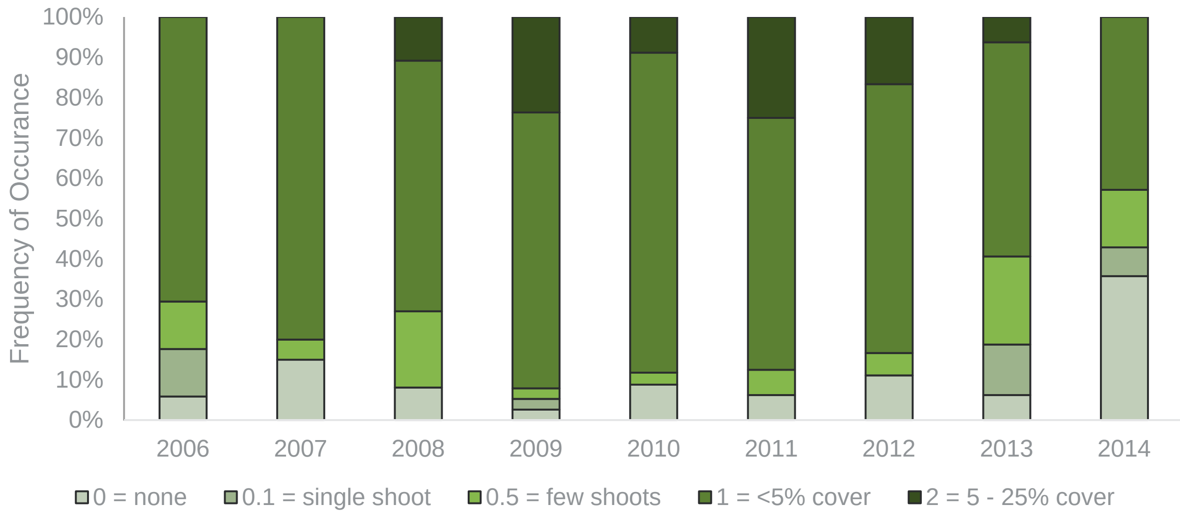


**Figure 4-2. Occurrence of each cover score category (Braun-Blanquet) for *H. wrightii* at fixed monitoring stations in Naples Bay, 2006–2014.**

### 4.1.3 Seagrass Density

Seagrass density (number of short shoots per square meter) was measured in each fixed quadrat sampling location during each survey event. When data from all transects and all survey events are considered together by year, it appears that *H. wrightii* was increasing in density until about 2011 and then began decreasing through 2014 (Figure 4-3). However, when the data are pooled by month, a trend of decreasing density as the growing season progresses becomes apparent (Figure 4-4). Naturally decreasing seagrass density as the season progresses from summer to winter is common in Southwest Florida bays (Yarbro and Carlson 2013). Because seagrass surveys were conducted only during the later months of the survey season between 2012 and 2014 (Table 4-1), it is difficult to separate a potential seasonal sampling bias from actual overall declines in seagrass in Naples Bay. It is also likely that water quality (*i.e.* nutrient and solids loading from the GGC) plays a role in the observed decreasing trend, and further investigation into the potential causes is warranted.

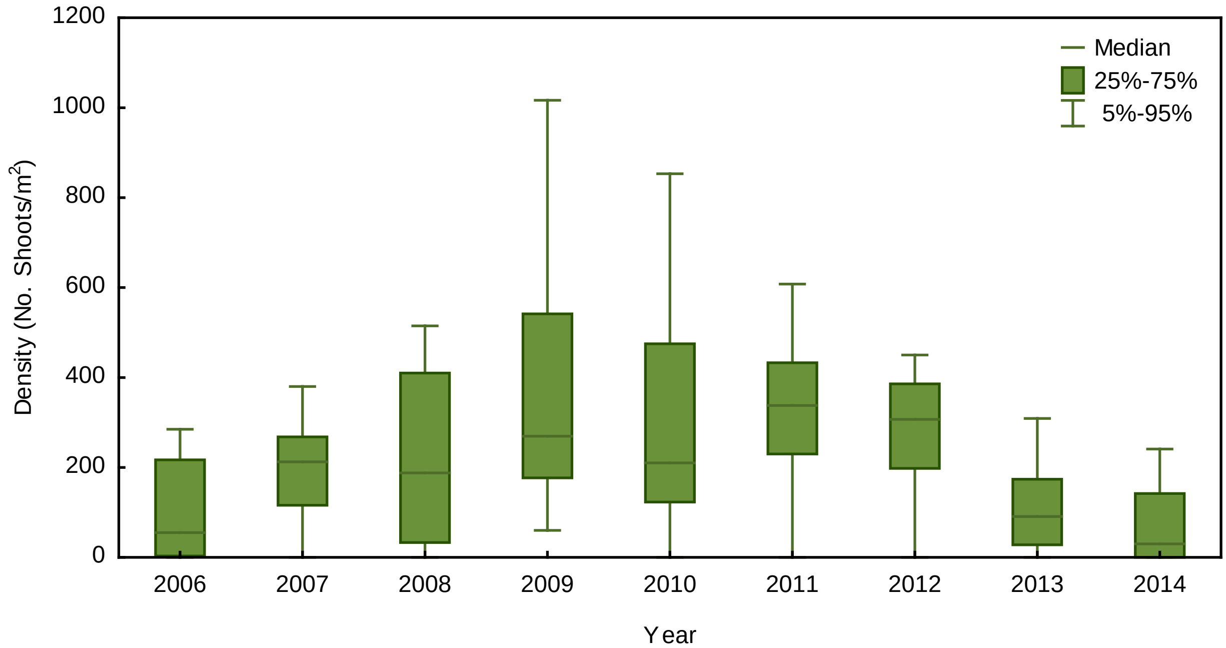


**Figure 4-3.** **Seagrass (*H. wrightii*) density (number of shoots/m²) in Naples Bay, 2006–2014.**

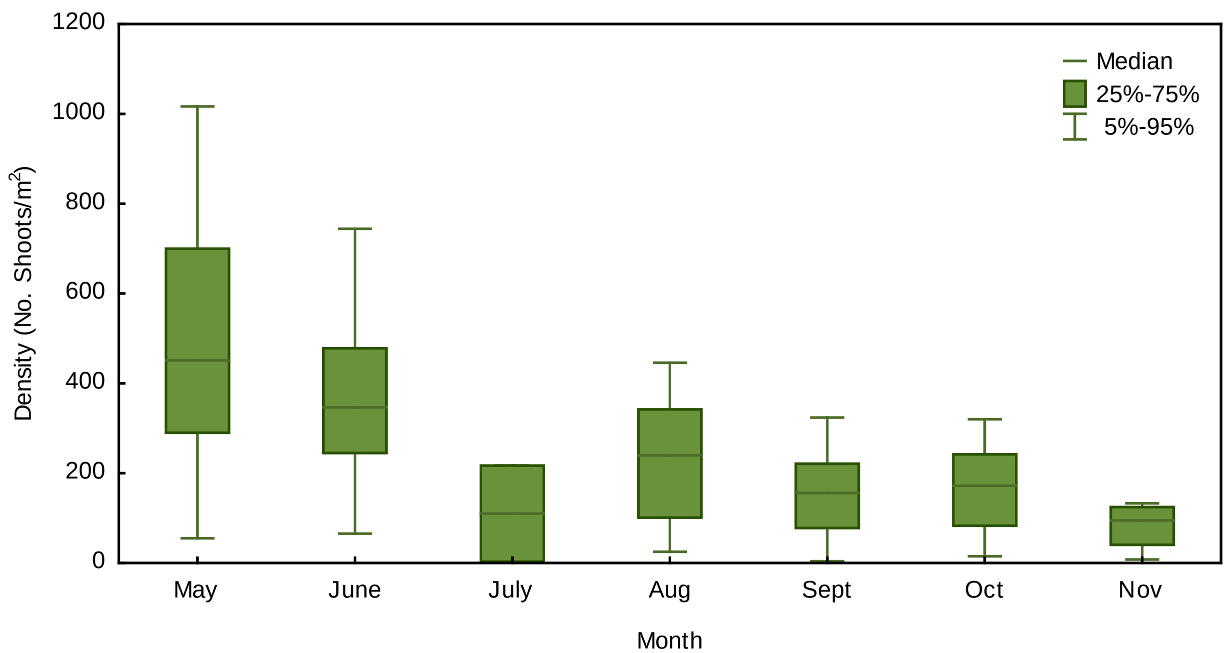


**Figure 4-4.** **Seagrass (*H. wrightii*) density (number of shoots/m²) by month in Naples Bay, 2006–2014**

### 4.1.4 Seagrass Depth Distribution

Water depths along the survey transects were standardized relative to mean high water (MHW) to eliminate tidal influence on water depth measurements. Of the three species of seagrass encountered during surveys in 2006–2014, *H. wrightii* grows at the widest range of water depths (Figure 4-5). *Halophila decipiens* and *Halophila engelmannii* were present only in slightly deeper water depths, in areas that are not likely to be exposed during low tides. The depth distributions are within the expected range for each species (see Section 4.4.1).


**Figure 4-5. Depth range of Naples Bay seagrasses (*H. wrightii*, *Halophila decipiens*, and *Halophila engelmannii*) and macroalgae (*Caulerpa* spp.). Green lines represent the mean depth for each species.**

In general, for transect seagrass surveys, changes in overall transect length from year to year can be an indicator of whether overall seagrass areal extent is increasing or decreasing. In addition, extension of the seagrass along the deep edge of the transect can be an indicator of improved water quality conditions. Likewise changes in the maximum depth of seagrass occurrence can signal changes in water quality, especially in terms of light availability.

Four of the five transects are located on relatively narrow shoals that run into the edge of a deep channel; thus, overall seagrass expansion on the deep edge of the bed is mostly likely limited by physical factors, not water quality. However, it is still useful to look at changes in transect length (measured as the distance from the landward seagrass edge to the furthest seaward seagrass location), in particular whether the transects are decreasing in length, as an indication of changes over seagrass area (Table 4-3). Notably, four out of five transects show relatively large drops in overall transect length in 2014 after several years of relatively little change. The fifth transect, NChannel, which is located on a much wider shoal, increased greatly in length starting in 2012, when seagrass colonized a gap between two previously discontinuous beds and the transect was extended to include the whole area. The overall transect length for NChannel was highest in 2014, but it should be noted that there were several areas along the transect with very little or no seagrass cover in 2014. Thus, seagrass appears to have expanded to a larger portion of the shoal but it may not be a continuous bed at this time.

The average depth of seagrass occurrence (Figure 4-6) was highest (maximum = 153 cm MHW) and most variable over time along the NChannel transect. The other four transects varied much less over time and generally averaged from 90 cm MHW to 115 cm MHW until 2011. After that, average seagrass depth declines along all four transects. As mentioned above, this could be related to physical factors rather than water quality changes and might be biased by differences in survey timing in more recent years.

**Table 4-3. Overall seagrass transect length at each survey in Naples Bay, 2006–2014. Bold values represent the minimum value for each transect.**

| Survey Event | BV1West | | BV2Mid | | BV3East | | NChannel* | | SPortRoyal | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Total Length (m) | % Change | Total Length (m) | % Change | Total Length (m) | % Change | Total Length (m) | % Change | Total Length (m) | % Change |
| 2006 | 20.7 | -- | 26.4 | -- | 26.8 | -- | 20 | -- | 30 | -- |
| Early 2007 | **11.3** | **-45** | 24.7 | -6 | | | | | 38.7 | 29 |
| Late 2007 | 13.8 | 22 | 24.4 | -1 | 29.7 | 11 | 22.6 | 13 | 31 | -20 |
| Early 2008 | 16 | 16 | 30.3 | 24 | 30.4 | 2 | 26.2 | 16 | 37.15 | 20 |
| Late 2008 | 16.2 | 1 | 20.1 | -34 | 29.3 | -4 | **21.7** | **-17** | 30 | -19 |
| Early 2009 | 16.2 | 0 | 31.4 | 56 | 30.9 | 5 | 28 | 29 | 37.3 | 24 |
| Late 2009 | 20.3 | 25 | 30 | -4 | 31.6 | 2 | 23 | -18 | 35.8 | -4 |
| Early 2010 | 14.3 | -30 | 27.9 | -7 | 31 | -2 | 22.5 | -2 | 35.3 | -1 |
| Late 2010 | 19.3 | 35 | 27.8 | 0 | 31 | 0 | 22.1 | -2 | 37.3 | 6 |
| Early 2011 | 19.8 | 3 | 27.8 | 0 | 30.8 | -1 | 26.6 | 20 | 37.3 | 0 |
| Late 2012 | 20 | 1 | 25.8 | -7 | 30.7 | 0 | 29.3* | 10 | 38.8 | 4 |
| Late 2013 | 20 | 0 | 25.2 | -2 | 32.1 | 5 | 175.3* | 498 | 35.3 | -9 |
| Late 2014 | 13.7 | -32 | **11.1** | **-56** | **23.6** | **-26** | 181.2* | 3 | **17.7** | **-50** |

*The number of sites along surveyed along this transect increased to cover a larger area. The large jump in transect length is due to the change in methodology.

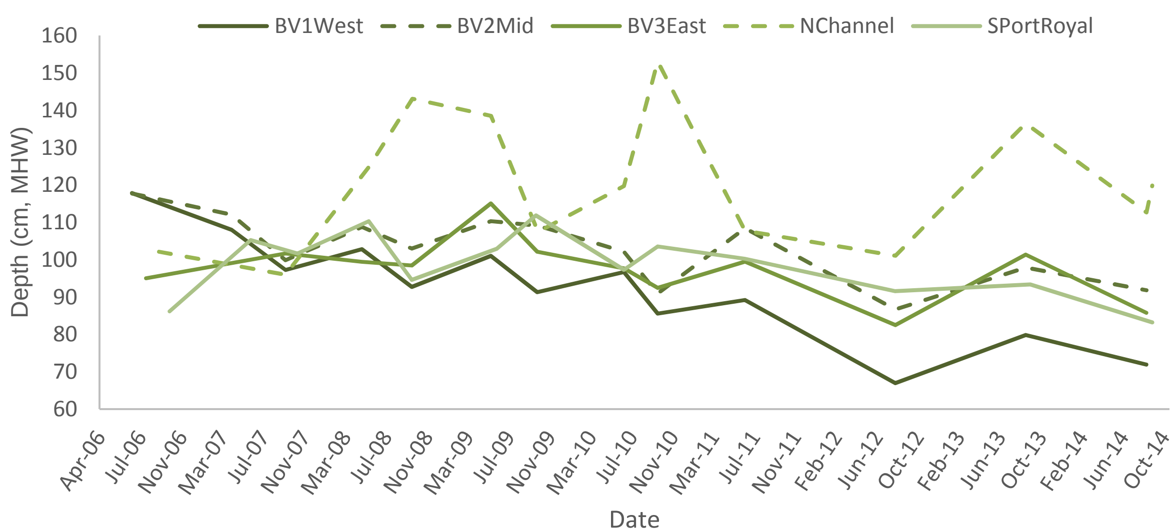


**Figure 4-6. Average depth (relative to MHW) of seagrass occurrence along transects in Naples Bay, 2006–2014.**

## 4.2 Fish Community

Fish sampling in Naples Bay was conducted using bottom trawls. Samples were collected approximately six times per year (generally every other month) with trawls in each of four zones in the bay (Figure 4-7) during each sampling event. From 2009 to August 2011, sampling was conducted at fixed transect sites. Starting in October 2011, sampling was conducted in one randomly selected grid within each zone at each sampling event. Fish species were identified, counted, and measured. Results of statistical analysis of fish community structure, diversity, richness, and abundance are presented in this section. Fish length data were not statistically analyzed, but are graphically summarized for the most common species in Appendix C.

### 4.2.1 Abundance and Species Composition

From April 2009 to September 2014, 132 bottom trawl samples were collected in Naples Bay: 33 samples from each of the four zones. A total of 32,036 individuals from 67 fish taxa and five invertebrate taxa were collected during the study (see Appendix C, Table C-1 and Table C-2 for a full list of taxa). Catch per trawl ranged from zero to 1,951 individuals. The number of different taxa per trawl ranged from zero to 21.

Mojarras (*Eucinostomus* spp.) and anchovies (*Anchoa* spp.) were the most numerous taxa collected, accounting for over 87 percent of the total catch from 2009–2014. They were also two of the most frequently caught taxa: occurring in 92 percent and 50 percent (respectively) of the trawl samples. In general, the other most frequently encountered species were also the most abundant overall (Table 4-4).Twelve taxa were only caught once, and in each case it was a single individual (Table 4-5).

**Table 4-4. Ten most commonly caught and most abundant taxa (grouped to Genus level) in Naples Bay bottom trawls, 2009–2014.**

| Most Common and Most Abundant | | | | | | | |
|---|---|---|---|---|---|---|---|
| | | Occurrence | | | Abundance | | |
| Taxa | Common Name | Rank | Number | % of Total | Rank | Number | % of Total |
| *Eucinostomus* sp.<br>*E. harengulus*<br>*E. gula* | Mojarra | 1 | 122 | 92.4 | 1 | 15025 | 46.9 |
| *Farfantepenaeus duorarum* | Pink Shrimp | 2 | 73 | 55.3 | 5 | 473 | 1.5 |
| *Anchoa* sp.<br>*A. hepsetus*<br>*A. mitchilli* | Anchovies | 3 | 67 | 50.8 | 2 | 12941 | 40.4 |
| *Ariopsis felis* | Hardhead Catfish | 4 | 61 | 46.2 | 9 | 158 | 0.5 |
| *Callinectes sapidus*<br>*C. similis* | Blue Crabs | 5 | 61 | 46.2 | 6 | 410 | 1.3 |
| *Synodus foetens* | Inshore Lizardfish | 6 | 60 | 45.5 | -- | 126 | 0.4 |
| *Lutjanus* sp.<br>*L. synagris*<br>*L. griseus* | Snappers | 7 | 54 | 40.9 | 7 | 197 | 0.6 |
| *Lagodon rhomboides* | Pinfish | 8 | 36 | 27.3 | 3 | 681 | 2.1 |
| *Cynoscion* sp.<br>*C. arenarius*<br>*C. nebulosus* | Seatrout | 9 | 32 | 24.2 | -- | 128 | 0.4 |
| *Prionotus scitulus*<br>*P. tribulus* | Searobins | 10 | 28 | 21.2 | -- | 89 | 0.3 |
| *Bairdiella chrysoura* | Silver Perch | -- | 21 | 15.9 | 10 | 139 | 0.4 |
| *Leiostomus xanthurus* | Spot | -- | 14 | 10.6 | 4 | 688 | 2.1 |
| Unidentified Family Portunidae | Swimming Crabs | -- | 10 | 7.6 | 8 | 184 | 0.6 |

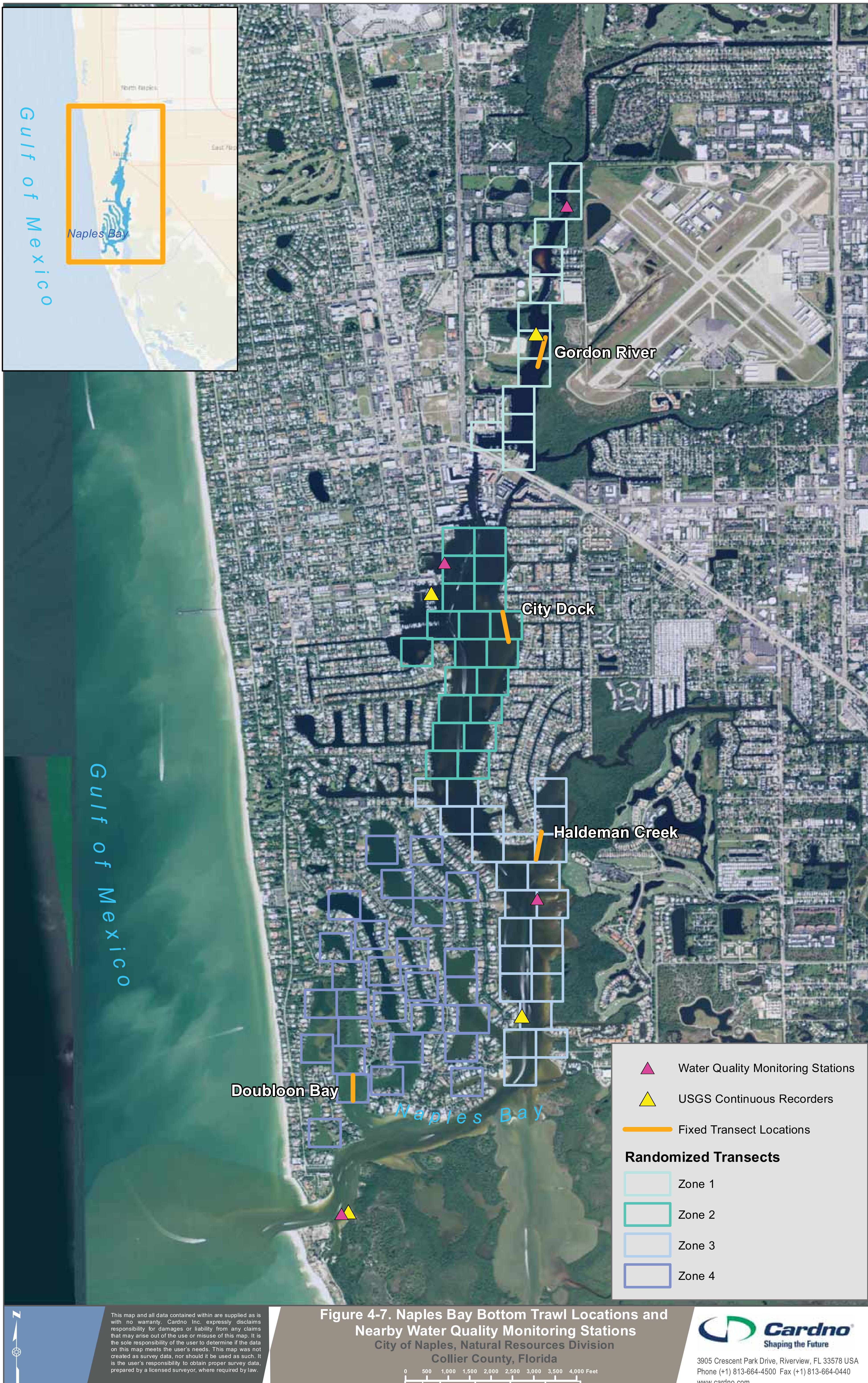




**Figure 4-7. Naples Bay Bottom Trawl Locations and Nearby Water Quality Monitoring Stations**
City of Naples, Natural Resources Division
Collier County, Florida

Cardno
Shaping the Future
3905 Crescent Park Drive, Riverview, FL 33578 USA
Phone (+1) 813-664-4500 Fax (+1) 813-664-0440
www.cardno.com

Date Created: 5/8/2015 Date Revised: 5/8/2015 File Path: Q:\UnitedStates\Florida\Tampa\City_Of_Naples\Water Quality Analysis Project\working\arcmap\Figure4_7_CON_BottomTrawl_20150505.mxd
GIS Analyst: adam.bausch

**Table 4-5. Least commonly caught and least abundant taxa (grouped to Genus level) in Naples Bay bottom trawls, 2009–2014.**

| Least Common and Least Abundant | | | |
|---|---|---|---|
| Taxa | Common Name | Number of Occurrences | Number of Individuals |
| *Acanthostracion quadricornis* | Scrawled Cowfish | 1 | 1 |
| *Albula vulpes* | Bonefish | 1 | 1 |
| *Ctenogobius smaragdus* | Emerald Goby | 1 | 1 |
| *Gobionellus oceanicus* | Highfin Goby | 1 | 1 |
| *Gymnura micrura* | Smooth Butterfly Ray | 1 | 1 |
| *Hypsoblennius hentz* | Feather Blenny | 1 | 1 |
| *Ophichthus gomesii* | Shrimp Eel | 1 | 1 |
| *Scorpaena brasiliensis* | Barbfish | 1 | 1 |
| Unidentified Family Clupeidae | Herrings | 1 | 1 |
| Unidentified Family Gobiidae | Gobies | 1 | 1 |
| Unidentified Family Sciaenidae | Croakers/Drums | 1 | 1 |
| Unidentified Suborder Pleuronectoidei | Flatfishes | 1 | 1 |

### 4.2.2 Diversity Indices

There are no significant differences in Shannon diversity, abundance, and number of taxa among the sampling zones (factorial ANOVA with season and zone; $p < 0.01$). However, there are obvious overall patterns between seasons and over the sampling period (Figure 4-8). Dry season samples have lower abundance and higher diversity than the wet season (one-way ANOVA of season pooled across all zones and years; $F = 420.8$, $p < 0.01$). In addition, all three metrics (in addition to other similar metrics) all appear to have a downward shift sometime in 2011 (Figure 4-8).

The timing of the overall downward shift in diversity, abundance, and richness in Naples Bay is very close to the timing of a change in the sample collection methodology. The sampling methodology changed from fixed transects to a randomized grid design starting with the October 2011 sampling event. Change-point analysis was used (Change-Point Analyzer v2.3 from Taylor Enterprises, Inc., Taylor 2000) to pinpoint the timing of this change in abundance, richness, and diversity graphically (Figure 4-9) and statistically, which allows for a comparison of the timing of the downward shift with respect to the timing of the change in methodology. Change-point analysis works by plotting the cumulative sum (CUSUM) over time of the differences between each observation and the average of all observations; changes in slope of the CUSUM plot indicate that a change in the mean of the observations has occurred (Figure 4-9). Bootstrapping the data is used to determine if the change in the CUSUM plot is statistically significant. The exact estimate of when the change occurred is given by moving the change point back and forth and minimizing the mean square error (MSE) of the two datasets on either side of the proposed change point. Once the change-points are defined (the first sampling event following the detected change), they are given a confidence level and confidence intervals (Taylor 2000).

For Naples Bay, the primary change points and confidence intervals from 2009 to 2014 were identified for five biological metrics (Figure 4-9 and 4-10). The primary change point is predicted just before the sampling methodology change for richness measures, just after the methodology change for abundance, and several months after the methodology change for diversity and evenness (Figure 4-10). In all cases, the metrics noticeably level off or begin to trend downward before the designated change-point and before the change in sampling methodology. This indicates the change in methodology may be coincidental and does not appear to be the cause of the downward trend. Further evidence for the conclusion that sampling methodology changes are not responsible for the observed decline in Naples Bay data is provided in Section 5.2.2.2 with a comparison of similar change points in other nearby estuaries.

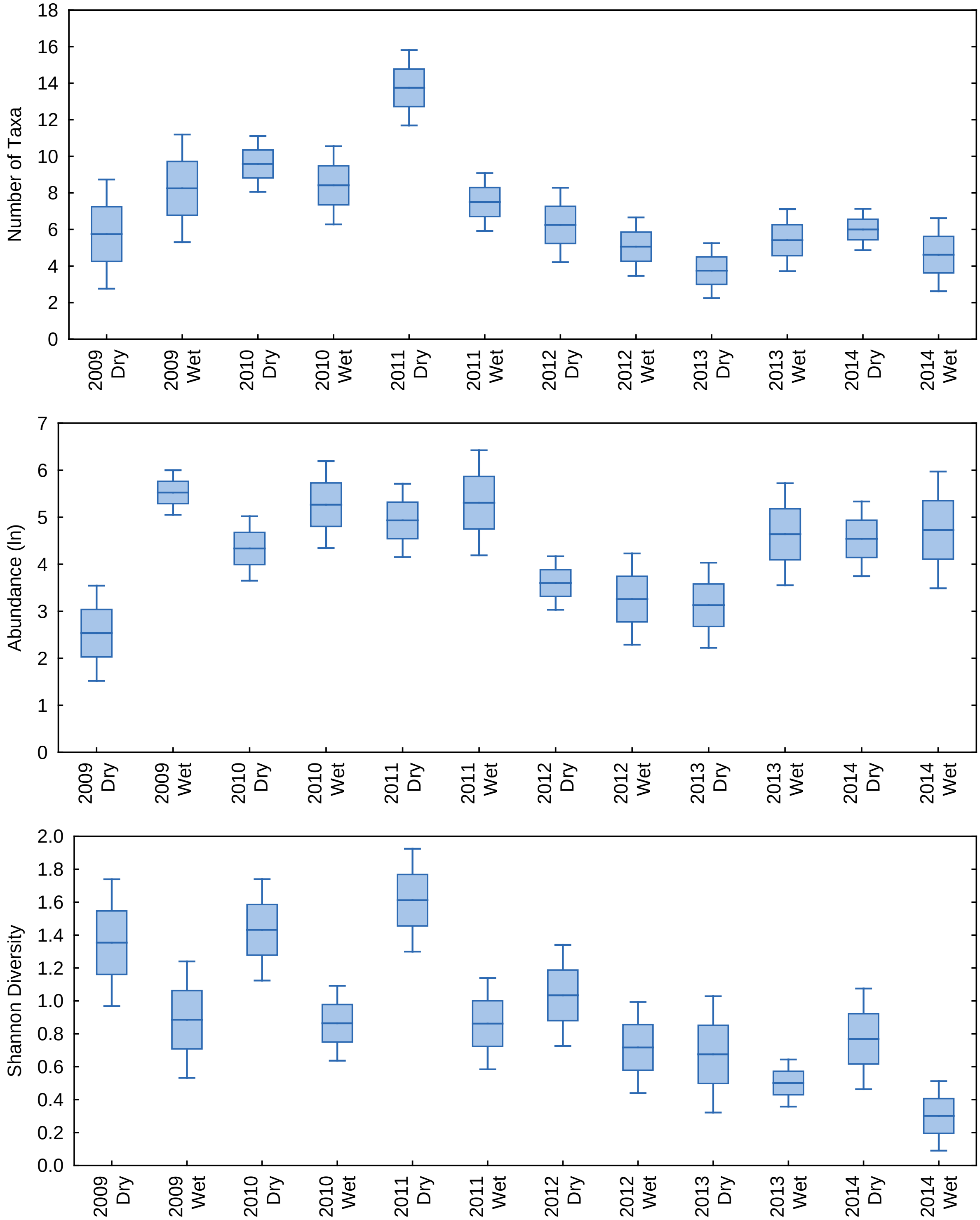


**Figure 4-8.** **Number of taxa, abundance, and diversity by season in Naples Bay bottom trawls, 2009–2014. (Mean, line; ±1SE, box; ±2SE, whiskers).**

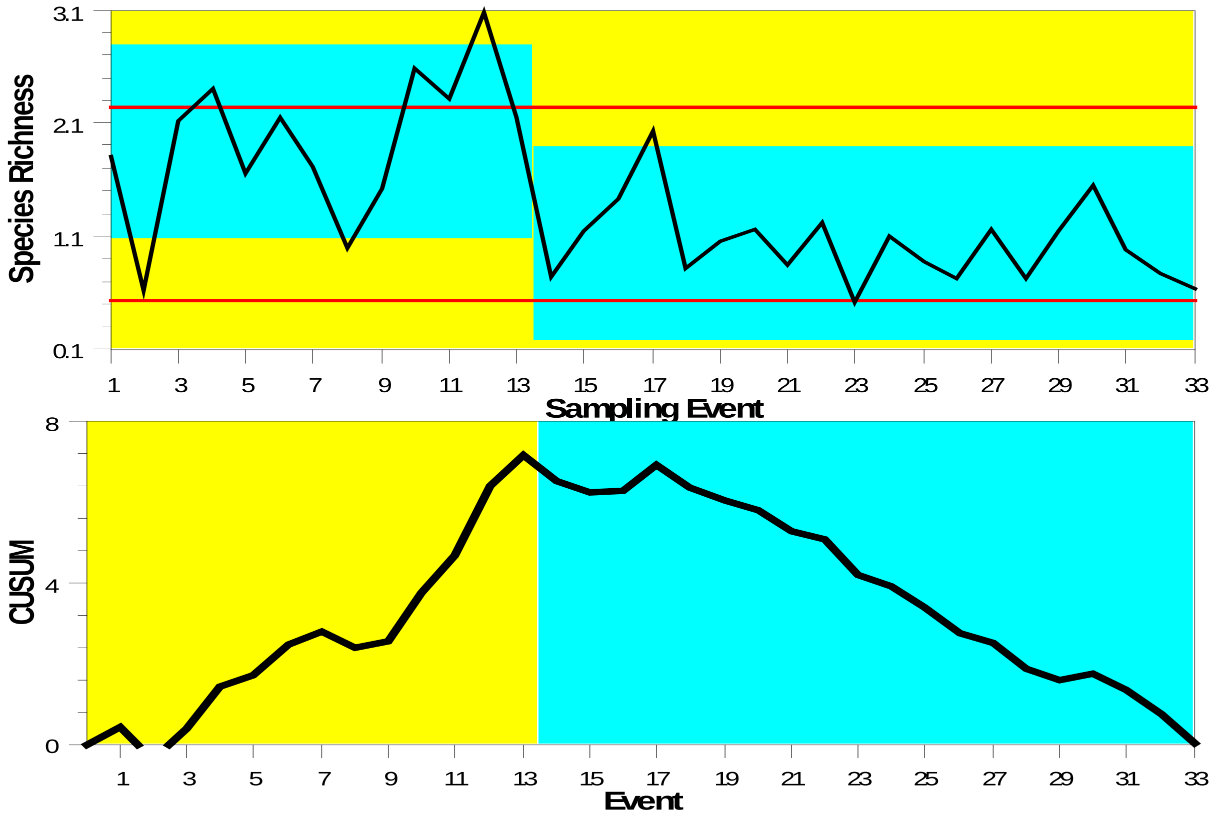


**Figure 4-9.** **Example of a change-point graph from Naples Bay with the time-series plot (top) and CUSUM plot (bottom) indicating a change-point in June 2011.**


**Figure 4-10.** **Change-points (blue lines) and 95 percent confidence intervals (grey dashed lines) for several fish diversity metrics in Naples Bay. The red line represents the date when the sampling methodology changed.**

### 4.2.3 Community Structure

Nonparametric multivariate analyses were used to assess similarity in species composition and abundance (‘community structure’). Analyses were conducted using PRIMER vC6 statistical software (Clarke and Gorley, 2006). Similarity was calculated using taxa abundance data for each sample (unless otherwise noted as pooled). Non-metric multidimensional scaling (MDS) was used for a visual depiction of the community structure relationship among samples. Statistical differences in community structure among or between groups of samples were identified using Analysis of Similarity (ANOSIM), and Similarity Percentage analysis (SIMPER) was used to identify which taxa were representative of dissimilarities among groups. The focus within Naples Bay was on differences among sampling zones between seasons and over time (among years).

*Differences Among Zones*

Species presence/absence data from the entire survey period (2009–2014) were pooled together by zone to give broad level picture of similarity in the species assemblages across zones. Overall, the similarity (Bray-Curtis) between zones ranged from 72 percent to 81.9 percent, with Zone 1 having the lowest similarity to the other zones and the lowest within-group similarity (Table 4-6). More simply put, of all the zones, Zone 1 had the most variable species assemblage from sample to sample and the least in common with other zones. In general, all four zones contain the same taxa (grouped to Genus level or higher); however, there are some taxa that are missing from or unique to a specific zone or are notably more abundant (contributing to ≥ 80 percent of total) in one zone than the others (Table 4-7).

**Table 4-6. Similarity of species assemblage between zones within Naples Bay, 2009–2014. (Bray-Curtis similarity, presence/absence data, pooled by zone).**

| Zone | 1 | 2 | 3 | 4 |
|---|---|---|---|---|
| 1 | | | | |
| 2 | 72.00 | | | |
| 3 | 72.46 | 76.32 | | |
| 4 | 73.68 | 81.93 | 80.52 | |

The one-way ANOSIM test for differences among samples (unpooled data, aggregated to Genus level, log (x+1) transformed, Bray-Curtis Similarity) from different zones shows that there are significant but very weak differences (ANOSIM Global $R = 0.07$, $p = 0.001$)) among zones: Zones 1, 2 and 3 are all different from one another, but Zone 4 is not different from any other. An MDS plot of these data does not show good separation among the zones (Figure 4-11), but does show that samples from Zone 1 are more widely scattered than those from Zone 2 or 3.

**Table 4-7. Taxa (grouped to Genus level or above) that are unique to, absent from, or most commonly associated with a specific zone in Naples Bay.**

| Zone | Genus | Species Common Name | Total Number |
|---|---|---|---|
| **Unique to Zone** | | | |
| 1 | *Ctenogobius* | Emerald goby | 1 |
| | *Sciaenops* | Red drum | 56 |
| | *Trinectes* | Hogchoker | 3 |
| | Clupeidae (Family) | Herrings | 1 |
| 2 | *Albula* | Bonefish | 1 |
| | *Ophichthus* | Shrimp eel | 1 |
| | *Selene* | Lookdown | 3 |
| | Gobiidae (Family) | Gobies | 1 |
| 3 | *Gymnura* | Smooth butterfly ray | 1 |
| | *Acanthostracion* | Scrawled cowfish | 1 |
| | Sciaenidae (Family) | Croakers/Drums | 1 |
| 4 | *Scorpaena* | Barbfish | 1 |
| | *Gobionellus* | Highfin goby | 1 |
| | *Hypsoblennius* | Feather blenny | 1 |
| | Pleuronectoidei (Suborder) | Flatfishes | 1 |
| **Absent from Zone** | | | |
| **Zone** | **Genus** | **Species Common Name** | **Total Number in Other Zones** |
| 1 | *Etropus* | Fringed flounder | 115 |
| | *Harengula* | Scaled sardine | 30 |
| | *Ogcocephalus* | Polka dot batfish | 47 |
| | *Opisthonema* | Atlantic thread herring | 9 |
| | Portunidae (Family) | Swimming crabs | 169 |
| 2 | *Chloroscombrus* | Atlantic bumper | 13 |
| | *Menticirrhus* | Kingfishes | 8 |
| 3 | *Achirus* | Lined sole | 80 |
| | *Opsanus* | Gulf toadfish | 15 |
| 4 | none | -- | -- |
| **More Common in One Zone** | | | |
| **Zone** | **Genus** | **Species Common Name** | **Number in Zone (Total Number)** |
| 1 | *Achirus* | Lined sole | 75 (80) |
| | *Chaetodipterus* | Atlantic spadefish | 4 (5) |
| | *Gobiosoma* | Code goby | 8 (9) |
| 2 | none | -- | -- |
| 3 | *Leiostomus* | Spot | 571 (688) |
| | *Ogcocephalus* | Polka dot batfish | 38 (47) |
| 4 | Portunidae (Family) | Swimming crabs | 159 (169) |

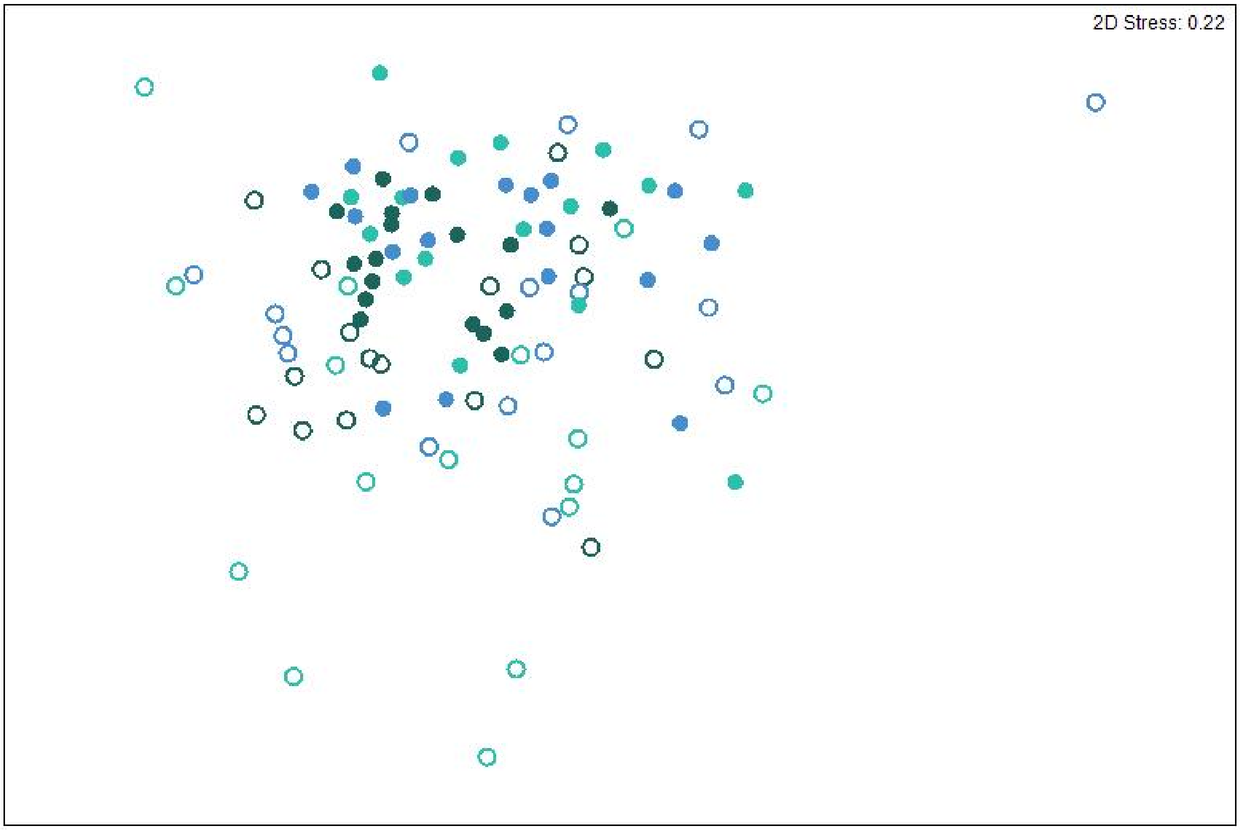


**Figure 4-11. nMDS ordination plot (Bray-Curtis similarity, log(x+1) transformed data) of the fish community structure within Naples Bay, 2009–2014. Light green = Zone 1; Dark green = Zone 2; Light blue = Zone 3. Open circles = dry season; Closed circles = wet season. Zone 4 is not shown for clarity.**

*Seasonal Effects*

When the points on the MDS plot are coded by season, a separation between wet and dry season samples is evident in the pattern (Figure 4-11). Considering season and zone together in a two-way ANOSIM test shows that there is a weak but significant difference between seasons (Global R = 0.117, p = 0.001) and the differences among the zones are a little weaker when season is taken into account (Global R = 0.061, p = 0.001). In addition to the pairwise differences noted in the one-way test, Zones 1 and 4 are significantly different from one another when season is a factor. A deeper look into differences among zones within each season reveals season-specific relationships between zones that are not evident in the one-way test: Zones 1 and 2 are only significantly different in the wet season, Zones 1 and 4 are only different in the dry season, and Zones 2 and 3 are only different in the dry season.

SIMPER analysis was used to quantify the average similarity among samples within season or zone, the average dissimilarity between seasons or zones, and which taxa contribute most to the similarity/dissimilarity. As noted above, most of the taxa found in Naples Bay are ubiquitous rather than limited to a specific zone. The same is true across seasons: there are few seasonal differences in which species are present. Thus, most of the similarities within and differences between seasons and zones is the result of differences in how species are assembled (which species co-occur) and differences in their overall abundance. The SIMPER results show that, for the most part, the same species are responsible for similarity within groups and dissimilarity between groups. Mojarras (*Eucinostomus* spp.) and anchovies (*Anchoa* spp.) are the largest contributors to dissimilarity in all pairwise comparisons, followed by pink shrimp (*Farfantepenaeus duorarum*), blue crabs (*Callinectes* spp.), pinfish (*Lagodon rhomboides*), and snappers (*Lutjanus* spp.) (Appendix C, Tables C-3 & C-4). Within-group similarity was lower in the dry season than the wet season and lower in Zone 1 than the other zones (Table 4-8); this indicates more variation in community structure among samples in those groups.

**Table 4-8. Average within group similarity (bold, italics) and between group dissimilarity for season and zones within Naples Bay, 2009–2014. (Bray-Curtis similarity, Genus-level, log(x+1) transformed data).**

| Average Similarity & Dissimilarity | | | | |
|---|---|---|---|---|
| **Between Seasons** | | | | |
| **Season** | Wet | Dry | | |
| Wet | ***40.16*** | | | |
| Dry | 67.31 | ***31.62*** | | |
| **Between Zones** | | | | |
| **Zone** | 1 | 2 | 3 | 4 |
| 1 | ***29.05*** | | | |
| 2 | 65.91 | ***44.42*** | | |
| 3 | 70.79 | 61.79 | ***35.32*** | |
| 4 | 69.55 | 60.78 | 64.85 | ***35.84*** |

*Interannual Patterns*

There were significant differences (ANOSIM, Global R = 0.108, p = 0.001) in community structure among years (Figure 4-12). Pairwise tests show two separate groups: 2010–2011 and 2012–2014. (2009 was excluded from this analysis because it did not have enough sampling events to fully represent both seasons in the year). This division between year groups occurs at about the same time frame as the change-points for the univariate diversity metrics (Section 4.2.2).

Average similarity within years was generally low (< 40 percent) and dissimilarity between years was generally as high as dissimilarity within a year (Table 4-9). The same species that are responsible for seasonal and zone differences account for the differences between the two year groups (2009–2010 vs. 2012–2014): mojarras (*Eucinostomus* spp.) and anchovies (*Anchoa* spp.) make the highest contributions to dissimilarity, followed by pink shrimp (*Farfantepenaeus duorarum*), blue crabs (*Callinectes* spp.), pinfish (*Lagodon rhomboides*), and snappers (*Lutjanus* spp.) (Appendix C, Table C-5). Notably, almost every species that contributes to the dissimilarity between the two groups has a lower average abundance during the 2012–2014 time frame.

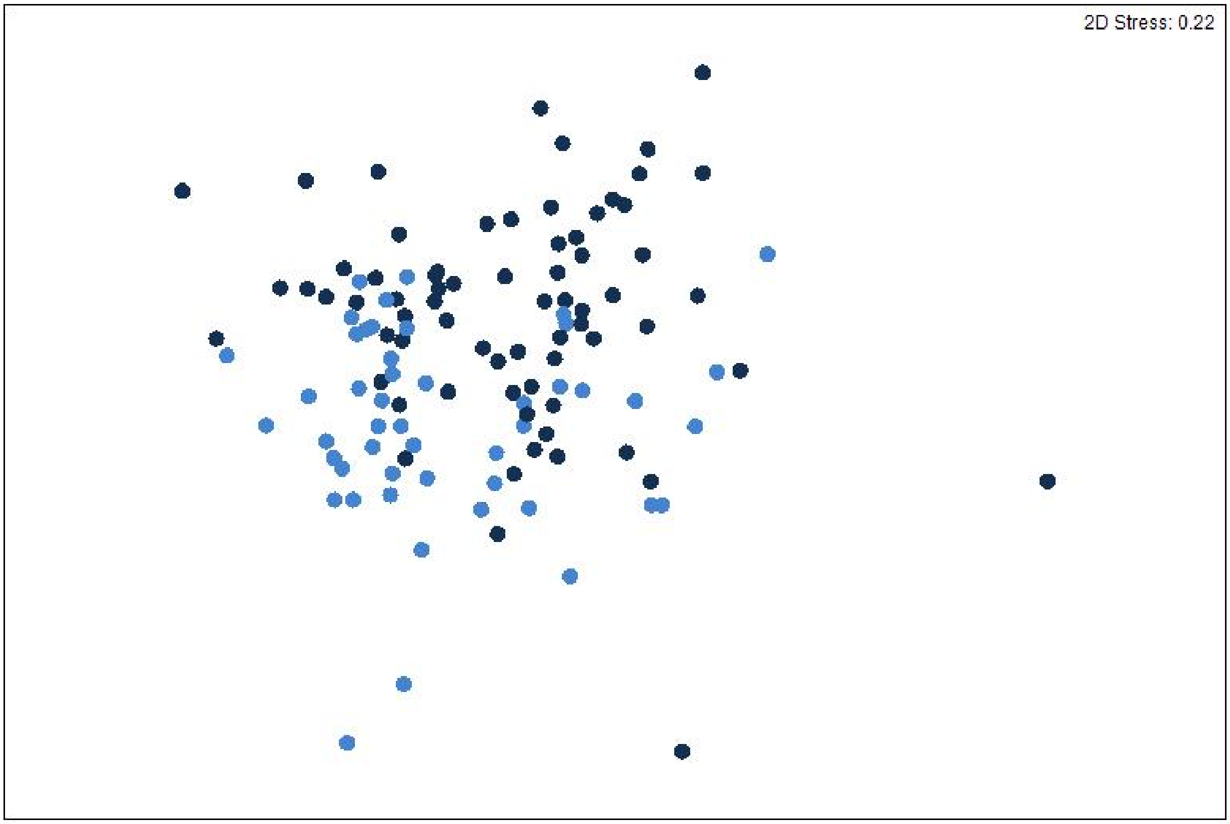


**Figure 4-12.** nMDS ordination plot (Bray-Curtis similarity, log(x+1) transformed data) of the fish community structure within Naples Bay, 2010–2014. Samples from 2010 and 2011 are represented by light circles and samples from 2012–2014 are represented by dark circles.

**Table 4-9.** Average within group similarity (bold, italics) and between group dissimilarity for years within Naples Bay, 2010–2014. (Bray-Curtis similarity, Genus-level, log(x+1) transformed data).

| Year | 2010 | 2011 | 2012 | 2013 | 2014 |
|---|---|---|---|---|---|
| 2010 | ***34.95*** | | | | |
| 2011 | 63.31 | ***38.05*** | | | |
| 2012 | 70.21 | 69.46 | ***32.39*** | | |
| 2013 | 71.09 | 69.31 | 67.91 | ***31.93*** | |
| 2014 | 67.37 | 66.51 | 65.83 | 67.20 | ***37.76*** |

# 5 Water Quality & Biological Interactions in Naples Bay

This section builds upon the two previous sections to attempt to identify potential causal links between the observed trends in water quality with the observed trends in biological communities in Naples Bay. While all water quality trends are considered here, we focus on attempting to identify the potential effects of salinity and GGC freshwater flow on biology because freshwater flow and salinity stress have been identified as the primary pollutants in Naples Bay (City of Naples 2010, Laakkonen 2014, Schmid *et al.* 2005, SFWMD 2007, FDEP 2010).

The following section describes the results of statistical and data analyses conducted to identify links between water quality and biological parameters of concern. Additionally, comparisons to other southwest Florida estuaries are included to determine if observed changes and potential effects of water quality are unique to Naples Bay or may have a regional connection.

## 5.1 Seagrass and Water Quality

Identifying the effects of water quality on a seagrass community is a complex undertaking since a many factors work in series or parallel to shape how and where a seagrass community will grow or thrive. Biological, chemical, and physical factors all play a role in shaping a seagrass community (Figure 5-1). This is especially true in Naples Bay. Schmid *et al.* (2005) reported an approximate 80 percent decrease in seagrass from the 1950s to 2003. The decline is presumed to be linked to a combination of channel dredging, urban buildup of the shoreline areas, and freshwater inputs, mostly from the GGC (Schmid *et al.* 2005).

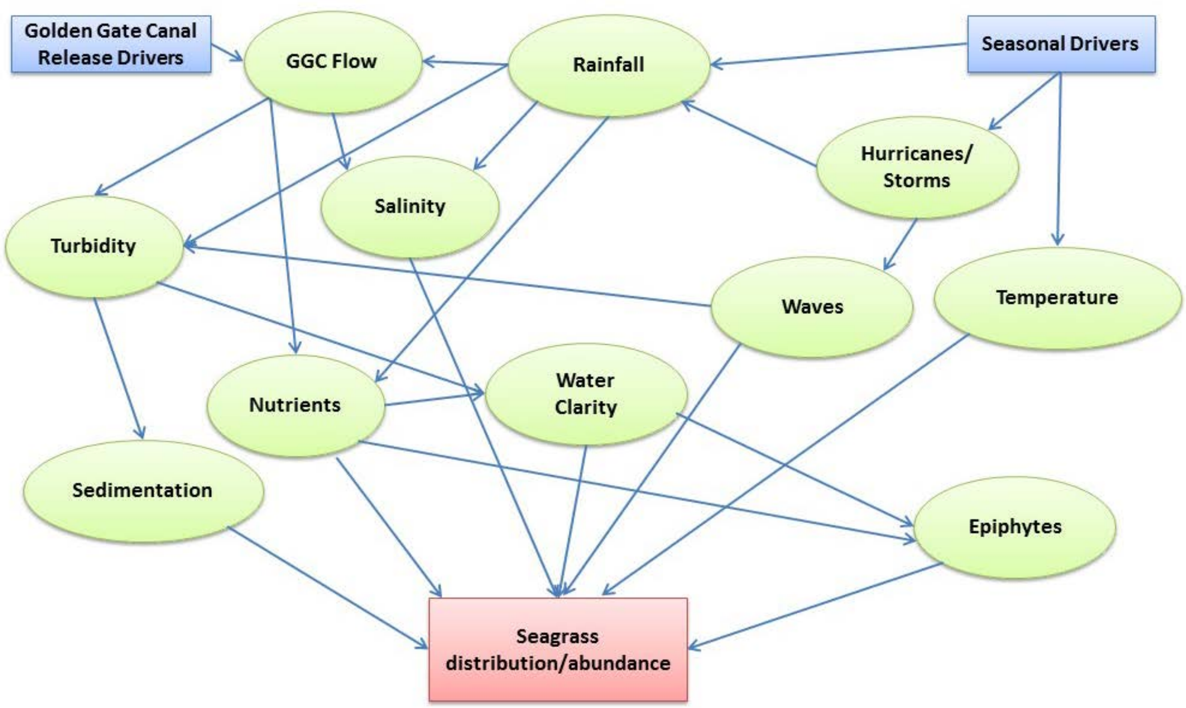


**Figure 5-1. Conceptual model of potential seagrass drivers in Naples Bay.**

In an ideal world, scientists would have perfect information for describing the casual relationships that underlay the issues of concern, such as the response of seagrasses to water quality in Naples Bay. But, in the real world examination of casual relationships is confounded by complex interactions among not only the causative effect of interest (in this case water quality), but the natural influences on seagrass biomass on a wide range of temporal and spatial scales (such as water clarity and physical stressors). The City's monitoring programs have generated a great deal of information that can be used in a decision analytic framework to evaluate the effect of water quality on seagrass. Examination of the strength of these sources of evidence can provide resource managers with a rigorous approach to teasing out the actual impact of water quality on seagrass biomass, and observed changes associated with other factors such as tidal flux, temperature, and salinity changes over space and time.

A Bayesian network (BN) is a tool for linking multiple lines of information, and examining the strength of complex environmental and effects-based relationships. A BN can be thought of as a graphical model with a series of nodes linked by arrows, where the arrows in a BN represent probabilities. The arrows indicate causal linkages among the nodes, and the nodes denote important system attributes. Each node is characterized by probabilities or probabilistic mathematical expressions that represent knowledge about these system attributes. The mathematical expressions may be 1) mechanistic descriptions such as chemical reaction kinetics, 2) empirical relationships such as linear regression models, or 3) relationships derived from expert judgment, depending on how much information we have about the relationships characterizing a particular node. The possible outcomes at each node are expressed probabilistically; thus a Bayes net is a set of conditional probabilities describing a set of likely system responses. The ability to incorporate mechanistic, empirical, and judgmental information makes the BN approach extremely flexible and facilitates an extension to non-traditional model endpoints (e.g., seagrass biomass) of concern. A full description of a BN network and the model set up and implementation for Naples Bay seagrass can be found in Appendix B, only the results are presented here.

Data are not available to represent all of the nodes in the conceptual model shown in Figure 5-1. However, we combined available data and believe the information is suitable for a draft assessment of the Naples Bay ecosystem, at least as a generalized approach. The chosen model output combined available data for the rainfall, flow, turbidity, salinity, total nitrogen, chlorophyll *a*, and seagrass biomass nodes (Appendix B). The results of the Naples Bay seagrass BN indicate, not surprisingly, that seagrass biomass is most influenced by a combination of salinity, turbidity, and chlorophyll *a*. In the BN model, the greatest possibility of a "good" seagrass state coincides with salinities at the higher end of normal estuarine conditions, low turbidity, and low chlorophyll *a*. The BN model predicts that because the effect of flow (GGC flow) on salinity is low in the southern portion of the Bay, changes in GGC flow are not expected to have a great deal of influence on seagrass biomass in the southern portion of the Bay. However, the predictive ability of this model is limited to the data that it includes. If historical data on seagrass, water quantity, and water quality was available for Naples Bay all the way back to 1950, then the relationships between each of the variables within the BN could change based on a wider range of physical conditions and biological responses prior to increased urbanization and freshwater stress.

While this model can be generally useful, we are careful not to over interpret the results because limited information is available to characterize all possible effects on seagrass in Naples Bay, which leads to uncertainty in the results. For example, the current seagrass beds in the southern portion of the Bay may be highly influenced by boat traffic, wave action, and sedimentation, about which little to no information are currently available. In the vicinity of the seagrass transects, a channel and marina were built during the monitoring period which can contribute to increased disturbance. Additionally, data concerning light attenuation and water clarity are lacking for the location of the current seagrass beds that would be valuable in determining causal links for important measures of water quality. Furthermore, this model cannot provide any information regarding the suitability of seagrass restoration for any other portions of the Bay, as it is restricted to the available information regarding the current seagrass in southern Naples Bay. This exercise provides an example of the type of analyses that can be used to link several potential factors that influence seagrass in Naples Bay and tease out which ones may play a more dominant role. The current application of the model for Naples Bay should be viewed as generally informative, but should not be used to draw any specific conclusions regarding causal factors of seagrass biomass or management decisions for seagrass restoration.

Section 4.1 describes the appearance of a declining trend in seagrass density over the 2012–2014 time period. This apparent declining trend is somewhat complicated by changes in sampling methodology that reduced sampling frequency and shifted sampling toward the end of the survey season when seagrass densities appear to be lower. It is difficult to separate a conclusion of actual decline in seagrass density over 2012–2014 from the possible seasonal sampling bias. While no definitive links could be drawn at this time, the decline in seagrass density during the summer months also coincides with the time period of substantial nutrient and solids loading from the GGC (see Section 3.1.1). On average, the GGC delivers approximately 90 tons of nitrogen and 355 tons of solids to Naples Bay each year, with over 90 percent of it delivered during the wet season (June–November). Relative to bay volume, the loadings from the GGC to Naples Bay are many times greater than total loadings to Tampa Bay, which has exhibited significant seagrass recovery. While further investigation into the loading events is warranted to identify concrete causal links, the available information is sufficient to conclude that loadings likely play a significant role in observed seagrass trends and should be an essential consideration for management and restoration activities.

Additional study on factors connected with water clarity and light attenuation are needed to identify the connections between turbidity, suspended solids loadings, localized perturbations from wave action and/or boat traffic, and seagrass growth and density in Naples Bay.

## 5.2 Fish and Water Quality

### 5.2.1 Within Naples Bay

Flow from the GGC was considered to be the most likely potential driver of water quality and biological changes in Naples Bay. Seasonal water quality changes related to GGC flow differ among the fish sampling zones within the Bay. Even though the current sampling program is designed to sample typical wet and dry seasons equally, it does not capture low/no flow and high flow conditions with the same frequency (Figure 5-2).

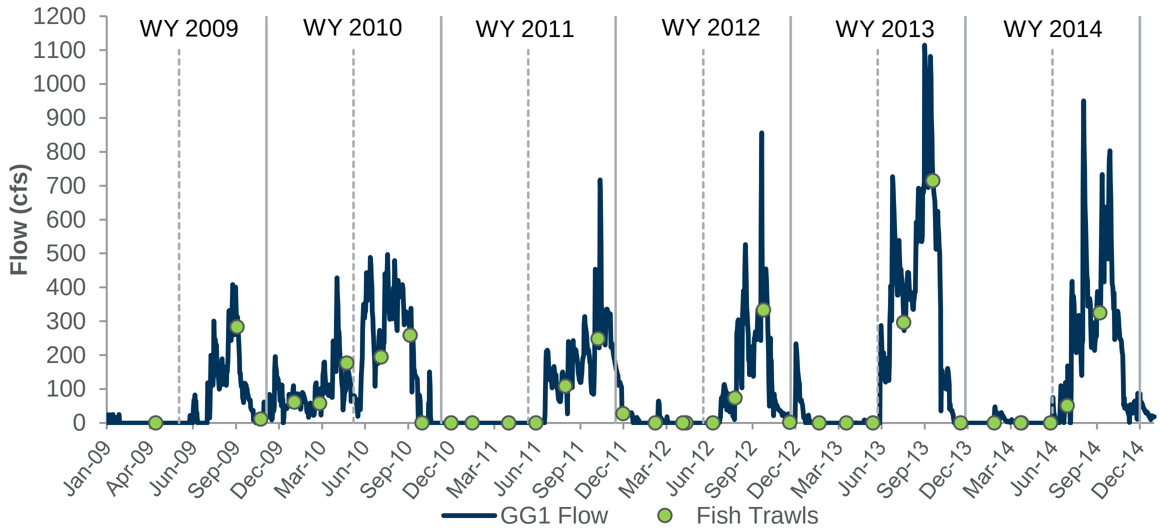


**Figure 5-2. Fish sampling events and Golden Gate Canal Flow by season.**

Several different methods were used to look for links between the fish community and water quality parameters. Initial, exploratory analyses were conducted using water quality measurements collected during trawling sampling events: bottom salinity, bottom temperature, and bottom DO. Several univariate diversity metrics such as number of taxa, species richness, abundance, and Shannon diversity were plotted against each of the three water quality parameters to see if correlations between the variables existed (see Appendix C, Figure C-1 for examples). No relationships were found over time or within or among zones.

The next step was to construct a water quality dataset, perform a Principle Components Analysis (PCA) on the data, and plot the univariate diversity metrics against the PC axis scores. Two different water quality datasets were constructed: one using the three variables measured during trawling events and one constructed from water quality variables from monitoring stations in the bay. The second dataset included measures of flow from the Golden Gate Canal, rainfall, salinity, water temperature, turbidity, TN, TP, chlorophyll a, and DO; data were from the 30 day period preceding each sampling event. Water quality variables were appropriately transformed and normalized before analysis. No relationships were found when the diversity metrics were plotted against the PC scores (see Appendix C, Figure C-2 for examples).

The third step was to use the BIO-ENV procedure in Primer. BIO-ENV selects the subset of water quality variables that maximizes correlation between the biotic and abiotic similarity matrices. Neither water quality dataset produced any variables that were well correlated to the fish community structure. The strongest correlation (Spearman coefficient = 0.1) was for temperature. Adding additional water quality variables did not increase the correlation. Limiting the biotic dataset to only the highest contributing taxa did not increase the correlation. Limiting the biotic dataset to only Naples Bay Zones 1, 2, and 3 resulted in a slightly increased correlation (Spearman coefficient = 0.135) for temperature and TN together.

The final comparisons between water quality variables and the fish community structure were made using the RELATE routine in Primer. The RELATE routine compares two resemblance matrices and measures how closely related the two sets of multivariate data are. No significant correlations between the water quality matrix and biological matrix were found, even if data were grouped by season and/or year or limited to only the highest contributing taxa. However, when a model matrix was constructed using latitude or distance to the where the Golden Gate Canal flow enters the Bay (and, thus, distance from Gordon Pass), there is a small but significant correlation ($\rho = 0.19$) between biological distance and geographic distance. (This analysis does not include samples in Zone 4 because of the difficulty in calculating over-water distances from the mouth of the Bay or the Canal). While this result does not help pinpoint which, if any water quality variables are related to the fish community structure, it does show that geographic position in the Bay (and relative to the Pass and Canal) is worth exploring further.

### 5.2.2 Regional Comparison

Even though no correlations were found between specific water quality variables and fish diversity metrics and community structure, there were general seasonal and annual patterns within the Naples Bay biological dataset. Comparing the communities and patterns in Naples Bay to other bays in the region can help determine whether those patterns are unique to Naples Bay or part of larger, regional environmental patterns. In addition, a bay to bay comparison will show how the Naples Bay community compares to other bays with different levels of human impact. Naples Bay is unique in southwest Florida in that it is the only estuary that experiences the extreme freshwater inflow from the GGC and resulting effect on salinity. The GGC effectively increases the natural Naples Bay drainage area by 10 fold, a condition not experienced in the other bays.

Five other bays in Southwest Florida have fish monitoring programs that use the same methodology employed in Naples Bay (NB) and were monitored during the same time period: Moorings Bay (MB), Rookery Bay (RB), Pumpkin Bay (PB), Faka Union Bay (FU), and Fakahatchee Bay (FH) (Figure 5-3). In general, Rookery, Pumpkin, Faka Union, and Fakahatchee Bays have considerably less local coastal development than Naples Bay. However, in terms of potential impacts to bay hydrology, Faka Union Bay receives large amounts of fresh water canal flow from developed land, Rookery Bay experiences an altered hydrological flow pattern based on flood control, while Fakahatchee and Pumpkin Bays have less direct anthropogenic impacts to hydrology relative to the other bays in the comparison. The monitoring program in Rookery Bay ended in 2010. Data collection in Moorings Bay does not occur as frequently as it does in Naples Bay. Some comparisons below will be limited to only the bays with comparable sampling dates or frequency.

#### *5.2.2.1 Abundance and Species Composition*

Overall, more taxa (grouped to Genus level or above) were caught in Naples Bay than in the other bays in this comparison (Table 5-1). However, higher species richness is most likely the result of comparing unequal numbers of trawls from each bay: increased trawling effort increases the likelihood of encountering rare species. Naples Bay also had higher catch per trawl than all other bays, with the exception of Moorings Bay. Higher overall catch numbers are likely linked to the predominance of small, schooling fish in Naples Bay, where just two taxa groups, mojarras (*Eucinostomus* spp.) and anchovies (*Anchoa* spp.) account for over 85 percent of the total catch. Both taxa are also abundant in other regional bays, but, except for Moorings Bay, do not contribute as highly to overall abundance (Table 5-2). In all other bays, the top 85 percent of total catch is also made up of pink shrimp (*Farfantepenaeus duorarum*) and pinfish (*Lagodon rhomboides*).While it appears that numerical dominance of mojarras and anchovies is a characteristic of the more developed bays like Moorings Bay and Naples Bay, it is not clear what is driving that pattern; habitat variables, water quality, hydrology, and bay morphology could all be contributing factors.

**Table 5-1. Summary of total abundance and total number of taxa by bay.**

| Summary Metric | NB | MB | RB | FH | FU | PB |
|---|---|---|---|---|---|---|
| Total Number of Individuals | 32036 | 23048 | 3472 | 20928 | 15278 | 21780 |
| Total Number of Taxa | 56 | 53 | 40 | 45 | 44 | 46 |
| Total Number of Trawls | 132 | 64 | 48 | 100 | 100 | 100 |
| Average Catch per Trawl | 242.7 | 360.1 | 72.3 | 209.3 | 152.8 | 217.8 |

**Table 5-2. Taxa contributing to the top 85 percent of abundance in each bay.**

| | Percentage of Total Abundance* | | | | | |
|---|---|---|---|---|---|---|
| Taxon | NB | MB | RB | FH | FU | PB |
| *Anchoa* spp. | 40.4 | 14.9 | 12.3 | 40.5 | 10.4 | 20.2 |
| *Eucinostomus* spp. | 46.9 | 77.3 | 47.4 | 23.3 | 41.9 | 45.5 |
| **Total** | **87.3** | **92.2** | **59.7** | **63.8** | **52.3** | **65.7** |
| *Lagodon rhomboides* | (2.1) | (0.4) | 13.4 | 5.1 | 14.6 | 11.4 |
| *Farfantepenaeus duorarum* | (1.5) | (0.6) | 8.6 | 17.8 | 15.4 | 9.7 |
| *Orthopristis chrysoptera* | (0.2) | (0.07) | 3.7 | (0.6) | (0.2) | (0.5) |
| *Symphurus plaguisa* | (0.2) | (0.06) | (1.0) | (2.2) | 5.4 | (2.4) |

*Numbers in parentheses are shown for comparison but do not contribute to the top 85% of individuals for that particular bay.

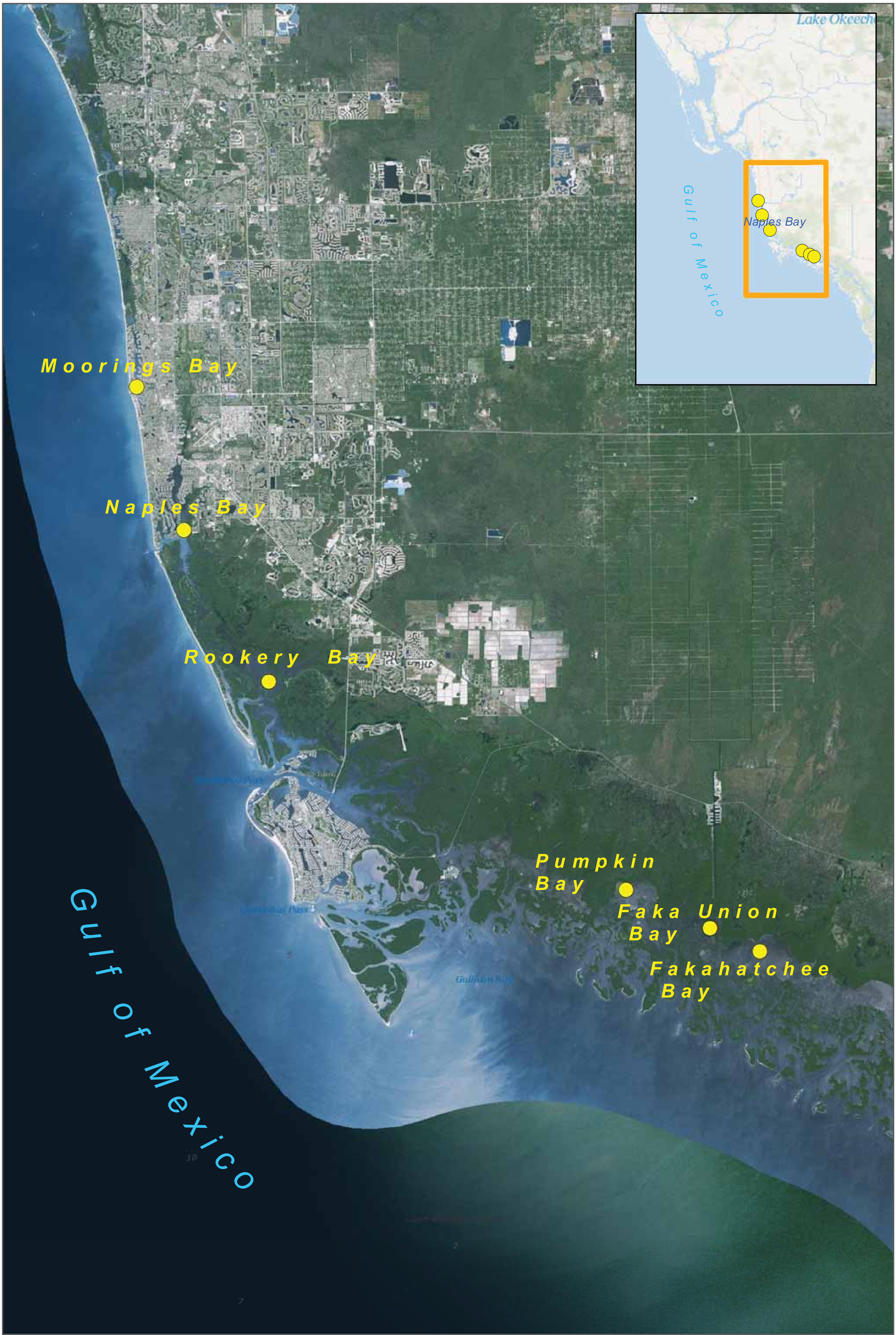




**Figure 5-3. Southwest Florida Bays with Fish Monitoring Programs Comparable to Naples Bay**
City of Naples, Natural Resources Division
Collier County, Florida

0 1 2 3 4 5 6 Miles
0 1 2 3 4 5 6 7 8 9 10 Kilometers

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Date Created: 5/8/2015 Date Revised: 5/8/2015 File Path: Q:\UnitedStates\Florida\Tampa\City_Of_Naples\Water Quality Analysis Project\working\arcmap\Figure5_3_CON_Bays_20150507.mxd
GIS Analyst: adam.bausch

#### 5.2.2.2 Diversity Indices

When the data for each bay are grouped by year and season, the same general patterns in diversity, richness, and abundance across seasons and years seen in Naples Bay are evident in other bays. In general, dry season samples have lower abundance and higher diversity than the preceding wet season. In addition, the downward shift in 2011 seen in Naples Bay is also apparent in other bays (Figure 5-4). Factorial ANOVA tests (with season and bay as factors) with post-hoc pairwise comparisons (Bonferroni tests, alpha = 0.05) were performed to look for differences in diversity, richness and abundance between Naples Bay and the other bays. Abundance in Naples Bay does not differ from any other bay ($p > 0.05$). Number of taxa is lower in Naples Bay than in Fakahatchee and Pumpkin Bays ($F = 9.32$, $p < 0.01$) and Shannon diversity is lower in Naples Bay than in Faka Union, Fakahatchee, and Pumpkin Bays ($F = 9.28$, $p < 0.01$).

Using change-point analysis, a major step change was detected in 2011 in Naples Bay (Section 4.2.2). The difference in diversity between Naples Bay and other bays only occurs after the 2011 change-point in Naples Bay (Factorial ANOVA on season and bay, 2009–2011, $F = 7.78$, $p < 0.01$); before the change-point, diversity in Naples Bay was not different from the other bays (Factorial ANOVA on season and bay, 2012–2014, $p > 0.05$). Change-point analysis univariate community metrics from the other bays shows that the timing of the overall downward shift in diversity, abundance, and richness in Faka Union and Fakahatchee Bays is very close to the timing of the change in Naples Bay (Figure 5-5). Pumpkin Bay and Moorings Bay did not have change-points at the same time; for Moorings Bay, this might be due to lower sampling frequency. The aligned timing of change across bays indicates that the driver for the change may be regional rather than localized to Naples Bay. Environmental factors such as temperature or rainfall might have impacted Naples Bay, Faka Union, and Fakahatchee Bays all at the same time. However, diversity is lower in Naples Bay than the other bays after the change point, suggesting that local factors within Naples Bay are also at work.

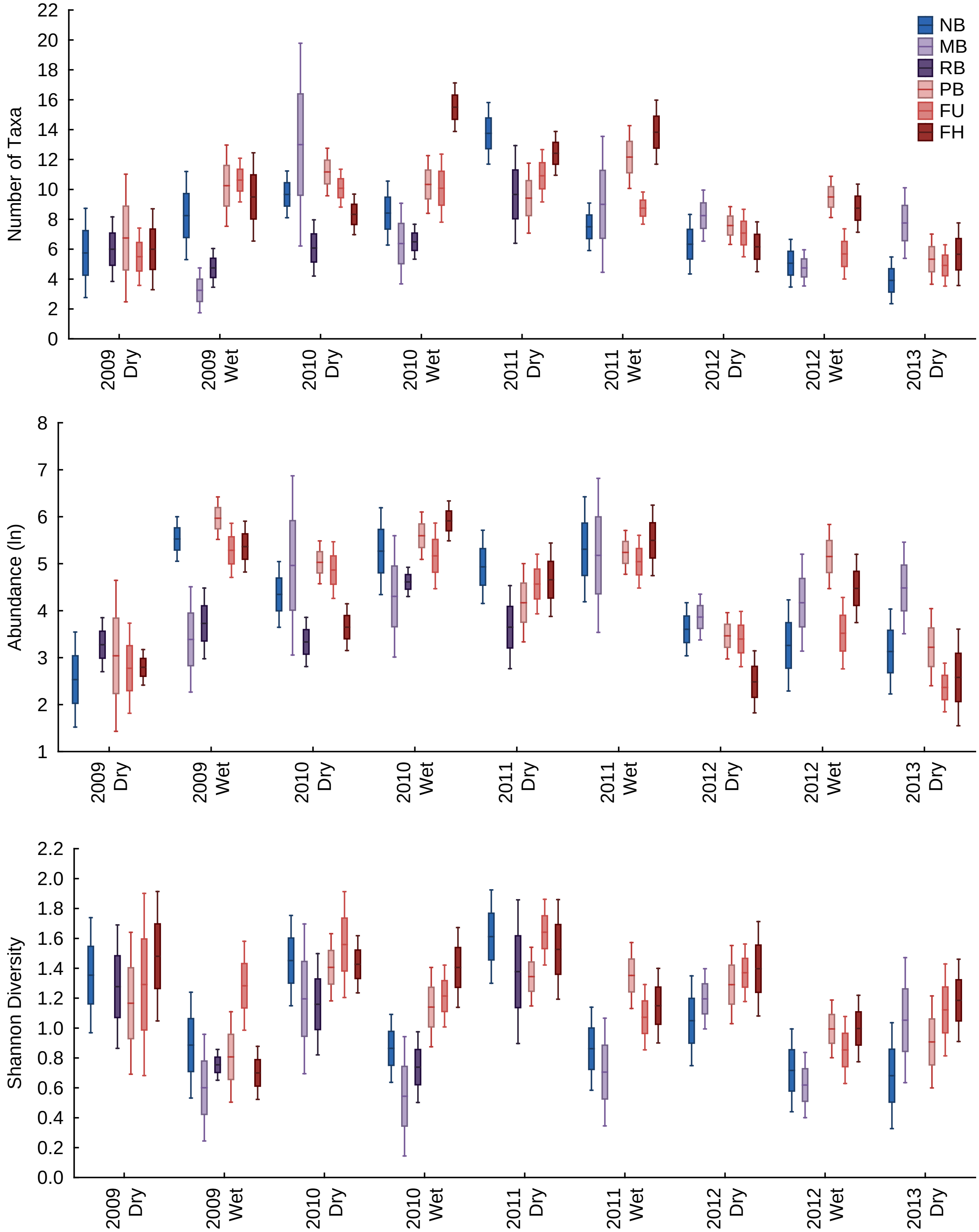


**Figure 5-4.** **Number of taxa, diversity, and abundance by season in bottom trawls Southwest Florida Bays, 2009–2013. (Mean, line; ± 1SE, box; ± 2SE, whiskers).**

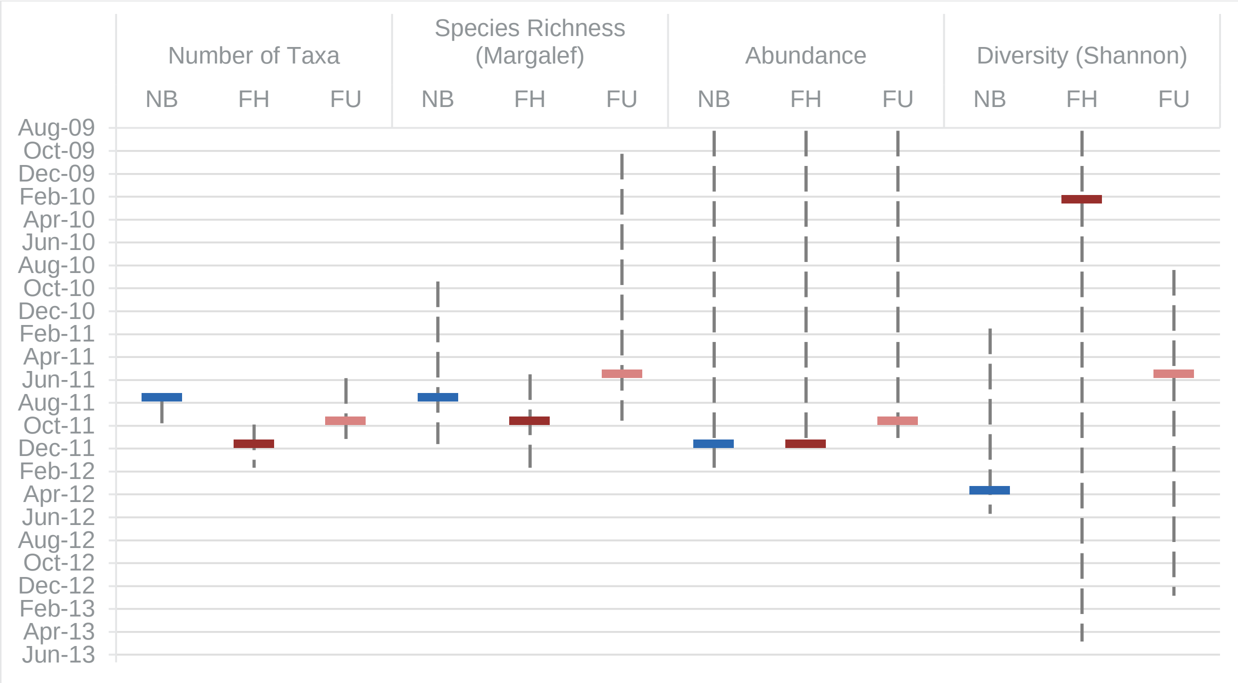


**Figure 5.5** **Change-points (colored lines) and 95 percent confidence intervals (grey dashed lines) for several fish diversity metrics in Naples, Fakahatchee, and Faka Union Bays. Pumpkin Bay did not show any change points for these indices.**

### *5.2.2.3 Community Structure*

Species presence/absence data from the entire survey period (2009–2014) were pooled together by bay to give broad level picture of similarity in the species assemblages across bays. Overall, the similarity (Bray-Curtis) between bays was high, ranging from 77 percent to 92 percent, (Table 5-3). Several taxa were not captured in Naples Bay but were present in other, less developed bays (Table 5-4). All taxa absent from Naples Bay were only found in relatively low abundances in the other bays.

**Table 5-3.** **Species assemblage similarity (Bray-Curtis, presence/absence, pooled by bay) between Bays.**

| Bay | MB | NB | RB | PB | FU | FH |
|---|---|---|---|---|---|---|
| MB | | | | | | |
| NB | 86.5 | | | | | |
| RB | 82.1 | 77.6 | | | | |
| PB | 80.0 | 77.7 | 80.5 | | | |
| FU | 76.8 | 80.4 | 81.4 | 83.5 | | |
| FH | 80.8 | 78.4 | 81.4 | 92.3 | 84.4 | |

**Table 5-4. Taxa (grouped to Genus level or above) that are absent from or unique to Naples Bay when compared to Rookery Bay, Fahakatchee Bay, Faka Union Bay, or Pumpkin Bay.**

| Taxa | Common Name |
|---|---|
| **Absent from NB** | |
| *Aluterus schoepfii* | Orange filefish |
| *Chasmodes saburrae* | Florida blenny |
| *Diplectrum formosum* | Sand perch |
| *Elops saurus* | Ladyfish |
| *Eugerres plumieri* | Striped mojarra |
| *Floridichthys carpio* | Gold spotted killifish |
| *Lucania parva* | Rainwater killifish |
| *Monacanthus ciliatus* | Fringed filefish |
| *Mugil* sp., *Mugil gyrans* | Mullet, Fantail mullet |
| *Pogonias cromis* | Black drum |
| *Rachycentron canadum* | Cobia |
| **Unique to NB** | |
| *Gobionellus oceanicus* | Highfin goby |
| *Gobiesox strumosus* | Skilletfish |
| *Hypsoblennius hentz* | Feather blenny |
| *Ophichthus gomesii* | Shrimp eel |
| *Sciaenops ocellata* | Red drum |
| *Selene vomer* | Lookdown |
| Order Teuthida | Squid |

Because ANOSIM tests on Naples Bay data showed a significant difference between seasons, season was used as a factor when looking for differences among bays. A two-way ANOSIM test among samples using bay and season as factors (unpooled data, aggregated to Genus level, log (x+1) transformed, Bray-Curtis similarity) shows that there are weak but significant differences among bays (ANOSIM Global R = 0.126, p = 0.001) and between seasons (Global R = 0.198, p = 0.001). Pairwise bay to bay comparisons show that Naples Bay, Moorings Bay, and Rookery Bay are not significantly different from one another but are different from the other three bays (Figure 5-6). Fakahatchee, Faka Union, and Pumpkin Bays are all significantly different from one another. The differences in community structure across years in Naples Bay was only evident in one of the two bays (Faka Union) that had similar change points in diversity metrics.

SIMPER analysis was used to quantify the average dissimilarity between Naples Bay and other bays and which taxa contribute most to the dissimilarity. As with the patterns within Naples Bay, most of the differences between bays is the result of differences in how species are assembled (which species co-occur) and differences in their overall abundance. The SIMPER results show that, for the most part, the same species are responsible for dissimilarity between bays: mojarras (*Eucinostomus* spp.) and anchovies (*Anchoa* spp.) are the largest contributors to dissimilarity in all pairwise comparisons, followed by pink shrimp (*Farfantepenaeus duorarum*), blue crabs (*Callinectes* spp.), pinfish (*Lagodon rhomboides*), lizardfish (*Synodus foetens*), snappers (*Lutjanus* spp.), hardhead catfish (*Ariopsis felis*), and blackcheek tonguefish (*Symphurus plagiusa*). (Appendix C, Table C6). When comparing Naples Bay to those bays with different community structure (Pumpkin, Faka Union, and Fakahatchee Bays), mojarras, anchovies and hardhead catfish are generally more abundant in Naples Bay; pink shrimp, and inshore lizardfish, blackcheek tounguefish and pinfish are generally less abundant in Naples Bay; and blue crabs and snappers show mixed results.

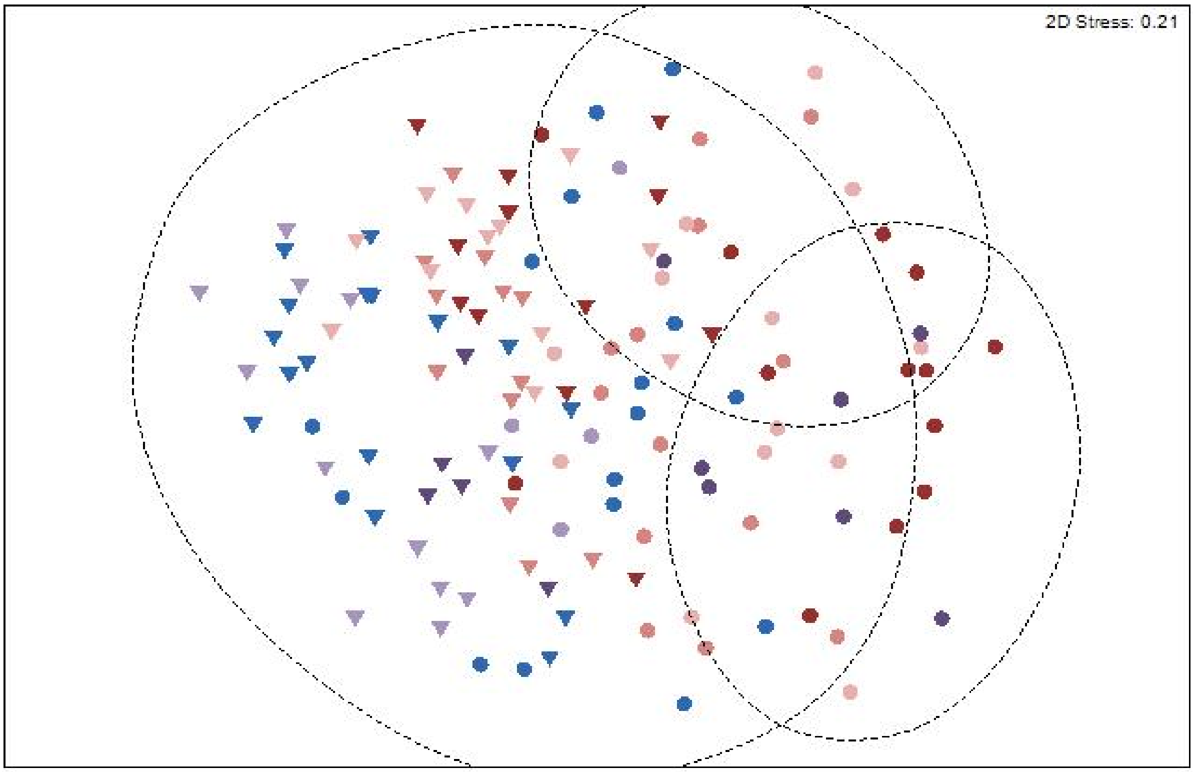


**Figure 5-6.** **MDS of fish community for all bays (Bray-Curtis similarity, log (x+1) transformed data) Data were pooled by month within year for illustration purposes only. FH, FU, and PB are represented in reds and pinks, MB and RB are violet, and NB is blue. Wet season samples are represented by triangles and dry season by circles. Dashed black circles enclose samples with at least 40 percent similarity.**

### 5.2.3 Discussion of Fish and Water Quality Relationship

As described above, no direct links or patterns between water quality and the biology data could be identified. Although some differences exist, as expected, the Naples Bay fish community is similar to that of the other southwest Florida bays. The high level of similarity indicates that large scale community shifts or adverse impacts that might be attributed to human induced impacts that Naples Bay has experienced are not apparent in the fish community data when compared to other southwest Florida estuaries that don't have the same level of human impact. This may indicate either the fish community is actually not affected by these variables in Naples Bay or perhaps the fish community is not sensitive enough to the impacts to be detected in the trawling dataset.

The downward shift in fish community univariate metrics (2011) observed in Naples Bay appears to also have occurred in other southwest Florida bays. However, after the shift, Naples Bay appears to have lower diversity and fewer taxa than the other bays in the comparison. Here we explore a potential role of the pattern of freshwater flow delivery to Naples Bay and its effect on salinity and hypothesize on how it may affect fish community.

During the dry season of 2010, approximately two to four times more rainfall occurred than other dry seasons during the time period of this study (see Section 3.1.1). This led to approximately 17 times more dry season flow from the GGC during the 2010 dry season than the average dry season flow during the time period for which GGC flow data are available (2009–2014). The substantially different rain and flow pattern during this time period altered the typical salinity and GGC flow pattern from the typical dry season “off” and wet season “on” pattern that was typical during the 2012–2014 time period.

The USGS continuous recorders in Naples Bay were not installed until summer of 2011 or early 2012 and, therefore, we do not have daily salinity data to represent the 2010 time period. However, the salinity pattern can be estimated by creating a spreadsheet model of the GGC flow and USGS salinity correlation in each section of the Bay (Section 3.2.2, Figure 3-11). The estimated daily average salinity during the 2010 time period was calculated from the correlation equation observed at each USGS continuous recorder location in Naples Bay (Figure 5-7). As the figure indicates, this simplistic model predicts daily average salinity fairly well at most locations, with the exception of the Gordon River location at Rowing Club Point. The model under-predicts salinity at this location when flow from the GGC is zero. However, the model is useful in estimating the different pattern of salinity that would have occurred as a result of the significantly different flow pattern during the 2009–2011 time period.

During 2010, the GGC contributed flow almost every day throughout the entire year. In contrast, the typical pattern of flow from the GGC is only during the wet season or after high dry season rain events. Although the dry season flows during 2010 were less in magnitude then typical wet season flows, the pattern significantly altered the salinity in the Bay at all locations from the normal dry season salinities. Dry season salinity was lower and more variable during 2010, but the pattern of salinity was more stable over the course of the entire year with less abrupt fluctuation between the dry and wet seasons when compared to the 2011–2014 time period.

The 2009–2011 time period saw a different pattern of flow and salinity in Naples Bay that happens to coincide with the highest fish diversity and richness. At this time we cannot identify any specific causal links between these two patterns, but believe further investigation is warranted. It’s possible that some dry season flows from the GGC, with a less abrupt change between dry and wet season flows, provides a more favorable environment for the fish community in Naples Bay. If a connection between flow pattern, salinity, and fish community can be identified for Naples Bay, this may inform potential management strategies for long-term restoration goals.

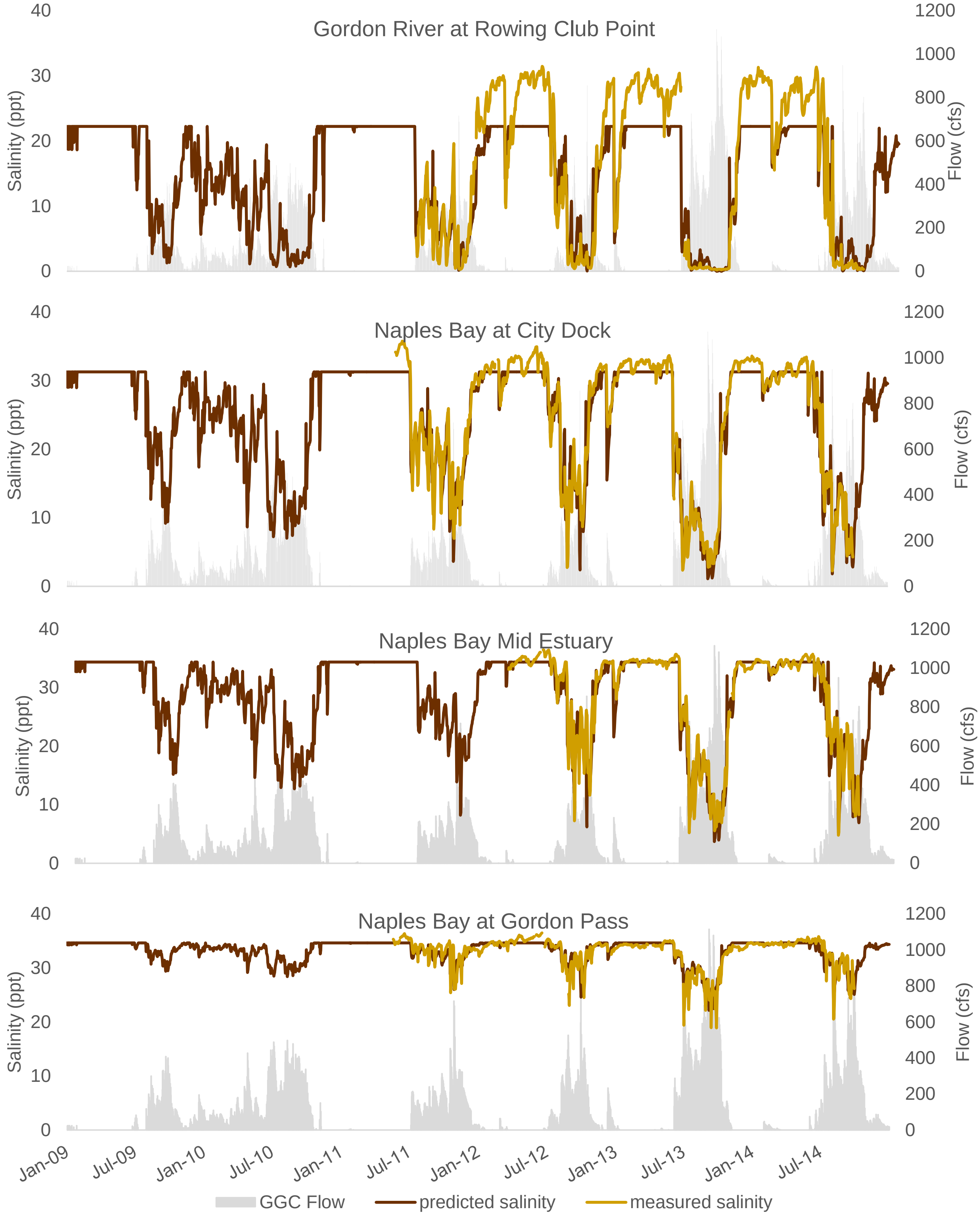


**Figure 5-7.** **Measured and predicted daily average salinity at USGS continuous recorder locations in Naples Bay, 2009–2014.**

# 6 Effect of Management Activities on Naples Bay

The City has undertaken many water quality management activities to improve water quality in the City's stormwater lakes and the downstream receiving waters of Naples and Moorings Bays. Most notably, these include the City wide fertilizer ordinance and installation of floating islands (floating wetlands) and aerators, along with educational programs designed to improve public awareness of local actions that can be taken to reduce runoff and improve water quality. Several communities across Florida have taken similar actions to improve stormwater runoff downstream water quality, but the City of Naples is taking the next step to attempt to quantify the improvements gained through the management programs. This information is valuable in making management decisions concerning cost effectiveness as well as calculating credit for programs that are working to improve water quality.

## 6.1 Fertilizer Ordinance

The City of Naples fertilizer ordinance (Code of Ordinances, Chapter 52, Article VII) was passed on March 5, 2008. The ordinance restricts fertilizer application from June through September each summer, limits the amount of phosphorus that can be applied, and requires at least 50 percent of nitrogen fertilizer to be slow-release, among other requirements. The ordinance also requires applicators to be certified.

As mentioned in section 3.2.4, a decreasing trend in nutrients (TN and TP) is observed at the long-term monitoring locations in the Bay. Since the fertilizer ordinance restricts wet season application of fertilizer, when stormwater runoff to the Bay is expected to highest, we conducted an additional Kendall Tau trend analysis on the wet season nutrient values over the period of record. The purpose was to determine if the observed decreasing trend could be identified during the wet season and if decreasing patterns could be temporally linked to the fertilizer ordinance. This analysis was conducted on the long-term Bay monitoring stations (GORDEXT/GORDPT, NBAYNL, NBAYWS, and GPASS6 (2005–2014)) as well as stations representing the major tributary and stormwater inputs to the Bay. Stations BC-2, BC-3, and HALDCRK have 2002-2014 data (or earlier); stations CURLEW, OYSBAY, HALDCR, and ROCKCR have 2011–2014 data.

Except for GORDEXT/GORDPT (Gordon River nearest the GGC), none of the stations show a declining trend in TN prior to the ordinance, but some stations do show a declining trend in some time period after 2008 when the ordinance was implemented. For wet season geometric mean TN, GORDPT/GORDEXT, the northernmost bay station, is significantly decreasing in the 2005–2014, 2008–2014, and 2010–2014 time periods (Figure 6-1, Table 6-1). Two southern bay stations, GPASS6 and NBAYWS, are also significantly decreasing in the 2010–2014 time period, with another bay station (NBAYNL) having a high correlation coefficient that was not statistically significant. The tributary stations CURLEW, HALDCR, and OYSBAY are significantly decreasing in the 2011–2014 time period; ROCKCR has a high correlation coefficient, but is not significant. However, stations located in the Gordon River (BC-2, BC-3) as well as upstream Haldeman Creek (HALDCRK) are not showing similar declining trends in wet season TN.

Statistically significant decreasing trends in wet season TP are found at several stations (Figure 6-1,Table 6-2). In the bay, three of the four stations show significant decreasing trends from 2008–2014 (GORDEXT/GORDPT, NBAYWS, GPASS6); NBAYWS is the only bay station to show a decreasing trend prior to the fertilizer ordinance. Tributary stations ROCKCR, CURLEW, and OYSBAY (2011–2014) had high correlation coefficients, but were not significantly declining because of small sample size. Stations located in the Gordon River (BC-2, BC-3) showed declines prior to the fertilizer ordinance (BC-3 2002-2014), but upstream Haldeman Creek (HALDCRK) did not show any declining trend in wet season TP.

Because the majority of declining trends in wet season TN are shown for periods after 2008, the results may indicate that the implementation of the fertilizer ordinance in 2008 is a contributing factor to the decreasing TN trend observed in Naples Bay. For wet season TP, some declining trends in tributary concentrations were evident prior to 2008, but the concentrations in the bay show the biggest trends from 2008-2014. Notably, TN and TP have decreasing trends at GGC stations, which are not impacted by the City's fertilizer ordinance, but may be impacted by the County's fertilizer ordinance. At this time we have no information regarding the potential reduction in fertilizer application timing or amounts within the City limits since the ordinance. However, with the statistically significant decrease in wet season nitrogen and phosphorus at some long-term Bay stations coinciding with the implementation of the fertilizer ordinance, we cannot discount the importance of the ordinance as a potential contributing factor to the decrease.

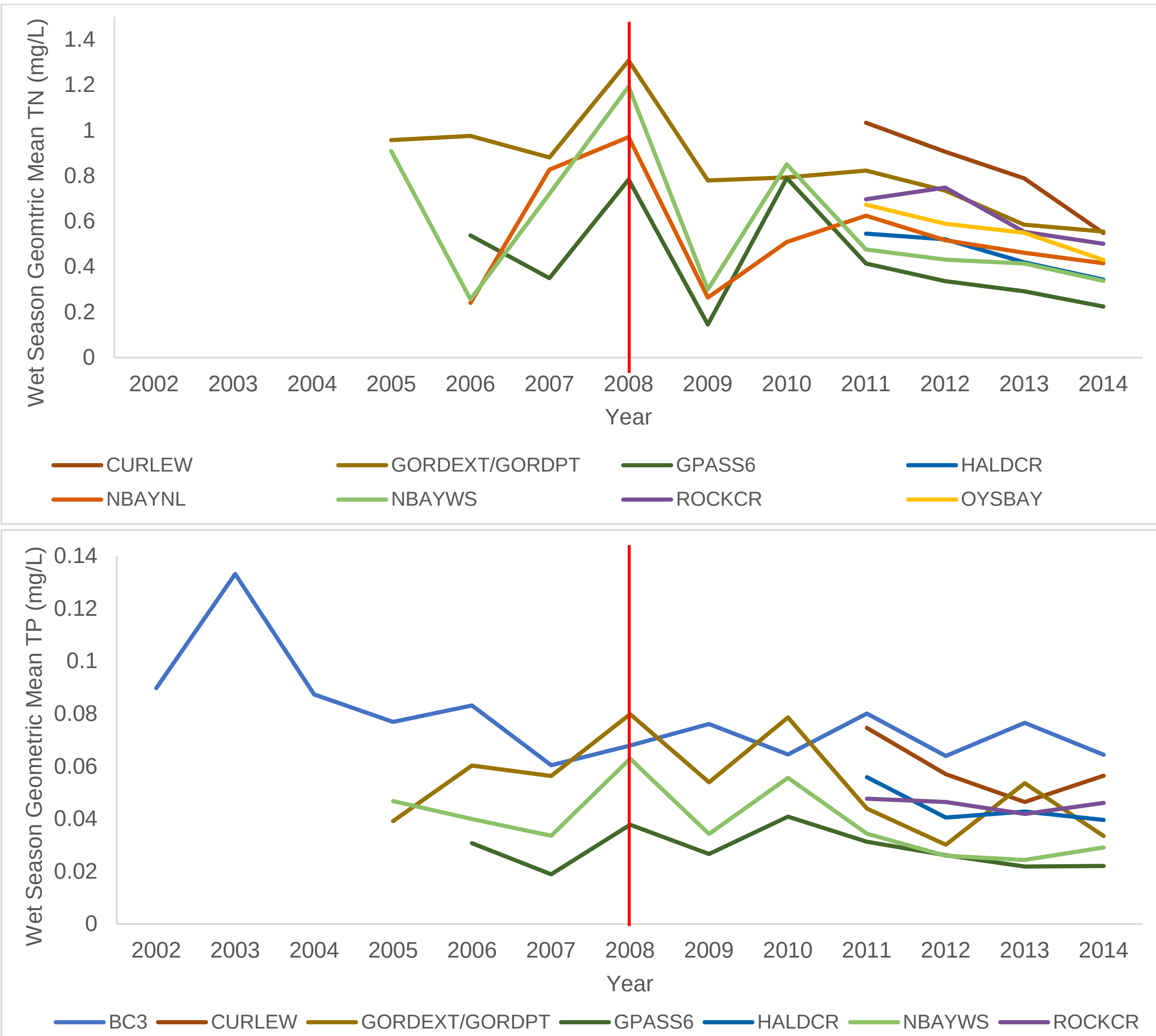


**Figure 6-1.** **Wet season nitrogen and phosphorus annual geometric mean concentrations for stations with significant trends in Naples Bay and tributaries. The red vertical line designates the implementation of the City's fertilizer ordinance.**

**Table 6-1. Kendall Tau correlation coefficients for wet season (June–November) geometric mean TN concentrations over various time periods in Naples Bay and its tributaries. Marked cells are significant at p<0.05.**

| Station | 2002–2014 | 2005–2014 | 2008–2014 | 2010–2014 |
|---|---|---|---|---|
| GORDEXT/GORDPT | -- | **-0.69** | **-0.71** | **-0.80** |
| NBAYNL | -- | -0.11 | -0.33 | -0.60 |
| NBAYWS | -- | -0.29 | -0.52 | **-1.00** |
| GPASS6 | -- | -0.39 | -0.43 | **-1.00** |
| BC-2 | 0.08 | 0.20 | -0.42 | 0.00 |
| BC-3 | 0.08 | 0.20 | 0.05 | 0.20 |
| HALDCRK | 0.30 | 0.37 | -0.04 | -0.40 |
| CURLEW | -- | -- | -- | **-1.00*** |
| OYSBAY | -- | -- | -- | **-1.00*** |
| HALDCR | -- | -- | -- | **-1.00*** |
| ROCKCR | -- | -- | -- | -0.67* |

* Data for these stations were from 2011-2014 only..

**Table 6-2. Kendall Tau correlation coefficients for wet season (June–November) geometric mean TP concentrations over various time periods in Naples Bay and its tributaries. Marked cells are significant at p<0.05.**

| Station | 2002–2014 | 2005–2014 | 2008–2014 | 2010–2014 |
|---|---|---|---|---|
| GORDEXT/GORDPT | -- | -0.33 | **-0.62** | -0.40 |
| NBAYNL | -- | -0.33 | 0.04 | 0.20 |
| NBAYWS | -- | **-0.47** | **-0.62** | -0.60 |
| GPASS6 | -- | -0.22 | **-0.62** | **-0.80** |
| BC-2 | -0.41 | -0.42 | -0.14 | 0.40 |
| BC-3 | **-0.51** | -0.20 | -0.14 | -0.20 |
| HALDCRK | 0.31 | 0.20 | -0.14 | -0.20 |
| CURLEW | -- | -- | -- | -0.67* |
| OYSBAY | -- | -- | -- | -0.33* |
| HALDCR | -- | -- | -- | -0.67* |
| ROCKCR | -- | -- | -- | -0.67* |

* Data for these stations were from 2011-2014 only.

## 6.2 Floating Islands

There are 28 stormwater lakes in the City of Naples which are intended to capture, retain, treat, and release stormwater prior to entering the receiving waters (Moorings Bay, Naples, or the Gulf of Mexico). They allow potential pollutants (nutrients, heavy metals, sediment) to settle out of the water column and are trapped in the lakes. Although their primary function is stormwater treatment, the aesthetics of these lakes are extremely important because they are located in and among the Naples community. The nutrients trapped in these lakes can cause algae blooms, which can lead to low dissolved oxygen levels and cause fish kills. Copper sulfate is typically applied to these lakes, many with long-term application histories, to control algae blooms.

In an effort to improve the treatment capacity and aesthetics of the stormwater lakes, the City has been engaging the public and Homeowners Associations in a proactive program to install floating habitat islands (floating wetlands). The islands are planted with native vegetation to increase the nutrient uptake capacity of the lakes. They also provide more area for wildlife. The plants are harvested once or twice per year, removing the nutrients from the stormwater system, lowering the nutrient concentrations in the lakes and reducing nutrient discharge to receiving waters.

The floating islands are also used in an effort to reduce copper concentrations in discharge to receiving waters. Naples Bay is currently listed as verified impaired for copper by the FDEP, therefore copper is an important concern for the City as well as the health of Naples Bay. The floating islands do not directly uptake copper, but are installed as part of an agreement between the City and Homeowners Associations (HOA) to stop using copper sulfate applications in lakes with floating islands. The lakes uptake nutrients and provide at least some shading to reduce or eliminate algae problems in the lakes, negating the need for the copper sulfate application. The reduced or eliminated copper sulfate applications can decrease copper delivered to receiving waters, thus improving water quality downstream.

To date, the City has installed floating islands in eight lakes beginning with North Lake (Lake 8) in March 2009 (Table 6-3). Six of the lakes currently have islands in them, while two lakes, Spring Lake and Lantern Lake, had the islands removed in March of 2012 and April of 2014, respectively. In order to attempt to characterize the potential these islands are having on water quality in the lakes or lake discharge, we paired the available data from the City's stormwater lake monitoring program with the timing of floating island installation and removal for each lake (Table 6-3). Of the eight lakes with islands, four lakes have available water quality data to represent the lake while the island is installed. However, only one lake (Swan Lake) has sufficient water quality data for a time period before and after island installation. Lake Manor, which had floating islands installed in two phases, does not have water quality data from the time period before any islands were installed but does have data from before and after the second group of islands was installed. Therefore, analysis of any changes or differences in water quality as a result of the islands was conducted on groupings of all lakes and an individual analysis of Swan Lake and Lake Manor.

An analysis of all the stormwater lakes was conducted to determine if lakes with floating islands exhibited statistically significant different water quality conditions than those without floating islands and aerators. The lakes were separated into three groups; water quality data from lakes that have never had floating islands and aerators; water quality data from lake that did/do have floating islands but data exists for the time period before/after the lakes were installed or removed; and water quality data from lakes with floating islands installed during the sampling events. A one-way ANOVA concluded that no statistically significant differences in water quality (TN, TP, and copper) existed between any groups of lakes ($p > 0.05$). This indicates stormwater lakes with islands do not exhibit statistically different water quality conditions from lakes without islands. However, this should not be interpreted to indicate the floating islands are not effective; the lack of significant difference could be related to the wide variation in water quality conditions among the lakes and relatively small datasets from which to conduct the analysis.

**Table 6-3. Floating island installation time periods in stormwater lakes and corresponding water quality data collection, City of Naples.**

| Lake Name (Number) | Date Installed | Date Removed | Water Quality Data Range | Number of Data Points |
|---|---|---|---|---|
| North Lake (8) | March 2009 | -- | December 2010–September 2012 | 5 |
| Spring Lake (11) | August 2009 | March 2012 | April 2012–February 2015 | 6 |
| Lake Manor (22) | August 2009* | -- | December 2010–February 2015 | 10 |
| Lake 25 | August 2011 | -- | N/A | 0 |
| East Lake (31) | March 2012 | -- | N/A | 0 |
| Lantern Lake (14) | March 2012 | April 2014 | February 2012–February 2015 | 26 |
| Lake 12 | April 2012 | -- | N/A | 0 |
| Swan Lake (2) | May 2013 | -- | December 2010–February 2015 | 10 |

* Lake Manor had one island installed in 2009 and three more installed in August of 2012.

Of the lakes with floating islands, only Swan Lake has sufficient pre-installation data to compare water quality before and after island installation. Lake Manor has sufficient data before and after the installation of some of its floating islands. While neither lake has sufficient data to conduct statistical analyses for the before and after island installation time periods, we plotted the available nutrient and copper data to determine if any visual patterns could be observed (Figure 6-2). There is no apparent change in nutrient or copper concentrations in the dataset for the time period after the floating islands were installed. There appears to be an increase in phosphorus concentrations throughout the December 2010–February 2015 time period.

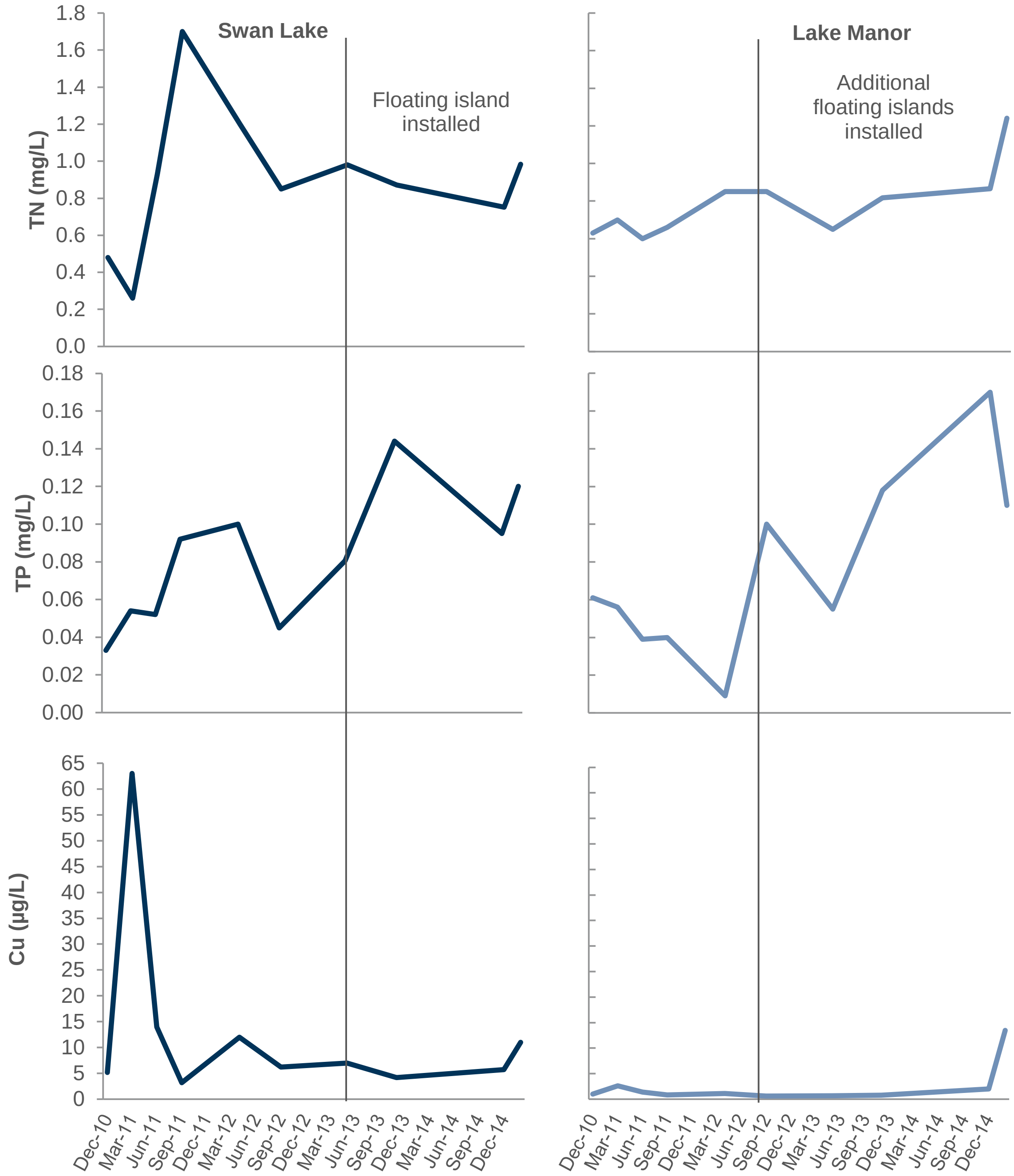


**Figure 6-2.** **Nutrient and copper results in Swan Lake and Lake Manor, before and after floating island and aerator installation, City of Naples.**

Although no statistical trends in the relatively small datasets for Swan Lake and Lake Manor are possible at this time and no apparent change in water quality is observed as a result of the island installation, qualitative evidence concerning their effectiveness at reducing algal blooms exists. Floating islands were placed in North Lake (Lake 8) and Lake 25 in March of 2009 and August of 2011, respectively. The City reports that in each case, within a few weeks of installation of floating islands and aerators, the algae disappeared (Figure 6-3).

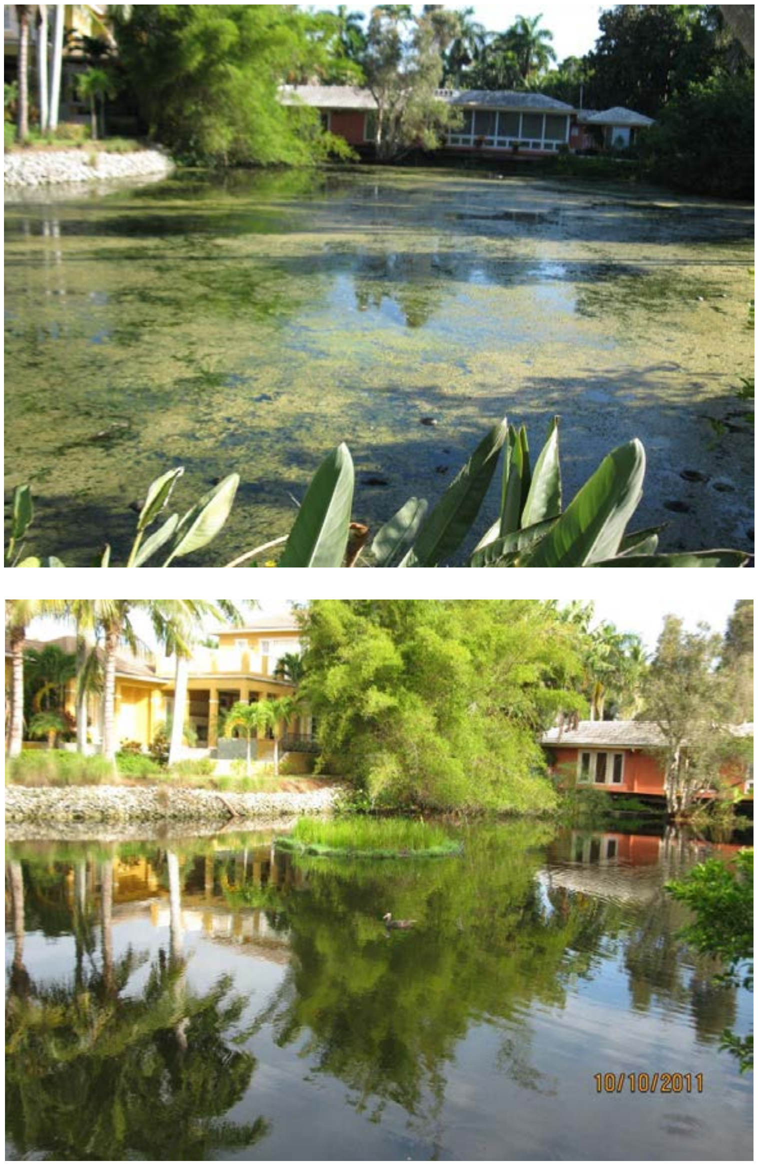


**Figure 6-3.** **City of Naples residential stormwater lake before and after installation of floating islands and aerators. (Photos courtesy of City of Naples Natural Resource Division.)**

A robust statistical survey design for the stormwater lakes is recommended and would likely be able to identify and quantify the improvements being observed by the installation of the floating islands and aerators. The Before/After Control/Impact (BACI) statistical survey model is recommended and useful for identifying when and where change occurs as a result of a resource management action, as well as being able to quantify the change. Specific recommendations for how to implement a statistically robust monitoring program designed for this purpose are provided in the accompanying *Naples Bay Monitoring Design* report.

The City is being proactive in its approach to managing and restoring Naples Bay. From water quality improvements through floating islands and aerators, fertilizer ordinances, and efforts to reduce copper sulfate applications, to mangrove, seagrass, and oyster restoration efforts, the City's programs are designed to improve the conditions of Naples Bay for the residents and guests of the area. Quantification of the improvements as a result of these programs is on the right track, but can still be improved to allow for accurate identification of the actual changes in water quality and biology.

# 7 Naples Bay Historical Comparison

Until 2006, when the City began its water quality and biological monitoring program, the only large scale, comprehensive monitoring, data analysis, and reporting effort in Naples Bay was completed in 1979 (Simpson *et al.* 1979). Data collection for this previous effort was completed over the course of a year from late 1976 through late 1977. This report concluded that the number one source of pollution in Naples Bay was the GGC freshwater flow and cessation of flow was the only course of action to restore the Bay (Simpson *et al.* 1979). The GGC flow brought enormous volumes of freshwater, silt, nutrients, and other pollutants that dramatically altered the water quality and biological community in Naples Bay (Simpson *et al.* 1979). These conclusions persist today in the discussion of the status of Naples Bay.

Now the opportunity exists to compare the current water quality and biological status of Naples Bay to the results of the 1979 study to determine what, if any, improvements have been realized in Naples Bay, and quantify any changes. Since the 1979 report several significant advancements in water resources have occurred that could play a role in any changes observed in Naples Bay from 1979 to current day. Most notably, the amendments to the Clean Water Act and adoption of water quality criteria, advanced wastewater treatment, stormwater management and treatment technologies, and specific to Naples Bay, the replacement and upgrades made to the GGC weir system. Although the sampling methodologies are not identical between the two time periods, there is significant overlap that allows for a meaningful comparison. Here we compare pertinent water quality and biological parameters from the 1979 Naples Bay Study to the current status observed in Naples Bay to identify and quantify changes and/or improvements since the last large scale monitoring effort was completed.

Comparisons are available between the Naples Bay Study and current data collection for some water quality and quantity parameters (nitrogen, phosphorus, chlorophyll *a*, salinity, rainfall, and GGC flow) as well as fish community abundance and diversity. These parameters were chosen because of their similar sampling methodologies and methods of reporting in the 1979 study. Specific information regarding how the comparisons were done and any necessary assumptions about the data are described in the following sections.

## 7.1 Golden Gate Canal Flow

Construction of the Golden Gate Main Canal system was completed in the late 1960s to drain upland areas, historically outside of the Naples Bay watershed, for residential development (City of Naples 2010, SFWMD 2007, FDEP 2010, and Simpson *et al.* 1979). When first constructed, the weir separating the GGC from the Gordon River was a concrete dam that allowed free flow over the weir whenever water exceeded the weir elevation (Simpson *et al.* 1979). This was the case at the time of the Naples Bay Study in the late 1970s. Since the mid-1980s, the SFWMD has been working to upgrade the GGC weir system and install structures to better manage flow, increase groundwater recharge, and improve water quality in Naples Bay by reducing freshwater inflow from the GGC, with the most recent improvements being implemented in 2012 (SFWMD 2012). Here we examine the differences in GGC flow magnitude and timing that occurred during the Naples Bay Study in the late 1970s (Simpson *et al.* 1979) and the current flow conditions that are observed today. This examination will serve as the basis for the subsequent discussions regarding water quality and biological comparisons between the historical and current time periods.

Simpson *et al.* (1979) reported daily flow from the GGC into the Gordon River during their study for water year 1977 (October 1976–September 1977). For the current time period, we calculated GGC daily flow for water years 2009–2014. Recorded GGC flow for water year 1977 averaged 193 mgd over the course of 335 days for which flow was recorded (Table 7-1). In comparison, the current years all showed fewer days of flow and less flow per day, with the exception of 2010 (358 days of flow) and 2013 (212 mgd on average). Flow during the Naples Bay Study was 33 to 85 percent greater than the flow recorded during the current time period. In 2010, the number of days of flow was greater, but the magnitude of flow was significantly reduced from the 1977 time period, and in 2013, the average daily flow was increased, but the number of days of flow was less than the historical time period.

**Table 7-1. Golden Gate canal flow comparison between water year 1977 (historical) and water years 2009–2014 (current). Historical data from Simpson *et al.* (1979); current data from SFWMD.**

| Time Period | Water Year | Total Flow (Million Gallons) | Days of Flow | MGD |
|---|---|---|---|---|
| Historical | 1977 | 65,656 | 335 | 193 |
| Current | 2009 | | | |
| | 2010 | 36,533 | 358 | 102 |
| | 2011 | 10,116 | 119 | 85 |
| | 2012 | 22,745 | 219 | 103 |
| | 2013 | 43,918 | 207 | 212 |
| | 2014 | 28,221 | 199 | 142 |

Since the GGC flow is heavily rainfall driven, we examined the rainfall conditions between the historical and current time periods to determine if rain patterns could explain the difference in flow pattern between the two time periods. Monthly and annual rainfall totals for both time periods were obtained from the National Oceanic and Atmospheric Administration (NOAA) rain gauge (GHCND:USC00086078) located in the Golden Gate area near Naples (see Figure 2-1). Rainfall in 1977 totaled 50.5 inches compared to total rainfall of between 52.7 and 64.5 inches for the current time period. This indicates rainfall during the historical period was similar to or even less than rainfall during the current time period.

This comparison indicates that GGC flow has been significantly reduced since the Naples Bay Study (1979). In fact, during 2013 when the highest rainfall (64.5 inches) and highest flow of the current time period (43.9 billion gallons) were recorded, flow was still 33 percent less than flow during the historical period. The GGC may have still been actively dewatering the upland areas during the late 1970s. In addition, control structure upgrades made to the canal system over the last 20 years are likely responsible for reduced flow observed since the Simpson *et al.* (1979) study. The potential effects of the reduced flow on Naples Bay water quality and biology are discussed below.

## 7.2 Water Quality

During the 1979 Naples Bay Study, water quality data were collected at nine locations throughout Naples Bay and the Gordon River (Marine Segment) monthly for one year (December 1976–November 1977) (Simpson *et al.* 1979). Four of those locations were in proximity to the City's current long-term sampling stations to allow for adequate comparisons of data (Figure 7-1). For grab sample data of TN, TP, and chlorophyll *a*, historical monitoring locations (Stations 10, 40, 50, and 70) were compared to current monitoring locations GPASS6, NBAYWS, NBAYNL, and GORDEXT/GORDPT, respectively. Simpson *et al.* (1979) conducted monthly diel monitoring for salinity, which allowed for comparison to USGS continuous recorder data from 2011 to 2014. Therefore, historical monitoring locations 10, 20, 50, and 70 were used to coincide with the USGS monitoring locations (Figure 7-1).

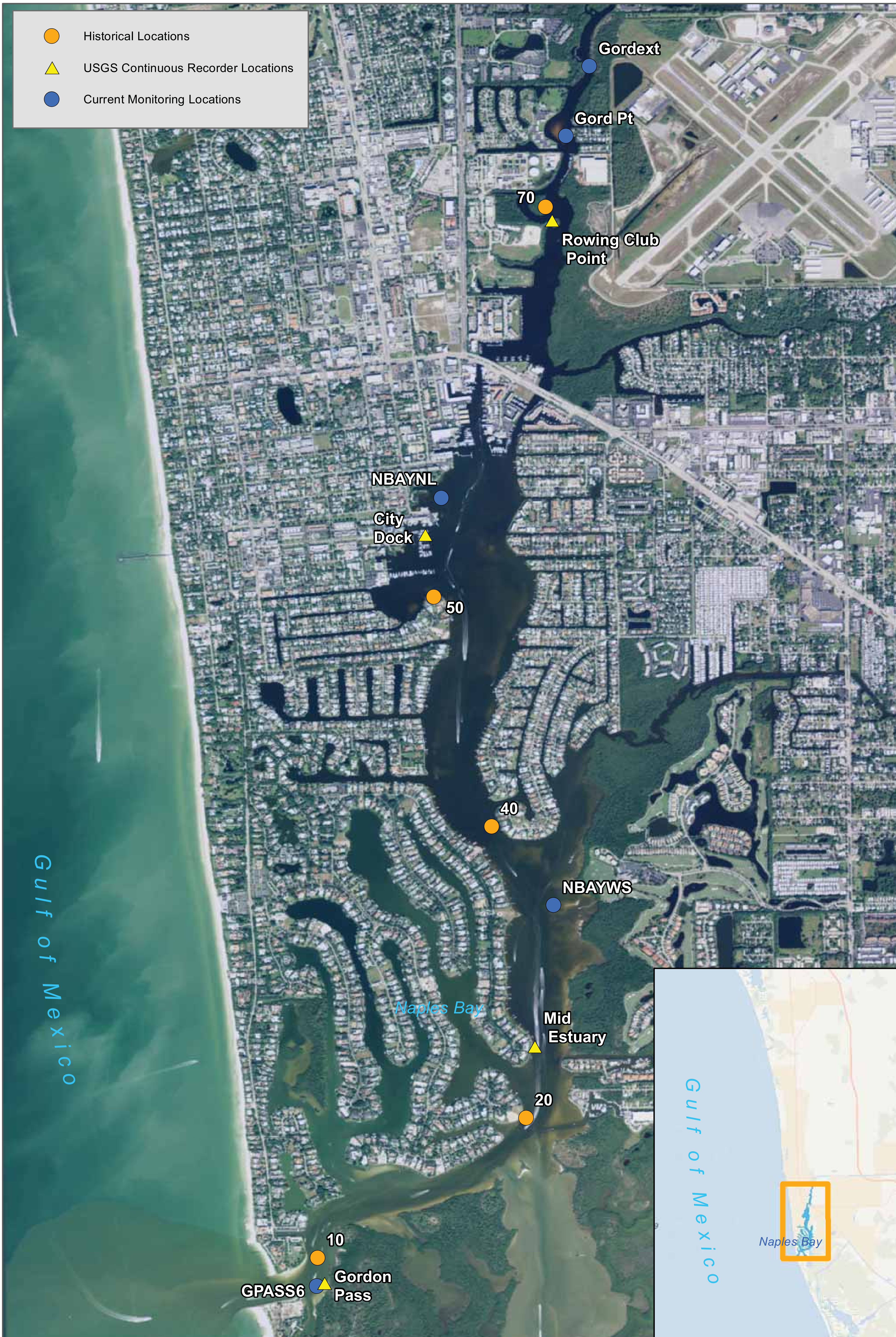


**Figure 7-1. Historical and Current Naples Bay Monitoring Locations for Comparison. Historical - Simpson et al 1979; Current - City of Naples**

City of Naples, Natural Resources Division
Collier County, Florida



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GIS Analyst: adam.bausch

For the current water quality data, monthly measurements at each station for the 2006–2014 time period were averaged to obtain a current representation for a given month and compared to the monthly measurement collected in 1976–1977. The individual diel sampling events at each station in the historical data were merged to generate a daily range in salinity for the surface and bottom and compared to surface and bottom salinity ranges observed in the current data. The results of this analysis are described below.

The current average surface salinity in Naples Bay and the Gordon River follows the same general pattern as the historical data with significant differences observed between the wet and dry seasons (Figure 7-2). The largest differences in average surface salinity are observed in the Gordon River and northern Naples Bay locations, with the current data showing increased salinity concentrations during most months, and especially during the wet season months. In addition, the daily range in salinity concentrations (shown in Figure 7-2 as the gray bars) is increased in the current data, with the two northern most stations (Gordon River and City Dock) showing the most significant change from the historical condition.

The shift in salinity regime in the current data can be attributed to the change in flow regime from the GGC canal between the two time periods. The significant reduction in GGC flow from the historical time period leads to greater overall salinity in the Bay, as well as allowing greater tidal influence further north in the Bay creating larger daily swings in salinity. This explains the increased daily salinity range, which is even more pronounced during the wet season when canal flow trends to be greater in magnitude and longer in duration. Although wet season flow from the canal is greater than during the dry season, the observed overall reduction in flow from the historical condition allows more tidal influence and therefore larger daily salinity swings. The observed daily range in salinity is less at the southern Bay locations (Mid Estuary and Gordon Pass) because the influence of the GGC canal flow on salinity is diminished.


**Figure 7-2.** **Mean and range salinity comparison between historical (1976–1977) and current (2011–2014) data in Naples Bay and Gordon River Marine Segment. Simpson *et al.* 1979; USGS 2011–2014. Blue = current data; Yellow = historical data; Gray Bars = daily salinity range.**

Throughout the Bay and Gordon River, significant changes in nutrients and chlorophyll *a* concentrations are also observed in the current data versus the historical data reported by Simpson *et al.* (1979). Average TN concentrations are 25 to 45 percent lower than reported in the 1976–1977 data, while average TP concentrations are 55 to 75 percent lower, and average chlorophyll *a* concentrations are 66 to 75 percent lower in the current data than reported in the historical data (Figure 7-3).

The observed decrease in nutrients and chlorophyll *a* concentrations in the Bay is not surprising given the multitude of advancements in water quality and water resources since the late 1970s including improved stormwater management, adoption and implementation of water quality criteria, and advanced wastewater treatment. All of these in combination would result in improved water quality in Naples Bay. Additionally, the reduction in GGC flow and loadings from the 1976–1977 would also contribute to the improved water quality. For the 2008–2014 time frame, the annual loadings to Naples Bay ranged from approximately 100,000–300,000 lbs of nitrogen and from approximately 3,500–11,000 lbs of phosphorus (see section 3.1.1). Nutrient loadings to Naples Bay during the December 1976–November 1977 time period were approximately 430,000 lbs and 17,500 lbs of nitrogen and phosphorus, respectively. As a result of the lower nutrient concentrations and reduced GGC flow, the maximum observed nutrient loadings in the current condition (2013, see section 3.1.1) represent a 30 percent reduction in nitrogen loading and approximate 40 percent reduction in phosphorus loading to Naples Bay from the historical condition.

This comparison indicates that water quality conditions (salinity, nutrients and chlorophyll *a*) in Naples Bay have improved significantly from the conditions observed by Simpson *et al.* in the late 1970s. Nutrient and chlorophyll *a* concentrations have been reduced and the salinity concentrations have somewhat increased with daily ranges that indicate more tidal influx in the upper Bay is occurring now. Several factors likely contribute to this improvement as mentioned above and we have no credible method of discerning which factors play the most influential role in this improvement with the data we have currently. However, is it reasonable to conclude that the observed reduction in flow from and loading from the GGC canal from the 1970s levels likely played a significant role in the Bay's observed water quality improvement.

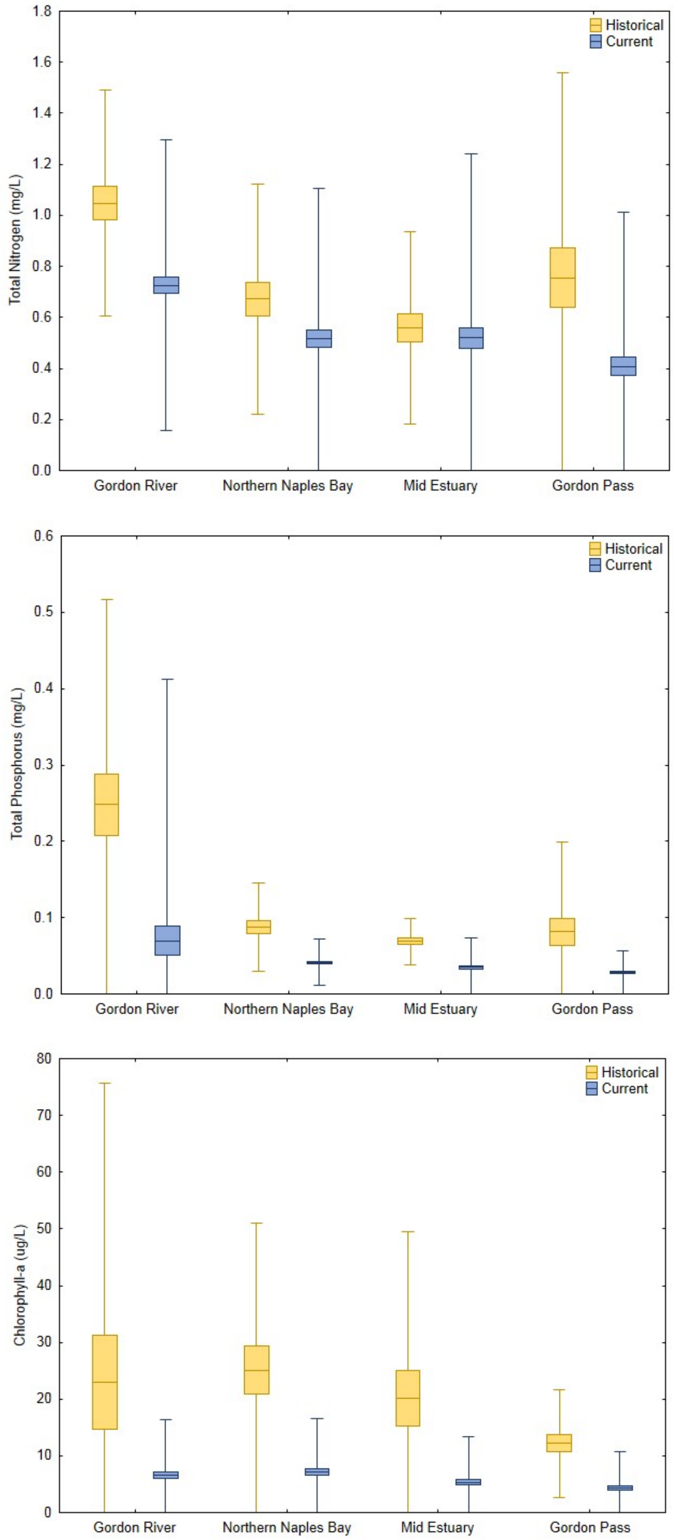


**Figure 7-3.** **Comparison of nutrient (nitrogen and phosphorus) and chlorophyll *a* concentrations between historical (1976–1977) and current (2008–2014) conditions. Historical data from Simpson *et al.* (1979); current data from City of Naples.**

## 7.3 Biology

Comparison of the biology community (fish) observed by Simpson *et al.* (1979) and the current data set are also possible. Although Simpson *et al.* (1979) also collected phytoplankton and benthic samples, this comparison will focus on fish as the current data set do not include benthic and phytoplankton data. Fish collection in the Naples Bay Study (1979) were conducted monthly using two 100m mid-water trawls pulled at each of 17 fixed locations throughout the Bay, Gordon River, tributary canals, and a control station in Dollar Bay. By comparison, trawling conducted by the City in the current dataset consists of bottom trawls at random locations within four different zones within the Bay, Gordon River, and Port Royal canal area.

Comparisons of fish results between the two time periods will focus on observed patterns across months and among zones of the Bay and not comparisons of the numbers themselves. We understand the difference in sampling methodologies and gear type play a significant role in understanding why the data may differ. Therefore, the comparison will focus on whether or not the current data exhibits the same pattern (seasonal and spatial) as observed in the Simpson *et al.* (1979) study. This type of comparison will allow us to understand whether changes have occurred over time in the fish communities, even though sampling methodologies were different.

Abundance and diversity data published in the Naples Bay Study (1979) were plotted by zone next to average data from the current Naples Bay monitoring program. Only stations that overlapped the current sampling zones were included in the comparison. Thus, historical station 90 was not part of this comparison. Station 90 was located in the Golden Gate Canal directly below the dam, and not within current sampling Zone 1. Notably, without station 90, the conclusions in the 1979 report about the diminished state of the Gordon River as a whole are not as apparent. The patterns showing large drops in diversity and abundance were completely driven by data from a station that is not representative of the rest of the Gordon River area. When patterns of abundance and diversity in 1977 are compared to patterns in the recent years (Figure 7-4 and Figure 7-5) a few differences are visible. First, the difference in abundance in Zones 2 and 3 is smaller in the recent dataset than it was in 1977. Second, the difference in abundance between Zone 1 and the other zones was greatest in the wet season during 1977, while it is greatest in the dry season in the recent data. This could be related to changes in flow between the two time frames: in 1977 the flow from the Golden Gate Canal was much higher than has been recently and did not completely shut off for several months at a time in the dry season as it has recently. In terms of diversity, there seems to be a stronger wet-dry seasonal trend in recent years that is not as evident in 1977. The current pattern shows gradually decreasing diversity as the summer and wet season progresses with an increase again in the drier winter months. In addition, there is less separation between Zone 1 and Zones 2 and 3 in the more recent dataset.

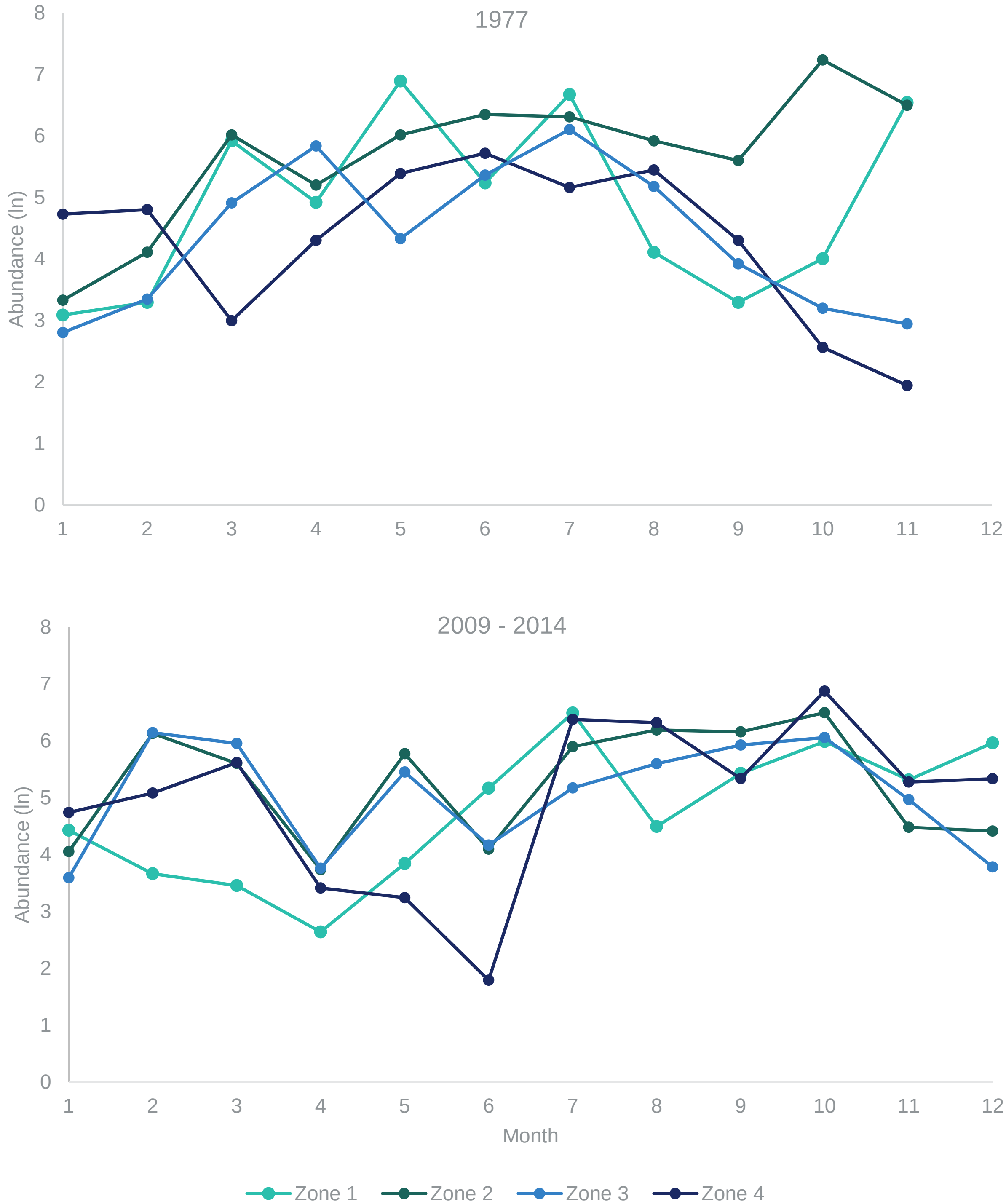


**Figure 7-4.** **Average fish abundance by zone for 1977 and 2009–2014 in Naples Bay.**

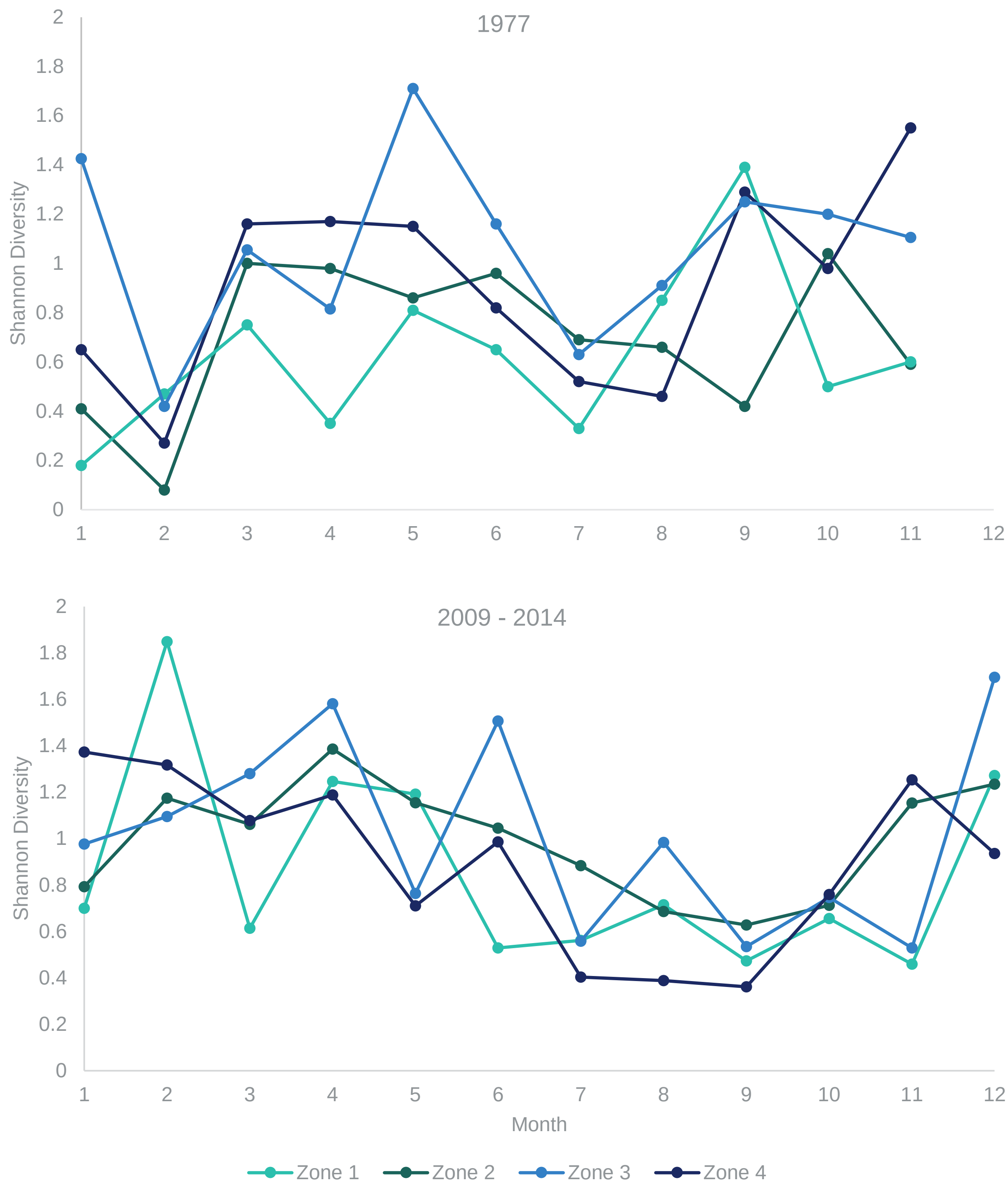


**Figure 7-5.** **Average fish diversity by zone for 1977 and 2009–2014 in Naples Bay.**

# 8 Conclusions

This effort provides a comprehensive look at the current status of water quality and biology in Naples Bay, along with comparisons to similar data from other southwest Florida estuaries and a historical comparison to the way Naples Bay looked in the late 1970s. The goal of this study is to provide the information and analysis necessary to make informed decisions regarding resource management and to determine what effect ongoing management activities are having on Naples Bay. Statistically significant trends in water quality and biological communities (fish and seagrass) were identified, and links between them that can inform management decisions were investigated. Inputs to Naples Bay (Golden Gate Canal, stormwater lakes, and pump stations) were quantified, where possible, and included in the investigation for their potential effect on Naples Bay. A summary of the major conclusions is provided below.

***Naples Bay Water Quality***

- Nutrients and chlorophyll *a* have significantly reduced since the late 1970s.
- In Naples Bay, total nitrogen and total phosphorus concentrations show a statistically significant decreasing trend at most long-term stations in the current dataset.
- Naples Bay Proper (south of SR 41) demonstrates compliance with the newly adopted NNC.
- Nitrogen and phosphorus data in the marine segment of the Gordon River (north of SR 41) indicate exceedances of the NNC for Naples Bay.
- Statistically significant increasing trends are observed for chlorophyll *a*, turbidity, copper, and bacteria.
- The current dataset indicates chlorophyll *a* and copper currently exceed their respective water quality standards in Naples Bay. Naples Bay is currently listed as impaired for copper and may be listed as impaired for chlorophyll *a* during future assessment cycles if the current trend continues.

***Golden Gate Canal***

- Freshwater inflow from the Golden Gate Canal plays a major role in shaping the water quality of Naples Bay. The canal flow affects salinity throughout the Bay, with the highest impacts observed in the northern region. In fact, the marine portion of the Gordon River above SR 41 shifts to a freshwater system virtually every summer.
- As a result of the weir replacement and upgrades made by the SFWMD, the WY2010–WY2014 canal flow was reduced by an average of 56 percent (range from 32 to 84 percent in the current dataset) from 1970s levels.
- The Golden Gate Canal delivers significant nutrient and solids loads to Naples Bay that can greatly affect water quality, biological communities, and management and restoration activities.
- Relative loads to Naples Bay from the GGC alone are many times greater than the sum of all loading sources to Tampa Bay, which has exhibited significant resource recovery.
- Management and restoration planning in Naples Bay must account for and address the significant loading issue to Naples Bay if water quality and biological community improvements will be successful.
    - Management options should weigh the costs and benefits of flow reduction from the GGC versus treatment to reduce concentrations given that concentrations of nutrients and solids are already relatively low and the magnitude of loadings to the Bay are the result of the significant volume contribution.

***Naples Bay Biological Communities***

> The fish community in Naples Bay are dominated by euryhaline and cosmopolitan species (anchovy and mojarra, making up greater than 85 percent of total catch) and are found in all zones of the Bay throughout the year, during times of significant canal flow as well as times of no flow.

> The fish community shows weak statistically significant differences from north to south between zones, mostly as a result of small changes in abundance of the most common species.

> Naples Bay fish abundance and species composition are generally similar to other southwest Florida estuaries (Rookery Bay, Pumpkin Bay, Fakahatchee Bay, and Faka Union Bay), which are located in areas with less urban development and direct stormwater runoff.

> In the current dataset, seagrass densities peaked in 2009/2010 and appear to have steadily decreased since then.

> The highest seagrass densities are observed during early growing season monitoring events (May–June), with significant declines in monitoring events conducted in August–November. In the more recent years of monitoring (2011–2014), all monitoring events were conducted later in the growing season, contributing to the decreasing trend over time.

***Naples Bay Water Quality - Biology Relationship***

> No statistically significant relationships between water quality parameters and the fish community were observed in the Naples bay trawling data.

> The similarity of the fish community in Naples Bay to other southwest Florida estuaries indicates the Naples Bay fish community does not appear to be sensitive to changes in water quality occurring in Naples Bay only (i.e. salinity).

> The significant loadings to Naples Bay likely play a role in the observed trend in seagrass density. Approximately 90 percent of the loads from the GGC are delivered during the seagrass growing season and likely contribute to the decreasing trend in seagrass density during the summer growing season.

> The link between Naples Bay loadings and biological communities warrants further investigation as seagrass and oyster restoration activities will be significantly affected if loadings are not addressed.

> The current data do not point to salinity as a potential seagrass limiting factor as the southern Bay does not experience the same extreme salinity shifts as the northern Bay and the dominant species of seagrass is well suited to a wide salinity range.

***City of Naples Management Activities***

> Statistically significant decreasing nutrient concentrations in the Bay appear to temporally coincide with the implementation of the City's fertilizer ordinance and may be a contributing factor in the observed decrease in nutrient concentrations.

> Sufficient data were lacking to conduct robust statistical analyses concerning changes in water quality as a result of the installation of floating islands in stormwater lakes.

> Although no change in water quality was observed, qualitative evidence exists that the islands are successful at reducing or eliminating algae in the stormwater lakes.

> One goal of the floating islands is to eliminate the need for private homeowners and HOAs to conduct repeated applications of copper sulfate that can discharge from the lakes into receiving waters. A more directed study would be needed to determine if this management action will be successful in addressing the Naples Bay copper impairment.

The City of Naples has developed and implemented a robust water quality and monitoring program. The program provides a consistent dataset that is successful in determining trends over time for many constituents of concern, including regulatory compliance. Enhancement of the program to answer the more complex questions regarding ecological systems interactions would be beneficial to achieve the resource management goals. More robust monitoring of seagrass (including measurements of water clarity and light attenuation, along with aerial mapping of seagrass extent) and oyster mapping and monitoring would link the specific water quality stressors (i.e. loadings; salinity) in space and time to their effect on the biological communities in Naples Bay. This information will allow identification and implementation of specific management activities to improve water quality and biology in Naples Bay.

This effort was successful in identifying statistically significant trends in water quality and biology in Naples Bay that will be useful to resource managers. The characterization of the current biological community provides a baseline for future management actions to measure progress and restoration goals. Specific recommendations for the ongoing monitoring programs to enhance understanding of the factors that affect water quality and biological communities in Naples Bay are provided in the accompanying document "*Naples Bay Monitoring Design*."

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Naples Bay Water Quality and Biological Analysis

APPENDIX

# A

RAW DATA SUMMARY

# Appendix A
# Raw Data Summary

## A.1 Stormwater Lake and Pump Station Results

**Table A-1. Summary of Raw Stormwater Lake Water Quality Data, City of Naples, December 2010–February 2015.**

| Lake Number | Parameter | Dec-10 | Mar-11 | Jun-11 | Sep-11 | Apr-12 | Sep-12 | May-13 | Nov-13 | Dec-14 | Feb-15 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 SE-B | Copper | | | | | | 17.0 | 45.6 | 11.4 | 127.0 | 40.6 |
| | Total Nitrogen | | | | | | 0.75 | 1.04 | 1.181 | 1.44 | 1.06 |
| | Total Phosphorus | | | | | | 0.08 | 0.07 | 0.09 | 0.09 | 0.03 |
| | Suspended Solids | | | | | | 24.0 | 13.0 | 1.6 | 1.6 | 1.0 |
| | Fecal Coliform | | | | | | 231 | 1000 | 40 | 290 | 50 |
| | Enterococci | | | | | | 100 | 80 | 14 | 50 | 80 |
| 2B | Copper | 5.2 | 63.0 | 14.0 | 3.2 | 12.0 | 6.2 | 7.0 | 4.2 | 3.5 | 11.0 |
| | Total Nitrogen | 0.48 | 0.26 | 0.93 | 1.70 | 1.20 | 0.85 | 0.98 | 0.87 | 0.75 | 0.98 |
| | Total Phosphorus | 0.03 | 0.05 | 0.05 | 0.09 | 0.10 | 0.05 | 0.08 | 0.14 | 0.10 | 0.12 |
| | Suspended Solids | 11.0 | 20.0 | 25.0 | 11.0 | 808.0 | 6.4 | 7.5 | 4.4 | 6.9 | 5.3 |
| | Fecal Coliform | 62 | 40 | 190 | 673 | 180 | 1840 | 3900 | 132 | 20 | 80 |
| | Enterococci | 961 | 1010 | 1990 | 100 | 461 | 961 | 400 | 17 | 40 | 10 |
| 3B | Copper | | | | | 5.6 | 2.8 | 10.3 | 3.5 | 6.5 | 14.3 |
| | Total Nitrogen | | | | | 1.10 | 1.00 | 1.05 | 1.12 | 1.13 | 0.95 |
| | Total Phosphorus | | | | | 0.11 | 0.13 | 0.14 | 0.12 | 0.17 | 0.08 |
| | Suspended Solids | | | | | 4.8 | 5.2 | 4.8 | 1.7 | 7.9 | 6.2 |
| | Fecal Coliform | | | | | 1440 | 259 | 3300 | 250 | 10 | 120 |
| | Enterococci | | | | | 140 | 47 | 156 | 33 | 2200 | 110 |
| 5B | Copper | 12.0 | 8.6 | 6.1 | 5.0 | 10.0 | 3.0 | 35.0 | 6.1 | 30.1 | 13.7 |
| | Total Nitrogen | 1.20 | 0.92 | 1.30 | 1.10 | 5.30 | 0.89 | 1.14 | 1.18 | 1.69 | 1.26 |
| | Total Phosphorus | 0.13 | 0.13 | 0.14 | 0.10 | 0.42 | 0.12 | 0.15 | 0.13 | 0.22 | 0.13 |
| | Suspended Solids | 5.2 | 4.0 | 2.8 | 4.0 | 17.0 | 4.8 | 4.8 | 5.6 | 11.0 | 10.2 |
| | Fecal Coliform | 88 | 58 | 220 | 200 | 270 | 310 | 188 | 44 | 20 | 70 |
| | Enterococci | 61 | 56 | 56 | 123 | 84 | 7 | 600 | 71 | 10 | 90 |
| 15B | Copper | 3.9 | 4.3 | 20.0 | 11.0 | 41.0 | 8.2 | 8.6 | 27.7 | 8.5 | 15.1 |
| | Total Nitrogen | 0.90 | 0.94 | 0.82 | 0.99 | 1.20 | 0.89 | 0.99 | 1.29 | 1.09 | 0.87 |
| | Total Phosphorus | 0.03 | 0.03 | 0.03 | 0.02 | 0.02 | 0.03 | 0.03 | 0.07 | 0.08 | 0.02 |

| Lake Number | Parameter | Dec-10 | Mar-11 | Jun-11 | Sep-11 | Apr-12 | Sep-12 | May-13 | Nov-13 | Dec-14 | Feb-15 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Suspended Solids | 5.6 | 5.2 | 4.4 | 2.8 | 4.4 | 4.8 | 7.3 | 1.3 | 4.0 | 2.6 |
| | Fecal Coliform | 380 | 86 | 755 | 200 | 100 | 230 | 2000 | 46 | 480 | 190 |
| | Enterococci | 204 | 83 | 579 | 29 | 46 | 17 | 85 | 8 | 140 | 260 |
| 19B | Copper | | | | | 1.2 | 0.4 | 0.5 | 0.5 | 2.0 | 7.4 |
| | Total Nitrogen | | | | | 2.40 | 1.20 | 1.21 | 1.28 | 1.24 | 1.37 |
| | Total Phosphorus | | | | | 0.06 | 0.05 | 0.05 | 0.14 | 0.24 | 0.13 |
| | Suspended Solids | | | | | 4.4 | 8.4 | 10.7 | 15.2 | 3.6 | 9.8 |
| | Fecal Coliform | | | | | 180 | 410 | 92 | 3 | 5 | 60 |
| | Enterococci | | | | | 313 | 27 | 298 | 6 | 40 | 200 |
| 6B | Copper | | | | | 0.6 | 0.5 | 0.5 | 0.7 | 2.0 | 9.4 |
| | Total Nitrogen | | | | | 0.83 | 1.20 | 0.97 | 0.73 | 0.93 | 1.38 |
| | Total Phosphorus | | | | | 0.05 | 0.13 | 0.26 | 0.03 | 0.12 | 0.11 |
| | Suspended Solids | | | | | 2.4 | 11.0 | 18.1 | 4.2 | 5.0 | 5.2 |
| | Fecal Coliform | | | | | 50 | 5200 | 96 | 133 | 70 | 90 |
| | Enterococci | | | | | 9 | 101 | 331 | 30 | 30 | 80 |
| 20B | Copper | 2.5 | 1.0 | 1.0 | 0.4 | 0.6 | 0.9 | 1.5 | 0.5 | 2.0 | 8.0 |
| | Total Nitrogen | 1.50 | 1.40 | 2.10 | 1.60 | 1.60 | 1.80 | 1.23 | 4.12 | 3.41 | 6.69 |
| | Total Phosphorus | 0.09 | 0.22 | 0.09 | 0.10 | 0.06 | 0.07 | 0.06 | 0.40 | 0.32 | 0.42 |
| | Suspended Solids | 6.0 | 20.0 | 11.0 | 14.0 | 8.4 | 13.0 | 69.7 | 8.0 | 26.4 | 56.0 |
| | Fecal Coliform | 370 | 118 | 520 | 410 | 50 | 4000 | 72 | 28 | 60 | 470 |
| | Enterococci | 111 | 164 | 1300 | 365 | 29 | 2420 | 57 | 1 | 40 | 170 |
| 22B | Copper | 1.0 | 2.6 | 1.4 | 0.8 | 1.1 | 0.6 | 0.7 | 0.8 | 2.0 | 13.5 |
| | Total Nitrogen | 0.63 | 0.70 | 0.60 | 0.66 | 0.85 | 0.85 | 0.65 | 0.82 | 0.87 | 1.24 |
| | Total Phosphorus | 0.06 | 0.06 | 0.04 | 0.04 | 0.01 | 0.10 | 0.06 | 0.12 | 0.17 | 0.11 |
| | Suspended Solids | 1.0 | 3.2 | 2.4 | 4.0 | 1.2 | 8.8 | 3.9 | 1.3 | 10.7 | 2.3 |
| | Fecal Coliform | 208 | 200 | 1660 | 1750 | 50 | 2340 | 128 | 54 | 90 | 150 |
| | Enterococci | 63 | 201 | 461 | 1300 | 8 | 378 | 132 | 35 | 140 | 130 |
| 9B | Copper | | | | | 11.0 | 3.1 | 54.6 | 164.0 | 4.8 | 47.2 |
| | Total Nitrogen | | | | | 1.30 | 1.10 | 3.78 | 1.22 | 1.28 | 3.23 |
| | Total Phosphorus | | | | | 0.17 | 0.05 | 0.42 | 0.08 | 0.24 | 0.56 |
| | Suspended Solids | | | | | 6.0 | 16.0 | 34.0 | 3.6 | 5.2 | 57.0 |
| | Fecal Coliform | | | | | 50 | 66 | 400 | 13 | 20 | 160 |
| | Enterococci | | | | | 34 | 49 | 208 | 4 | 40 | 130 |
| 10B | Copper | | 7.8 | 0.8 | | 1.9 | 1.8 | 3.9 | 1.2 | 0.6 | 1.0 |
| | Total Nitrogen | | 0.70 | 1.10 | | 1.60 | 1.10 | 1.83 | 1.64 | 1.10 | 1.05 |

| Lake Number | Parameter | Dec-10 | Mar-11 | Jun-11 | Sep-11 | Apr-12 | Sep-12 | May-13 | Nov-13 | Dec-14 | Feb-15 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Total Phosphorus | | 0.06 | 0.04 | | 0.10 | 0.03 | 0.11 | 0.09 | 0.17 | 0.13 |
| | Suspended Solids | | 26.0 | 22.0 | | 9.6 | 8.0 | 79.4 | 11.4 | 6.3 | 3.9 |
| | Fecal Coliform | | 40 | 40 | | 721 | 374 | 23 | 128 | 20 | 5 |
| | Enterococci | | 2420 | 100 | | 182 | 186 | 1 | 81 | 50 | 10 |
| 26B | Copper | | | | | 57.0 | 61.0 | 55.4 | 76.7 | 73.2 | 50.0 |
| | Total Nitrogen | | | | | 0.59 | 0.76 | 0.87 | 0.56 | 1.26 | 1.21 |
| | Total Phosphorus | | | | | 0.04 | 0.07 | 0.06 | 0.03 | 0.13 | 0.06 |
| | Suspended Solids | | | | | 1.6 | 6.0 | 8.2 | 6.8 | 11.6 | 5.2 |
| | Fecal Coliform | | | | | 180 | 890 | 290 | 42 | 90 | 90 |
| | Enterococci | | | | | 68 | 2 | 1 | 35 | 200 | 140 |
| 11B | Copper | | | | | 4.9 | 3.0 | 5.0 | 10.6 | 6.3 | 13.0 |
| | Total Nitrogen | | | | | 1.20 | 0.99 | 0.84 | 0.67 | 0.87 | 0.84 |
| | Total Phosphorus | | | | | 0.06 | 0.11 | 0.10 | 0.09 | 0.16 | 0.08 |
| | Suspended Solids | | | | | 3.6 | 3.6 | 10.4 | 1.6 | 4.1 | 5.1 |
| | Fecal Coliform | | | | | 50 | 489 | 645 | 132 | 560 | 350 |
| | Enterococci | | | | | 93 | 194 | 649 | 40 | 160 | 290 |
| 14B | Copper | | | | | 3.4 | 2.3 | 5.0 | 2.7 | 7.1 | 11.9 |
| | Total Nitrogen | | | | | 0.76 | 1.90 | 1.56 | 1.16 | 1.32 | 1.25 |
| | Total Phosphorus | | | | | 0.89 | 0.22 | 0.54 | 0.21 | 0.46 | 0.70 |
| | Suspended Solids | | | | | 7.2 | 14.0 | 28.0 | 23.2 | 16.0 | 9.3 |
| | Fecal Coliform | | | | | 50 | 1 | 178 | 430 | 80 | 210 |
| | Enterococci | | | | | 372 | 142 | 1120 | 164 | 140 | 330 |
| 24B | Copper | | | | | | 2.4 | 14.9 | 17.5 | 4.6 | 4.4 |
| | Total Nitrogen | | | | | | 2.70 | 3.33 | 2.35 | 3.26 | 3.26 |
| | Total Phosphorus | | | | | | 1.30 | 0.62 | 1.42 | 2.38 | 1.56 |
| | Suspended Solids | | | | | | 14.0 | 82.0 | 30.0 | 10.6 | 28.5 |
| | Fecal Coliform | | | | | | 3200 | 520 | 76 | 270 | 130 |
| | Enterococci | | | | | | 42 | 980 | 132 | 360 | 200 |

**Table A-2. Summary of Raw Pump Station Water Quality Data, City of Naples, December 2010–February 2015.**

| Pump Station | Parameter | Dec-10 | Mar-11 | Jun-11 | Sep-11 | Apr-12 | Jul-12 | Sep-12 | Dec-12 | May-13 | Jun-13 | Aug-13 | Nov-13 | Feb-14 | Dec-14 | Feb-15 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| PW-Pump | Copper | 8.6 | | | | 2.0 | 8.2 | 38.0 | 1.3 | 6.3 | | 14.6 | 3.9 | 6.9 | 10.4 | 21.4 |
| | Total Nitrogen | 1.60 | | | | 1.30 | 1.19 | 1.10 | 1.40 | 1.17 | | 1.08 | 1.17 | | 1.90 | 1.49 |
| | Total Phosphorus | 0.22 | | | | 0.07 | 0.08 | 0.09 | 0.10 | 0.08 | | 0.17 | 0.09 | 0.09 | 0.27 | 0.10 |
| | Suspended Solids | 8.0 | | | | 2.8 | 7.6 | 4.8 | 1.2 | 1.3 | | 1.8 | | 1.2 | 2.1 | 4.0 |
| | Fecal Coliform | 855 | | | | 3400 | 1980 | 4200 | 5200 | 18 | | 470 | 5400 | 800 | 20 | 110 |
| | *Enterococci* | 1300 | | | | 870 | 500 | 516 | 437 | 594 | | 140 | 49 | 5400 | 200 | 790 |
| 11-Pump | Copper | 1.4 | 1.6 | 2.7 | 1.2 | 1.7 | 2.9 | 3.2 | 1.1 | | 1.4 | 1.7 | 2.2 | 1.0 | 2.0 | 15.5 |
| | Total Nitrogen | 1.50 | 1.70 | 1.80 | 1.30 | 1.60 | 1.52 | 1.80 | 1.81 | | 1.31 | 1.26 | 1.54 | | 1.56 | 1.57 |
| | Total Phosphorus | 0.13 | 0.11 | 0.21 | 0.15 | 0.12 | 0.14 | 0.60 | 0.13 | | 0.12 | 0.15 | 0.12 | 0.14 | 0.21 | 0.12 |
| | Suspended Solids | 1.6 | 1.6 | 13.0 | 2.8 | 3.6 | 4.0 | 5.2 | 2.8 | | 2.5 | 1.1 | | 1.2 | 0.7 | 0.6 |
| | Fecal Coliform | 390 | 82 | 18700 | 11200 | 9910 | 112000 | 4700 | 450 | | 61 | 210 | 115 | 918 | 50 | 350 |
| | *Enterococci* | 215 | 1000 | 510 | 1800 | 1730 | 200 | 127 | 501 | | 472 | 60 | 961 | 96100 | 540 | 2900 |
| 14-Pump | Copper | 5.0 | | | | 2.9 | 45.0 | 3.6 | 2.2 | | 3.2 | 2.5 | 3.9 | 3.0 | 5.6 | 2.1 |
| | Total Nitrogen | 1.60 | | | | 1.10 | 0.91 | 1.10 | 1.92 | | 0.79 | 1.22 | 1.50 | | 1.75 | 1.81 |
| | Total Phosphorus | 0.48 | | | | 0.83 | 0.15 | 0.16 | 0.40 | | 0.15 | 0.21 | 0.26 | 0.68 | 0.55 | 0.55 |
| | Suspended Solids | 2.4 | | | | 4.8 | 54.0 | 74.0 | 4.0 | | 26.3 | 8.6 | 8.6 | 3.1 | 8.6 | 1.3 |
| | Fecal Coliform | 360 | | | | 4000 | 1350 | 220 | 360 | | 2000 | 800 | 16 | 918 | 330 | 2300 |
| | *Enterococci* | 1730 | | | | 300 | 1200 | 333 | 550 | | 1400 | 3400 | 961 | 96100 | 880 | 2200 |

Naples Bay Water Quality and
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# APPENDIX B

## SUMMARY OF STATISTICAL ANALYSES

# Appendix B
# Summary of Statistical Analyses

## B.1 Water Quality Statistical Analyses

### B.1.1 Salinity/Flow Model

Cardno developed a model designed to evaluate the effect of freshwater flow emanating from the Golden Gate Canal on salinity concentrations at downstream areas (see Section 3.2.1). Sensitivity testing of several time series model forms was implemented, including the use of autoregressive integrated moving average (ARIMA) models, autoregressive error models, and general linear models with trigonometric functions. We also evaluated the use of daily, weekly, and monthly data. The final model was fit to three years of monthly data (8/2011–7/2014, n = 36) for each of four downstream locations (Gordon River, City Dock, Mid-estuary, and Gordon Pass). The use of monthly data effectively smoothed the model development data set and provided a reasonable model fit for all areas.

The time series data for flow and salinity showed strong lag one autocorrelation (i.e., the value at time t is correlated with the value at time t-1) with the daily data, but there were a great deal of missing information (i.e., measurements on consecutive days) using either monthly or weekly data points. Using monthly means negated the need for a lagged relationship between salinity and flow, and resulted in a reasonable model. We discovered that the best model for the three years of monthly data was simply:

$$\text{Salinity}_t = B_0 + B_1 * \ln(\text{flow}_t) + E_t$$

Effectively, the model predicts salinity at any month (t) as a function of the natural log of flow in the month. The degree of response that salinity has to flow decreases as the distance from GGC increases. Graphics illustrating the predictive ability of the models for each area are below.

The model was developed to estimate the change in salinity in Naples Bay as a result of potential GGC flow reduction scenarios. Three scenarios were chosen to represent a 30, 50, and 70 percent reduction in GGC flow. Graphics of the model results are also provided below.

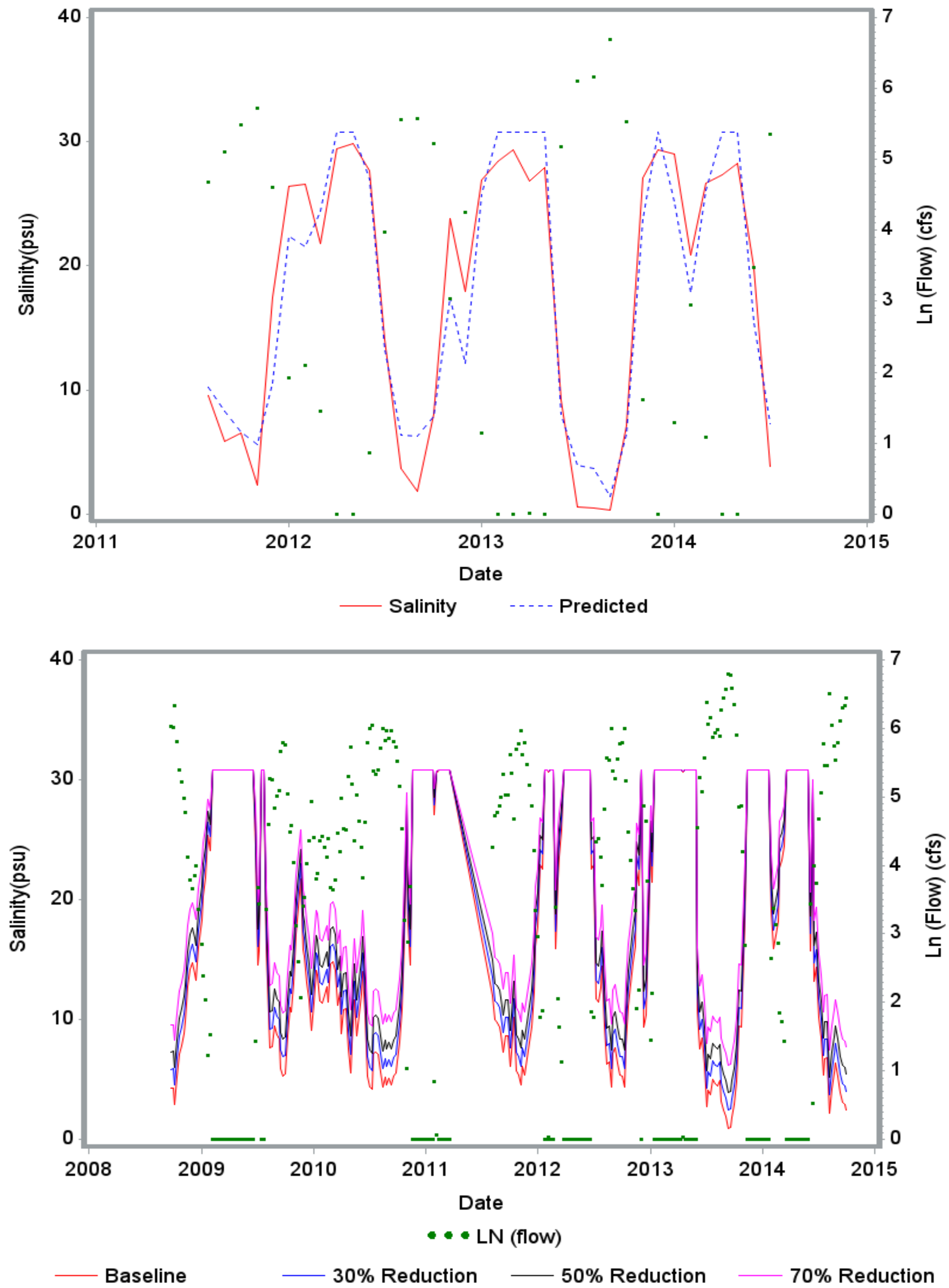
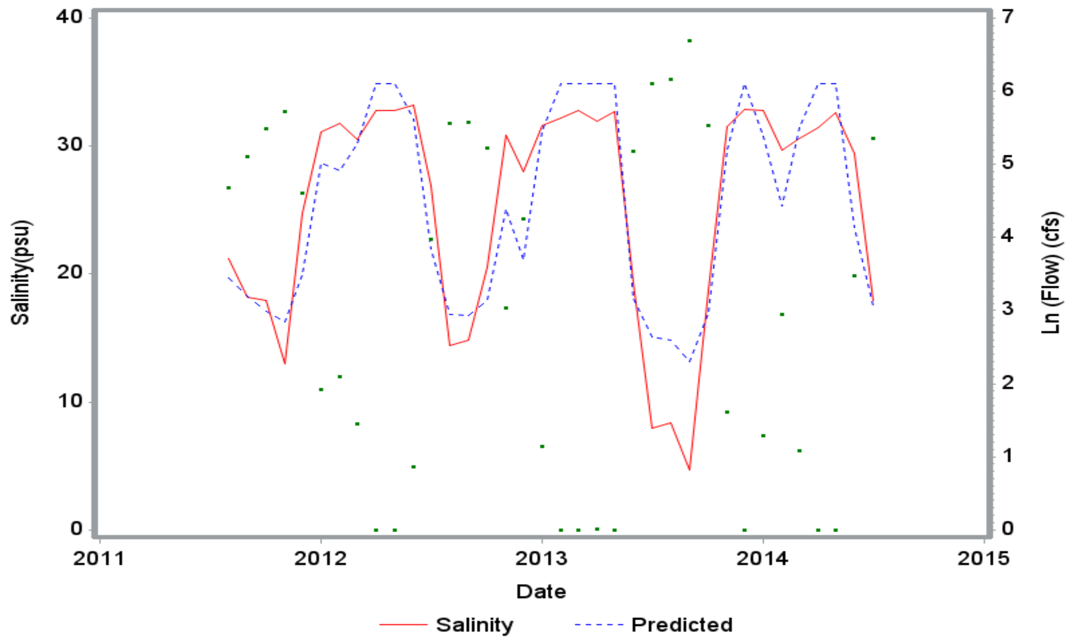



**Figure B-1.** **Relationship of observed and predicted salinity, including flow magnitude (green dots) for Gordon River (top). Model results showing estimated salinity from flow reduction scenarios (bottom).**



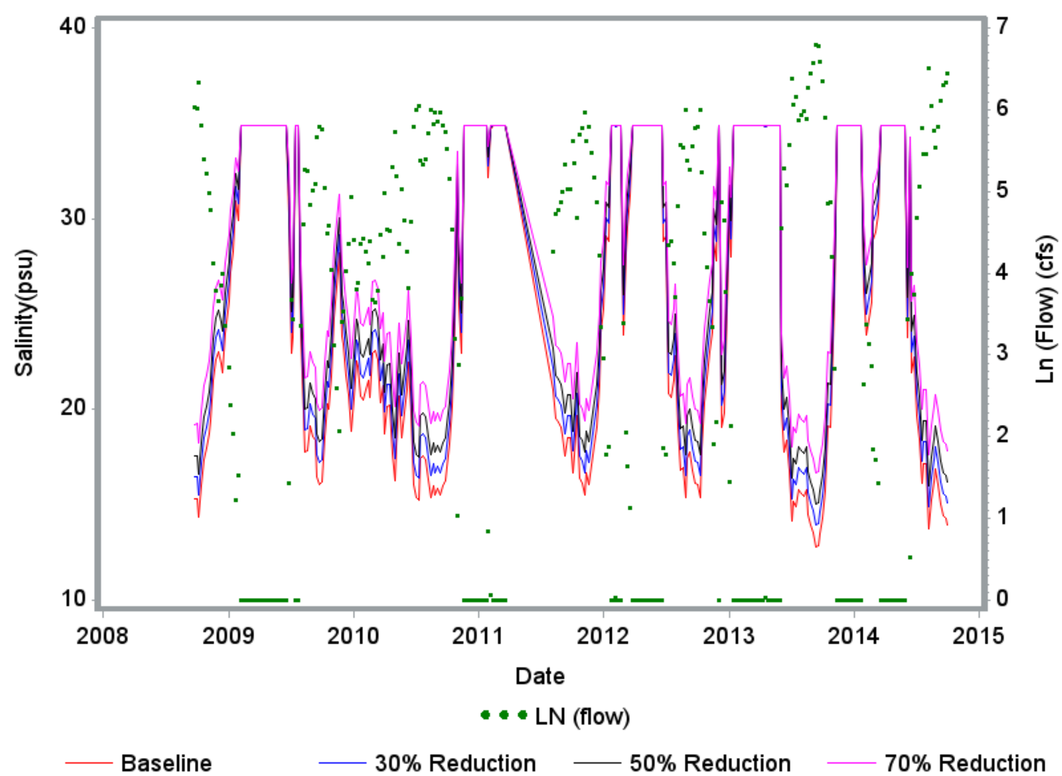
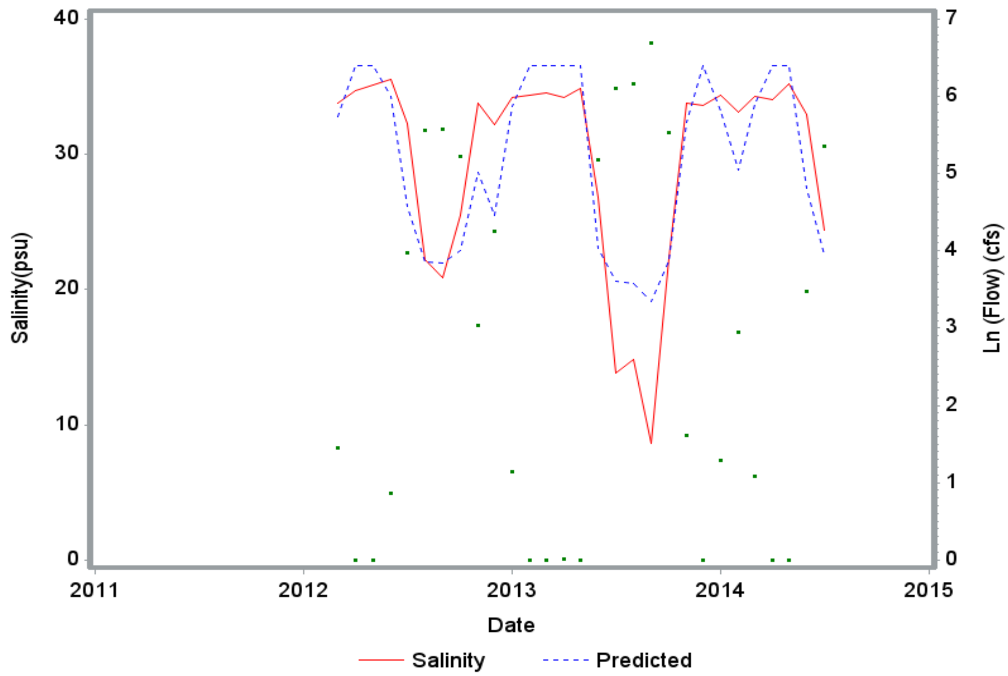




**Figure B-2.** **Relationship of observed and predicted salinity, including flow magnitude (green dots) for Naples Bay at City Dock (top). Model results showing estimated salinity from flow reduction scenarios (bottom).**



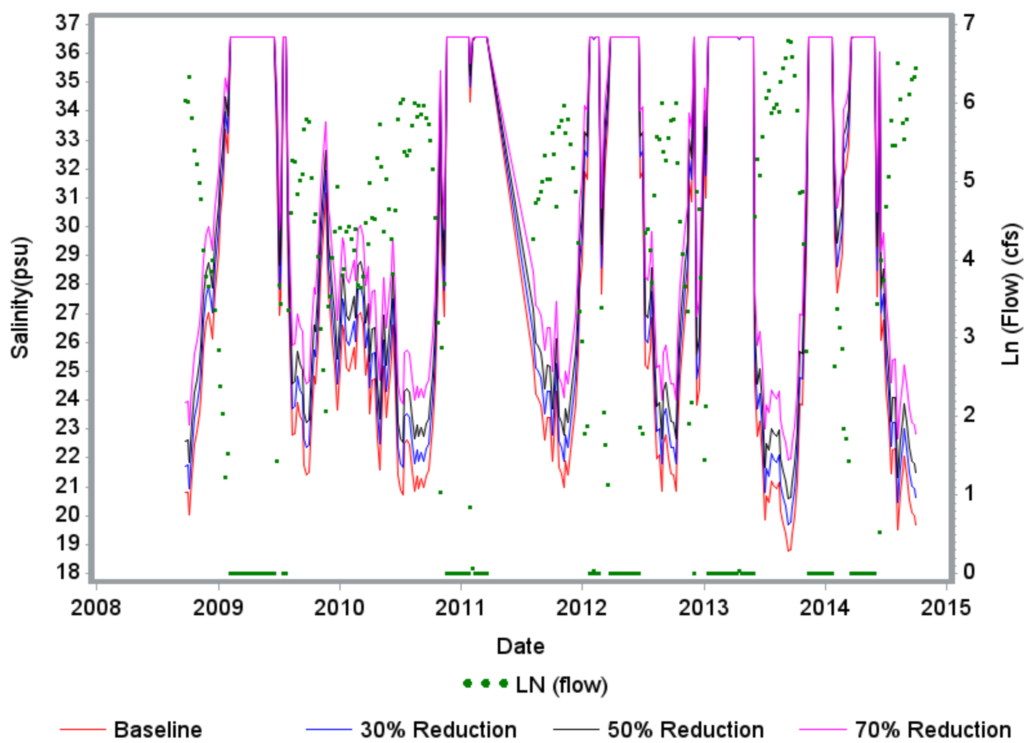
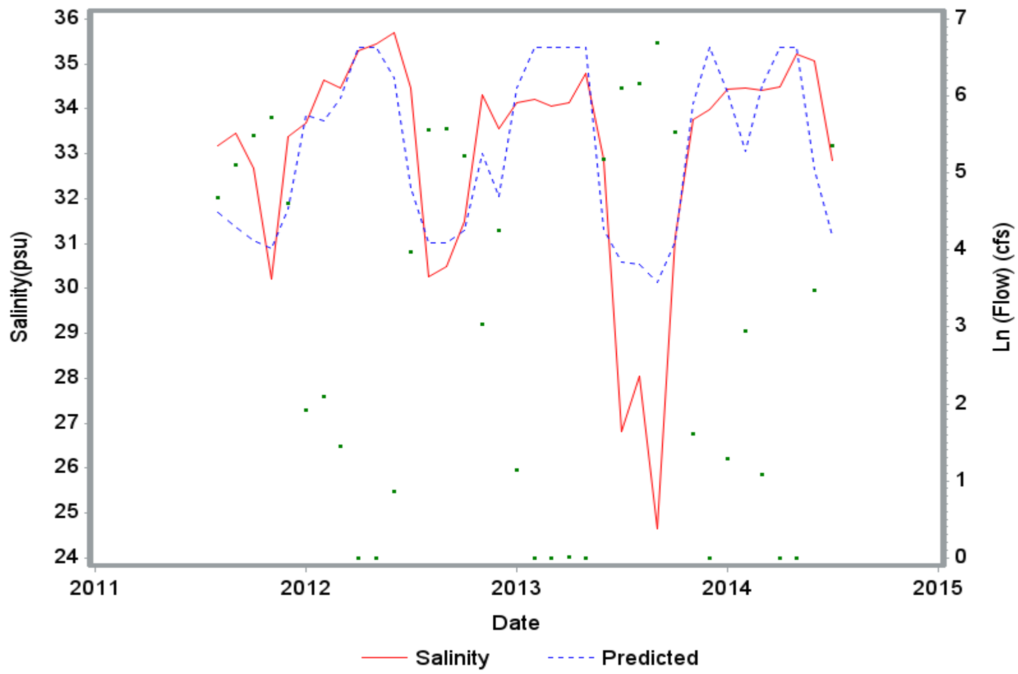




**Figure B-3. Relationship of observed and predicted salinity, including flow magnitude (green dots) for Naples Bay Mid Estuary (top). Model results showing estimated salinity from flow reduction scenarios (bottom).**



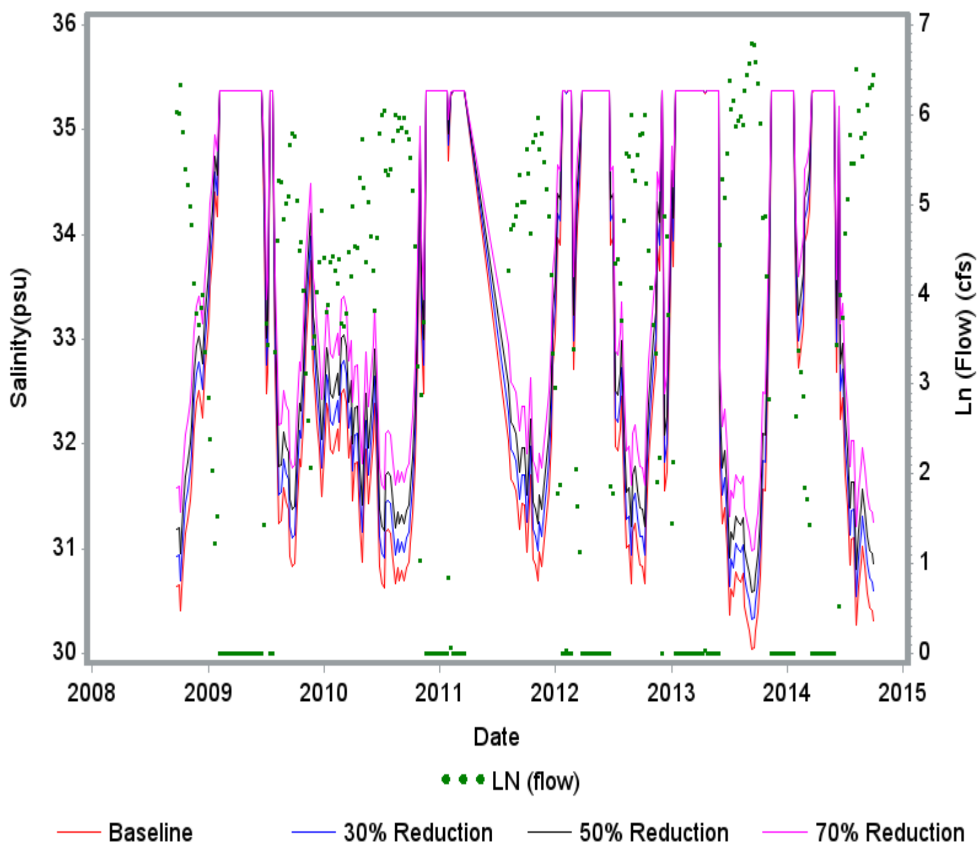




**Figure B-4.** **Relationship of observed and predicted salinity, including flow magnitude (green dots) for Naples Bay at Gordon Pass (top). Model results showing estimated salinity from flow reduction scenarios (bottom).**

## B.2 Biological Statistical Analyses

### B.2.2 Seagrass BayesNet Analytical Framework

A Bayesian network (BN) is a tool for linking multiple lines of information, and examining the strength of complex environmental and effects-based relationships. A BN can be thought of as a graphical model with a series of nodes linked by arrows, where the arrows in a BN represent probabilities. The arrows indicate causal linkages among the nodes, and the nodes denote important system attributes. Each node is characterized by probabilities or probabilistic mathematical expressions that represent knowledge about these system attributes. The mathematical expressions may be 1) mechanistic descriptions such as chemical reaction kinetics, 2) empirical relationships such as linear regression models, or 3) relationships derived from expert judgment, depending on how much information we have about the relationships characterizing a particular node. The possible outcomes at each node are expressed probabilistically; thus a Bayes net is a set of conditional probabilities describing a set of likely system responses. The ability to incorporate mechanistic, empirical, and judgmental information makes the BN approach extremely flexible and facilitates an extension to non-traditional model endpoints (e.g., seagrass biomass) of public concern.

Bayesian model building often begins with a graphical model that consists of boxes and arrows characterizing key relationships (see Figure 5-1). The lines of evidence are complex, with several intertwining interactions finally leading to a possible effect on seagrass. By displaying the conceptual model in this fashion, changes in seagrass caused by sources other than water quality can be evaluated. The advantage of this approach, relative to standard regression models, is that each link within a complex ecological systems can be modeled.

**Model Implementation**

An appropriate data set for implementing the Bayes Net consists of columns containing information for each node of the net, and rows representing repeated measures of the each variable. Combining the various variables of interest can be challenging, in that each row should represent consistent information in both space and time. Information available to Cardno does not represent all of the conceptual model nodes shown above. However, we combined available data and believe the information is suitable for a draft assessment of the Naples Bay ecosystem, at least as a generalized approach.

Using the available data, we constructed and implemented several versions of possible decision diagrams. Sensitivity testing of the possible models resulted in the following final BN model:

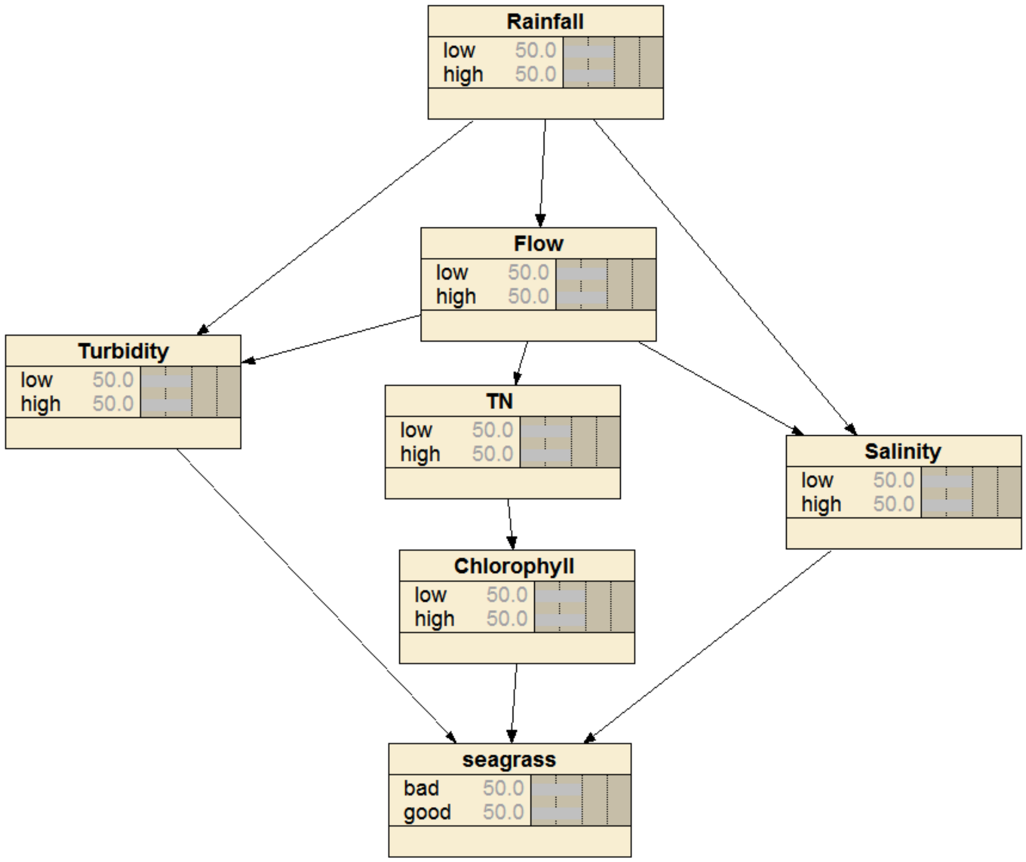



**Figure B-5. Reduced Naples Bay model focused on available information from the Naples Bay monitoring program and other sources.**

Once a reasonable and functional conceptual model is developed (Figure B-5), each node in the model must be categorized (termed *state* generation). The categorization process reflects the statistical concepts inherent in conditional probability (and not discussed here), but from a practical perspective, each node is represented by a probability distribution and an associated set of states, where each state has a probability of occurrence. Cardno used a simple approach for generating these distributions and states. As shown in Figure B-5, two possible states (high and low) were assigned to each node (or variable). Before examination of the actual Naples data, the probability that each state occurred in nature was assumed uniform (i.e., 50 percent for each of the 2 states). In other words, no prior understanding of the probability that each state occurred was used in the modeling process. The uniform probability are shown in Figure B-5, with the probability of each state shown to be 50(percent).

Prior to implementing the BN, each of the states (low and high) for each node is defined based on data ranges specifically for each variable. The following ranges were established for each node/state combination:

**Rainfall**
Low: 0–15 inches
High 15–35 inches

**Flow**
Low: 0–100 cfs
High: 100–800 cfs

**Turbidity**
Low: 0–20 NTU
High: 20–100 NYU

**Total Nitrogen**
Low: 0–0.57 mg/L
High: 0.57–1.1 mg/L

**Salinity**
Low: 0–25 ppt
High: 25–40 ppt

Chlorophyll
Low: 0–4.5 µg/L
High: 4.5–11 µg/L

**Seagrass biomass**
Bad: 0–211 counts (measured as number of shoots)
Good: 211–2000 counts (measured as number of shoots)

The Figure B-5 Bayes Net was implemented using the above data and model setup. The results of the model are shown in Figure B-6. The probability of each parent node reflect the likelihood of occurrence for each state conditional on the data. The probability of each child node reflects the probability of the parent nodes, and the information in the data. Therefore, the shown probabilities reflect the information in all linked parent nodes as well as the likelihood of the data as found in the data set. The child node reflects the complex interactions in the linkages up the decision tree. At the bottom of each node, the weighted mean of the variable (e.g., the weighted mean of rainfall is 17.6, where the weights are taken to be the state-specific probabilities).

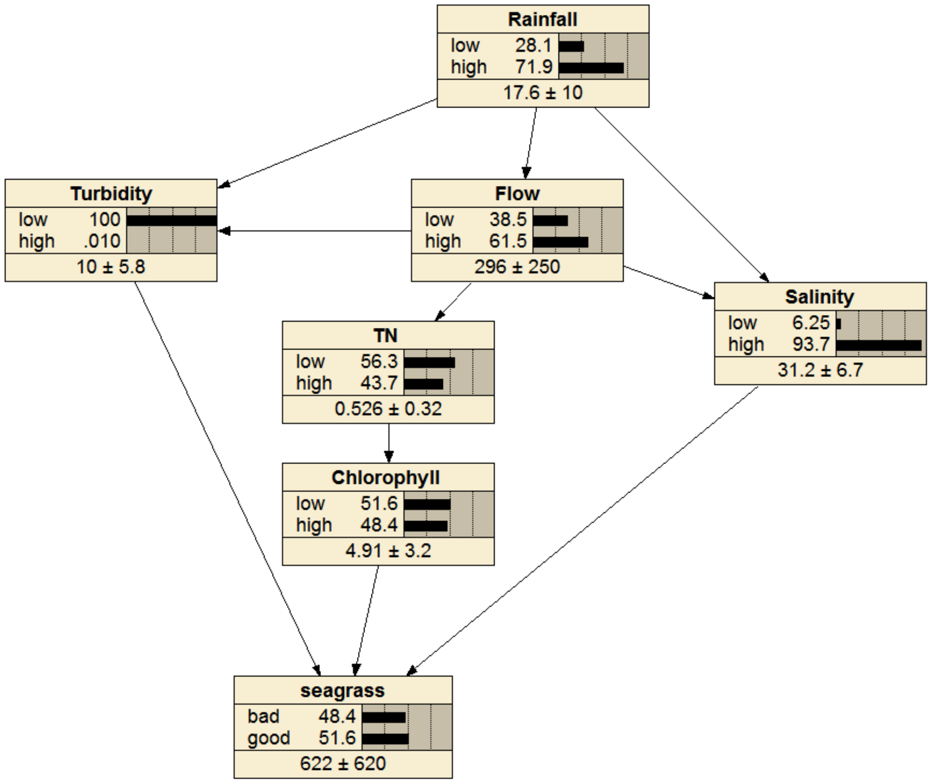
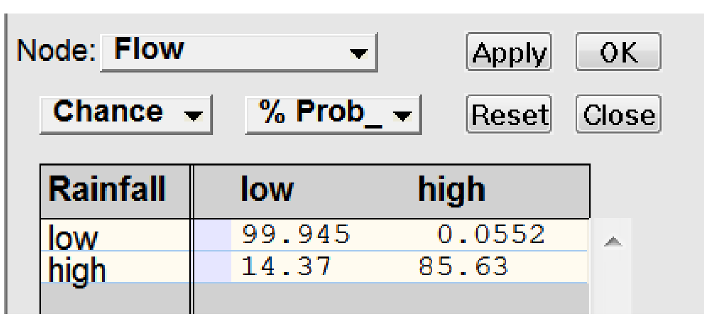



**Figure B-6.** **Final probability for each node/state combination conditional of the Naples Bay monitoring data.**

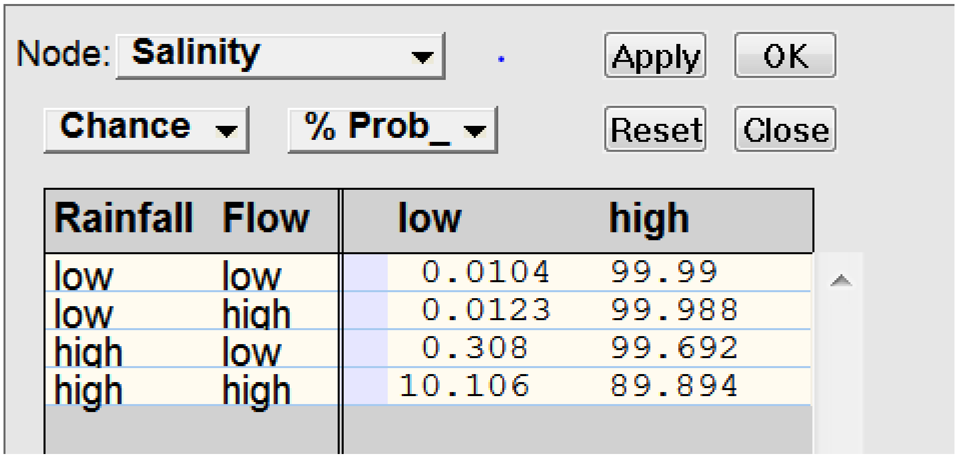
Below, we illustrate and interpret the findings at selected nodes. The child node, flow, is shown in Figure B-7 We use this figure to illustrate how to interpret the findings. When rainfall is low the probability of low flow is 99.9 percent. When rainfall is high, the probability of low flow changes to 14.4 percent, while the probability of high flow increases to 85.6 percent. In other words, rainfall and flow are correlated, but are not directly proportional.




| Rainfall | low | high |
|---|---|---|
| low | 99.945 | 0.0552 |
| high | 14.37 | 85.63 |

**Figure B-7.** **Table of probabilities for the child node flow.**

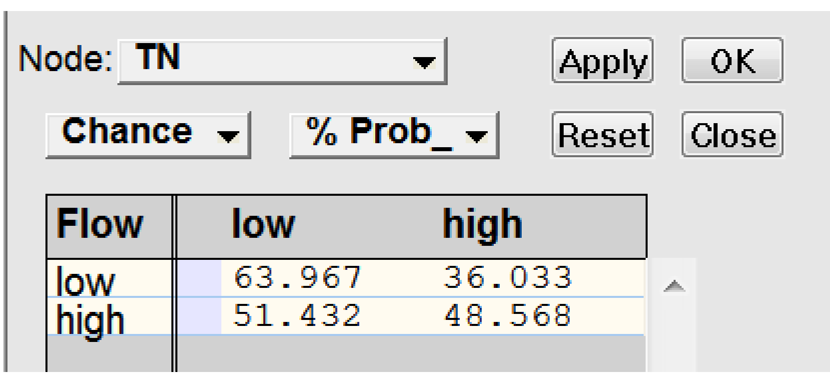
Figure B-8 shows the table for the child node salinity. Salinity has two inputs, rainfall and flow. The joint probability of rainfall and flow, along with information on salinity from the data set, result in the probabilities shown in Figure B-8. Generally, rainfall and flow states have little impact on the expected salinity state. This finding is consistent with the flow-salinity regression model described in Section 3.2.2, where the impact of freshwater inflow is shown to be only marginally correlated with salinity concentrations in the southern part of Naples Bay.




| Rainfall | Flow | low | high |
|---|---|---|---|
| low | low | 0.0104 | 99.99 |
| low | high | 0.0123 | 99.988 |
| high | low | 0.308 | 99.692 |
| high | high | 10.106 | 89.894 |

**Figure B-8. Table of probabilities for the child node salinity**

Figure B-9 displays the probability table for the child node total nitrogen (TN). As shown in the table, the effect of flow on expected nitrogen levels is marginal, with only a 10 percent or so change in probability due to increasing high flows.




| Flow | low | high |
|---|---|---|
| low | 63.967 | 36.033 |
| high | 51.432 | 48.568 |

**Figure B-9. Table of probabilities for the child node total nitrogen.**

Seagrass biomass and population status is represented in Figure B-10. The expected biomass of seagrass is influenced by salinity, chlorophyll a, and turbidity (see Figure B-6). The largest probability of a good seagrass state is associated with salinity = high, turbidity = low, and chlorophyll a = low. And since the effect of flow on salinity in the southern part of Naples is marginal (see Figure B-8), the model does not predict large fluctuations in seagrass biomass based on changes in flow. A state of high chlorophyll a is generally associated with a state of “bad” seagrass biomass, but the degree of association is at best mild.

Node: ShootCount Apply OK

Chance % Prob_ Reset Close

| Salinity | Turbidity | Chlorophyll | bad | good |
|---|---|---|---|---|
| low | low | low | 49.74 | 50.26 |
| low | low | high | 66.692 | 33.308 |
| low | high | low | 52.661 | 47.339 |
| low | high | high | 52.166 | 47.834 |
| high | low | low | 29.976 | 70.024 |
| high | low | high | 66.692 | 33.308 |
| high | high | low | 48.748 | 51.252 |
| high | high | high | 57.885 | 42.115 |

**Figure B-10. Table of seagrass probabilities as a result of the salinity, turbidity, and** chlorophyll ***a*** **parent nodes.**

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Naples Bay Water Quality and
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# APPENDIX C

## SUPPLEMENTAL BIOLOGICAL TABLES AND FIGURES

# Appendix C
# Supplemental Biological Tables and Figures

**Table C-1. Fish taxa* caught in Naples Bay bottom trawls, 2009–2014.**

| Scientific Name | Common Name | Species Code |
|---|---|---|
| *Acanthostracion quadricornis* | Scrawled cowfish | LACT QUAD |
| *Achirus lineatus* | Lined sole | ACHI LINE |
| *Albula vulpes* | Bonefish | ALBU VULP |
| *Anchoa hepsetus* | Striped anchovy | ANCH HEPS |
| *Anchoa mitchilli* | Bay anchovy | ANCH MITC |
| *Anchoa* spp. | Anchovies | ANCH SPP |
| *Ancylopsetta ommata* | Ocellated flounder | ANCL QUAD |
| *Archosargus probatocephalus* | Sheepshead | ARCH PROB |
| *Ariopsis felis* | Hardhead catfish | ARIU FELI |
| *Bagre marinus* | Gaftopsail catfish | BAGR MARI |
| *Bairdiella chrysoura* | Silver perch | BAIR CHRY |
| *Brevoortia smithi* | Yellowfin menhaden | BREV SMIT |
| *Chaetodipterus faber* | Atlantic spadefish | CHAE FABE |
| *Chilomycterus schoepfii* | Striped burrfish | CHIL SCHO |
| *Chilomycterus* spp. | Burrfishes | CHIL |
| *Chloroscombrus chrysurus* | Atlantic bumper | CHLO CHRY |
| *Ctenogobius smaragdus* | Emerald goby | GOBI SMAR |
| *Cynoscion arenarius* | Sand seatrout | CYNO AREN |
| *Cynoscion nebulosus* | Spotted seatrout | CYNO NEBU |
| *Cynoscion* sp. | Seatrout | AYNO F |
| *Dasyatis sabina* | Atlantic stingray | DASY SABI |
| *Etropus crossotus* | Fringed flounder | ETRO CROS |
| *Eucinostomus gula* | Silver jenny | EUCI GULA |
| *Eucinostomus harengulus* | Spotfin mojarra | EUCI HARE |
| *Eucinostomus* spp. | mojarra species | EUCI SPP |
| Family Clupeidae | Herrings | CLUPEIDAE |
| Family Gobiidae | Gobies | GOBI R |
| Family Sciaenidae | Croakers/Drums | SCIAENIDAE |
| *Gobiesox strumosus* | Skilletfish | GOBI STRU |
| *Gobionellus oceanicus* | Highfin goby | GOBI OCEA |
| *Gobiosoma robustum* | Code goby | GOBI ROBU |
| *Gymnura micrura* | Smooth butterfly ray | GYMN MICR |
| *Harengula jaguana* | Scaled sardine | HARE JAGU |
| *Hippocampus erectus* | Lined seahorse | HIPP EREC |

**Table C-1. Fish taxa* caught in Naples Bay bottom trawls, 2009–2014.**

| Scientific Name | Common Name | Species Code |
|---|---|---|
| *Hypsoblennius hentz* | Feather Blenny | HYPS HENT |
| *Lagodon rhomboides* | Pinfish | LAGA RHOM |
| *Leiostomus xanthurus* | Spot | LEIO XANT |
| *Lutjanus griseus* | Mangrove Snapper | LUTJ GRIS |
| *Lutjanus* spp. | Snappers | LUTJ GRIS/APO |
| *Lutjanus synagris* | Lane Snapper | LUTJ SYNA |
| *Menticirrhus americanus* | Southern Kingfish | MENT AMER |
| *Menticirrhus* spp. | Kingfishes | MENT SPP |
| *Microgobius gulosus* | Clown Goby | MICR GULO |
| *Microgobius microlepis* | Banner Goby | MICR MICR??? |
| *Microgobius thalassinus* | Green Goby | MICR THAL |
| *Nicholsina usta* | Emerald Parrotfish | NICH USTA |
| *Ogcocephalus cubifrons* | Polka-Dot Batfish | OGCO CUBI |
| *Ophichthus gomesii* | Shrimp Eel | OPHI GOME |
| *Opisthonema oglinum* | Atlantic Thread Herring | OPIS OGLI |
| *Opsanus beta* | Gulf Toadfish | OPSA BETA |
| *Orthopristis chrysoptera* | Pigfish | ORTH CHRY |
| *Paralichthys albigutta* | Gulf Flounder | PARA ALBI |
| *Prionotus scitulus* | Leopard Searobin | PRIO SCIT |
| *Prionotus tribulus* | Bighead Searobin | PRIO TRIB |
| *Sciaenops ocellata* | Red Drum | SCIA OCEL |
| *Scorpaena brasiliensis* | Barbfish | SCOR BRAS |
| *Selene vomer* | Lookdown | SELE VOME |
| *Sphoeroides nephelus* | Southern Puffer | SPHR NEPH |
| *Sphoeroides spengleri* | Bandtail Puffer | SPHR SPEN |
| *Stephanolepis hispidus* | Planehead Filefish | MONA HISP |
| Suborder Pleuronectoidei | Flatfishes | FLOUNDER? |
| *Symphurus plagiusa* | Blackcheek Tounguefish | SYMP PLAG |
| *Syngnathus louisianae* | Chain Pipefish | SYNG LOUI |
| *Syngnathus scovelli* | Gulf Pipefish | SYNG SCOV |
| *Synodus foetens* | Inshore Lizardfish | SYNO FOET |
| *Trinectes maculatus* | Hogchocker | TRIN MACU |
| *Urophycis floridana* | Southern Hake | UROP FLOR |

*Leptocephalus larvae were not included in analysis presented in this report.

**Table C-2. Invertebrate taxa caught in Naples Bay bottom trawls, 2009–2014.**

| Scientific Name | Common Name | Species Code |
|---|---|---|
| Included in Analysis | | |
| *Callinectes sapidus* | Blue Crab | CALI SAP |
| *Callinectes similis* | Lesser Blue Crab | CALI SIM, CALI SIMILUS, SALI SIM |
| *Farfantepenaeus duorarum* | Pink Shrimp | PENA SPP |
| Family Portunidae | Swimming Crabs | SWIM CRABS |
| Order Teuthida | Squids | SQUID |
| Excluded from Analysis | | |
| *Menippe mercenaria* | Stone Crab | MENI MERC |
| *Aplysia* sp. | Seahares | APLYSIA SEAHARE |
| *Libinia* sp. (?) | Spider Crabs | SPIDER CRAB |
| Order Stomatopoda | Mantis Shrimp | MANTIS SHRIMP |
| Family Xanthidae | Mud Crabs | MUD CRAB, MUD CRABS |
| Superfamily Majoidea | Decorator Crab | DECORATOR CRAB |
| *Melongena corona* | Crown Conch | CROWN CONCH |
| Family Inachidae | Arrow Crabs | ARROW CRAB |
| *Hepatus epheliticus* | Calico Box Crab | CALICO CRAB |
| Superfamily Paguroidea | Hermit Crabs | HERMIT CRAB |
| *Luidia* sp. | Nine-Armed Sea Star | 9 ARM SEA STAR, 9ARM |
| Class Asteroidea | Five-Armed Sea Star | 5 ARM SEA STAR |
| Class Ophiuroidea | Brittle Stars | BRITTLE STAR |
| *Limulus polyphemus* | Atlantic Horseshoe Crab | HORSESHOE CRAB |
| Order Neogastropoda | Whelk Egg Case | WHELK EGG CASE |
| Order Decapoda | Purple Crab | PURPLE CRAB |
| Order Anaspidea | Seahares | SEA HARES |
| *Bursatella leachii* | Ragged Seahare | RAGGED SEA HARES |
| *Aplysia fasciata* | Mottled Seahare | MOTTLED SEA HARE |

**Table C-2. SIMPER two-way (season and zone) results for Naples Bay sampling zones. All taxa that contributed to dissimilarity (up to 90 percent) between groups are listed here in order of greatest contribution to overall difference among groups.**

| Taxon | Average Group Abundance | | | | % Contribution to Dissimilarity | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Zone 1 | Zone 2 | Zone 3 | Zone 4 | Zones 1 & 2 | Zones 1 & 3 | Zones 1 & 4 | Zones 2 & 3 | Zones 2 & 4 | Zones 3 & 4 |
| *Anchoa* | 1.17 | 2.86 | 1.73 | 2.29 | 17.32 | 13.53 | 16.34 | 17.03 | 17.93 | 16.87 |
| *Eucinostomus* | 2.78 | 3.54 | 3.13 | 3.35 | 15.90 | 16.32 | 20.07 | 11.88 | 13.12 | 15.34 |
| *Farfantepenaeus* | 0.83 | 0.95 | 0.90 | 0.93 | 6.78 | 7.02 | 7.95 | 6.30 | 6.87 | 7.06 |
| *Callinectes* | 0.72 | 1.14 | 0.46 | 0.53 | 7.20 | 5.00 | 5.02 | 6.37 | 6.72 | 3.99 |
| *Lagodon* | 0.62 | 0.95 | 0.30 | 0.25 | 6.69 | 4.99 | 4.67 | 5.83 | 5.75 | 2.87 |
| *Lutjanus* | 0.13 | 0.69 | 0.87 | 0.38 | 4.59 | 6.71 | 2.82 | 5.56 | 4.51 | 6.25 |
| *Ariopsis* | 0.54 | 0.54 | 0.32 | 0.41 | 4.79 | 4.85 | 4.97 | 3.59 | 3.91 | 3.42 |
| *Synodus* | 0.23 | 0.75 | 0.50 | 0.42 | 4.96 | 3.94 | 3.51 | 4.35 | 4.73 | 4.00 |
| Order Teuthida | 0.04 | 0.43 | 0.46 | 0.30 | 2.88 | 3.20 | 2.29 | 4.30 | 3.60 | 4.02 |
| *Cynoscion* | 0.34 | 0.38 | 0.34 | 0.27 | 3.37 | 3.52 | 3.26 | 3.23 | 2.89 | 3.00 |
| *Bairdiella* | 0.02 | 0.42 | 0.30 | 0.18 | 2.87 | 1.72 | 1.10 | 3.72 | 3.20 | 2.38 |
| *Ogcocephalus* | 0.00 | 0.04 | 0.50 | 0.15 | | 4.38 | 1.00 | 3.77 | 1.05 | 4.27 |
| *Prionotus* | 0.14 | 0.14 | 0.30 | 0.39 | 1.29 | 2.19 | 2.73 | 1.93 | 2.36 | 3.07 |
| *Etropus* | 0.00 | 0.02 | 0.43 | 0.44 | | 2.41 | 2.34 | 2.48 | 2.41 | 3.47 |
| *Leiostomus* | 0.11 | 0.40 | 0.40 | 0.04 | 2.23 | 1.96 | | 3.02 | 1.98 | 1.72 |
| *Symphurus* | 0.08 | 0.19 | 0.22 | 0.25 | 1.21 | 1.24 | 1.60 | 1.69 | 2.04 | 2.08 |
| *Orthopristis* | 0.06 | 0.37 | 0.08 | 0.11 | 2.32 | | 0.90 | 2.30 | 2.52 | 1.10 |
| Family Portunidae | 0.00 | 0.03 | 0.21 | 0.29 | | 1.31 | 1.31 | 1.37 | 1.42 | 2.29 |
| *Achirus* | 0.29 | 0.05 | 0.00 | 0.04 | 2.06 | 1.88 | 2.21 | | | |
| *Microgobius* | 0.06 | 0.06 | 0.05 | 0.13 | 1.29 | 1.12 | 1.53 | | 1.00 | |
| *Archosargus* | 0.12 | 0.07 | 0.04 | 0.04 | 1.18 | 1.05 | 1.20 | | | |
| *Chloroscombrus* | 0.04 | | 0.11 | 0.08 | | 1.04 | 1.00 | | | 1.23 |
| *Sciaenops* | 0.23 | 0.00 | 0.00 | 0.00 | 1.01 | 1.01 | 1.04 | | | |
| *Opsanus* | 0.08 | 0.13 | | 0.09 | 0.93 | | 0.89 | | 1.08 | |
| *Harengula* | | 0.04 | 0.16 | 0.09 | | | | 0.88 | | 1.19 |
| *Menticirrhus* | | | 0.10 | 0.08 | | | | | | 1.04 |
| *Gobiosoma* | 0.15 | | | 0.02 | | | 0.87 | | | |
| *Syngnathus* | 0.13 | | | 0.05 | | | | | 0.81 | |
| *Chilomycterus* | | 0.02 | 0.10 | | | | | 0.77 | | |
| *Opisthonema* | 0.04 | | | 0.08 | | | | | 0.76 | |

**Table C-4. SIMPER two-way (season and zone) results for Naples Bay season groups. All taxa that contributed to dissimilarity (up to 90 percent) between groups are listed here in order of greatest contribution to overall difference.**

| Taxon | Average Group Abundance | | % Contribution to Dissimilarity |
|---|---|---|---|
| | Dry | Wet | |
| *Eucinostomus* | 2.42 | 3.94 | 17.60 |
| *Anchoa* | 1.70 | 2.30 | 15.41 |
| *Farfantepenaeus* | 0.95 | 0.86 | 6.90 |
| *Callinectes* | 1.00 | 0.44 | 6.37 |
| *Lagodon* | 0.89 | 0.19 | 5.68 |
| *Ariopsis* | 0.52 | 0.43 | 4.43 |
| *Lutjanus* | 0.29 | 0.73 | 4.30 |
| *Synodus* | 0.51 | 0.45 | 3.77 |
| *Cynoscion* | 0.18 | 0.48 | 3.34 |
| Order Teuthida | 0.30 | 0.31 | 3.10 |
| *Prionotus* | 0.40 | 0.12 | 2.32 |
| *Etropus* | 0.39 | 0.06 | 2.25 |
| *Bairdiella* | 0.27 | 0.19 | 2.18 |
| *Leiostomus* | 0.40 | 0.08 | 1.98 |
| *Ogcocephalus* | 0.19 | 0.15 | 1.75 |
| *Symphurus* | 0.21 | 0.16 | 1.58 |
| *Orthopristis* | 0.19 | 0.12 | 1.51 |
| Family Portunidae | 0.24 | 0.03 | 1.32 |
| *Achirus* | 0.06 | 0.13 | 1.30 |
| *Microgobius* | 0.13 | 0.08 | 1.08 |
| *Archosargus* | 0.11 | 0.03 | 0.99 |
| *Chloroscombrus* | 0.02 | 0.10 | 0.83 |
| *Paralichthys* | 0.07 | 0.05 | 0.79 |

**Table C-5. SIMPER results for Naples Bay year groups. All taxa that contributed to dissimilarity (up to 90 percent) between groups are listed here in order of greatest contribution to overall difference.**

| Species | Average Group Abundance | | % Contribution to Dissimilarity |
|---|---|---|---|
| | 2010 & 2011 | 2012-2014 | |
| *Anchoa* | 2.72 | 1.49 | 15.01 |
| *Eucinostomus* | 3.40 | 2.99 | 14.14 |
| *Farfantepenaeus* | 1.45 | 0.52 | 7.96 |
| *Lagodon* | 1.02 | 0.25 | 6.50 |
| *Callinectes* | 1.16 | 0.42 | 6.29 |
| *Lutjanus* | 0.59 | 0.46 | 4.05 |
| *Cynoscion* | 0.56 | 0.13 | 3.89 |
| *Ariopsis* | 0.54 | 0.38 | 3.70 |
| *Synodus* | 0.61 | 0.39 | 3.69 |
| *Leiostomus* | 0.64 | 0.01 | 2.91 |
| Order Teuthida | 0.35 | 0.29 | 2.85 |
| *Prionotus* | 0.50 | 0.08 | 2.61 |
| *Etropus* | 0.42 | 0.10 | 2.53 |
| *Bairdiella* | 0.44 | 0.11 | 2.52 |
| *Ogcocephalus* | 0.18 | 0.19 | 1.87 |
| *Symphurus* | 0.34 | 0.06 | 1.77 |
| Family Portunidae | 0.28 | 0.02 | 1.52 |
| *Orthopristis* | 0.18 | 0.16 | 1.39 |
| *Achirus* | 0.07 | 0.13 | 1.20 |
| *Archosargus* | 0.10 | 0.05 | 1.10 |
| *Microgobius* | 0.13 | 0.10 | 1.09 |
| *Paralichthys* | 0.13 | 0.02 | 0.98 |
| *Opsanus* | 0.13 | 0.03 | 0.84 |

**Table C-6. SIMPER two-way (season and zone) results for Southwest Florida bays. All taxa that contributed to dissimilarity (up to 90 percent) with Naples Bay are listed here in order of greatest contribution to overall difference.**

| Taxon | Average Group Abundance | | | | | | % Contribution to Dissimilarty with NB | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | NB | MB | RB | PB | FU | FH | MB | RB | PB | FU | FH |
| *Anchoa* | 2.15 | 1.52 | 1.05 | 1.38 | 1.09 | 1.29 | 16.97 | 13.6 | 12.07 | 12.1 | 11.47 |
| *Eucinostomus* | 3.02 | 3.71 | 2.51 | 3.09 | 2.89 | 2.67 | 16.1 | 12.98 | 10.96 | 12.24 | 10.7 |
| *Farfantepenaeus* | 0.99 | 0.6 | 1.11 | 2.04 | 2.13 | 1.79 | 6.73 | 8.21 | 8.95 | 9.97 | 8.1 |
| *Lagodon* | 0.63 | 0.38 | 0.99 | 1.61 | 0.86 | 1.42 | 4.13 | 8.05 | 8.96 | 6 | 8.21 |
| *Callinectes* | 0.81 | 0.23 | 0.52 | 0.95 | 0.67 | 0.57 | 4.43 | 6.04 | 6 | 5.26 | 5.01 |
| *Synodus* | 0.45 | 0.43 | 0.69 | 0.57 | 0.82 | 0.69 | 4.08 | 5.24 | 3.45 | 4.8 | 3.85 |
| *Lutjanus* | 0.54 | 0.44 | 0.34 | 0.7 | 0.26 | 0.61 | 5.49 | 3.79 | 3.73 | 3.71 | 3.58 |
| *Ariopsis* | 0.53 | 0.51 | 0.23 | 0.23 | 0.28 | 0.21 | 4.58 | 4.08 | 3.4 | 3.73 | 3.55 |
| *Symphurus* | 0.22 | 0.15 | 0.28 | 0.86 | 1.24 | 0.79 | 1.72 | 2.31 | 4.02 | 6.14 | 3.63 |
| *Bairdiella* | 0.28 | 0.17 | 0.28 | 0.47 | 0.38 | 0.59 | 2.32 | 3.01 | 3.05 | 3.03 | 3.68 |
| *Cynoscion* | 0.38 | 0.29 | 0.19 | 0.3 | 0.43 | 0.33 | 3.8 | 2.84 | 2.46 | 2.97 | 2.36 |
| *Etropus* | 0.27 | 0.33 | 0.29 | 0.18 | 0.2 | 0.41 | 2.77 | 2.83 | 2.08 | 2.35 | 3.11 |
| *Prionotus* | 0.3 | 0.24 | 0.2 | 0.35 | 0.33 | 0.49 | 2.17 | 2.3 | 2.67 | 2.82 | 3.1 |
| *Ogcocephalus* | 0.18 | 0.18 | 0.16 | 0.23 | 0.05 | 0.12 | 2.34 | 2.26 | 2.28 | 1.49 | 1.83 |
| *Leiostomus* | 0.31 | 0.37 | 0.14 | 0.07 | 0.12 | 0.12 | 2.46 | 2.19 | 1.58 | 1.73 | 1.72 |
| Order Teuthida | 0.28 | 0.22 | 0 | 0 | 0 | 0 | 3.02 | 1.59 | 1.33 | 1.44 | 1.32 |
| *Sphoeroides* | 0.08 | 0.07 | 0.14 | 0.21 | 0.26 | 0.32 | 0.85 | 1.45 | 1.33 | 1.84 | 2.14 |
| *Orthopristis* | 0.13 | | 0.36 | 0.25 | | 0.46 | | 3.09 | 1.8 | | 2.71 |
| *Syngnathus* | 0.07 | | | 0.64 | 0.2 | 0.49 | | | 3.33 | 1.29 | 2.48 |
| *Paralichthys* | 0.08 | 0.09 | 0.16 | 0.19 | 0.25 | 0.25 | 0.93 | 1.47 | 1.21 | 1.66 | 1.72 |
| *Achirus* | 0.05 | | | 0.38 | 0.59 | 0.37 | | | 1.72 | 2.98 | 1.64 |
| *Gobiosoma* | 0.05 | | | 0.36 | | 0.62 | | | 1.9 | | 3.1 |
| *Microgobius* | 0.09 | 0.14 | 0.07 | | 0.41 | | 1.2 | 0.89 | | 2.11 | |
| *Opsanus* | 0.09 | | | 0.19 | 0.13 | 0.33 | | | 1.32 | 1.03 | 1.65 |
| *Archosargus* | 0.07 | 0.08 | | 0.2 | | | 0.98 | | 1.35 | | |
| *Chloroscombrus* | 0.06 | 0.21 | | | | | 1.9 | | | | |
| *Menticirrhus* | 0.07 | | 0.14 | | | | | 1.13 | | | |
| Family Portunidae | 0.17 | | 0 | | | | | 1.03 | | | |
| *Bagre* | 0.06 | 0.06 | | | | | 0.92 | | | | |
| *Sciaenops* | 0.07 | 0.05 | | | | | 0.9 | | | | |

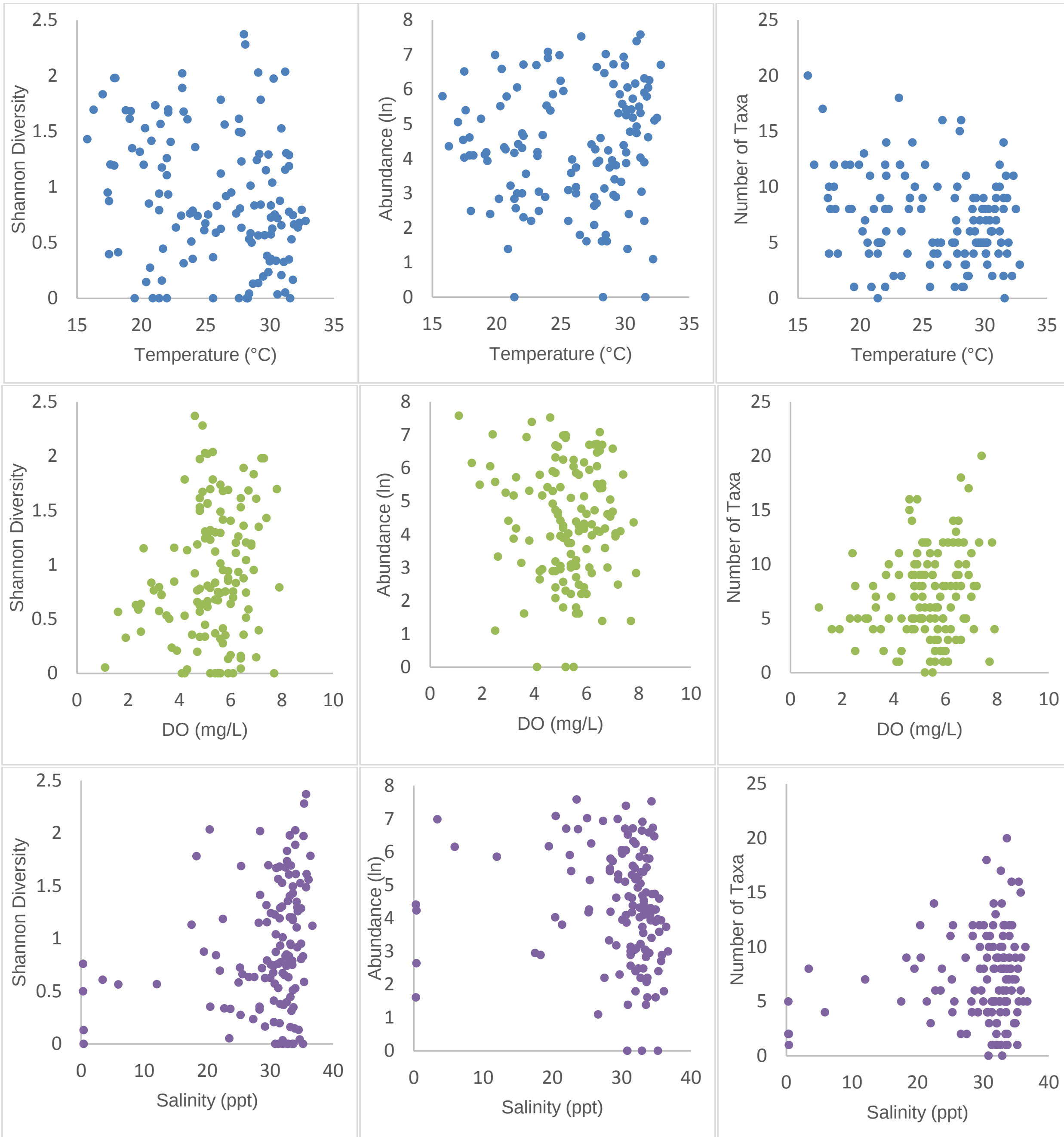


**Figure C-1. Examples plots of diversity metrics against field water quality measurements.**

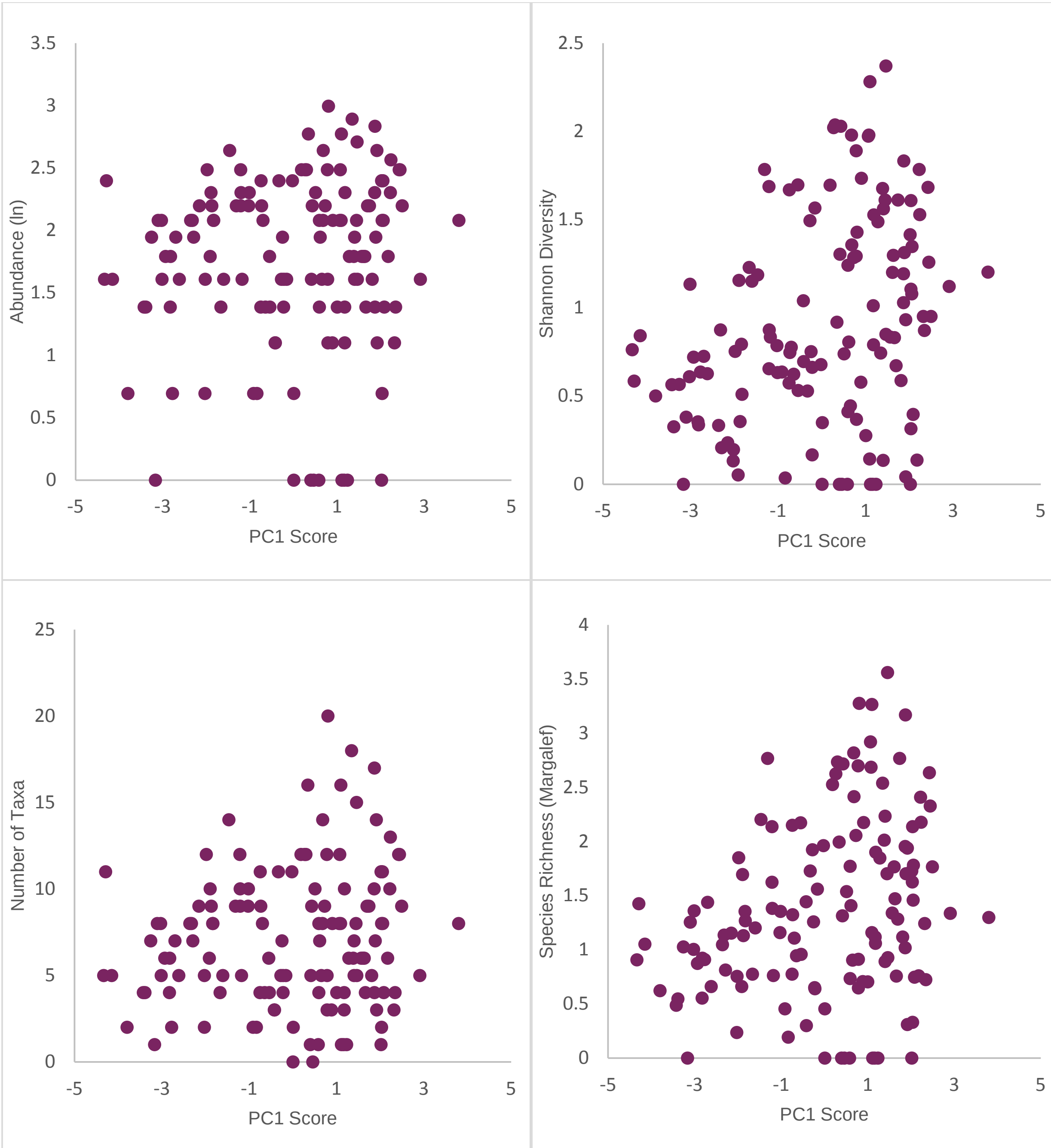


**Figure C-2.** **Examples plots of diversity metrics against PC scores from the water quality dataset.**

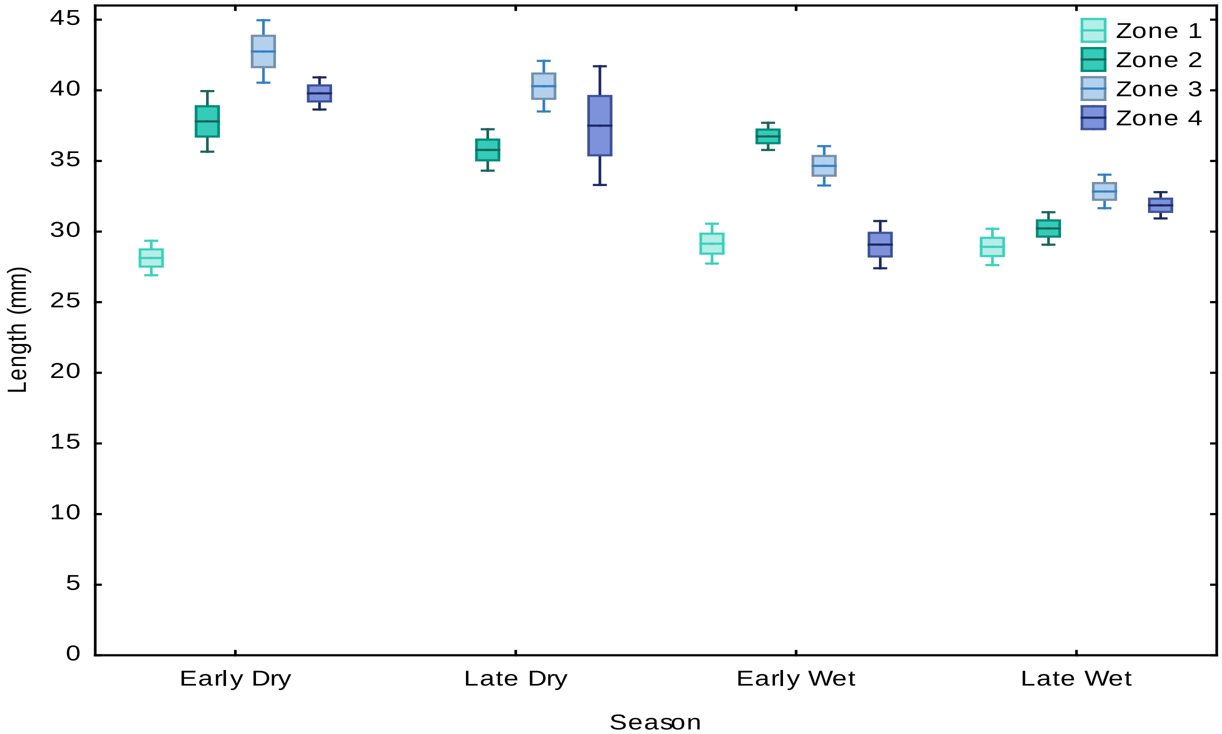


**Figure C-3.** ***Anchoa mitchilli*** **length in Naples Bay 2009–2014 (mean ± 1SE and ±2 SE).**

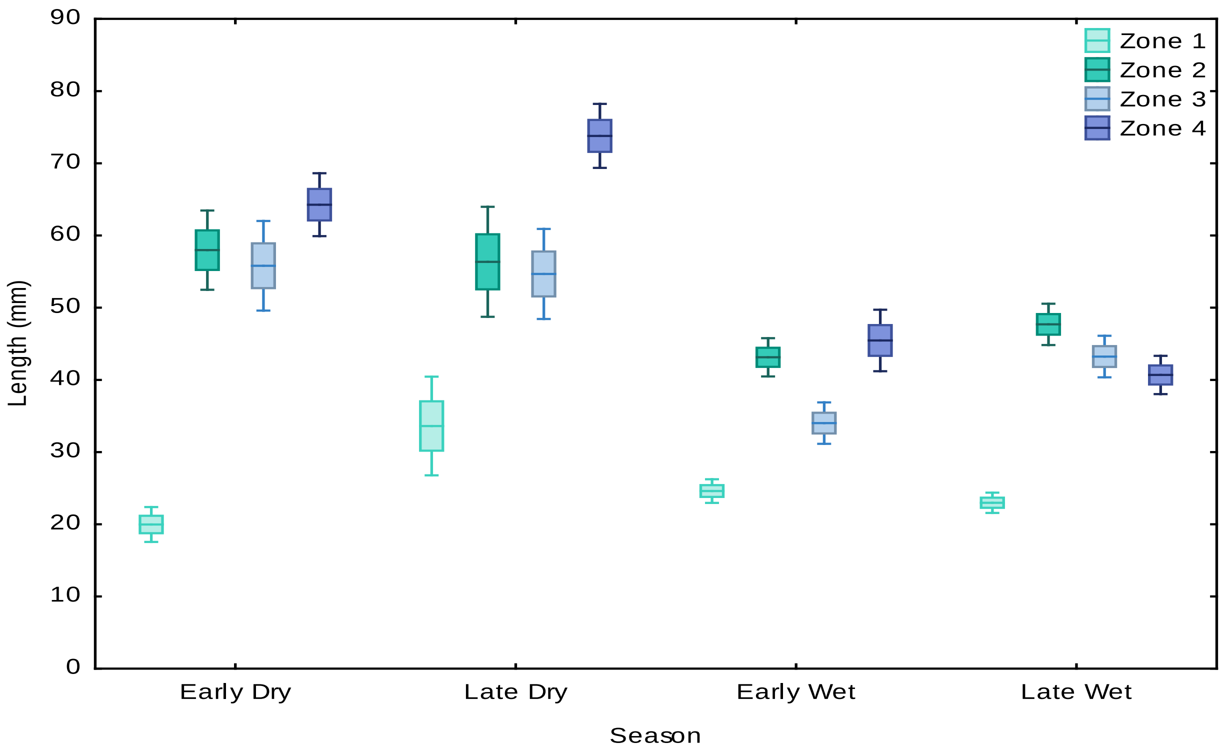


**Figure C 4.** ***Eucinostomus*** **spp. length in Naples Bay 2009–2014 (mean ± 1SE and ± 2SE).**

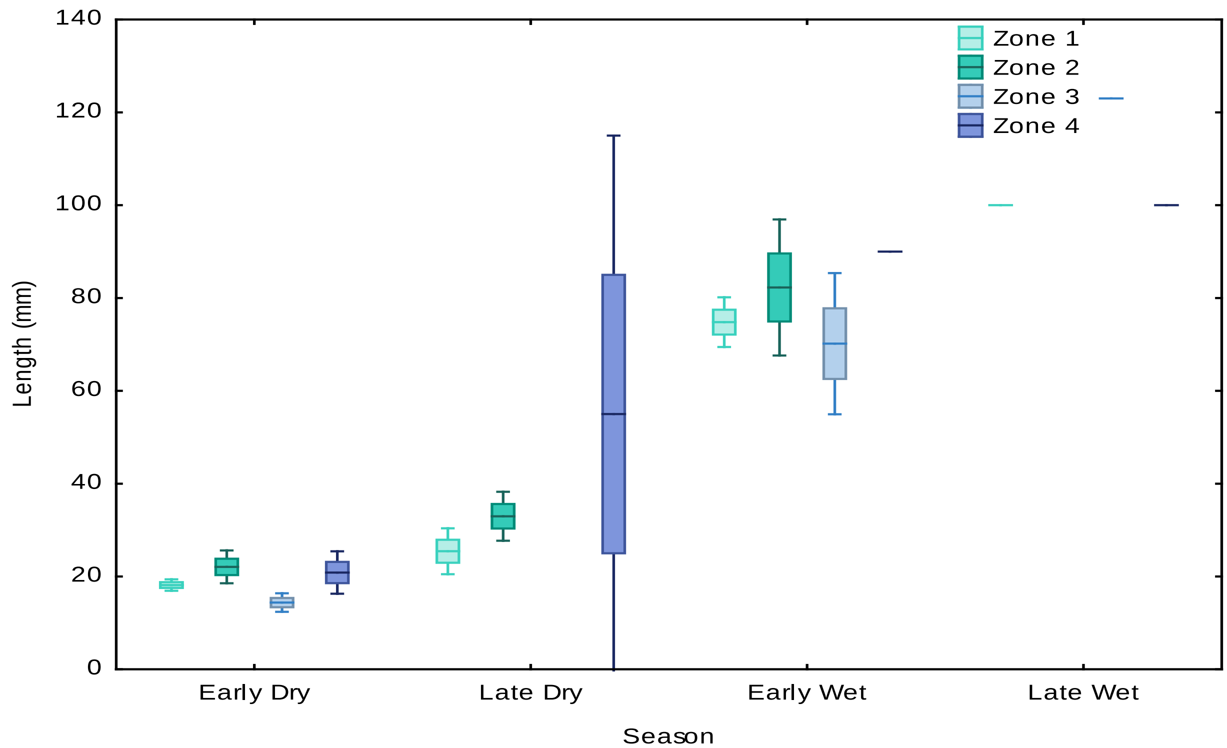


**Figure C-5.** ***Lagodon rhomboides*** **length in Naples Bay 2009–2014 (mean ± 1SE and ± 2SE).**

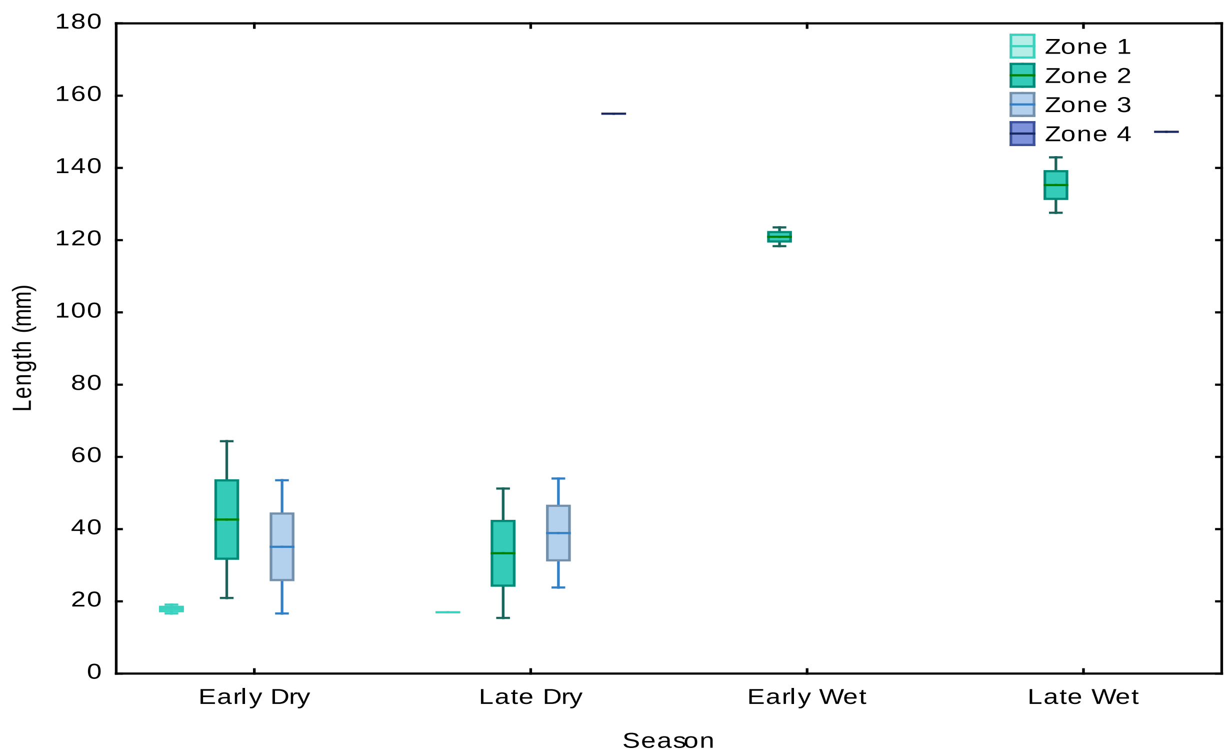


**Figure C-6.** ***Leiostomus xanthurus*** **length in Naples Bay 2009–2014 (mean ±1 SE and ±2 SE).**

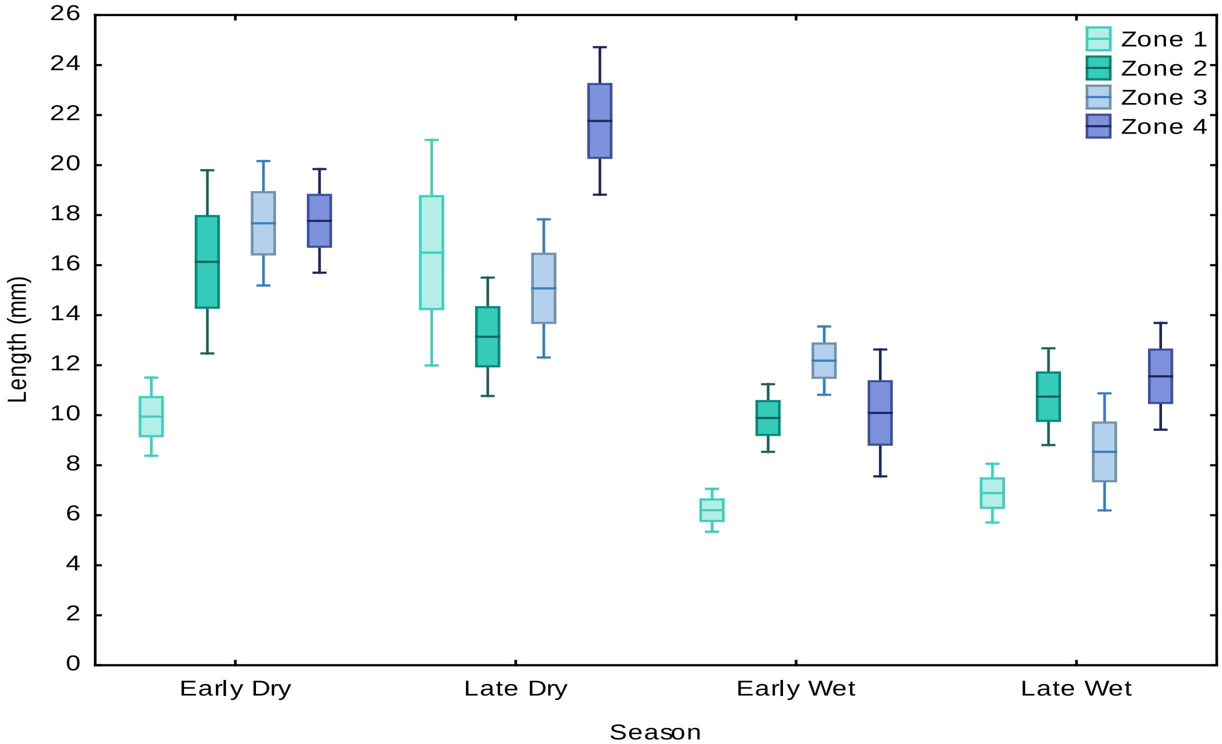


**Figure C-7.** ***Farfantepenaeus duorarum*** **length in Naples Bay 2009–2014 (mean ± 1SE and ± 2SE).**

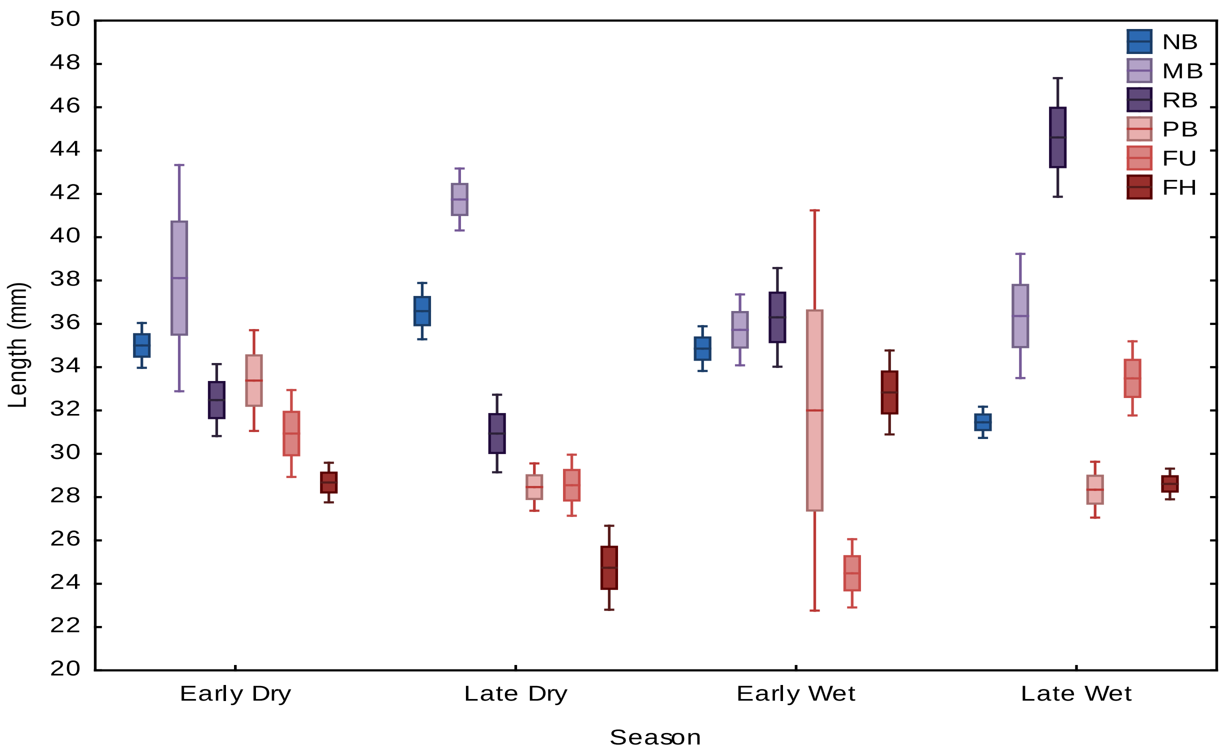


**Figure C-8.** ***Anchoa*** **spp. length in Southwest Florida Bays 2009–2014 (mean ± 1SE and ± 2SE).**

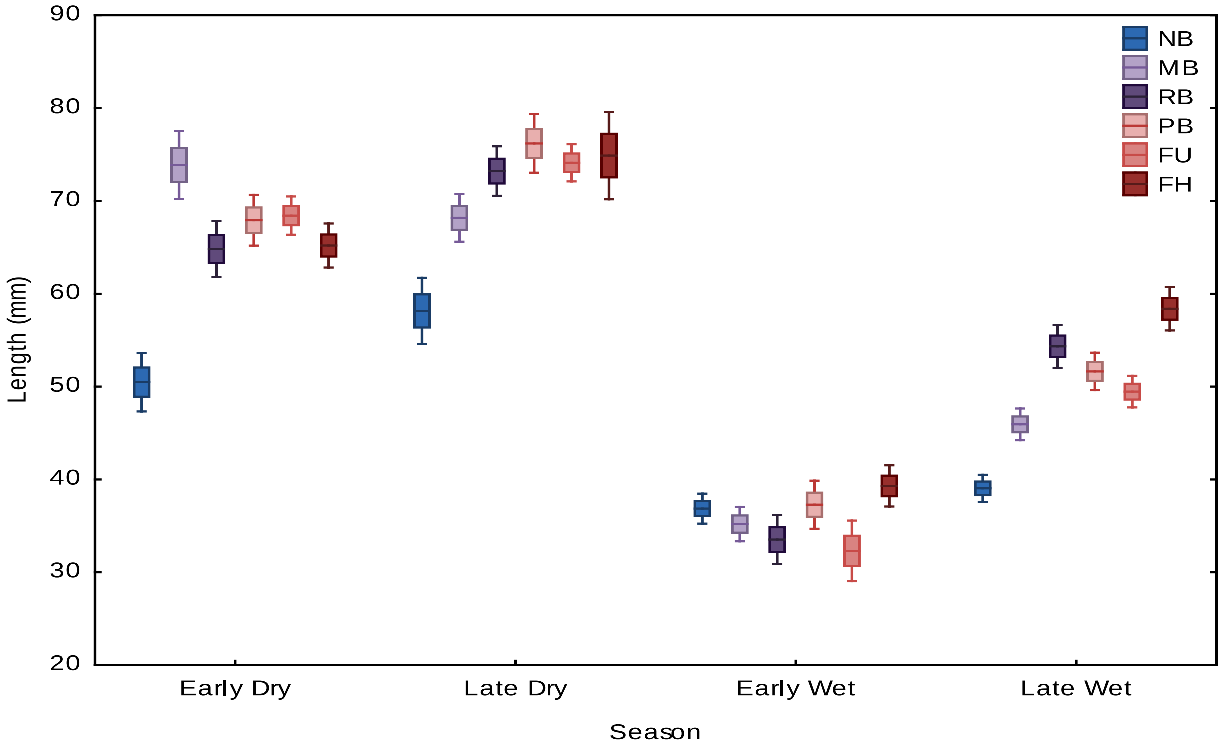


**Figure C-9.** ***Eucinostomus*** **spp. length in Southwest Florida Bays 2009–2014 (mean ± 1SE and ± 2SE).**

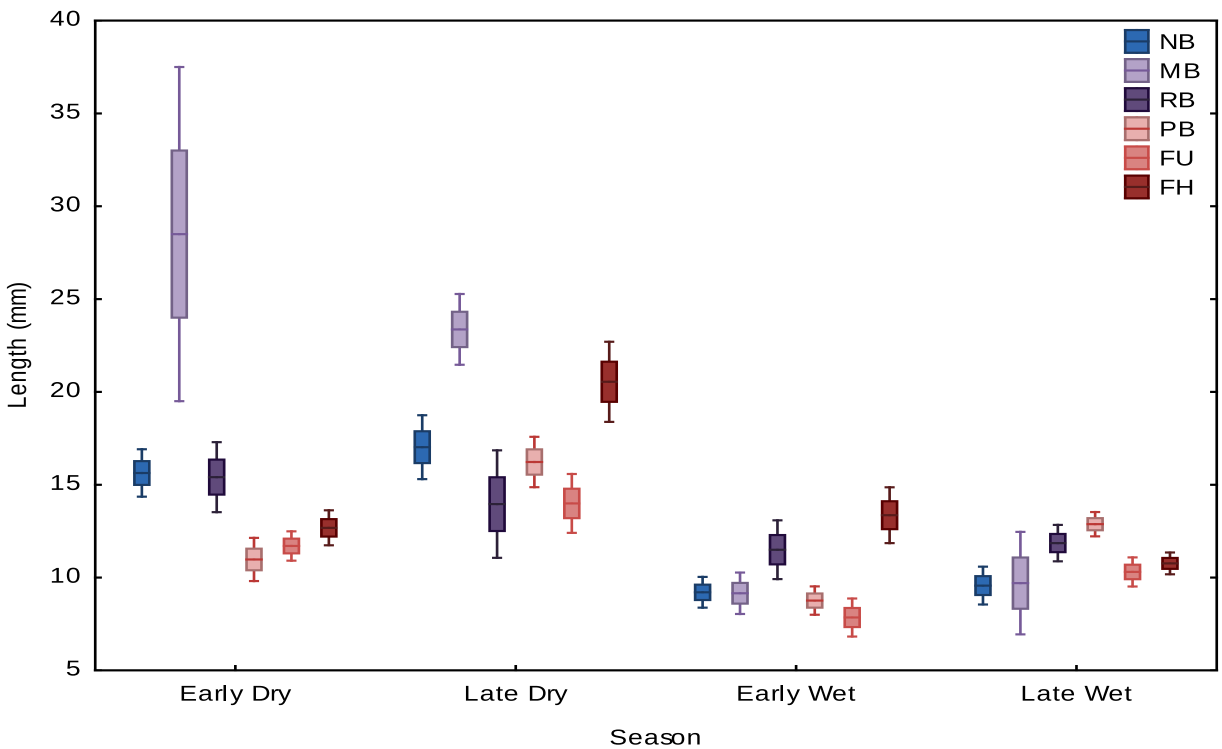


**Figure C-10.** ***Farfantepenaeus duorarum*** **length in Southwest Florida Bays 2009–2014 (mean ± 1SE and ± 2SE).**

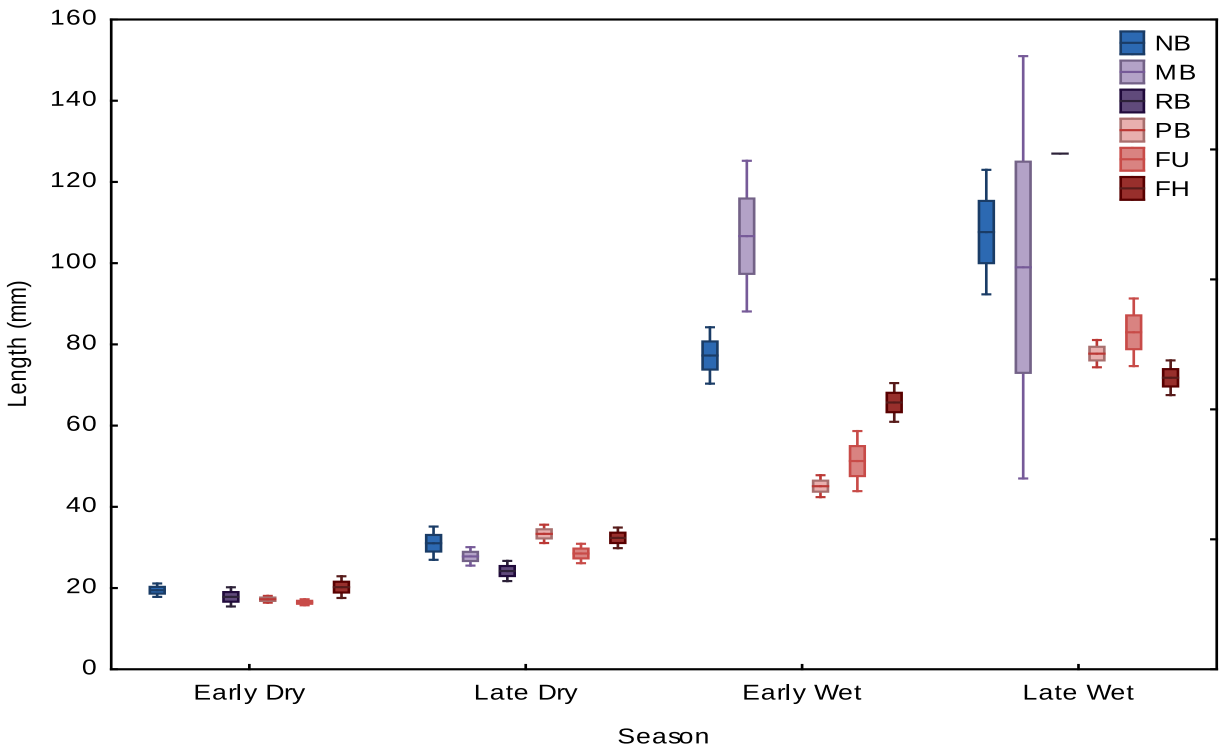


**Figure C-11.** ***Lagodon rhomboides*** **length in Southwest Florida Bays 2009–2014 (mean ± 1SE and ± 2SE).**

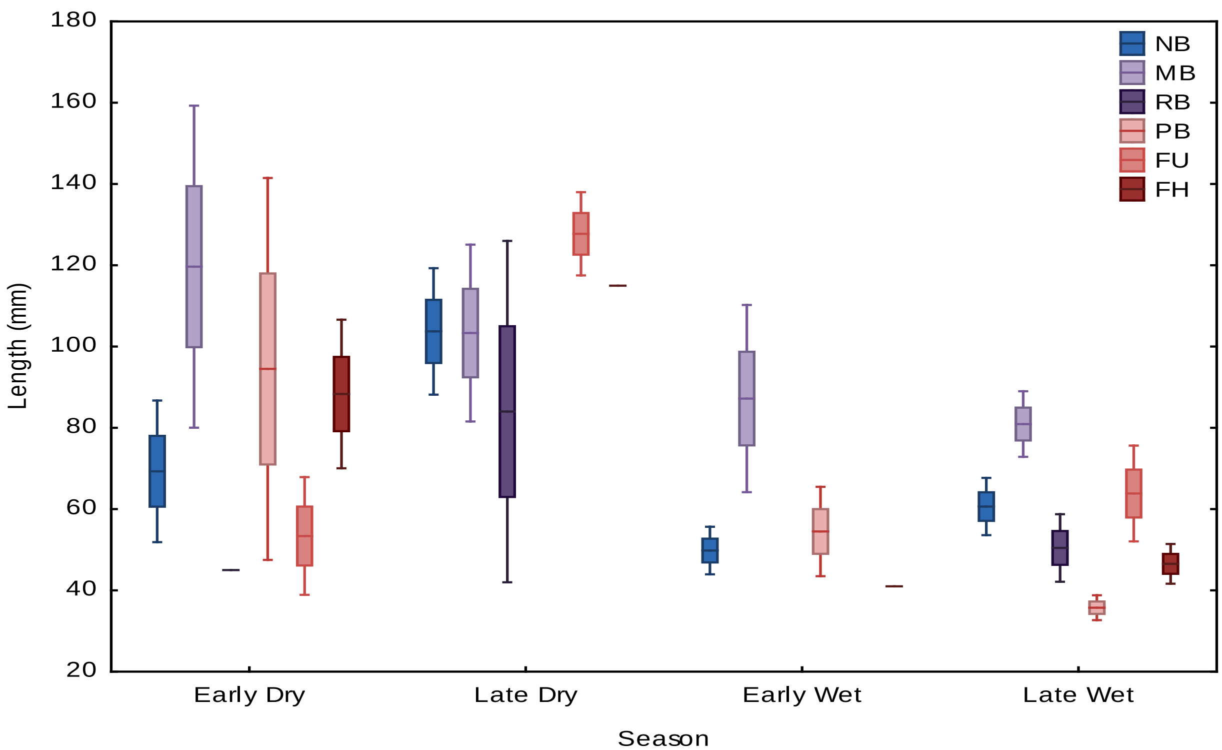


**Figure C-12.** ***Lutjanus synagris*** **length in Southwest Florida Bays 2009–2014 (mean ±1 SE and ± 2SE).**

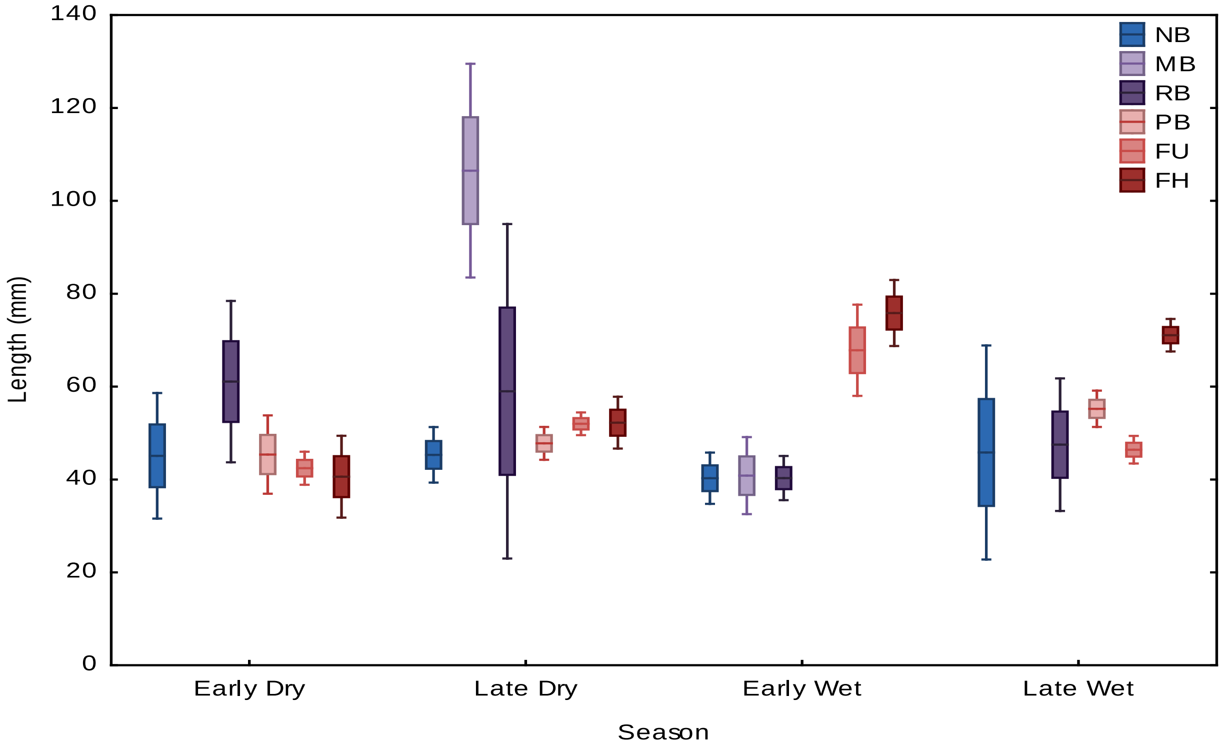


**Figure C-13.** ***Symphurus plagiusa*** **length in Southwest Florida Bays 2009–2014 (mean ±1 SE and ± 2SE**

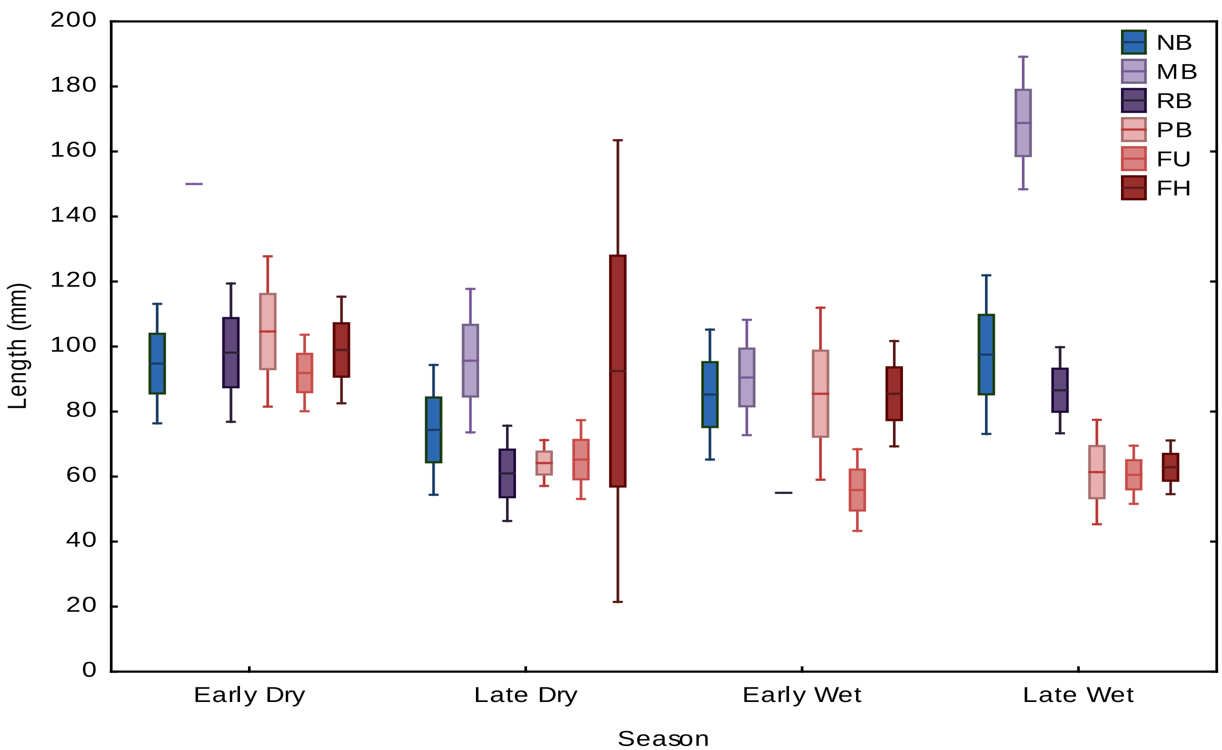


**Figure C-14.** ***Synodus foetens*** **length in Southwest Florida Bays 2009–2014 (mean ±1 SE and ± 2SE**