Roberts Bay North

WATERSHED MANAGEMENT PLAN

Sarasota County

Southwest Florida Water Management District

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Chapter 4

Water Quality

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4.0 WATER QUALITY

Water quality is a key indicator of the environmental health of estuaries and watersheds. Good water quality promotes a diverse and sustainable natural biota and minimizes risks to human health. Among others, the primary water quality constituents of interest in this Watershed Management Plan (WMP) include salinity, dissolved oxygen (DO), nitrogen, phosphorus, chlorophyll, and coliform bacteria. The "quality" of water is largely estimated by the concentrations (or loads) of these constituents. These constituents, in turn, are largely affected by anthropogenic influences throughout the watersheds of most coastal communities. For instance, coastal development has altered the natural hydrology of most coastal watersheds by increasing the amount of impervious surfaces and fragmenting the drainage basins of tidal tributaries, resulting in increased surface water runoff and increased "flashiness" of freshwater inputs into tidal tributaries. These watershed alterations have affected the volume and timing of freshwater inflows into coastal basins, altering natural estuarine salinity patterns and increasing the mass (load) of nutrients and other pollutants into estuarine tributaries. Increased nutrient loads can increase primary production (chlorophyll *a*) in freshwater and estuarine systems and can lead to eutrophication (low DO and high chlorophyll *a*), an indicator of ecosystem degradation. A major goal of this WMP is to characterize water quality throughout the Roberts Bay North watershed, identify degraded waters, and evaluate how to improve observed problems within Roberts Bay North.

This chapter provides detailed information on the water quality of Roberts Bay North including spatial and temporal trends, water quality conditions of concern, water quality targets (levels of service) for water quality indicators, analysis of pollutant loadings, response to pollutant loading, pollutant-loading targets, and recommended actions for the proper stewardship of Roberts Bay North water quality.

Current water quality monitoring programs conduct monthly sampling events in both the watershed drainage basins and the estuary. The estuarine water quality has been routinely sampled since 1995, while the watershed monitoring program has only been in place since 2006. Historical data were collected in the watershed; however, these programs were discontinued in 1992. Although these historical data are described in this chapter, the relevance of these data to current conditions as well as consistency in methods used in data collection between periods are suspect, and therefore the focus of the water quality assessment is based on recent data collected between 1998 through 2007. The assessment begins with evaluation of the current conditions and spatial and temporal trends, identifies water quality indicators of concern, and develops water quality targets for these indicators. Assessment of pollutant-loading targets (levels of service [LOS]) and recommended actions complete the evaluation of how Sarasota County can help to ensure proper stewardship of the valuable natural resources by protecting water quality conditions in Roberts Bay North.

4.1 STATUS AND TRENDS

The status and trends of water quality in Roberts Bay North proper and in its major tributaries are discussed in this section.

4.1.1 Estuarine Water Quality

Estuarine water quality is principally a function of nutrient loading from the watershed and atmospheric deposition. Effects of nutrient loading are mediated by the mixing with the Gulf. Parts of Roberts Bay North may not be well-mixed. The nearest inlet (Big Sarasota Pass) is north and oriented northeast/southwest, which facilitates exchange with Big Sarasota Bay (Figure 4-1). Tidal exchange with Roberts Bay is restricted by Fishery Point on the eastern side on Siesta Key. To the south, the closure of Midnight Pass has likely also resulted in increased residence times in Roberts Bay North. This section introduces exploratory data analysis of water quality in Roberts Bay North by examining descriptive plots and descriptive statistics that summarize the spatial distribution patterns within the estuary. Time series plots are used to explore temporal trends in water quality, and the Kendall Tau trend test (Reckhow, 1993) is used to objectively assess temporal changes that have taken place in the estuary over the past 10 years using a statically sound and robust method.

4.1.1.1 Status

The water quality in the Roberts Bay North estuary was evaluated by examining the distribution of values for several water quality indicators of estuarine health and calculating statistics from those distributions over varying temporal scales. Box and whisker plots were generated that compare the overall distribution for water quality constituents within each stratum of the Roberts Bay estuarine sampling segmentation scheme (Figure 4-2). The box and whisker plots display the preponderance of the distribution beginning with the 5% percentile shown as the lower whisker of the plot as identified in the example provided in Figure 4-3. The 25th percentile is identified by the lower bound of the box, while the center horizontal line represents the median value. The 75th percentile and 95th percentile values are correspondingly represented by the upper bound of the box and whisker, respectively. The box and whisker plots allow the reader to distinguish many characteristics of the data distribution.

The distributions of four common *in situ* recorded physical water quality constituents in Roberts Bay North for a 10-year period from 1998 through 2007 are provided in Figure 4-4. Each water quality monitoring stratum within Roberts Bay North is represented in the boxplot. Water temperature is evidently quite similar among strata while salinity, bottom DO, and pH exhibit spatial differences. The influence of Phillippi Creek is evident in these plots as salinity, DO, and pH are reduced in Stratum 13-3, which encompasses the mouth of Phillippi Creek.

Figure 4-1 Waterbodies near Roberts Bay North Watershed

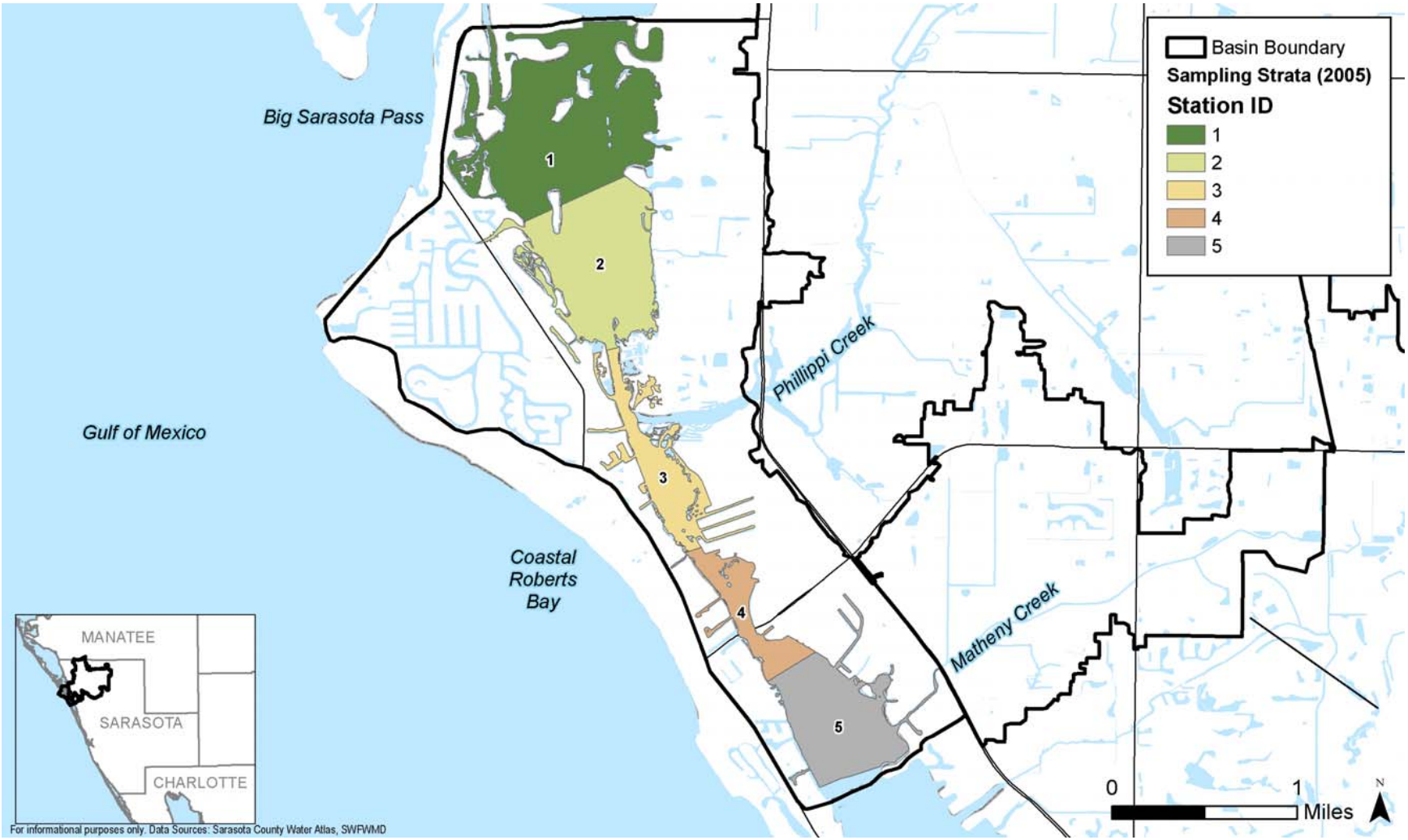

Figure 4-2 Roberts Bay North Water Quality Sampling Strata

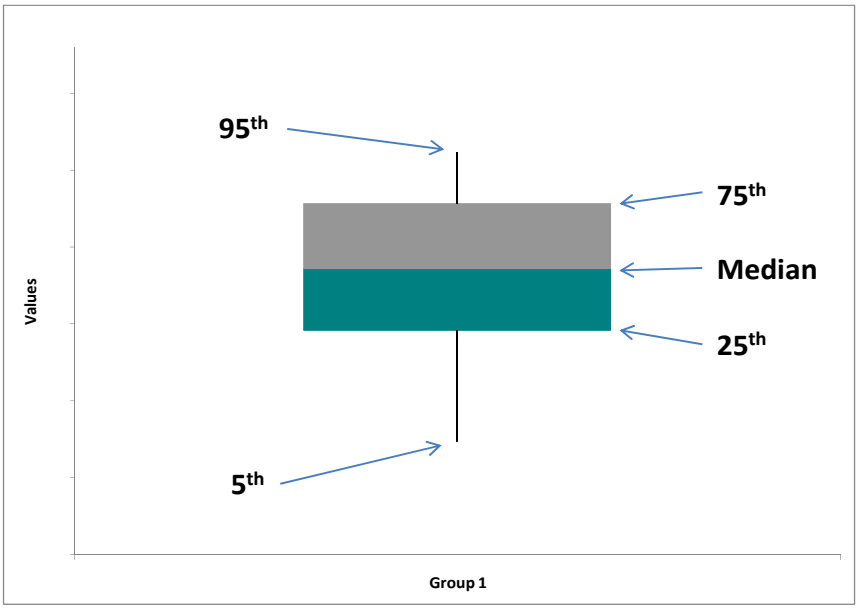

Figure 4-3 Example of a Boxplot Illustrating Aspects of the Data Distribution

Interestingly, DO and pH show nearly identical spatial trends, decreasing in Strata 13-3 and 13-4 and then increasing again in the northern portion of Little Sarasota Bay (Stratum 13-5). Salinity also tends to decrease moving south through the estuary, indicating that the freshwater inflows from Phillippi Creek and Matheny Creek tend to effect salinity in the lower portion of the estuary more than the northern portions (i.e., Strata 13-1 and 13-2).

Water quality constituents that represent nutrients (nitrogen and phosphorus) and biological effects (chlorophyll production and light attenuation) showed similar and expected effects of freshwater inflows with increased nutrient concentrations and increased light attenuation in the stratum directly influenced by Phillippi Creek (Figure 4-5). Interestingly, while Total Nitrogen (TN) and Total Phosphorus (TP) concentrations increased in the southern estuarine strata, chlorophyll *a* and light attenuation were more similar across strata.

The closure of Midnight Pass in Little Sarasota Bay has likely increased residence times in the southern strata and resulted in increased influence of Phillippi and Matheny Creeks on the lower strata of Roberts Bay North. Water normally moving south from Phillippi Creek that would have been flushed on outgoing tides now may be trapped in the northern portion of Little Sarasota Bay.

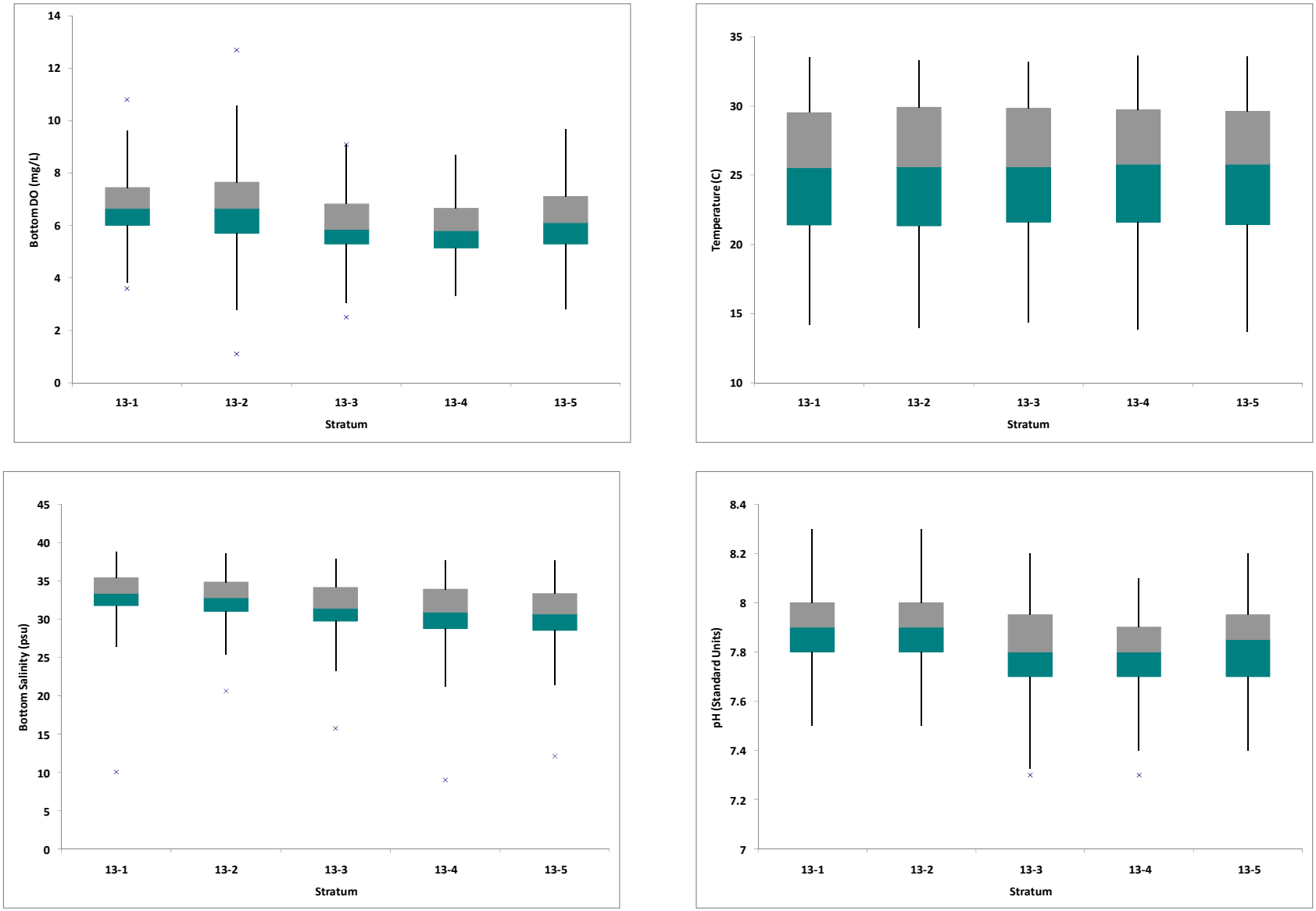

Figure 4-4 Distribution of Physical Water Quality Constituents by Stratum in Roberts Bay North

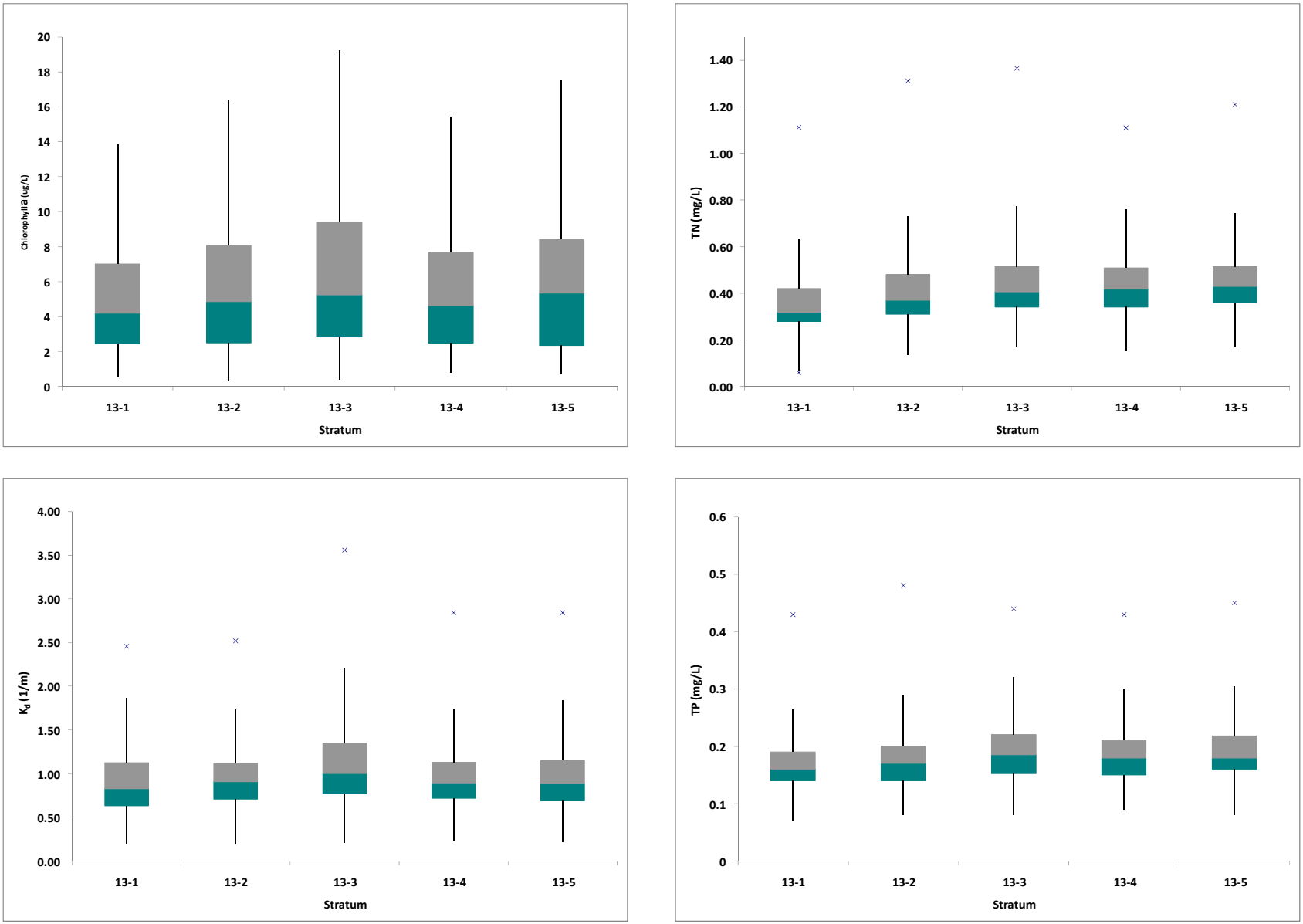

Figure 4-5 Distribution of Nutrient and Biologically Related Water Quality Constituents by Stratum in Roberts Bay North.

4.1.1.2 Trends

Temporal trends in water quality were assessed using graphical plots and the seasonal Kendall Tau trend test (Reckhow, 1993). The Kendall Tau is a nonparametric test that estimates the median slope from all pair-wise comparisons in a time series of data. The statistical test accounts for seasonality and serial autocorrelation before evaluating the statistical significance of the trend in the time series. Therefore, the Kendall Tau is a sophisticated and robust method to evaluate trends in water quality data that often do not fit the assumptions necessary for the use of parametric statistics.

Time series trends provide information on the temporal variations in water quality and elucidate how changes in environmental conditions such as interannual variation in freshwater inflows impact the water quality constituent of interest. The time scale over which the trend is assessed is important when assessing trends. We chose the last 10 years of data to analyze to this assessment for the following reasons:

- FDEP evaluation uses the previous 7.5 years for evaluation of water quality data for Impaired Waters Rule (IWR) calculations except when assessing conditions relative to historical values.
- Previous analysis suggested that data collected before 1998 in Sarasota County was suspect with respect to several parameters including chlorophyll and light attenuation.
- The data from 1998–2007 were collected by a consistent field crew and analyzed by the same laboratory (Mote Marine Laboratory).

The following water quality constituents were included in the time series analysis:

- Bottom DO
- Surface salinity
- Bottom salinity
- Vertically averaged salinity
- Color
- Biological Oxygen Demand (BOD)
- Corrected chlorophyll *a*
- Light extinction coefficient (Kd)
- Total nitrogen (TN)
- Total phosphorus (TP)
- Turbidity

Results of the Kendall Tau test suggested that 5-day BOD was significantly improving with a declining slope of 0.029 mg/L (Table 4-1). Chlorophyll and Turbidity also improved over this time series with negative slopes, indicating reduced concentrations.

Table 4-1 Results of Seasonal Kendall Tau Trend test for Selected Constituents in Roberts Bay North Based on Data Collected from 1998 through 2007

Parameter	Kendall Tau Slope
Biochemical oxygen demand (BOD) (mg/L)	-0.029
Bottom salinity (ppt)	0.000
Surface salinity (ppt)	0.000
Mean salinity (ppt)	0.000
Bottom Dissolved Oxygen (mg/L)	0.086
Color (PtCo units)	0.000
Chlorophyll *a* (µg/L), corrected	-0.294
Light extinction coefficient (K_d) (1/m)	0.000
Total nitrogen (TN) (mg/L)	0.000
Total phosphorus (TP) (mg/L)	0.000
Turbidity (NTU)	-0.175
TSS (mg/L)	0.000

*Shading indicates improved water quality.

Time series plots with significant slopes and salinity are provided in Figure 4-6 to compare the time series to relevant hydrologic conditions. The smoothed time series trend line is shown on these plots to aid the reader in identifying changes in the moving average value for the water quality constituent. While the moving average trend line is not necessarily linear, the Kendall Tau test is testing for a monotonic trend in the time series. Plots of nutrient and biologically based constituents are provided in Figure 4-7. While none of these constituents exhibited significant trends in the time series, the plots are informative for examining the covariance of these parameters over time such as the relationship between chlorophyll *a* and light attenuation.

4.1.2 Watershed Water Quality

As part of Sarasota County's proactive approach to stewardship of their water quality, the Sarasota County Water Resources Department currently monitors surface water quality at 45 sites within the watershed: two sites in the Matheny Creek basin, five in the Coastal basin, and 17 (38 in 1996) in the Phillippi Creek basin as shown in Figure 4-8. The water quality monitoring program has been ongoing since 2006. Samples are collected monthly with the objective of supporting monitoring efforts related to Total Maximum Daily Loads (TMDL) for tributaries. Additionally, the County conducted a "special longitudinal study" of Phillippi Creek for coliform bacteria in support of the septic replacement program in the Phillippi Creek watershed. Forty stations along the creek mainstem and Main A are sampled monthly for this program.

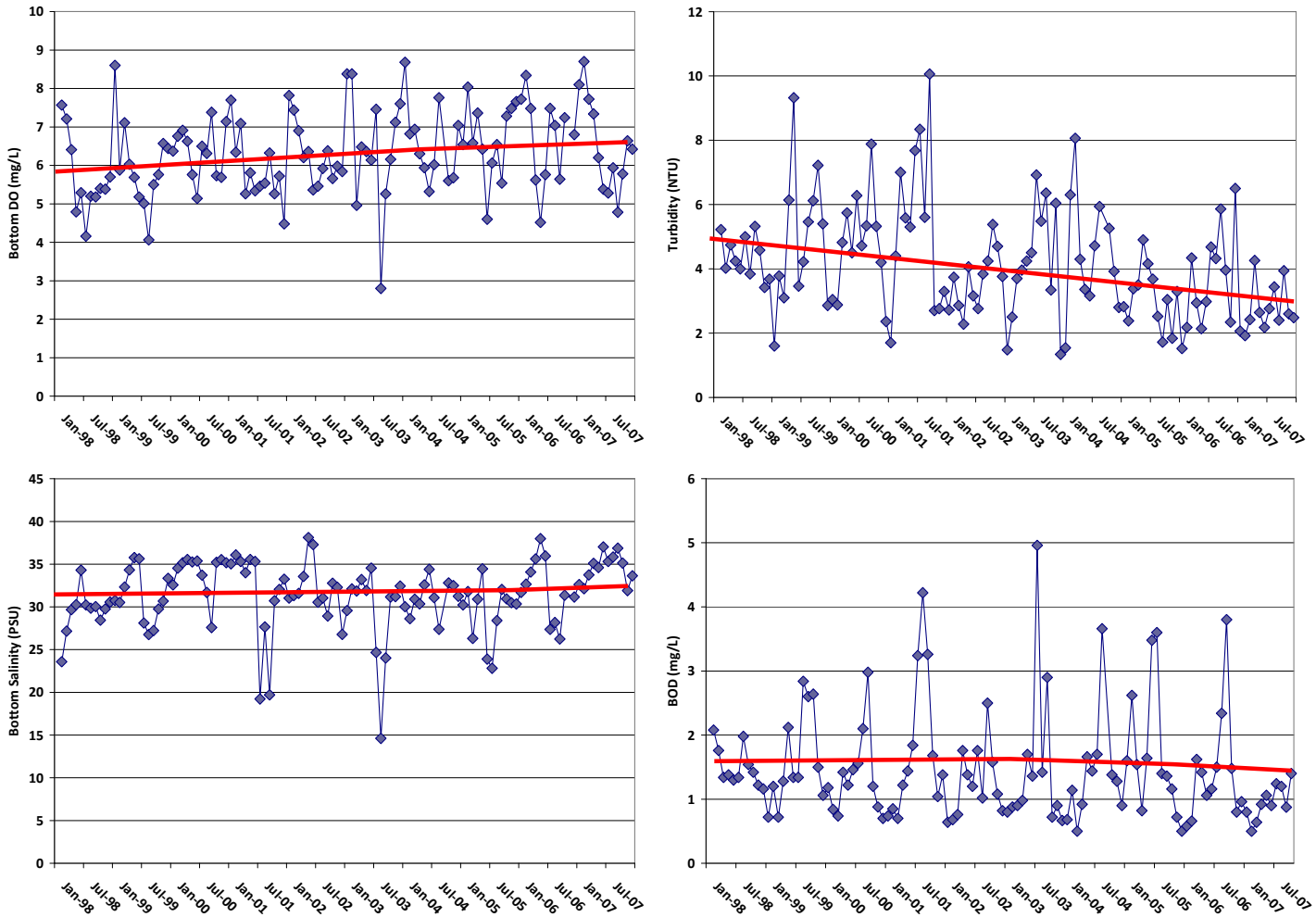

Figure 4-6 Time Series Plots and Smoothed Trend (Line) for Dissolved Oxygen, Turbidity, Salinity and Biochemical Oxygen Demand for Data Collected from 1998 through 2007 in the Roberts Bay North Estuary

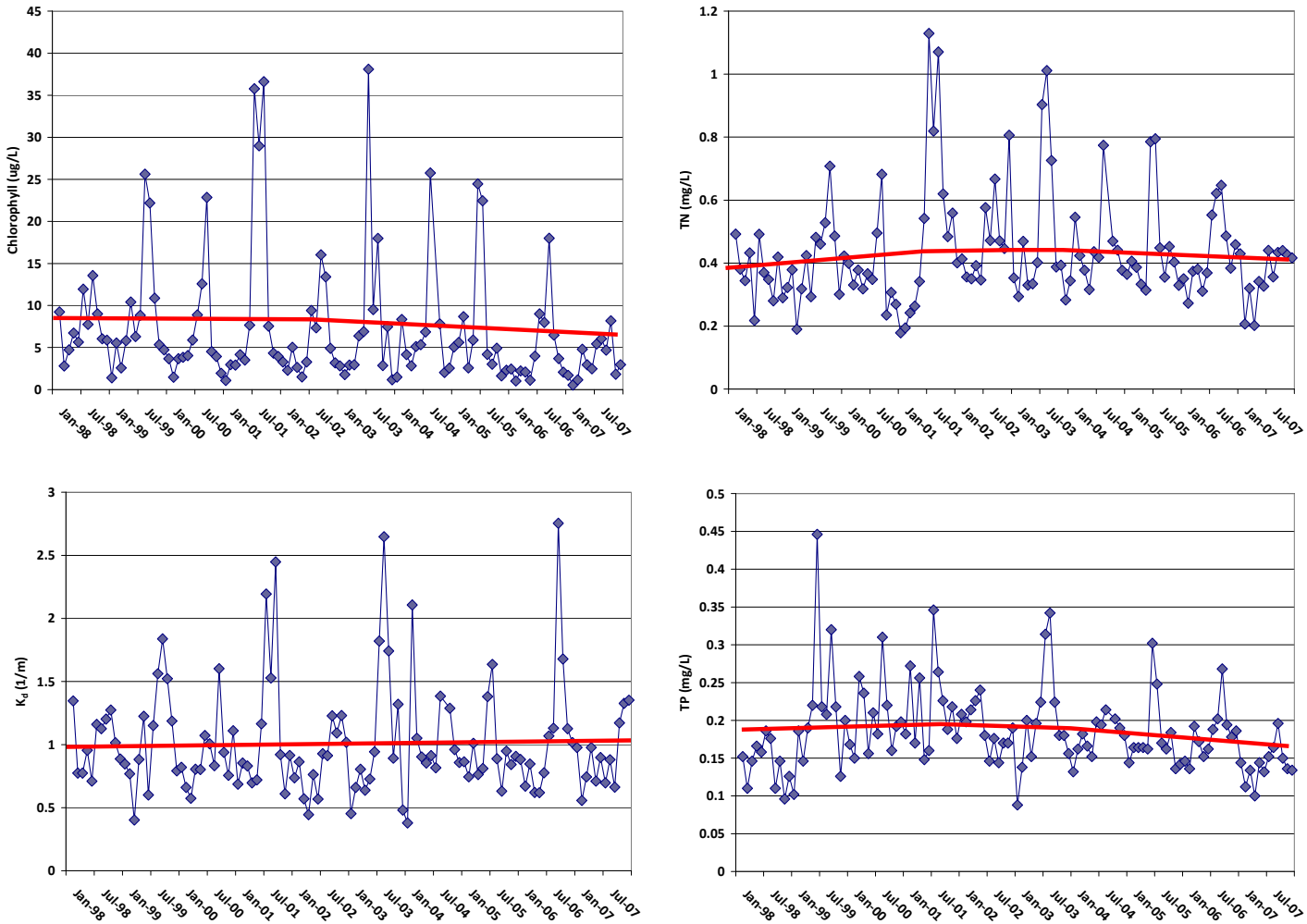

Figure 4-7 Time Series Plots and Smoothed Trend (Line) for Nutrient and Biologically Relevant Parameters for Data Collected from 1998 through 2007 in the Roberts Bay North Estuary

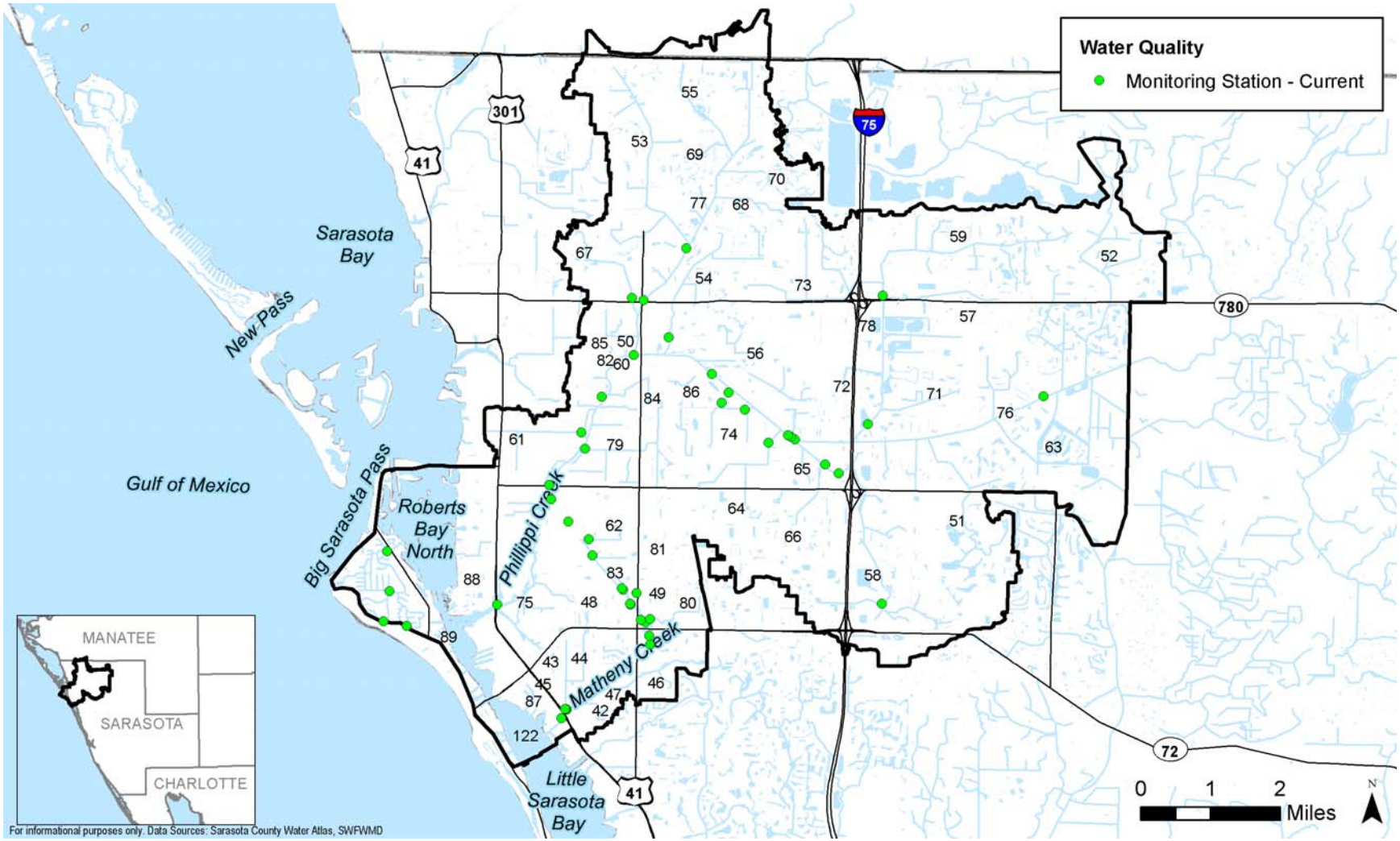

Figure 4-8 Current Roberts Bay Watershed Water Quality Monitoring Stations

Historically, other agencies have also conducted sampling in the watershed. A review of the Sarasota County Water Resources Atlas shows that the following sample sites, as shown in Figure 4-9, have been used:

- U.S. Geological Survey (USGS) – 18 sites.
- Florida Department of Environmental Protection (FDEP) – 41 sites.
- Mote Marine Laboratory – 13 sites.
- Southwest Florida Water Management District (SWFWMD) – 7 sites.

These sampling programs were initiated after the passage of the Federal Clean Water Act of 1972 and sampled approximately quarterly between 1973 and 1992. One site in the Matheny Creek basin, two sites in the Coastal basin, and 21 sites in the Phillippi Creek basin have consistent data records. Sampled parameters were similar to those currently sampled. No consistent water quality monitoring data in Phillippi Creek were collected between 1992 and 2006.

Four representative sites (Figure 4-10) were chosen to compare summary statistics for selected parameters between historical and more recent data collection efforts. These sites include one station in Matheny Creek at US 41 and three stations in Phillippi Creek: downstream at US 41, and two in the Main A canal—one at Palmer Blvd and one at Cattleman Road. Fecal coliform, TN, and TP were compared.

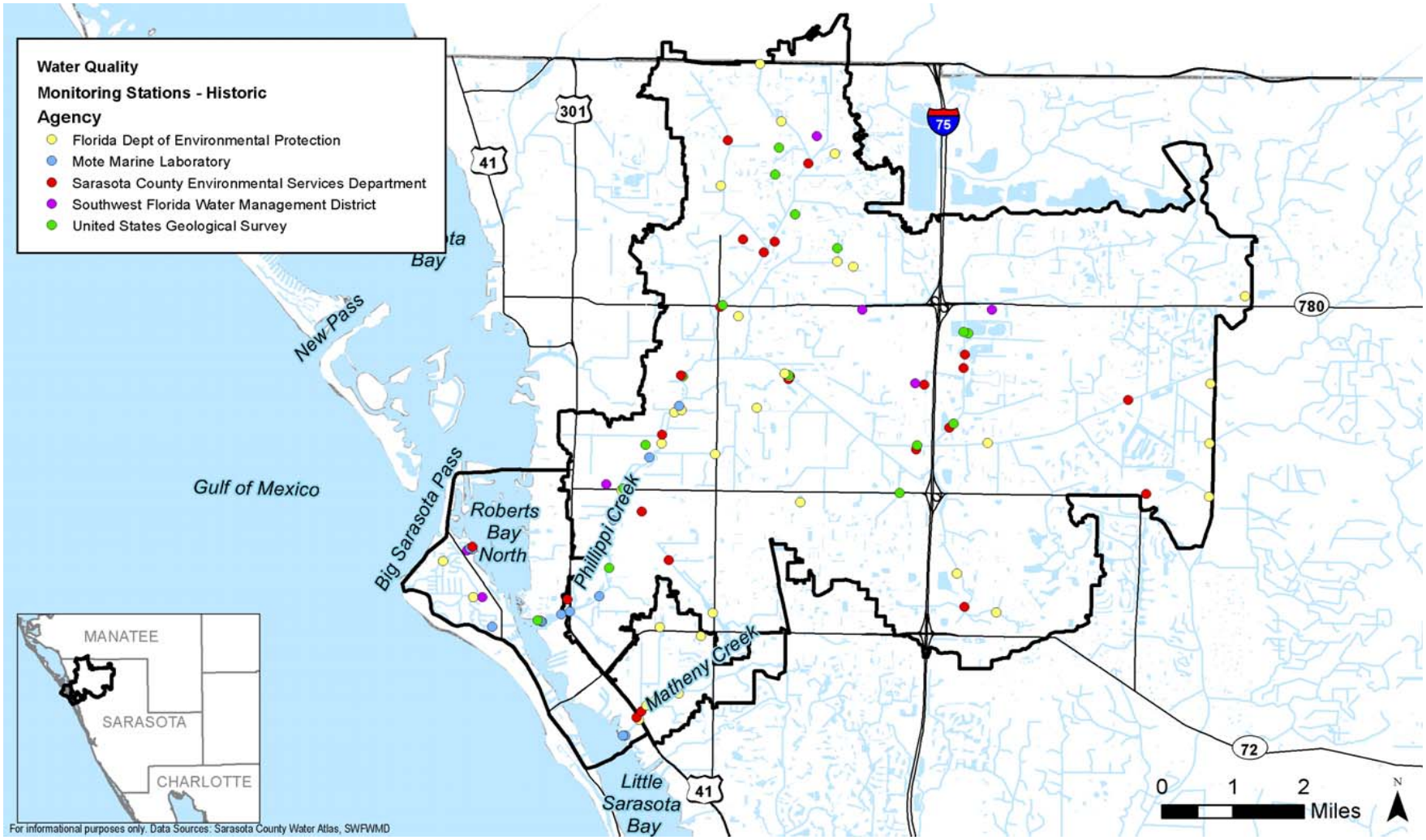

Figure 4-9 Historical Roberts Bay Watershed Water Quality Monitoring Stations

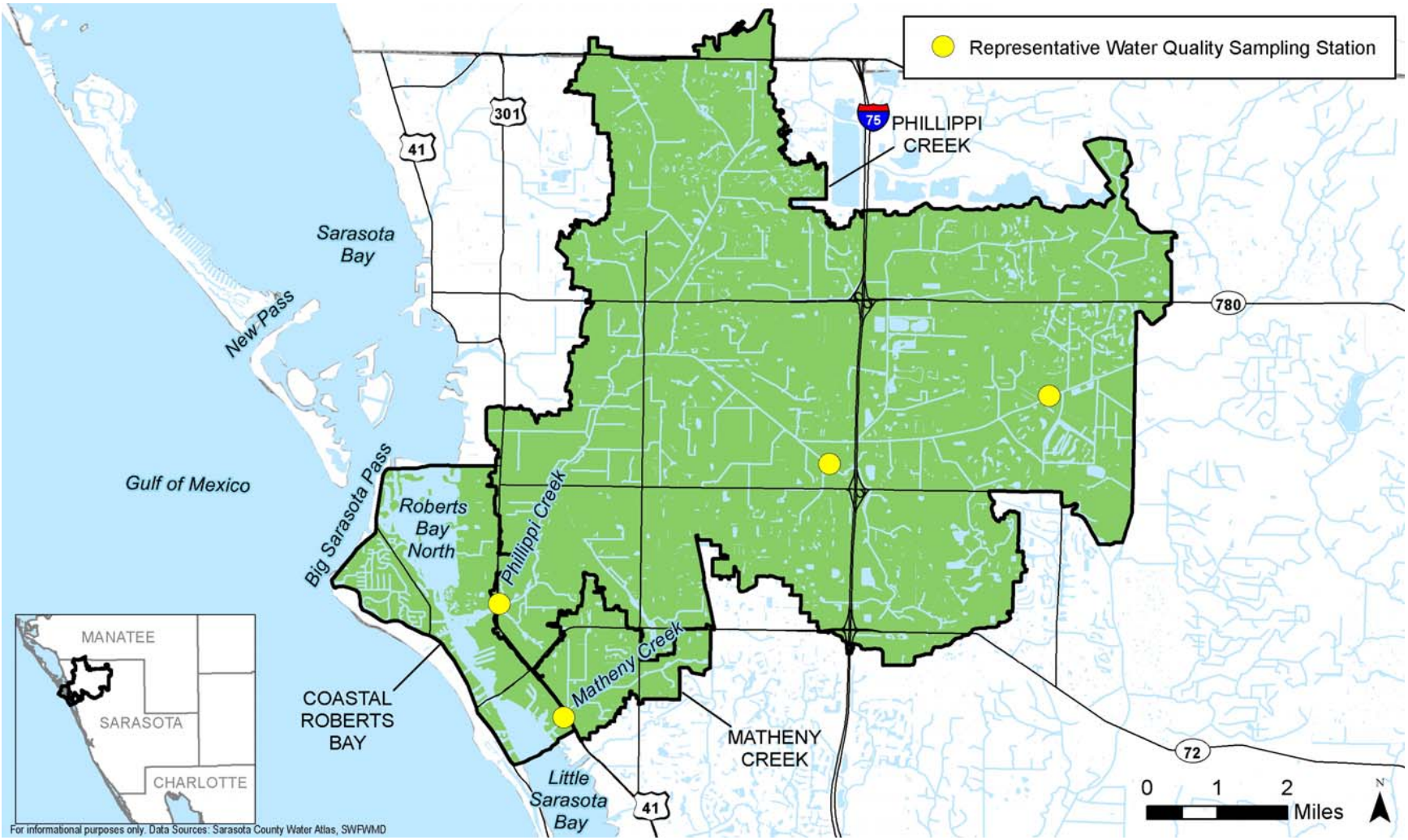

Figure 4-10 Four Representative Sampling Sites Used to Compare Historical and Current Water Quality

The State water quality standard for fecal coliform bacteria in Class III fresh and marine waters is 800 mpn (most probable number) on any day (Chapter 602-302.530, FAC). On average, fecal coliform concentrations were historically below the State standard in Phillippi Creek, but in the more recent data collected as part of the Phillippi Creek septic replacement program monitoring, values exceeded the State standard on average (Tables 4-2 and 4-3). Matheny Creek fecal coliform values in both historical and recent data were higher on average than the State standard.

Total coliform concentrations in Phillippi and Matheny Creek were also historically much higher than the State standard, but data on total coliform concentrations are not available for the recent monitoring activity. There are no nutrient criteria currently established under State statute; however, recent concentrations of TN and TP were approximately half of their historical values on average except in Matheny Creek where the average concentrations have approximately doubled.

As part of Sarasota County's proactive approach to stewardship of their water quality, many capital improvement projects are currently taking place with the aim to reduce anthropogenic sources of nutrient inputs into Roberts Bay North and improve water quality conditions. Wastewater treatment plants that discharge into Phillippi Creek and Matheny Creek are being taken offline and the septic tanks are being replaced. A sediment management plan is being implemented to reduce sediment loads into estuarine receiving bodies. Identifying water quality conditions of concern and developing criteria for these indicators that allow for changes in water quality to be tracked through time as a measure of the success of watershed management efforts are critical to evaluating the success of these watershed management actions.

Table 4-2 Summary Statistics for Historical Water Quality (1973–1992) at Selected Sites in the Roberts Bay North Watershed

Station	Value	Chlorophyll (µg/L)	Conductivity (µmho/cm)	DO (mg/L)	Fecal Coliform (col/100ml)	pH	Total Coliform (col/100ml)	TN (mg/L)	TP (mg/L)
Phillippi Creek at US41	Mean	10.98	16992.08	5.57	610.72	7.57	2828.42	1.54	0.46
	Min	3	380	2.5	10	6.5	100	0.2	0.09
	Max	18.72	41680	10	11000	8	59000	3.44	1.3
Main – A at Cattleman	Mean	4.8	771.78	5.47	525.86	7.35	6911.21	1.52	0.53
	Min	3	2.5	0	10	6.42	100	0.26	0.1
	Max	6.8	1370	11	12000	8.7	190000	2.86	1.27
Main A at Palmer	Mean	9.26	884.02	5.05	633.38	7.44	7448.4	1.57	0.37
	Min	1	280	0.3	40	4.8	100	0.24	0.06
	Max	16.05	1800	9.4	10000	8.52	340000	4.35	0.97
Matheny Creek at US41	Mean	26.16	1289.58	5.04	2119.94	7.51	10062.72	1.22	0.29
	Min	26.16	330	0.8	10	6.03	100	0.49	0.06
	Max	26.16	12850	11	20000	7.9	160000	1.82	0.73

Table 4-3 Summary Statistics for Recent Water Quality (2006–2007) at Selected Sites in the Roberts Bay North Watershed

Station	Value	Fecal Coliform (col/100ml)	TN (mg/L)	TP (mg/L)
Phillippi Creek at US41	Mean	7050.95	0.61	0.24
	Min	10	0.13	0.11
	Max	51000	1.18	0.48
Main – A at Cattleman	Mean	1334.71	0.75	0.22
	Min	60	0.45	0.1
	Max	5000	1.18	0.53
Main A At Palmer	Mean		0.89	0.1
	Min		0.61	0.08
	Max		1.32	0.144
Matheny Creek at US41	Mean	2356.67	1.99	0.53
	Min	10	0.51	0.12
	Max	15900	0.61	0.24

4.1.3 Water Quality Conditions of Concern

As part of the Sarasota Bay system, Roberts Bay North has been designated an Outstanding Florida Water (OFW) and is listed in Chapter 62-302.700(i), FAC (FDEP, 2009b). An OFW is a waterbody designated as worthy of special protection because of its natural attributes. This special designation is intended to protect existing good water quality, i.e., no degradation of water quality is permitted. Most OFWs are areas managed by the state or federal government such as parks, wildlife refuges, preserves, marine sanctuaries, estuarine research reserves, waters within state or national forests, scenic and wild rivers, or aquatic preserves. Generally, the waters within these managed areas are OFWs because the managing agency has requested this special protection.

As mandated by the Federal Clean Water Act and the U.S. Environmental Protection Agency (EPA), FDEP has established criteria for evaluating water quality throughout Florida using a waterbody classification system and evaluative criteria for a host of water quality constituents (Chapter 62-302.530, FAC). FDEP compiles surface water quality data collected throughout Florida using its STORET database and its Waterbody Identification (WBID) system to assess water quality impairment of WBIDs under the IWR Chapter 62-302.530, FAC).

A TMDL is a scientific determination of the maximum amount of a given pollutant that a surface water body can absorb and still meet water quality standards (FDEP, 2009c). The basic steps in the TMDL program are as follows:

1. Assess the quality of surface waters—are they meeting water quality standards?
2. Determine which waters are impaired—that is, which ones are not meeting water quality standards for a particular pollutant or pollutants.
3. Establish and adopt by rule a TMDL for each impaired water for the pollutants of concern—the ones causing the water quality problems.
4. With extensive local stakeholder input, develop, a Basin Management Action Plan (BMAP) that summarizes what actions will be taken by whom to correct impairments.
5. Implement the strategies and actions in the BMAP.
6. Measure the effectiveness of the BMAP, both continuously at the local level and through a formal re-evaluation every five years.
7. Change the plan and actions if things are not working.
8. Reassess the quality of surface waters periodically.

The following includes a summary for those waterbodies that have existing TMDLs and a summary of those waterbodies that have been verified as impaired but have no existing TMDL.

4.1.3.1 Existing TMDLs in the Roberts Bay North Watershed

TMDLs have been established by EPA for three WBIDs in the Roberts Bay North Watershed (Figure 4-11):

- Roberts Bay North proper (WBID 1968D) (FDEP, delisted in 2010)
- Phillippi Creek (WBID 1937) (EPA)
- West Clark Lake (WBID 1971) (EPA)

Currently, Roberts Bay North is a Class III waterbody with designated uses of Recreation, Propagation and Maintenance of a Healthy, and Well-Balanced Population of Fish and Wildlife. The State assesses nutrient impairment using chlorophyll levels in two ways: if an annual average chlorophyll value exceeds 11 µg/L or if the chlorophyll *a* values in 2 consecutive years exceed historical values by more than 50%.

The FDEP established a TMDL for Roberts Bay North for nutrients in 2005 (FDEP, 2005). Impairment was based on exceedances of the historical chlorophyll *a* criterion, which was developed from the 1992 through 1996 data. The Roberts Bay North site-specific historical chlorophyll *a* threshold is 4.8 µg/L. Annual averages exceeded the threshold in 1998, 1999, 2001, 2003, and 2005 and ranged close to the threshold in 2000, 2002, and 2004. Annual averages in 2006 and 2007 were much lower (Figure 4-12). This corresponds to a reduction in freshwater inflows associated with an extended drought period in southwest Florida. Chlorophyll *a* levels in Roberts Bay North were more than 50% above historical levels in 6 of the 8 years between 1997 and 2004. Therefore, it was verified as impaired.

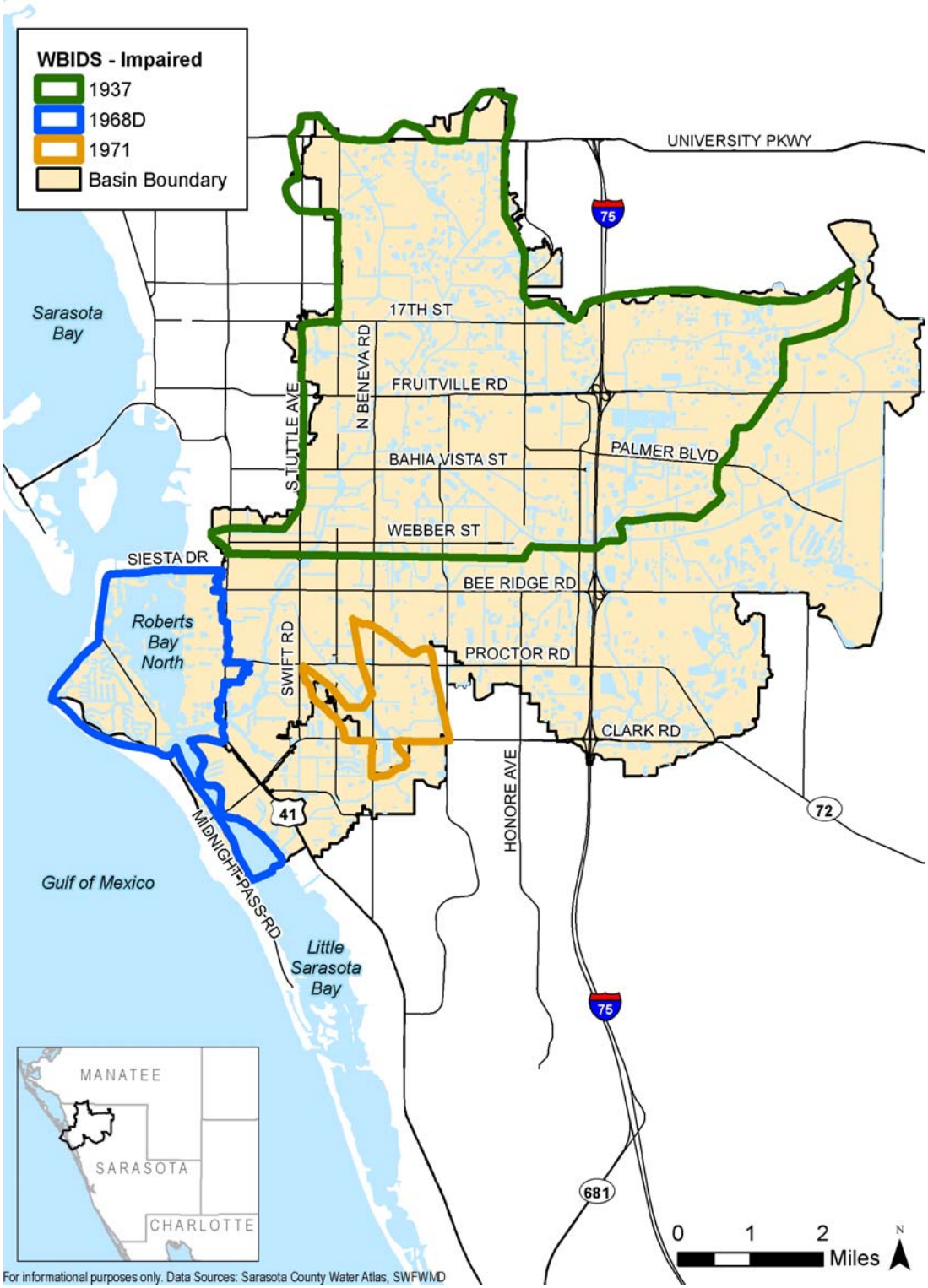

Figure 4-11 Roberts Bay North Impaired WBIDs

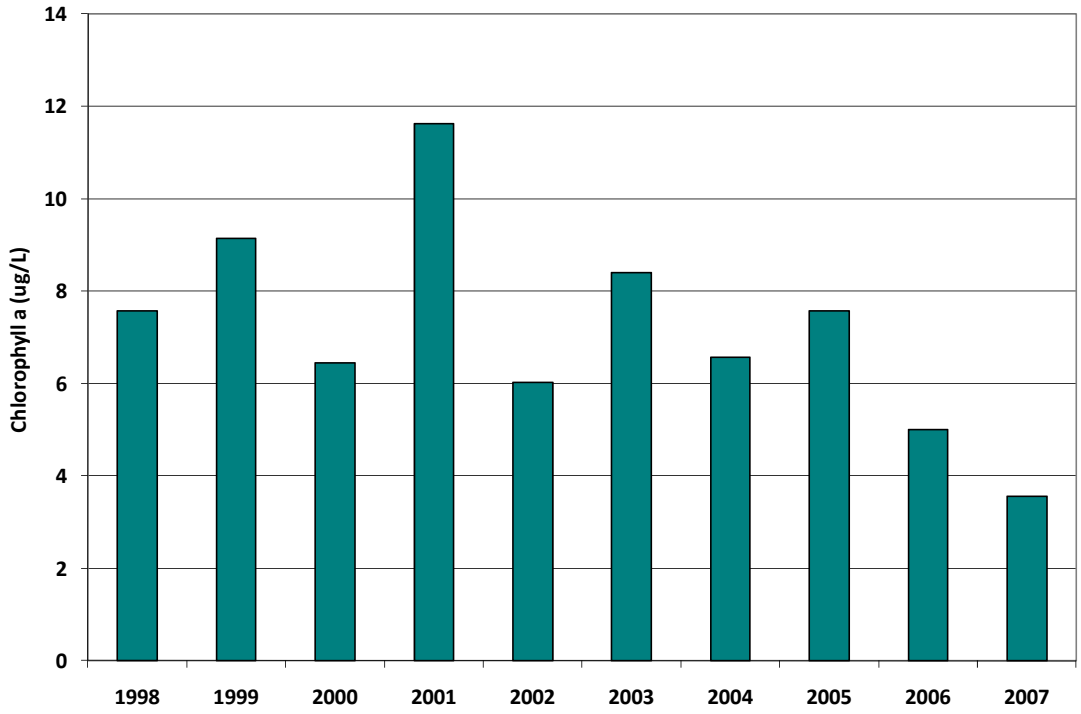

Figure 4-12 Annual Average Chlorophyll *a* Values from 1998 through 2007 in Roberts Bay North Estuary (WBID 1968D)

To determine the appropriate TMDLs for Roberts Bay North proper (FDEP 2005), loadings estimates were developed for point sources, municipal separate storm sewer systems (MS4s), atmospheric deposition, groundwater, septic systems, and nonpoint sources for the historical period (1992–1996). The average annual TN load in 1992–1996 was 679,090 lbs/year. The average estimated load for 1997–2004 was 683,219 lbs/year. FDEP concluded that a 0.6% reduction in TN loads annually is required to return to historical levels.

Although Roberts Bay North proper was listed as impaired for historic chlorophyll *a* during the original TMDL basin analyses, it is no longer considered impaired. Using more recent data there are now more than three consecutive years with chlorophyll *a* annual means below the chlorophyll *a* threshold value of 11 µg/L.

A TMDL for West Clark Lake for nutrients (TN and TP) was published in September 2005 (EPA, 2005). Although the TMDL was developed by FDEP it was published by EPA Region 4 and is not on the FDEP verified list. Impairment was due to exceedances in Trophic State Index (TSI). The TSI weighs chlorophyll *a*, TN, and TP concentrations to determine a score that can be compared to the IWR standard. If the average TSI value for an individual year exceeds 60 and color > 40 PtCo units, a WBID can be classified as verified as impaired under the IWR. The 2003 TSI for West Clark Lake exceeded 60 in all four quarters and averaged 69.

To determine the appropriate TMDLs for West Clark Lake, a land use-based model similar to the one employed for Roberts Bay North proper was used. However, components for point sources, atmospheric deposition, and septic systems were not included in the estimation of 2003 loads to West Clark Lake. The TN load was 12,842 lbs/year and the TP load was 1,660 lbs/year. To achieve a TSI of 60 for West Clark Lake, the TMDL recommends loadings reductions to 7,611 lbs/year of TN and 1,480 lbs/year of TP. These figures are equivalent to loadings reductions from 2003 levels of 59.3% for TN and 10.8% for TP.

4.1.3.2 Other Impairments within the Roberts Bay North Watershed

Group 3 basins are those watersheds that are assessed for TMDLs during the third year of FDEP's five-year cyclic Basin Assessment program. Group 1 basins were first assessed in 2002, and Group 3 in 2005. Group 3 basins are now undergoing their second assessment cycle in 2010.

In 2005, Group 3 Basins were evaluated for exceedances of FAC water quality criteria and, when deemed to be verified as impaired, were prioritized as High, Medium, or Low for TMDL development. Those WBIDs categorized as High Priority were slated for immediate TMDL development in the first cycle of TMDLs in 2005, whereas Medium priority TMDLs for other impaired WBIDs, including Phillippi Creek, were slated for TMDL development in the second cycle in 2010.

Proposed TMDLs for DO, nutrients, and fecal coliforms were published for Phillippi Creek (WBID 1937) in September 2009. The proposed TMDLs for DO and nutrients were developed because of exceedance of the Class III freshwaters criterion for DO. The criterion for DO is "Shall not be less than 5.0 mg/L. Normal daily and seasonal fluctuations above these levels shall be maintained." [2-302.530 (31), FAC] The water quality assessment was performed using data from the FDEP IWR Version 35 database. Constituents related to DO and nutrients evaluated were DO, BOD, total organic carbon (TOC), pH, chlorophyll *a*, and nutrients (nitrogen and phosphorus).

The proposed TMDLs for DO and nutrients were stated as percentage reductions in loadings of TN, TP, and BOD. The proposed reductions of 70% in loadings of these constituents were based on the reductions deemed necessary to maintain DO levels in the WBID above 5.0 mg/L at all times (EPA, 2009a). The basis for the percentage reductions was a combined hydrodynamic and water quality model of Phillippi Creek developed by that indicates that 70% reductions in loadings of TN, TP, and BOD are necessary to achieve water quality standards with respect to DO.

The appropriateness of the existing DO standards has been a subject of much discussion recently. These standards are viewed by many as not be reasonable given the physical nature of many of Florida's water bodies. Investigation of this issue is merited.

4.2 WATER QUALITY TARGETS

Arguably the single-most important element of an effective WMP is setting resource protection targets. There are four common approaches to setting targets (Janicki Environmental, 2002):

- Targets based on historical conditions.
- Targets based on reference system conditions.
- Targets based on regulatory standards.
- Targets based on the environmental requirements of critical resource(s).

Although one approach may be used by itself, a preferred method is to develop potential targets using more than one approach and to look for unifying results among these approaches to guide water quality target selection (Janicki Environmental, 2002, 2003). The following discusses each of the potential approaches:

- Historical conditions—If data that describe water quality in historical (undegraded) conditions exist, then that condition can be used as a restoration target. The use of this approach is typically desirable when historical data are available. However, data comparability is often limited due to sample design and methodological differences between historical and current data.
- Water quality standards—For this approach water quality monitoring data are compared to established standards to identify any samples failing to meet the standards. The presumption is that if water quality does not meet standards, then there is a problem. This approach is straightforward and is acceptable if adopted standards are appropriate. However, target setting using the standards approach is much less definitive if non-numeric standards are used, for example Florida's nutrient standards. To improve this situation, FDEP is currently in the process of establishing numeric standards for nutrients (FDEP, 2009a).
- Reference sites—For the reference site method, conditions at an area of interest are compared to similar but undegraded sites. This method is also useful but is difficult to implement, partially because it is often not easy to identify a suitable reference site and real differences between the sites must be identified. EPA uses the reference site method frequently, most often for freshwater systems. The FDEP Stream Condition Index (SCI) is a reference site example (FDEP, 2007). The benefit of this bioassessment approach is that multiple site characteristics (hydrology, water quality, habitat disturbance, etc.) are integrated.
- Resource-based—Resource-based target setting is widely accepted as the preferable approach as it directly ties water quality to the resource of concern. Resource-based targets have been set for many waterbodies both locally (Tampa Bay, Indian River Lagoon, Caloosahatchee River) (Greening and Janicki, 2006; Tomasko et al., 2001; Steward et al., 2005) and nationally (e.g., Chesapeake Bay Program).

Effective watershed management is typically based on preserving existing features or on restoring degraded areas to desirable conditions. A critical initial step in this process is to determine what resources are most beneficial and should therefore receive priority attention. A resource of concern should be desirable and representative of a larger habitat or system. Its extent and status should be measureable and manageable; that is, there should be an available suite of actions that can be used to foster the resource of concern's sustainability.

Given the importance of seagrasses in the Roberts Bay North estuary, setting water quality targets based on the requirements for their growth and reproduction is preferred. Seagrass meets all of the criteria above. Seagrasses serve significant functions within the estuarine ecosystem. They help maintain water clarity by trapping fine sediments and particles with their leaves and stabilizing the estuarine sediments with their roots. Seagrasses are very effective at removing dissolved nutrients from water that can enter from land runoff. The removal of sediment and nutrients improves water clarity, thereby improving overall ecosystem health. Seagrasses provide nursery habitats for fish, crustaceans, and shellfish, providing a nursery ground for many recreationally and commercially valuable species. They are also food for organisms that inhabit them and marine mammals such as manatees and waterfowl such as ducks. Human activities can harm seagrasses by degrading estuarine water quality and promoting physical disturbances and algal blooms. Reductions in light availability associated with nutrient inputs and sediments can damage or eliminate seagrass habitat. If seagrass is thriving, then it is likely that the system is general healthy and extensive (and expensive) monitoring of other indicators may not be necessary. Seagrass can be mapped through field reconnaissance and aerial mapping to track its extent over time. Also, the spatial extent of seagrass growth depends on water clarity, which depends on other water quality parameters, including chlorophyll *a*, turbidity, and color.

Seagrass targets for Roberts Bay North have been established by the Sarasota Bay Estuary Program (SBEP). The process for defining targets for each of the SBEP segments was based on a comparison of the historical (ca. 1950) seagrass coverage to recent surveys conducted by the SWFWMD. A description of the District mapping effort can be found in Kaufman (2006). The SBEP defined the seagrass target as the larger of either the historical cover or the average of the last two seagrass surveys. Figure 4-13 presents the seagrass cover data used to establish the Roberts Bay North target. The seagrass target for Roberts Bay North is 348 acres, which is the average of the 2004 and 2006 coverages.

The seagrass target for Roberts Bay North is 348 acres.

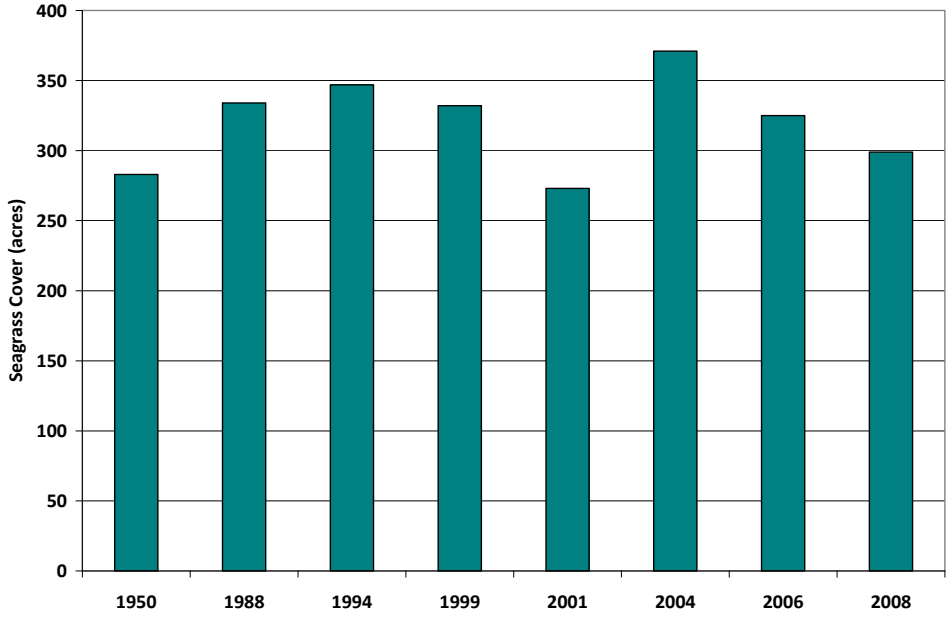

Figure 4-13 Seagrass Cover (Acres) from the Historical and Recent Surveys in Roberts Bay North

In the following discussion, water quality targets based on seagrass success and desirable salinity conditions and meeting DO standards in Roberts Bay North are defined. These targets will be applied to loading-water quality response models to estimate the loading targets to be addressed by the watershed projects and programs.

4.2.1 Seagrass-Related and Water Quality Standard-Based Targets

Given the seagrass target for Roberts Bay North, the next step in the target setting process is to determine the water quality conditions that are conducive to the protection and restoration of seagrasses. Water clarity, a measure of the amount of sunlight that can penetrate the water, is a significant determinant of seagrass success in a given estuary (Dawes et al., 2004). Clear waters are indicative of a healthy estuary, although many factors impact water clarity. Excess suspended sediments from runoff can negatively impact water clarity. Nutrients, mainly nitrogen and phosphorus, can fuel the growth of photosynthesizing algae. High chlorophyll *a* concentrations can also decrease water clarity. In turn, decreased water clarity can negatively impact seagrass cover, reducing habitat availability to the hundreds of species that depend on them.

Examination of the ambient water quality data shows the interrelationships among chlorophyll, light attenuation, and turbidity in Roberts Bay North (Figures 4-14 and 4-15). While light attenuation declines with both increasing chlorophyll and turbidity, more of the variation in light attenuation is related to variation in chlorophyll *a* concentrations as evidenced by the respective coefficients of determination (r^2).

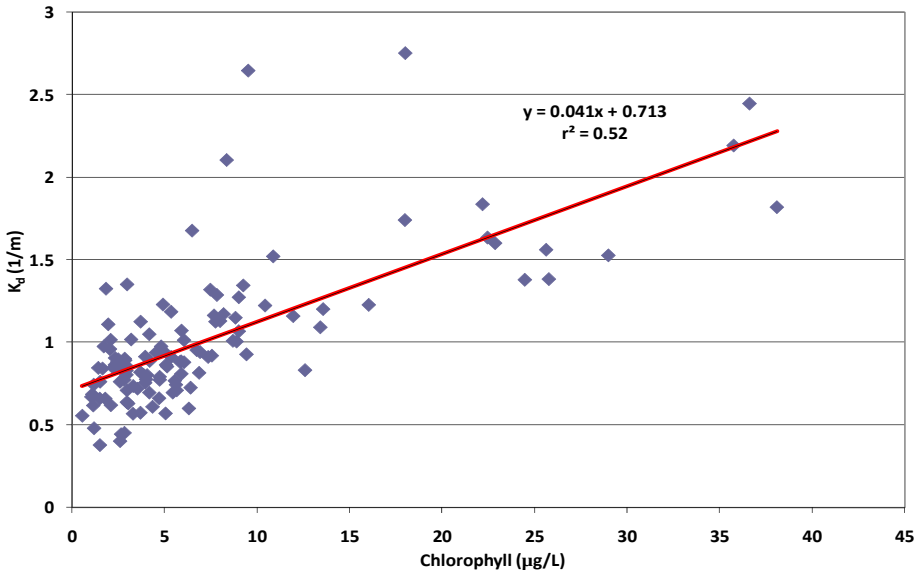

Figure 4-14 Relationship Between Chlorophyll *a* Concentrations and Light Attenuation in Roberts Bay North (1998–2007)

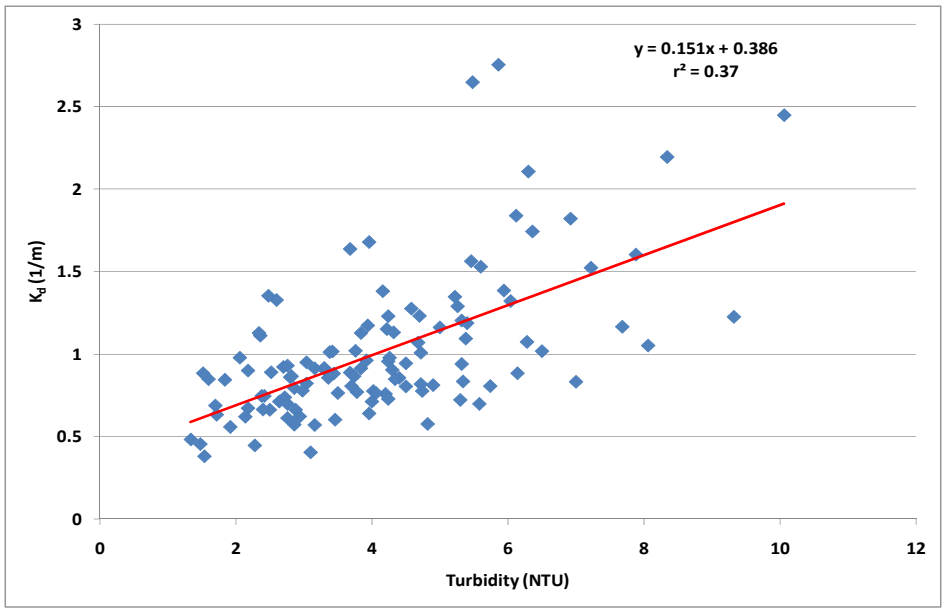

Figure 4-15 Relationship Between Turbidity and Light Attenuation in Roberts Bay North (1998–2007)

Water clarity targets were recently suggested for the estuarine waters of Sarasota County (Wessel et al., 2007). These targets were based on a light attenuation target identified by Corbett and Hale (2006) that is protective of seagrasses. Linking this target to the spatial and depth distributions of seagrasses provided segment-specific water clarity targets. This work was completed before the recent establishment of seagrass targets by the SBEP.

As discussed above, the recent seagrass coverage in Roberts Bay North meets or exceeds that estimated for the historical period (ca. 1950). This observation leads to either of two conclusions: the recent water clarity conditions in Roberts Bay North are conducive to seagrass growth and reproduction in those waters, or water clarity is not a critical determinant of seagrass cover in Roberts Bay North. Since the latter conclusion is not likely, it is reasonable to conclude that the recent water clarity conditions in Roberts Bay North are conducive to seagrass growth and reproduction in those waters.

In October 2009, the SBEP Policy and Management boards directed the Technical Advisory Committee to initiate the development of numeric nutrient criteria for the estuarine waters of the area. A subcommittee of the TAC began the process by reviewing the existing seagrass and chlorophyll *a* data and proposing a set of chlorophyll endpoints to support the development of the numeric nutrient criteria. Given that the recent extents of seagrasses are meeting the established targets, the subcommittee determined that the recent chlorophyll *a* concentrations and resultant water clarity are protective of the seagrasses in each of the SBEP segments. Upon review of the chlorophyll *a* concentration data, it was deemed appropriate to include not only the data from the 2004–2005 period but also from several antecedent years (2001–2003). The resultant mean chlorophyll *a* concentrations from this overall period (2001–2005) were established as the targets for each segment.

The subcommittee further recognized that there may be years in which these targets may be exceeded without causing significant reductions in seagrass cover. This means that there is some allowable amount of variation that should not elicit a significant degradation in water quality and therefore seagrass coverage. The subcommittee defined this level of variation as the standard deviation around the mean annual chlorophyll *a* concentrations in each segment for the entire period of record. Therefore, a distinction is made between a **target, i.e., a desired chlorophyll *a* concentration** and a **threshold**, **i.e., a chlorophyll *a* concentration above which undesirable chlorophyll *a* concentrations exist**. The chlorophyll *a* threshold for each segment is the sum of the target and the standard deviation around the mean annual chlorophyll *a* concentrations in each segment.

The chlorophyll *a* target for Roberts Bay North is 8.2 µg/L. The among-year standard deviation is 2.8 µg/L. Therefore, the threshold chlorophyll *a* concentration for Roberts Bay North is 11 µg/L. The K_d levels are the mean conditions during the 2001–2005 period that coincide with the period during which the seagrass targets have been set and represent the recent wide range in rainfall in this region. The K_d target for Roberts Bay North is 1.0 m^{-1}. The among-year standard deviation is 0.09 m^{-1}. Therefore, the K_d threshold is 1.09 m^{-1}. The DO target is a water quality standard-based target for estuarine waters. (See water clarity section in Chapter 9 for more detailed information on K_d)

The following Roberts Bay North water quality targets and standard deviations (for chlorophyll *a* and Kd) are:

- Chlorophyll *a* concentration – 8.2 µg/L and 2.8 µg/L.
- Kd – 1.0 (mL) and 0.1 (mL).
- DO – 4 mg/L.

The targets and standard deviations defined above have been applied in the development of the watershed report card discussed in Chapter 9.

4.2.2 Salinity Targets

4.2.2.1 Understanding of Estuarine Salinity Dynamics

To establish meaningful targets for salinity and eventually freshwater inflows in Roberts Bay North, an understanding of how freshwater inflows affect salinity is important. Estuaries are semi-enclosed coastal bodies of water that have at least one river or stream flowing into them and a connection to the sea. Salinity in estuaries varies spatially from fresher water at the point of the freshwater inflow in the upstream portion of the estuary to more saline water in the downstream portion where the estuary connects to the sea. Circulation patterns, both horizontal and vertical, also influence the spatial variation in salinity observed in estuaries (Figures 4-16 and 4-17).

As expected, increases in freshwater inputs from the watershed result in lower salinities in the estuary, while decreases in freshwater flows result in higher salinities in the estuary. Therefore, estuaries typically have seasonal patterns of higher salinities during the lower flow dry season and lower salinities during the higher flow wet season.

In addition to the seasonal pattern of salinity in estuaries, there is also a daily variation due to the tides. As the tide rises, salinities in the estuary increase as more saline water enters the system from the sea; as the tide falls, salinities decrease (Hardisty, 2007).

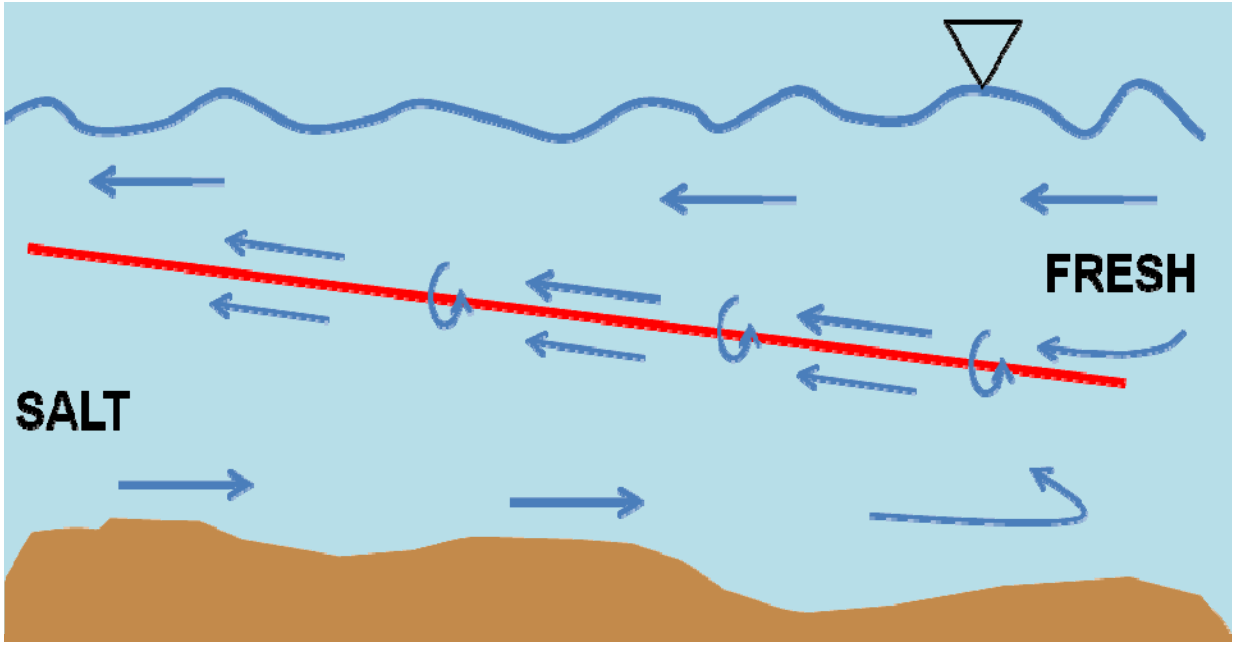

Figure 4-16 Conceptual Depiction of Vertical Tidally-Averaged Circulation Pattern (Modified from Goodwin, 1987)

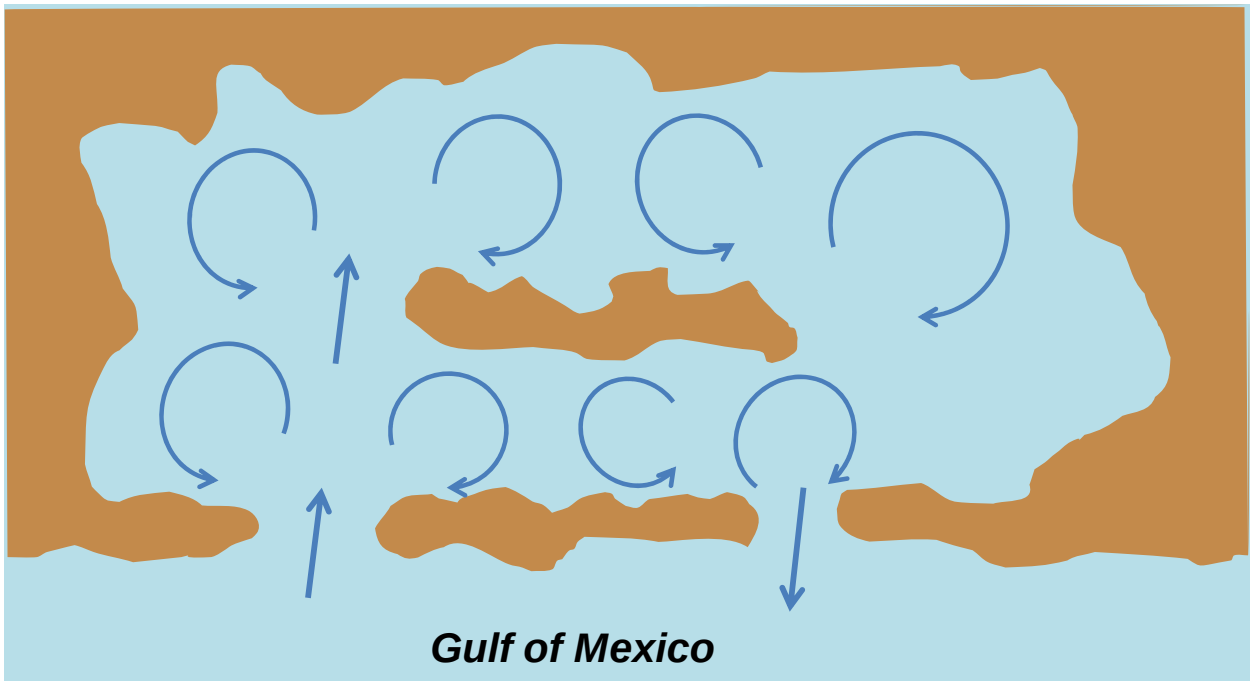

Figure 4-17 Conceptual Depiction of Horizontal Tidally-Averaged Circulation Pattern (Modified from Goodwin, 1987).

In addition to influencing salinities in an estuary, freshwater inflows also influence residence time. Residence time represents the amount of time that it takes for the water in the estuary to be replaced. Increases in residence time can result in depleted DO levels and increased accumulation of sediments (Nedwell and Raffaelli, 1999; Wolanski, 2007). Changes in residence

time resulting from temporal variation on freshwater inputs have been shown to affect the likelihood of excessive algal blooms (SWFWMD, 2008a; Janicki Environmental, 2008b).

Estuaries provide habitat for many organisms including fishes and benthic macroinvertebrates and therefore are characterized by their high diversity and primary production (Hobbie, 2000). Because salinity in estuaries varies considerably on daily and seasonal time frames, many organisms that inhabit estuaries can tolerate large variations in salinity. However, many of these organisms cannot tolerate completely fresh or very saline water, which is why they inhabit the brackish water of estuaries.

Temporal and spatial variations in salinity can have a direct impact on the composition and distribution of biota within an estuary (Hobbie, 2000; Wolanski 2007)—for example, fishes (Janicki Environmental 2004a and 2008a, SWFWMD 2008a) and benthic macroinvertebrates (Janicki Environmental 2007a and 2008b).

Human activity has significantly impacted many estuaries in the United States, often resulting in less available estuarine habitat because of pollution and physical alteration of systems (NRC, 1994). Human activities can lead to either reductions or increases in freshwater inflows to an estuary. Two examples in southwest Florida are the estuarine portion of the lower Hillsborough River and Dona Bay.

Recent analysis has shown that human activity has led to a decline in freshwater inflows to the lower Hillsborough River (SWFWMD, 2008b). The decline in freshwater inflows led to a loss of oligohaline habitat (water less than 5 ppt) in the lower Hillsborough River. To address the reduction in oligohaline habitat in the lower Hillsborough River, the minimum flow for the system was modified to maintain sufficient oligohaline habitat.

In Dona Bay, canal construction in the watershed adjacent to the historical Dona Bay watershed resulted in a large seasonal increase in freshwater inflows to Dona Bay (SWFWMD, 2009). The increase in freshwater inflows has negatively impacted seagrass and oyster populations in Dona Bay. The draft minimum flow for Dona Bay has recommended Minimum Flows Levels that would allow small flow reductions in Fox and Salt Creek.

Since salinity can vary significantly over a wide range of temporal and spatial scales and many estuarine organisms can tolerate large variations in salinity, defining a salinity *target* must necessarily account for these givens. Therefore, we recommend that a target salinity regime that accounts for these givens be defined. Freshwater input targets can then be defined based on the empirical relationship between salinity and freshwater inflows.

4.2.2.2 Relationship between Flows and Salinity in Roberts Bay North

In Section 4.1 we described the individual status and trends of the primary water quality parameters affecting the health and productivity of the Roberts Bay North estuary. Many of the

natural systems described in Section 3.2.1 have preferred conditions for success within the natural variation in estuarine systems. For example, the preferred range of salinity for the health and success of oysters has been identified as 14–28 ppt (Kennedy et al., 1996). Salinities less than 10 ppt inhibit the success of oyster larvae, while salinities higher than 30 ppt decrease growth rates and increase the likelihood of parasitic infection (Stanley and Sellers, 1986). Turtle grass, *Thalassia testudinum*, is another species that has salinity preferences within estuarine environments and generally prefers salinities above 20 ppt (Zieman and Zieman, 1989). Many estuarine fish taxa that use the Roberts Bay North estuary have preferential salinities as well (Serviss and Sauers, 2002). Therefore, the timing and volume of freshwater inputs into the Roberts Bay North estuary are important to providing one of the primary environmental requirements for the success of these important natural resources.

To evaluate the effects of hydrologic loadings on estuarine salinities, monthly freshwater volume estimates from the *Spatially Integrated Model for Pollutant Loading Estimates* (SIMPLE) model were related to empirical data on salinities from the ambient monitoring program. The sum of all monthly freshwater volumes from all basins in the watershed (including direct rainfall to the estuary) was calculated for each month in the time series from 1995 through 2007. These freshwater volumes were then matched to the empirical data averaged monthly across all measurements. The objectives of this process were to:

- Relate hydrologic volumes from SIMPLE model output to estuarine salinities.
- Identify differences in hydrologic loading between historical, current, and future conditions (See the Water Budget Section in Chapter 3 for a description of conditions used in the SIMPLE model).
- Estimate differences in salinities between historical, current, and future conditions.
- Establish potential hydrologic loading targets protective of salinity regimes.

To accomplish this, a predictive linear regression model was developed that estimated the bay-wide average salinity as a function of inflow volumes from the watershed. The regression included antecedent freshwater inputs including the freshwater volume loading to the estuary in the month preceding the salinity measurement as well as the current month's freshwater volume input. A seasonality term was also included to account for the differential effects of freshwater inputs throughout the year because of evapotranspiration, mixing, and differences in tidal amplitude as the result of the mixed semi-diurnal nature of tides in southwest Florida.

Based on empirical data, the regression relationship developed was used to predict salinities during historical and future conditions such that these hydrologic scenarios could be compared with respect to estimating the changes in estuarine salinity regimes in Roberts Bay North based on anthropogenic alterations to land-use characteristics that altered the natural hydrology.

Monthly average salinities in the Roberts Bay North estuary ranged from 10 ppt to 38 ppt with a median salinity of 31.5 ppt based on empirical data from 1998 through 2007. Model predictions

suggested that every 1000 acre-feet of freshwater introduced into Roberts Bay North monthly would decrease the bay-wide average salinity by approximately 0.5 ppt (Figure 4-18). While Figure 4-18 displays the generalized relationship between freshwater inflows and predicted salinities, the regression equation also depended on the freshwater volume reaching the estuary in the month preceding the salinity measure as well as the time of year when the salinity measurement was taken. The model performed reasonably well for its intended purpose with an r^2 value of 0.69 and 85% of the differences between observed and predicted salinities (i.e., the residuals) were less than 2.5 ppt (Figure 4-19).

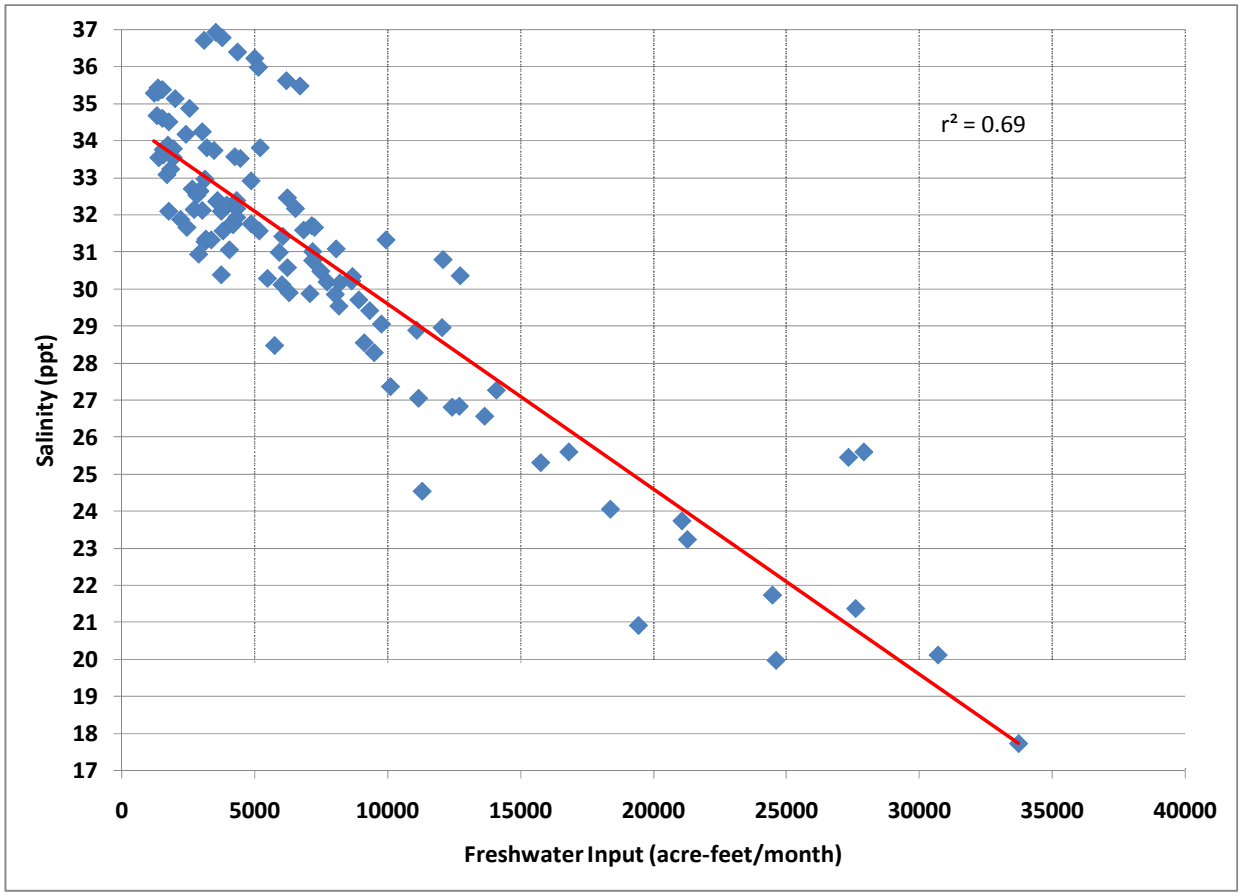

Figure 4-18 Relationship Between Freshwater Inputs (acre-feet/month) and Bay-Wide Average Predicted Salinities in the Roberts Bay North Estuary

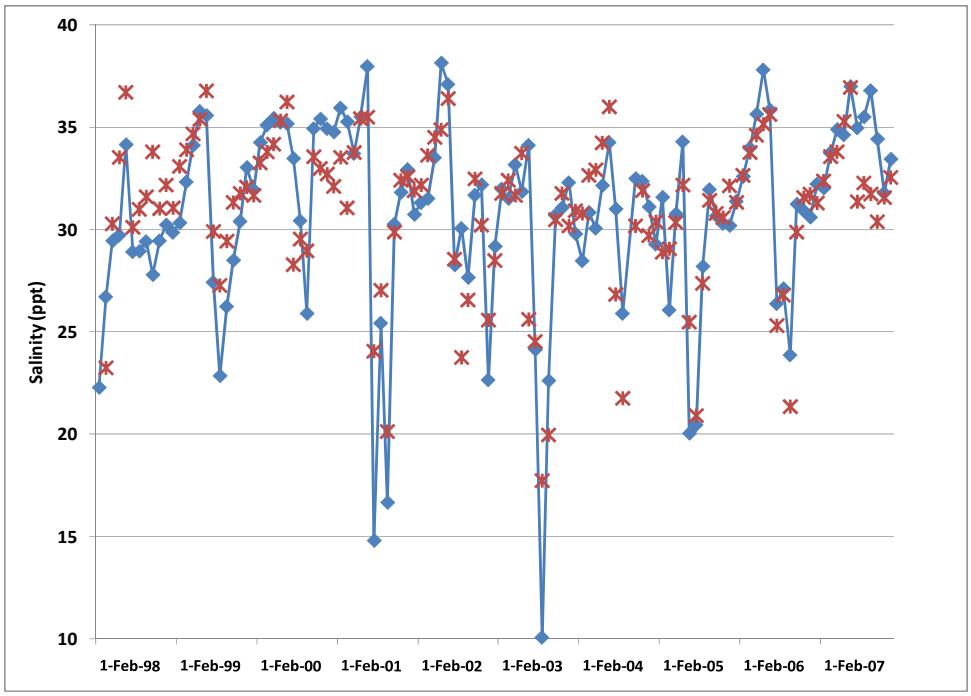

Figure 4-19 Time Series of Predicted (line) and Observed (Star) Bay-Wide salinity values in the Roberts Bay North estuary between 1998 and 2007.

The regression described above was used to hindcast the salinity distributions in Roberts Bay North under the historical conditions defined in Chapter 3. A cumulative distribution curve was produced to present the historical salinity distributions in Roberts Bay North (Figure 4-20). This curve represents the target salinity regime for Roberts Bay North.

Cumulative distribution curves were also produced to describe the differences between the historical and current distributions of hydrologic volumes. The SIMPLE model predictions indicate that current freshwater inputs to the bay tended to exceed the historical inputs (Figure 4-21) and resulted in a predicted difference in salinity in the estuary. The cumulative distribution function plots of the predicted salinities (Figure 4-22) suggested that historical salinity values were typically higher than current salinities and that the distribution of salinities has shifted by ca. 1.5 ppt between historical and current conditions.

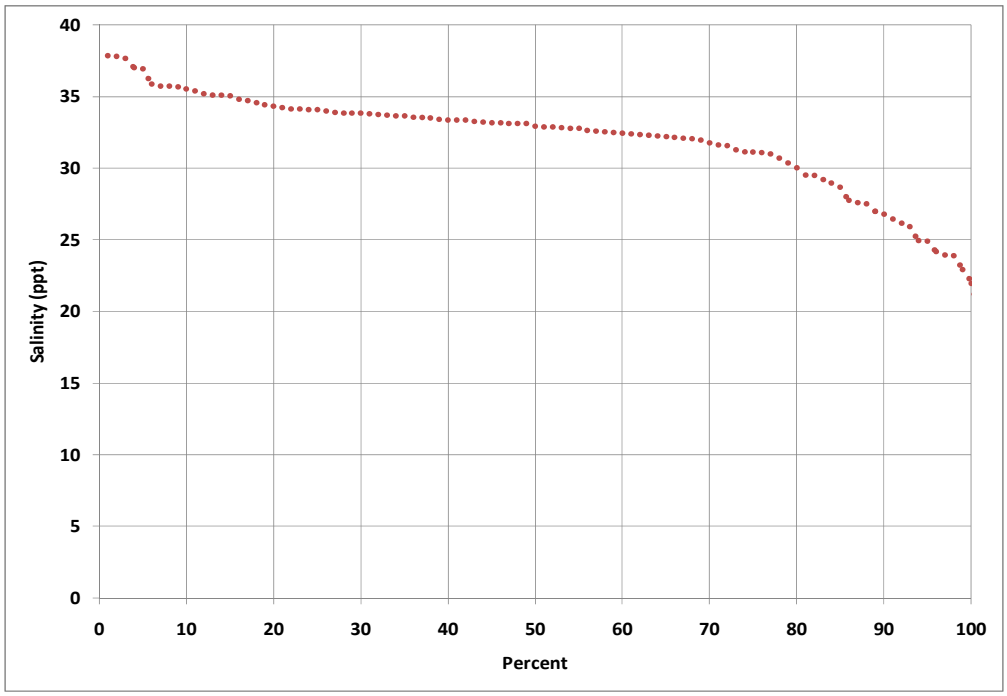

Figure 4-20 Hindcast of Historical Salinity Regime Based on the Relationship between Historical Flows and Bay-Wide Salinity in the Roberts Bay North Estuary

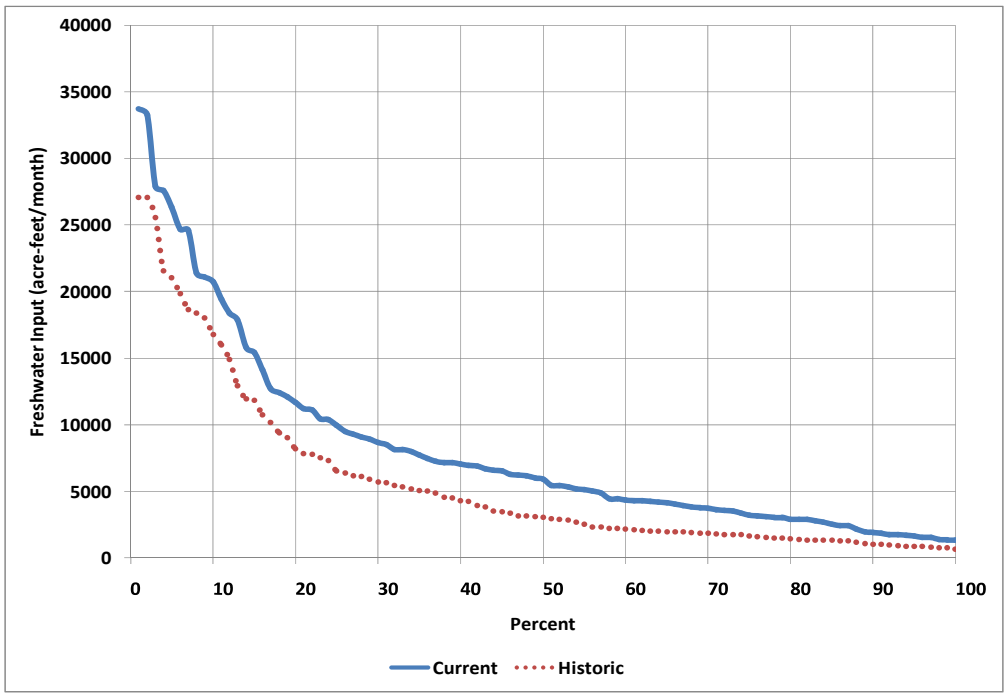

Figure 4-21 Comparison of Historical and Current Freshwater Input Distributions

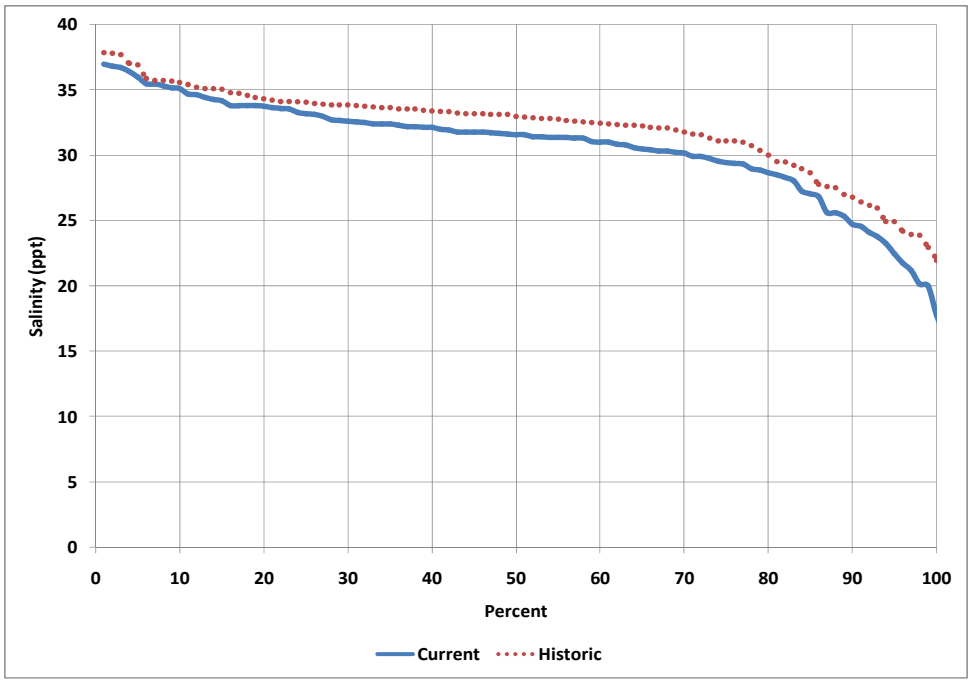

Figure 4-22 Comparison of Historical and Current Salinity Distributions for the Roberts Bay North Estuary

A difference of 2.5 ppt was chosen as a conservative estimate of a biologically relevant change in salinity in an estuarine environment, given the dynamic nature of salinity variation in an estuarine system. This difference also corresponds with a difference that would be outside the uncertainty of the regression model predictions. The differences between the historical and current conditions were calculated for each date in the time series and tabulated to define the proportion (percent) of days in a month when the difference was larger than 2.5 ppt. Differences in salinity greater than 2.5 ppt occurred primarily in the wet season between August and October, indicating that the greatest changes to estuarine salinities were decreased salinities in the wet season (Figure 4-23). Therefore, if the target salinity regime presented in Figure 4-20 is to be attained, then actions to reduce the wet season flows will be necessary.

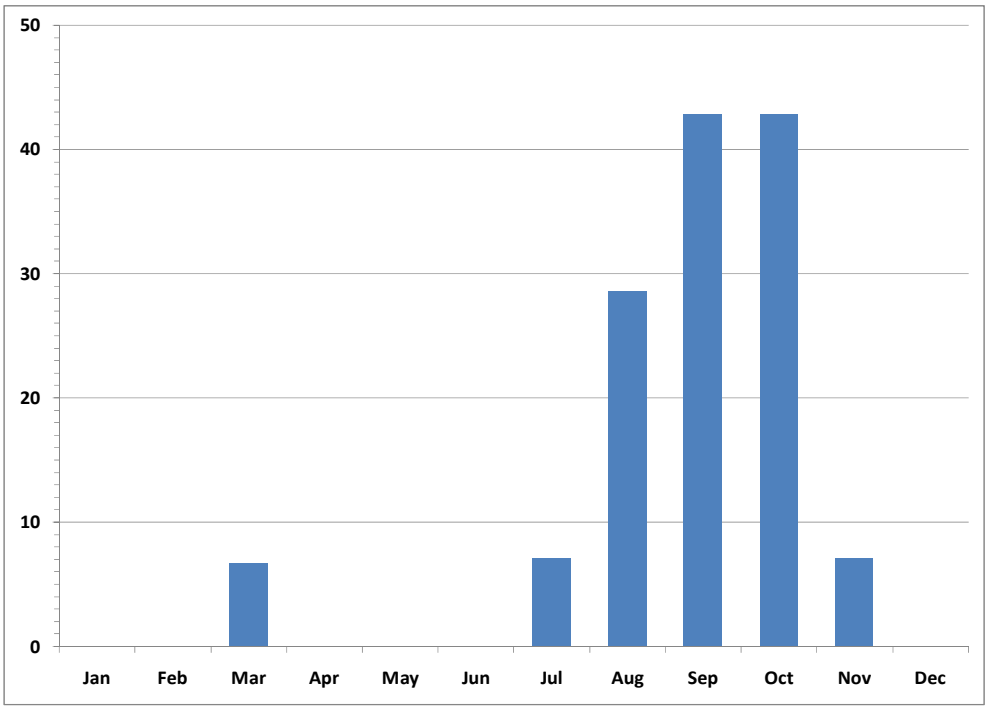

Figure 4-23 Percent of Differences Between Historical and Current Salinities Greater Than 2.5 ppt over a 14-year Simulated Rainfall Record

Despite the observation that salinities were different between historical and current conditions and that those differences appeared to be largest during the summer, the current salinities in Roberts Bay North remained in the polyhaline to euryhaline range with summertime median and average salinities above 25 ppt throughout Roberts Bay North (Figure 4-24). While spatial differences exist with respect to the influence of freshwater volume loadings into Roberts Bay North with lower salinities found in the southern portions of the estuary, these salinities do not appear to be detrimental to the critical natural resources inhabiting the estuary (e.g., mangroves, seagrasses, and oysters). Attempts to mitigate the effects on increased freshwater volumes entering Roberts Bay North for retaining historical salinity regimes should concentrate on capturing wet season discharges from the watershed. These aspects of the water budget are described in detail in the watershed portion of the natural systems section dealing with the water budget (Chapter 3, Section 3.1.2).

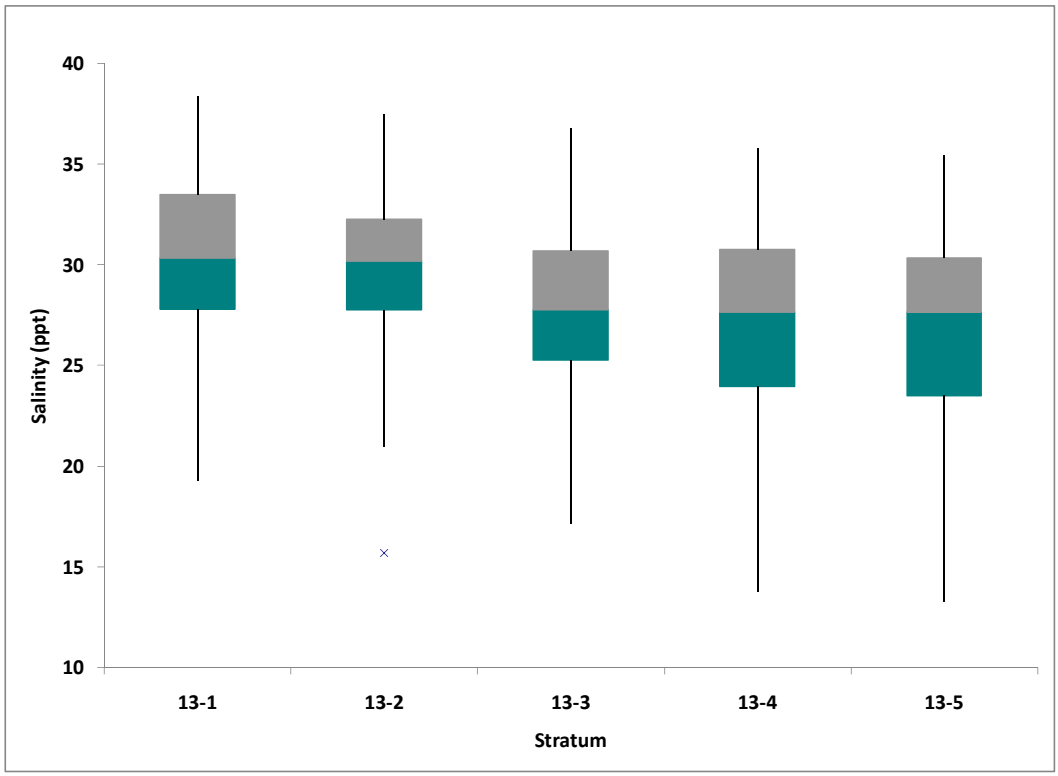

Figure 4-24 Distribution of Observed Wet Season (i.e., July–October) Salinities in Roberts Bay North by Stratum

4.3 POLLUTANT-LOADING ANALYSIS

A thorough understanding of the nature, sources, and spatial and temporal variability in pollutant loads is necessary if an effective WMP is to emerge. This understanding will aid in a further understanding of the manner and degree to which the receiving waters will respond to the pollutant loadings.

A generalized conceptual relationship between watershed inputs and water quality responses is provided in Figure 4-25. Altered freshwater inputs can significantly alter salinity patterns in estuaries and the community structure of biota within the system. Additionally, estuarine residence time depends on freshwater inputs and can influence the water quality responses in the estuary to changes in watershed loadings.

Since the current freshwater inputs are somewhat higher than historical, the current salinities as a result are correspondingly lower, especially in the summer months.

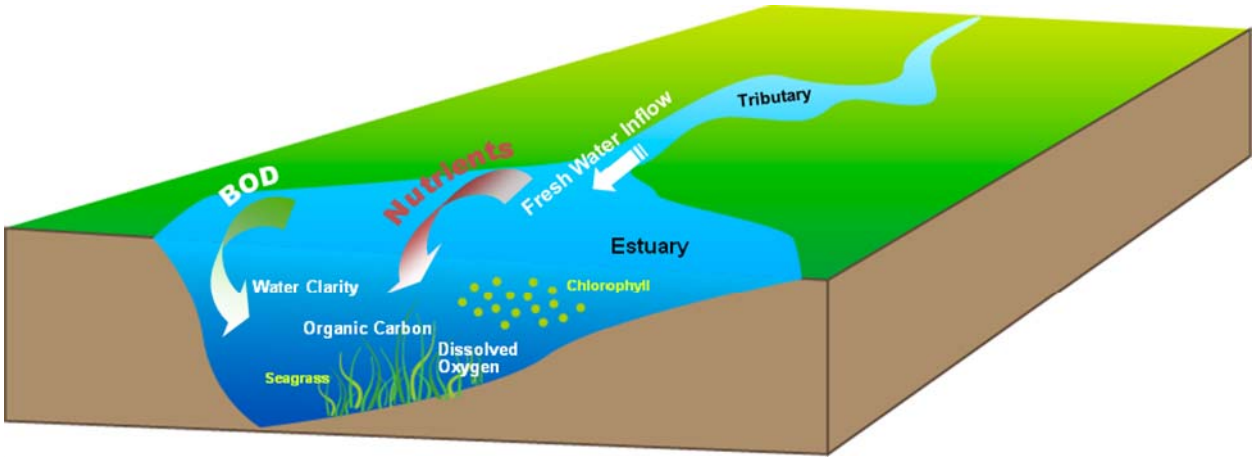

Figure 4-25 Conceptual Illustration of Watershed Loadings and Principal Indicators of Estuarine Health in Florida Estuaries

Water quality in a waterbody is influenced by the pollutants that reach the water body. Pollutants come from many sources, including runoff from land, groundwater flows, atmospheric deposition, and point sources. To improve water quality, managers must identify the pollutants that are responsible for the degradation in water quality. For example, in nearby Tampa Bay nitrogen was identified as the pollutant that was significantly contributing to decreases in water quality. This decline in water quality resulted in numerous adverse impacts. The elevated nitrogen loadings contributed to increased chlorophyll concentrations in the bay and a corresponding reduction in water clarity and seagrass abundance. Thus, management actions have been taken to reduce nitrogen inputs, and these management actions have contributed to an improvement in water quality.

Before management actions are implemented, resource managers must identify the pollutants that are responsible for the degradation of water quality. After the pollutants have been identified, the sources of these pollutants must be identified and quantified. The quantification of loading sources allows managers to focus their resources on those sources that make the greatest contribution to the problem. As expected, not all pollutant sources will be easy to manage. For example, loadings from atmospheric deposition can often originate outside the watershed and can therefore be difficult to manage. Point sources, on the other hand, are discrete sources of pollutant loadings that can generally be located and quantified with certainty.

With many watersheds, direct runoff (also known as nonpoint source runoff) represents a significant amount of the total load from the watershed. Direct runoff is the result of rainfall and is affected by land use and soils. Management of direct runoff is complicated due to the nature of runoff and the number of entities involved. Therefore, to control direct runoff, actions must be taken in concert with landowners and land custodians. This involves individuals from the owner of a single-family home to the city, county, state, and federal governments who are responsible for huge tracts of land including roadways, recreation areas, and conservation areas. State and local governments can also have a significant impact on direct runoff through the adoption of ordinances that relate to construction projects. For example, by requiring adequate water

retention areas (retention ponds, swales, etc.) as part of new construction, direct runoff can be greatly reduced as a portion of rainfall is sequestered and allowed to infiltrate the soil instead of directly running off into surface water ways.

4.3.1 Estimation of Pollutant Loading to Roberts Bay North

To better understand the influence of loadings to Roberts Bay North, a pollutant-loading model, the SIMPLE, was developed for the Roberts Bay North watershed in Sarasota County. Sarasota County contracted with Jones Edmunds & Associates, Inc. to determine hydrologic yield and loading estimates for a wide array of pollutants, including nutrients, metals, coliforms, and—specific to the present analysis—TN loads, BOD loads, and total suspended solids (TSS) loads, throughout the watershed. The model's spatial domain is divided into basins and subbasins throughout the watershed as seen in Figure 4-26. The temporal range for the model's application was from 1995 to 2007, with output produced at monthly intervals, which is roughly equivalent to the response time to these pollutant loads observed in Sarasota County's bays and estuaries (Jones Edmunds, 2008). An in-depth description of the model can be found in Jones Edmunds (2008).

The SIMPLE estimates loads from the following sources:

- Hydrologic model: The SIMPLE incorporates a hydrologic engine originally used in the Braden River Surface Water Resource Assessment (Jones Edmunds, 1997). Input data requirements for the SIMPLE hydrologic model include freshwater flows, NEXRAD rainfall, evapotranspiration rates, water surface elevations, land use, soils, and groundwater data.
- Baseflow module: Baseflow was calculated as part of the hydrologic model and was determined as a function of each unique NEXRAD pixel/land use/soil combination, as described in the direct runoff module. This module also includes an evapotranspiration term.
- Direct runoff module: To calculate loads based on direct runoff, data on NEXRAD rainfall, land use, soils, and best management practices (BMPs) were integrated into the SIMPLE. Land use data from 1990 and 2004 were used to estimate temporal change in the watershed and to determine runoff coefficients between pre-development and development conditions. Soils were used to estimate infiltration and runoff characteristics in the watershed. The BMP spatial data, like the land use component, were constructed to reflect temporal changes in their coverage between the pre-development and developed conditions. Each unique NEXRAD pixel/land use/soil combination was joined with Event Mean Concentrations to determine loadings estimates.

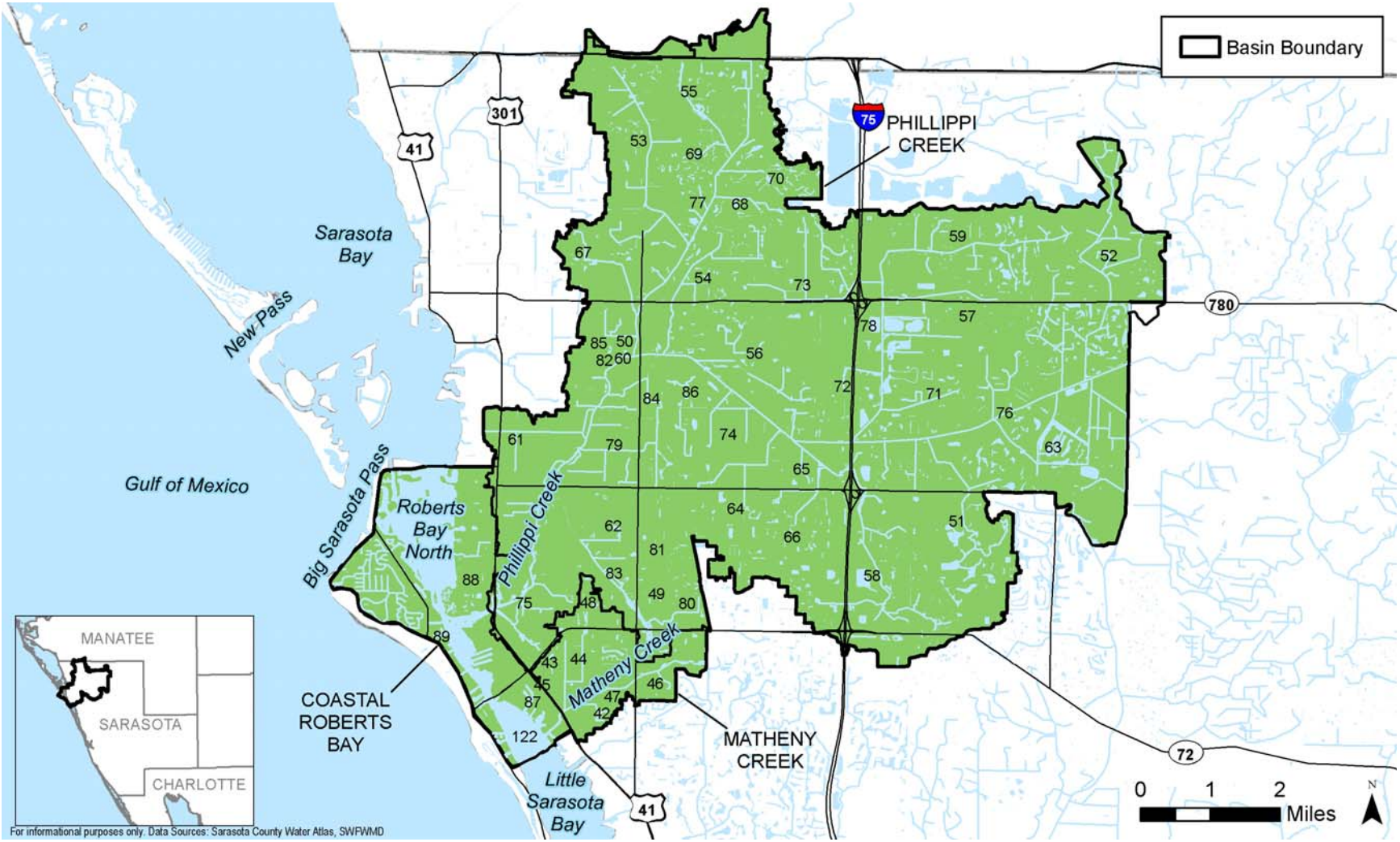

Figure 4-26 Overview of the Roberts Bay North Watershed Showing the Locations of the SIMPLE Basins and Subbasins

- Irrigation module: This module considers three sources of irrigation water: groundwater/potable, stormwater, and reclaimed water, with different concentrations used for each source. The potable and reclaimed water concentrations were set based on FDEP requirements, while stormwater, which is not yet regulated, was assumed to have concentrations similar to baseflow. The SIMPLE assumed that all residential, agricultural, commercial, and golf course land uses were irrigated.
- Point-source module: This module considers 38 non-delegated wastewater treatment plants (WWTPs), averaging less than 0.05 MGD, and 17 larger, delegated WWTPs, which discharge between 0.1 and 6.0 MGD, in the watershed. The smaller facilities typically serve small communities, campgrounds, and parks, while the delegated point sources serve larger municipalities. The method of calculating point source loadings was based on flow and concentration. Monthly data received from Sarasota County (non-delegated) and FDEP (delegated) were used to calculate loadings for the point source module. An overview of the point sources of the watershed shown with hydrology is presented in Figure 4-27.
- Septic tank module: Sarasota County provided Jones Edmunds with the spatial location of the approximately 45,000 septic tanks in the County. However, 80,000–90,000 septic tanks are estimated; the undocumented septic tanks were accounted for based on current septic and sewer coverages and the Sarasota County parcel coverage. Average flow rates were based on land use, either residential or non-residential, while three concentration levels were assigned (high, medium, and low), depending on soil type, the presence of BMPs, and the distance from the nearest conveyance.

Nitrogen loadings due to atmospheric deposition were estimated as follows. *Total atmospheric deposition* is defined as the sum of wet deposition (rainfall) and dry deposition (gaseous constituent interaction and dust fallout) directly to the surface of the bay. Deposition of pollutants to the watershed of the bay is incorporated into nonpoint source loading estimates.

Three data types are needed to estimate total atmospheric deposition:

- An estimate of the hydrologic load directly to the surface of the bay via precipitation.
- An estimate of the pollutant concentration in that precipitation.
- An estimate of dry deposition, either from empirical data or model-based estimates.

The hydrologic loads to the surface of the bay via precipitation were estimated in the same manner as for the hydrologic modeling effort. NEXRAD-derived rainfall provided by SWFWMD was used to derive monthly rainfall totals to the bay surface.

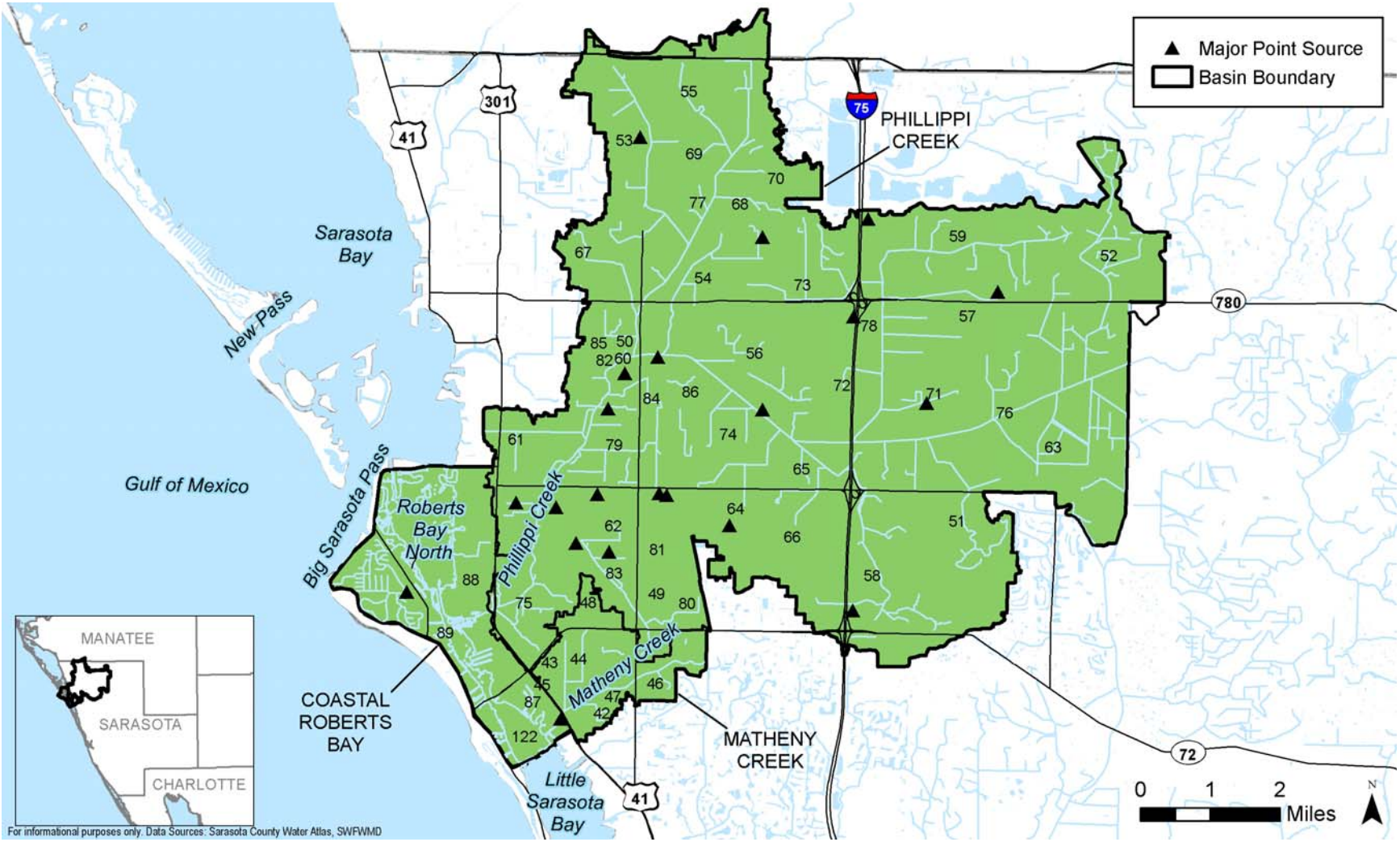

Figure 4-27 Overview of the Location of Major Point Sources within the Roberts Bay North Watershed

Precipitation-weighted mean monthly rainfall TN concentration data were obtained from the National Atmospheric Deposition Program (NADP) Verna Wellfield site in Sarasota County. The TN loadings from precipitation were estimated by multiplying the monthly precipitation-weighted mean TN concentrations from the Verna site and the monthly bay surface hydrologic loads to estimate monthly wet TN loads to the bay.

An estimate of dry deposition was also needed to develop total atmospheric deposition to the bay surface, as the total deposition is the sum of wet (rainfall) and dry deposition. The Sarasota Bay National Estuary Program initiated an intensive atmospheric deposition monitoring program in September 1998 that lasted for 1 year. From the atmospheric nitrogen concentration data collected during this 1-year monitoring period, dry deposition was estimated to make up approximately 29% of the total atmospheric deposition directly to the surface of Sarasota Bay (SBNEP, undated).

Another estimate of atmospheric deposition TN loading to the surface of Sarasota Bay was provided by a modeling effort using the CALMET/CALPUFF modeling system (Poor, 1999). The model results predicted that approximately 89% of the total nitrogen deposition to the surface of Sarasota Bay was from dry deposition. The predicted wet deposition to the surface of the bay was an order of magnitude less that that measured at the nearby Verna NADP site (Poor, 1999). Importantly, the modeling effort indicated that Sarasota Bay and Tampa Bay shared the same airshed (EPA, 2000).

Since a longer term record of atmospheric deposition data collection exists for Tampa Bay and since the two bays share the same airshed, dry deposition data collected as part of the Tampa Bay Atmospheric Deposition Study (TBADS) were used for this effort. This study was conducted for a 10-year period (August 1996 through June 2006) and included sampling elements for both wet and dry atmospheric deposition at an intensive monitoring site located on the Gandy Bridge Causeway. The data available from TBADS have been used to estimate atmospheric deposition to Tampa Bay. These data include precipitation nitrogen concentration data, wet and dry deposition rates, and an estimate of the ratio of dry:wet deposition (Poor, 2000; Pribble et al., 2001). Seasonal ratios of dry:wet deposition were derived from the TBADS data, with the wet season ratio of 0.66 indicating that dry deposition makes up approximately 40% of the total deposition in the wet season, and the dry season ratio of 1.05 indicating that dry deposition makes up approximately 51% of the total deposition in the dry season. Both of these seasonal proportions are greater than that from the 1-year Sarasota Bay study, which found 29% of the total deposition was due to dry deposition. However, the lower value from the 1-year Sarasota Bay study may be an artifact of the much shorter data-collection period, and the longer-term record from the TBADS study is assumed to provide a more accurate representation of the typical contribution from dry deposition over a longer period of time for the airshed including Sarasota Bay and Tampa Bay.

Using monthly precipitation nitrogen concentrations from the Verna NADP site and the NEXRAD-derived monthly rainfall, the equation for wet deposition of nitrogen is as follows:

$$Nwet_m = [N]_m * H_m,$$

where:

$Nwet_m$ = wet deposition of nitrogen for each month *m*,

$[N]_m$ = mean precipitation-weighted nitrogen concentration in the rainfall measured at the Verna Wellfield for each month *m*, and

H_m = estimated hydrologic load from rainfall for each month *m* to the bay surface.

Dry deposition was estimated using the TBADS-derived seasonal dry:wet deposition ratio, which was 1.05 for the dry season (months 1-6, 11, and 12) and 0.66 for the wet season (months 7-10), as follows:

$$Ndry_m = \text{Seasonal Deposition Ratio} * Nwet_m,$$

where:

$Ndry_m$ = dry deposition of nitrogen for each month *m*, and

$Nwet_m$ = wet deposition of nitrogen for each month *m*.

The total atmospheric deposition to a surface of the bay was given as the sum of the wet and dry deposition, as follows:

$$Ntot_m = Nwet_m + Ndry_m,$$

where:

$Ntot_m$ = total atmospheric deposition of nitrogen for each month *m* to the surface of the bay.

The monthly TN loadings were then summed over each year to provide annual loadings from atmospheric deposition directly to the surface of the bay.

4.3.2 Analysis of the Sources and Temporal and Spatial Variability in Pollutant Loadings to Roberts Bay North

An understanding of the relative importance of the sources of pollutant loads to Robert Bay North and the spatial and temporal and temporal variability in these loads provide a critical basis for the WMP development. Given limited resources, knowledge of "How much" and "Where" necessitates the appropriate prioritization of management actions.

4.3.2.1 Source Attribution

The majority of the TN loading to Roberts Bay North from 1995 through 2007 was from direct runoff (62%), base flow (23%), and point sources (7%) (Figure 4-28). The remaining TN loadings were from septic, irrigation, and atmospheric deposition, accounting for 4%, 2%, and 2%, respectively. There was clear intra-annual variation on the relative contributions of TN loads (Figure 4-29). Direct runoff contributions were greatest during the summer months concurrent with the highest seasonal freshwater inputs. Conversely, during the dry season both base flow and point source contributions were greatest.

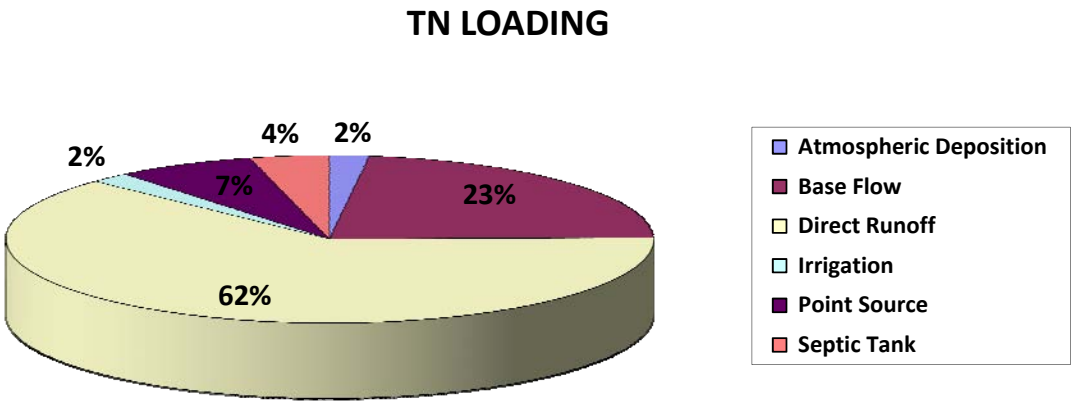

Figure 4-28 Relative Contributions from Each Source of TN Loads to Roberts Bay North (1995–2007)

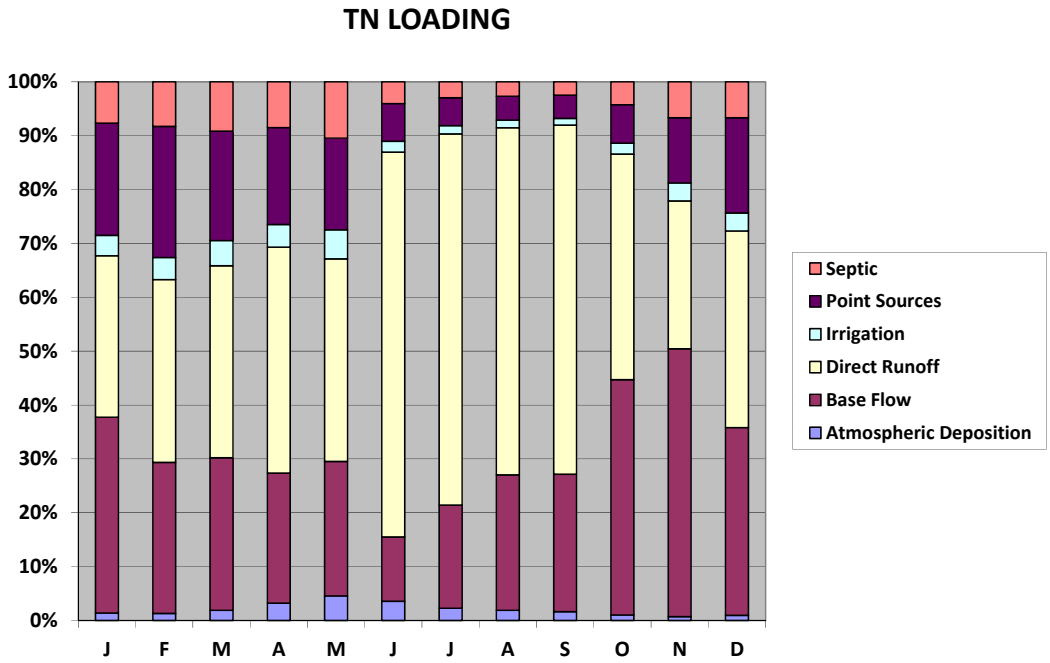

Figure 4-29 Monthly Variation in the Relative Contributions from Each Source of TN Loads to Roberts Bay North (1995–2007)

Similar analyses of the source attribution of TSS and BOD loads were completed. The majority of the TSS loading was from direct runoff (83%) and base flow (15.0%) (Figure 4-30). The remaining TSS loadings were from septic, irrigation, and point sources, accounting for 1%, 1%, and <1%, respectively. Seasonally, direct runoff contributions were greatest in the summer while base flow TSS loads were greatest during the dry season (Figure 4-31).

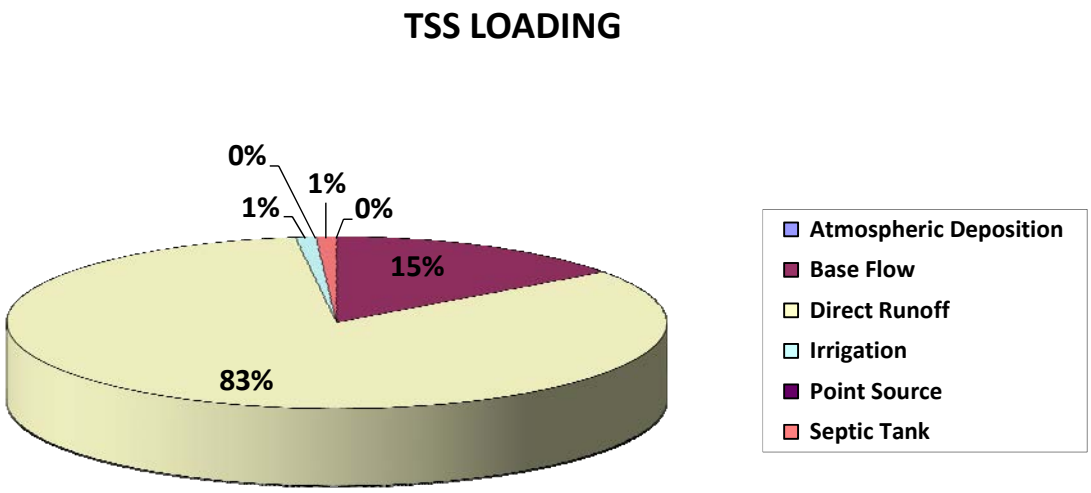

Figure 4-30 Relative Contributions from Each Source of TSS Loads to Roberts Bay North (1995–2007)

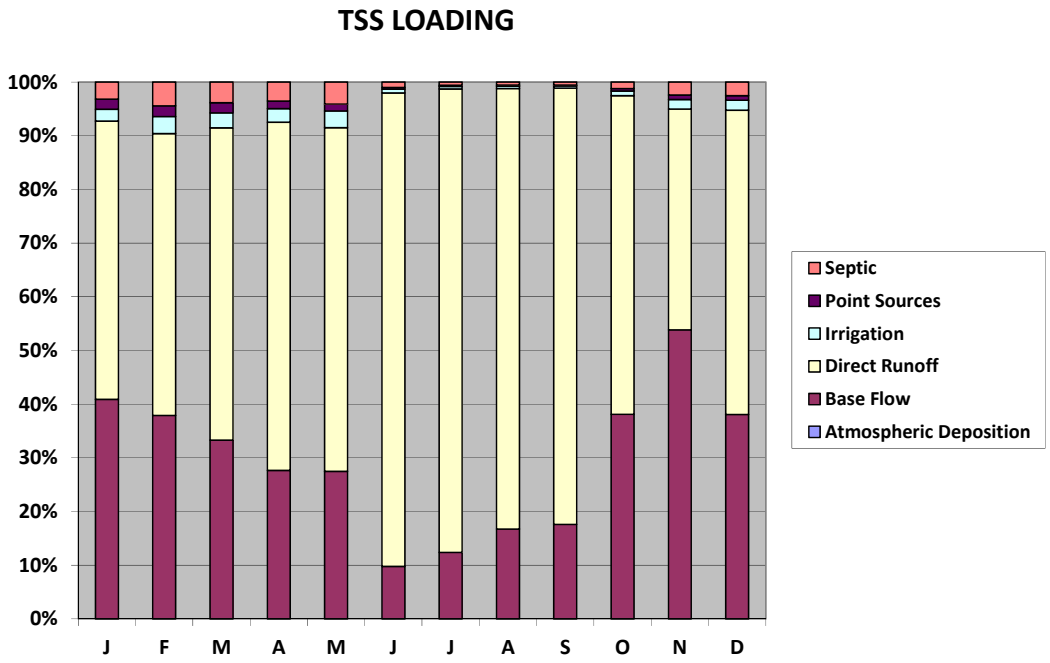

Figure 4-31 Monthly Variation in the Relative Contributions from Each Source of TSS Loads to Roberts Bay North (1995–2007)

The majority of the BOD loading was from direct runoff (70%) and base flow (20%) (Figure 4-32). The remaining BOD loadings were from septic, point sources, and irrigation, accounting for 6%, 3%, and 1%, respectively. Seasonal variation in BOD loads from direct runoff and base flow was similar to that observed for both TN and TSS (Figure 4-33).

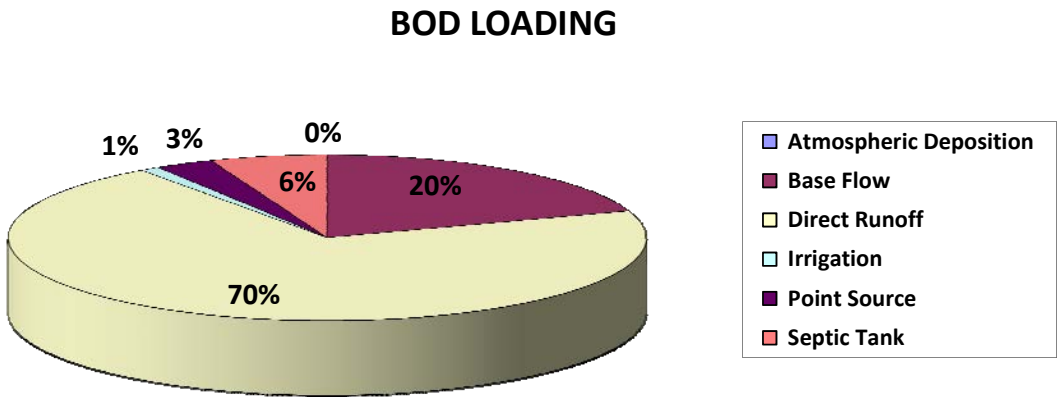

Figure 4-32 Relative Contributions from Each Source of BOD Loads to Roberts Bay North (1995–2007)

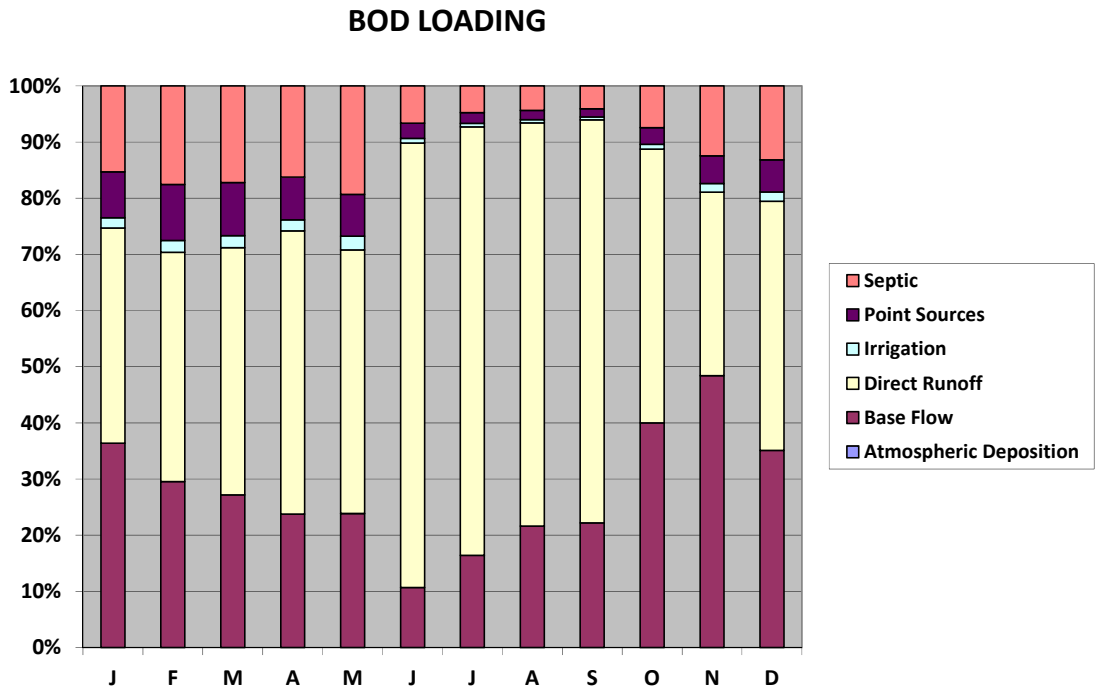

Figure 4-33 Monthly Variation in the Relative Contributions from Each Source of BOD Loads to Roberts Bay North (1995–2007)

4.3.2.2 Temporal Variability in Pollutant Loads to Roberts Bay North

Pollutant loads can vary significantly over time, and an understanding of this temporal variability is essential. Longer-term trends in loads can indicate changes in the nature of the watershed draining to the waterbody of concern. Seasonal variation in loads can also be an important determinant of the water quality responses in the receiving waterbody.

The total annual TN loads to Roberts Bay North varied significantly from a maximum of 285 tons in 1995 to a minimum of 82 tons in 2007 (Figure 4-34). The average annual TN load to Roberts Bay was 185 tons per year. Since direct runoff is the largest contributor to TN loads, large variations in annual loads are expected as rainfall varies from year to year. As a result of the seasonal variation in rainfall, TN loads are typically higher in the wetter summer months (Figure 4-35).

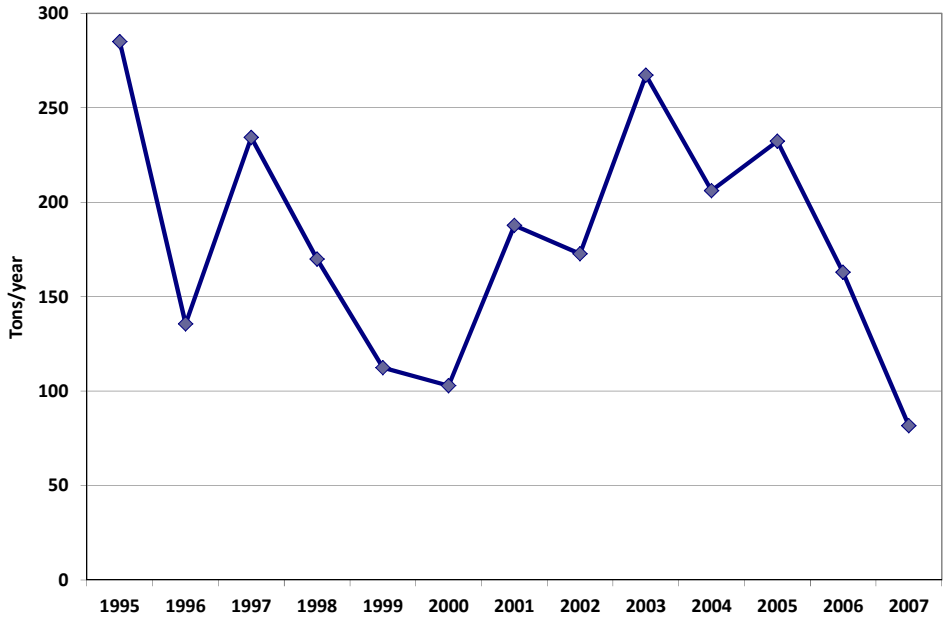

Figure 4-34 Interannual Variation in TN Loads to Roberts Bay North (1995–2007)

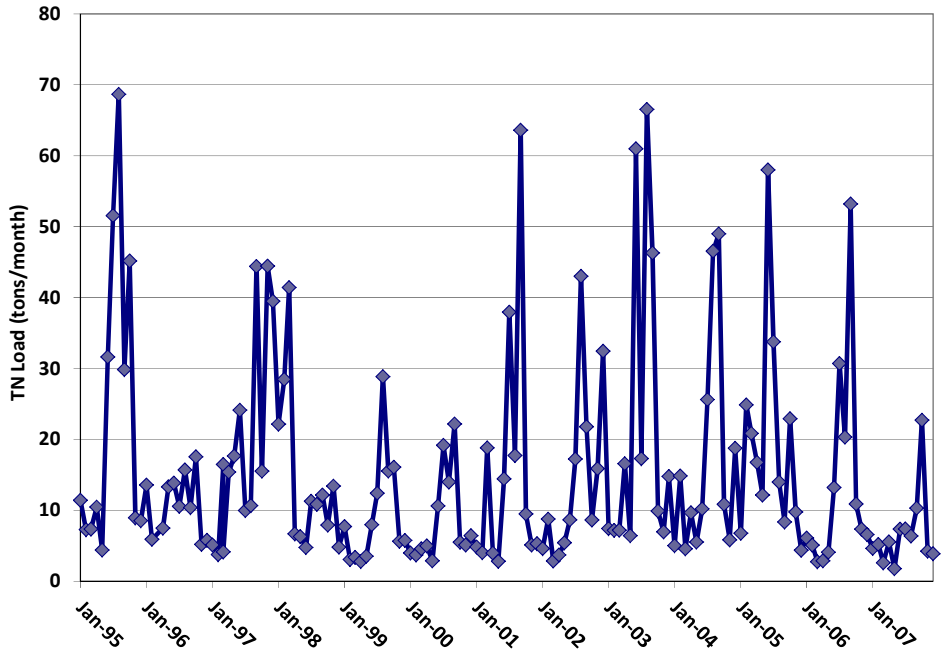

Figure 4-35 Monthly TN Loads to Roberts Bay North (1995–2007)

The total annual BOD loads to Roberts Bay North varied significantly from a maximum of 87415 tons in 1995 to a minimum of 269 tons in 2007 (Figure 4-36). The average annual BOD load to Roberts Bay was 560 tons per year. Since direct runoff is the largest contributor to BOD loads, large variations in annual loads are expected as rainfall varies from year to year. As a result of the seasonal variation in rainfall, BOD loads are typically higher in the wetter summer months (Figure 4-37).

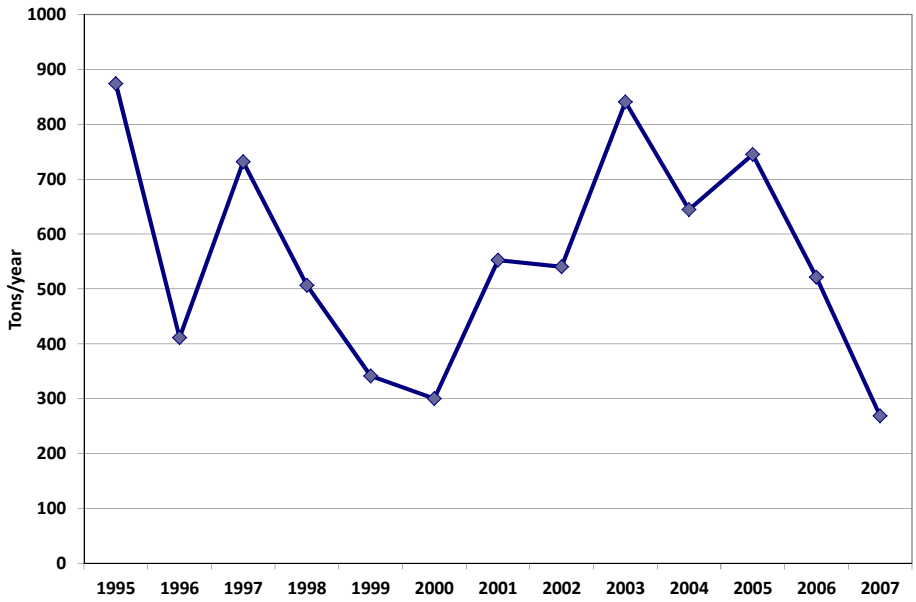

Figure 4-36 Interannual Variation in BOD Loads to Roberts Bay North (1995–2007)

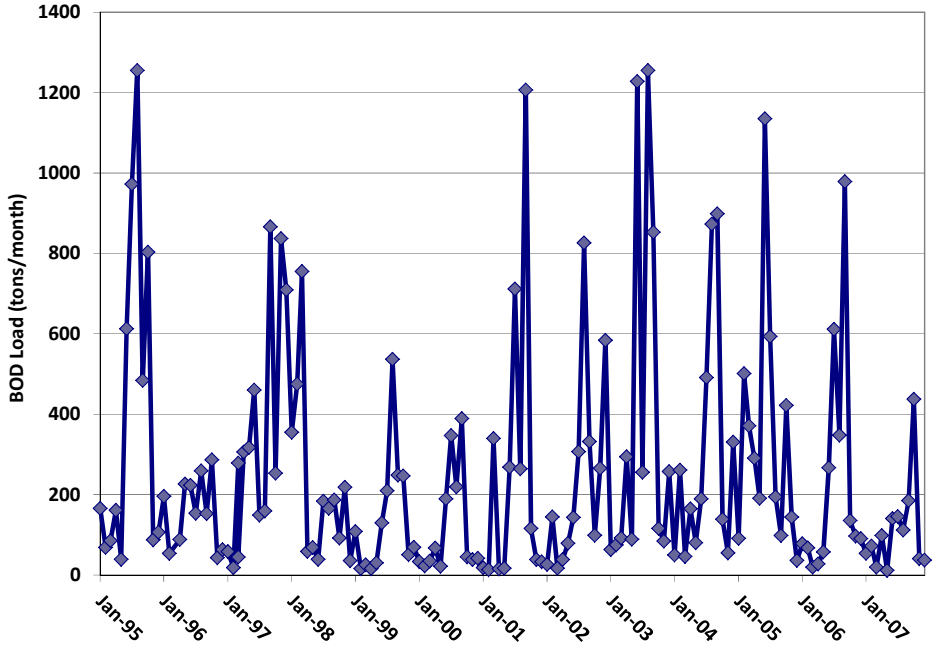

Figure 4-37 Monthly BOD Loads to Roberts Bay North (1995–2007.

The total annual TSS loads to Roberts Bay North varied significantly from a maximum of 4839 tons in 1995 to a minimum of 1351 tons in 2007 (Figure 4-38). The average annual TSS load to Roberts Bay was 3011 tons per year. Since direct runoff is the largest contributor to TSS loads, large variations in annual loads are expected as rainfall varies from year to year. As a result of the seasonal variation in rainfall, TSS loads are typically higher in the wetter summer months (Figure 4-39).

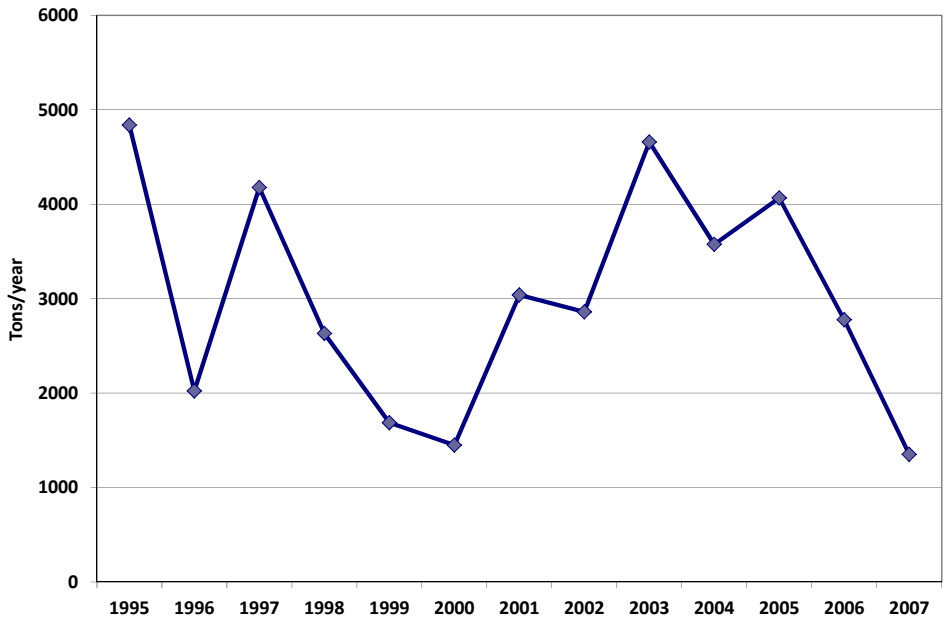

Figure 4-38 Interannual Variation in TSS Loads to Roberts Bay North (1995–2007)

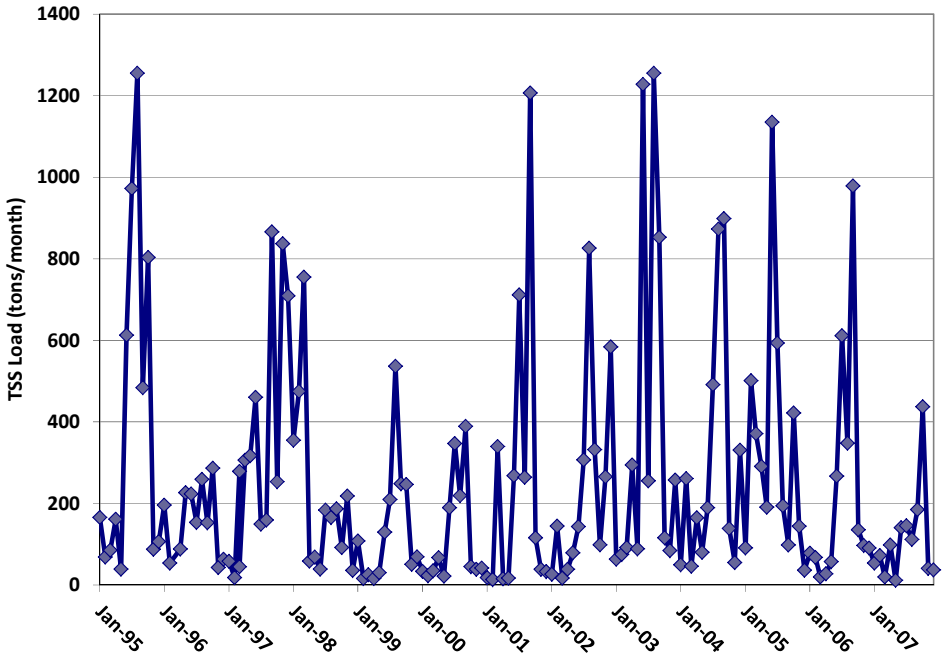

Figure 4-39 Monthly TSS Loads to Roberts Bay North (1995–2007)

4.3.2.3 Spatial Variability in Pollutant Loads to Roberts Bay North

In addition to an understanding of the temporal variability in pollutant loads, an understanding of the spatial variability in these loads is critical. With this understanding comes the focus for the potential projects and programs to address these loads.

The following loading estimates provided by the SIMPLE model are analyzed:

- TN loads.
- BOD loads.
- TSS loads.

The spatial variation in the pollutant-loading estimates is examined in two ways. First, the average annual total loadings (expressed as tons/year) from each basin in the Robert Bay North watershed (Figure 4-40) are discussed. Secondly, unit area loads (expressed as lbs/acre/year) from each subbasin are presented and examined.

Understanding of the spatial variability in pollutant loadings depends upon knowledge of the spatial variability of the volume of water delivered to the bay from various portions of its watershed. Chapter 3 shows that 84% of the total freshwater input to the bay originates in the Phillippi Creek basin. Figure 3-6 presents the normalized (ac-ft/ac) total volume of water for each subbasin within the Roberts Bay North watershed. The subbasins within the central portion of the Phillippi Creek basin and in the Coastal basin deliver the highest normalized (i.e., ac-ft/ac/yr) freshwater inputs to Roberts Bay North.

A. TN Loads

The average annual TN loads to Roberts Bay North are presented in Figure 4-40. More than 80% of the total TN load to the bay was generated in the Phillippi Creek basin.

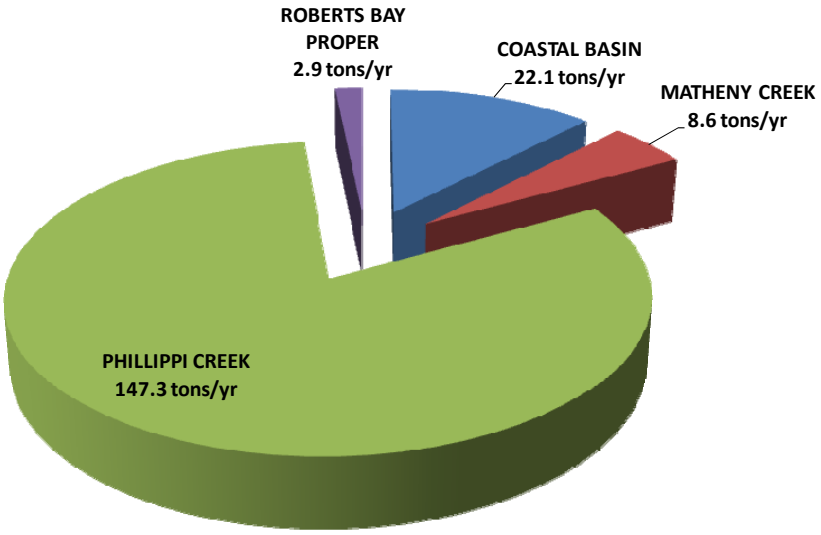

Figure 4-40 Average Annual TN Loads by Basin to Roberts Bay North (1995–2007)

Average annual unit area TN loads (lbs/acre/year) are highest in the Coastal Roberts Bay Subbasins 88 and 89 and in Phillippi Creek Subbasin 85, all of which are located in the watershed's most urbanized regions (Figure 4-41). As Chapter 1 shows, agricultural land uses are most predominant in the eastern half of Phillippi Creek. Four subbasins (52, 57, 76, and 71) in particular contain the highest proportion of agricultural land uses and each contribute moderate amounts of TN per unit area (Figure 4-41). Unit area TN loads from subbasins within the Matheny Creek basin were relatively low to moderate. The lowest unit area TN loads are found in Subbasins 4, 5, and 6 (these subbasins are small islands in Roberts Bay). We included these as part of Siesta Key, which had the highest unit area TN loads. The lowest unit area TN loads otherwise are Subbasins 46 and 70, which are Matheny Creek and Phillippi Creek, respectively, in the Coastal basin, where the largest proportion of upland vegetation land cover exists.

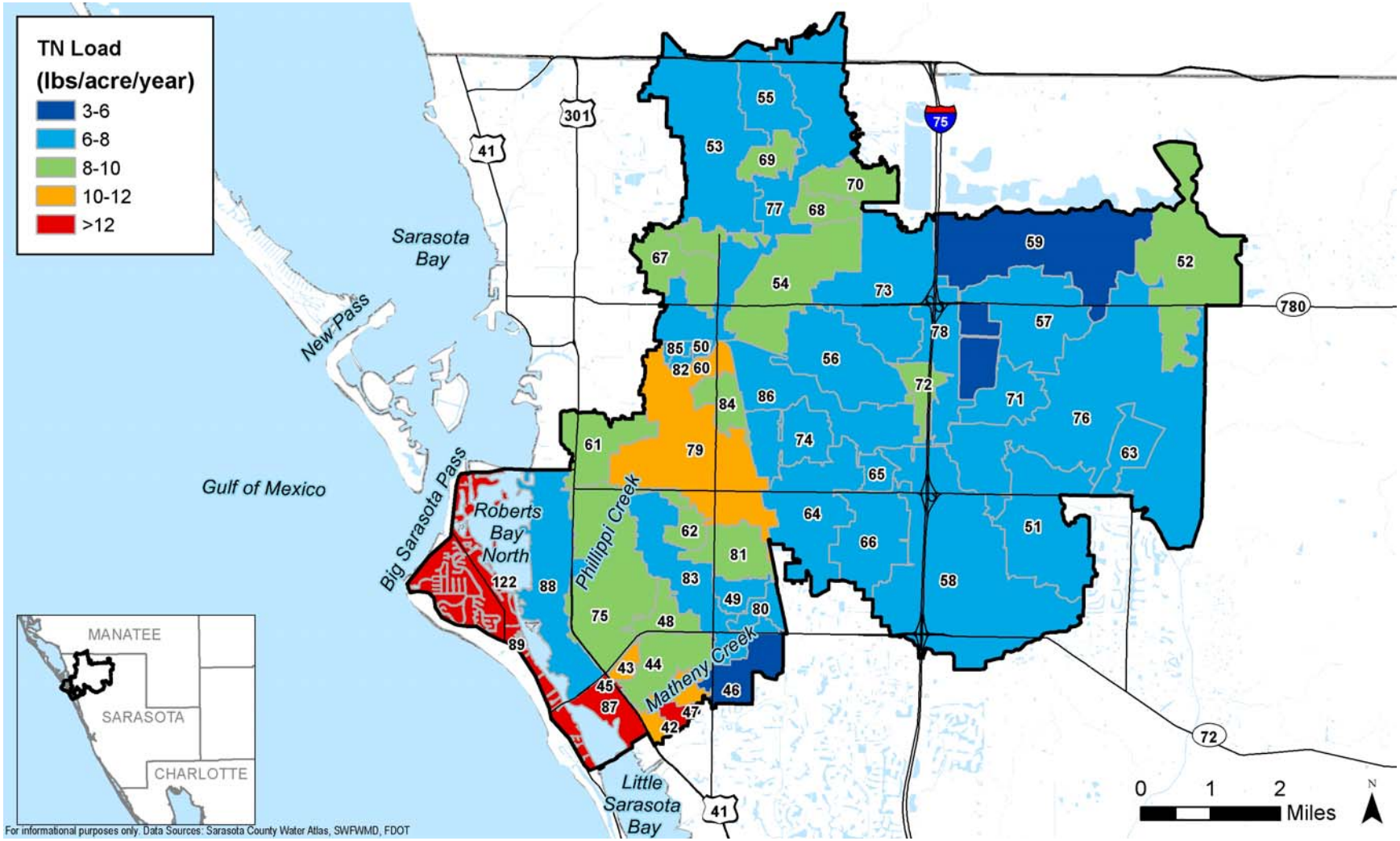

Figure 4-41 Average Annual Unit Area TN Loads (lbs/ac/year) by Subbasin in the Roberts Bay North Watershed (1995–2007)

B. BOD Loads

The average annual BOD loads to Roberts Bay North are presented in Figure 4-42. More than 80% of the total BOD load to the bay was generated in the Phillippi Creek basin.

Average annual unit area BOD loadings estimates are presented in Figure 4-43. The highest unit area BOD loadings are found in the watershed's most urbanized subbasins, generally located in the western (downstream) half of the Phillippi Creek basin and throughout the Matheny Creek basin. The four highest per unit area BOD loads were estimated in decreasing order in Subbasins 72, 43, 54 and 84.). Each of these subbasins has greater than 90% of their land use attributed to the urban land use classification. The smallest contributions of BOD loads came from the Coastal Roberts Bay island subbasins (4, 5, and 6), same as TN—we included these subbasins with Siesta Key), while low to moderate unit area contributions are identified in the upstream, more undeveloped subbasins in the Phillippi Creek basin. These results suggest that urbanized subbasins are more likely to contribute higher BOD loads than those of a more agricultural or natural character.

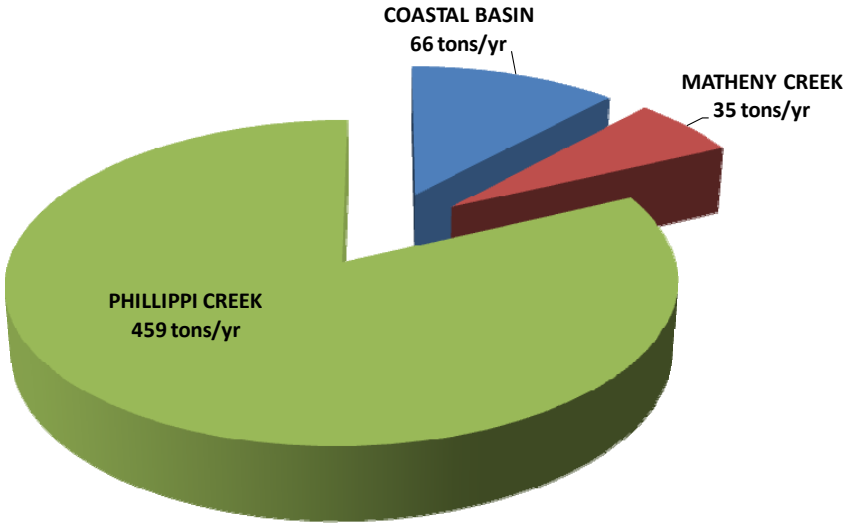

Figure 4-42 Average Annual BOD Loads by Basin to Roberts Bay North (1995–2007)

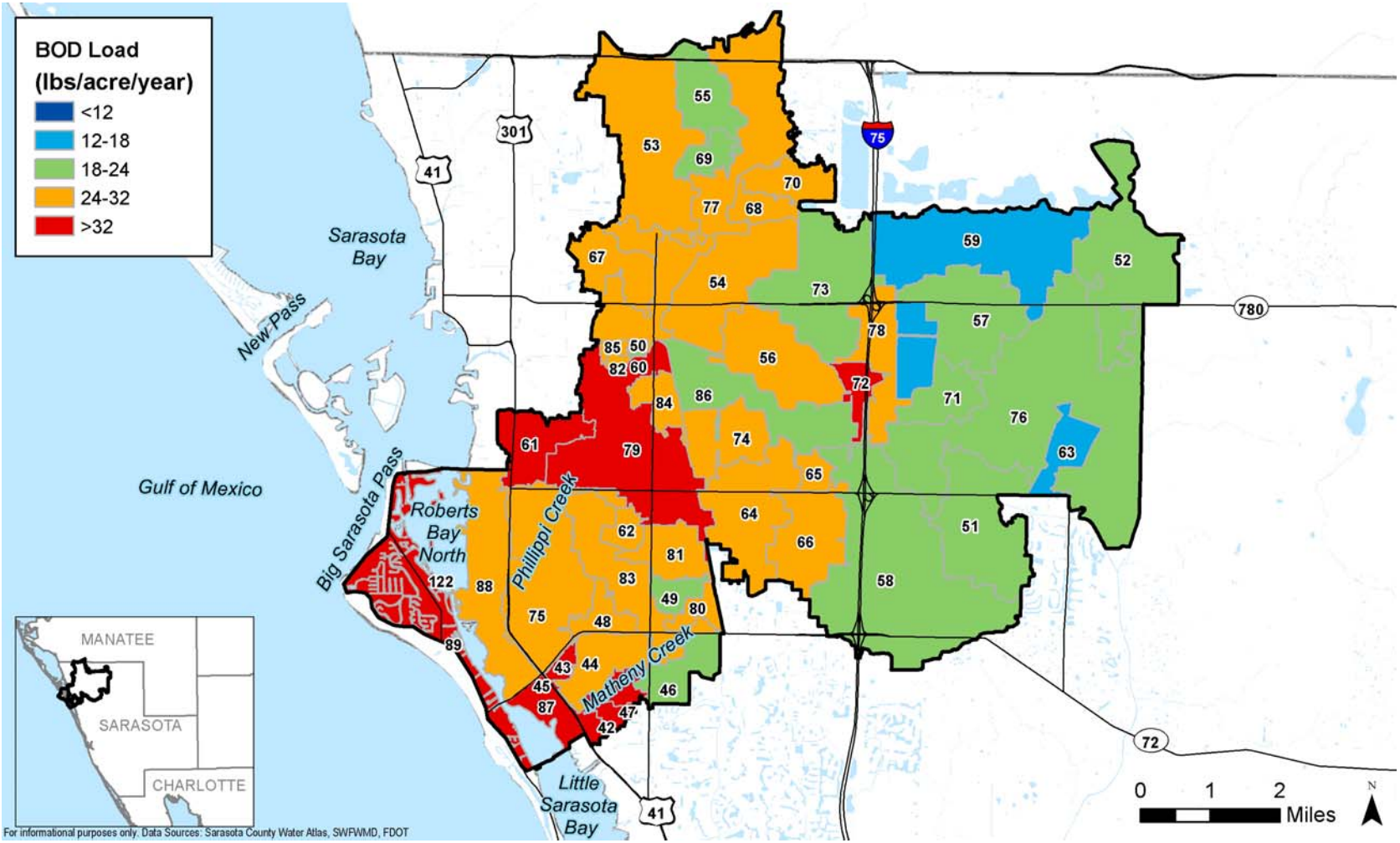

Figure 4-43 Average Annual Unit Area BOD Loads (lbs/ac/year) by Subbasin in the Roberts Bay North Watershed (1995–2007)

C. TSS Loads

The average annual TSS loads to Roberts Bay North are presented in Figure 4-44. Nearly 90% of the total TSS load to the bay was generated in the Phillippi Creek basin.

The annual average unit area TSS loadings are shown in Figure 4-45. Five of the six subbasins that had the highest unit area BOD loads are also among the six highest contributors of TSS loads. Subbasins 84, 54, and 72 in Phillippi Creek and Subbasin 43 in the Matheny Creek basin are among the top six for unit area loadings for both BOD and TSS. These results suggest that urbanization may be a key indicator for likely high values of both constituents, as all five of these subbasins have urban land use proportions in excess of 90% of their areas. However, unlike BOD loadings, moderately high per unit area TSS loads can be observed in the more agricultural subbasins within the watershed as well. Subbasins 52, 57, and 71 are among the four subbasins with the highest percentage of their land use classified as agriculture yet also have TSS loads ranked in the top 20 overall. The smallest per unit area TSS load is seen in Subbasins 6, 7 (same as TN and BOD—we grouped these with Siesta Key), and 59, where wetlands and other non-urban land uses are in generally higher proportions than observed elsewhere in the Roberts Bay North watershed.

The following conclusions can be drawn from the observations of spatial variability in pollutant loadings:

- Generally, the largest subbasins and in particular Subbasins 40 and 23 in the Phillippi Creek basin are consistently the largest contributors of hydrologic yields and pollutant loads.
- The subbasins in the western (downstream) portions of the watershed generally have the highest unit area hydrologic yields and loads, which reflect their small, yet highly urbanized character.
- The highest unit area TN loads are observed in the urbanized subbasins of Phillippi Creek and Coastal Roberts Bay, although moderate loads are seen farther upstream in the more agricultural portions of the watershed.
- High values of both BOD and TSS unit area loads are seen in the most urbanized portions of the watershed, whereas elevated TSS load estimates are seen in the more agricultural regions, but BOD levels are lowest in these more rural areas.
- Subbasins 1 and 3, which form the immediate eastern and western boundaries of Roberts Bay Proper, display consistently high unit area loads.

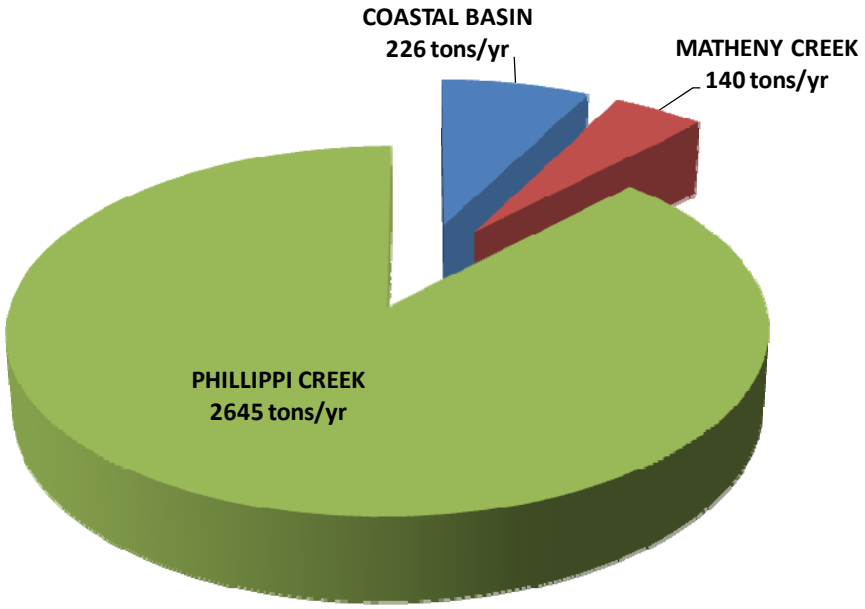

Figure 4-44 Average Annual TSS Loads by Basin to Roberts Bay North (1995–2007)

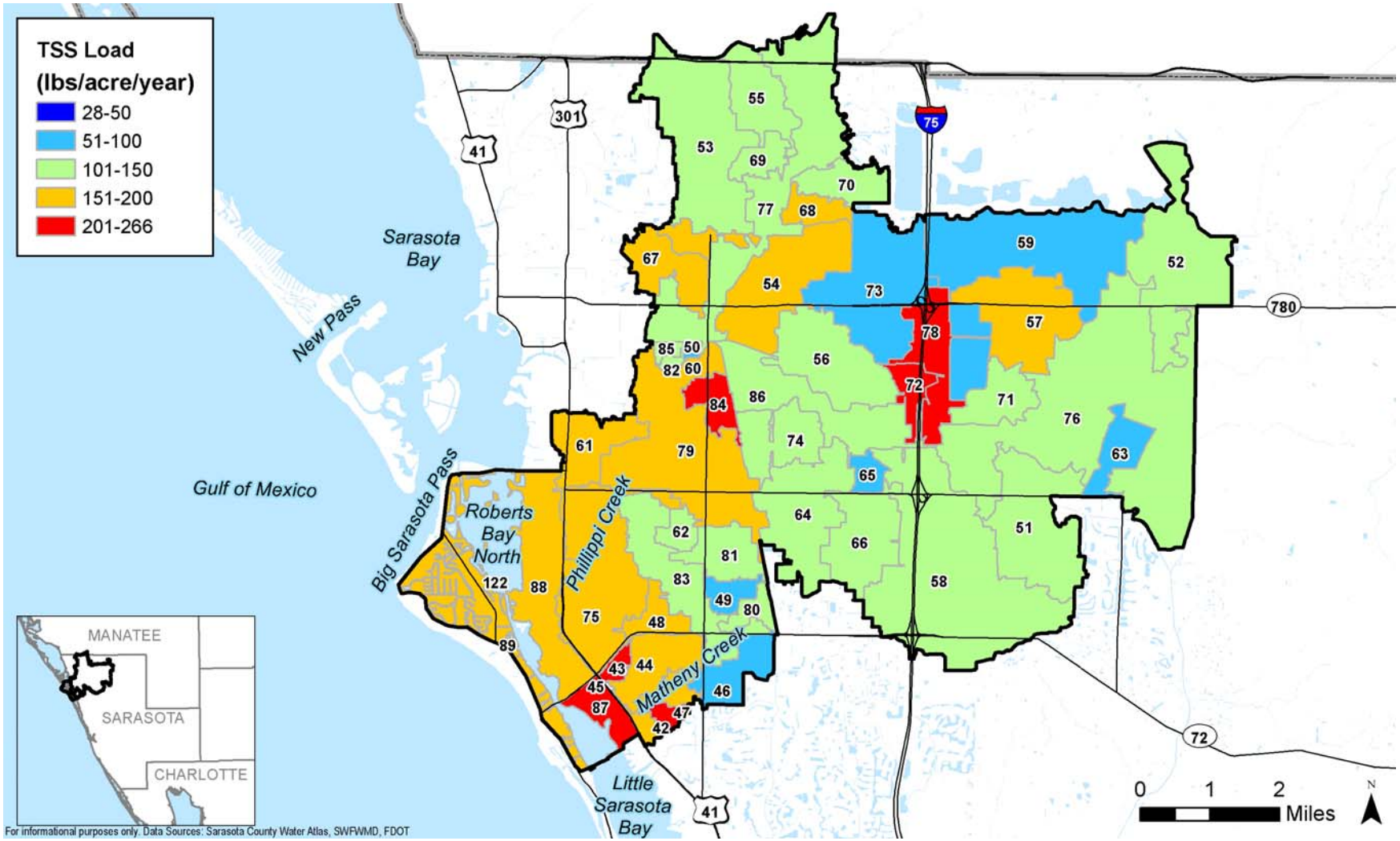

Figure 4-45 Average Annual Unit Area TSS loads (lbs/ac/year) by Subbasin in the Roberts Bay North Watershed (1995–2007)

4.4 ANALYSIS OF THE RESPONSES IN ROBERTS BAY NORTH TO POLLUTANT LOADINGS

4.4.1 Nutrient Loading to Estuaries

The consequences of increased nutrient loading to an estuary include increased episodes of noxious blooms, reductions in aquatic macrophyte communities, and hypoxia and/or anoxia, often leading to substantial shifts in ecosystem processes. Nitrogen and phosphorus are the nutrients of greatest concern because they most often control eutrophication and their inputs are often anthropogenic (Paerl et al., 2003). The single largest global change in the N cycle results from synthetic inorganic fertilizers that became widely used after the 1950s. In addition to widespread use of fertilizers, increased use of fossil fuels and production of N-fixing crops have dramatically increased nitrogen loading across the globe.

Before the 1990s in the United States, phosphorus loading was dominated by point sources, specifically wastewater. With the successful effort to reduce P loading in wastewater, non-point-source loading has increased in significance. As in most estuarine systems (National Research Council, 2000), N is the limiting nutrient in Tampa Bay. Strong empirical evidence based on annual water quality sampling in the region and bioassay results points to the importance of nitrogen in controlling algal biomass and growth in this estuary. Therefore, the focus of nutrient reduction in Tampa Bay is N loading. Currently, no specific nutrient-reduction laws are mandated by any U.S. government agency, although certain mandates under the Clean Water Act are acting to implement water quality standards and reduce TMDLs (Boesch, 2002). Every watershed is unique, and standards must account for the individual characteristics of each. This makes enacting and implementing nutrient-reduction strategies very difficult, especially given the need to determine how to achieve locally desired resource-management goals.

The EPA's National Estuary Programs have been instrumental in establishing site-specific goals and implementing these goals through the participation of national, regional, and local agencies; governments; and private entities. The central process of eutrophication is not a single focused issue but rather a multitude of factors that combine to cause water quality issues that change depending on ecosystem location and sources of pollution. One commonly used way to assess and control eutrophication is to identify indicators, such as seagrass growth and coverage and primary production, for managing estuarine systems. Light availability is the principal factor limiting seagrass distribution. Managing primary production as a result of increased nitrogen loading has a direct effect on surface irradiation depth. For example, in the Chesapeake Bay, Dennison and others (1993) established habitat requirements for submerged aquatic vegetation based on TSS, chlorophyll *a* concentrations, and median photosynthetically active radiation. A similar management approach was also used in the Indian River Lagoon and Tampa.

The SBEP is developing a scope of work that will define the methodology to be used to set water-quality targets for the Sarasota Bay system. In addition, FDEP will be establishing numeric nutrient criteria for estuarine waters over the next year. We expect these criteria to be

expressed as either concentrations or loadings and may include an adjustment for variation in residence times or other confounding factors.

4.4.2 Influence of Circulation and Residence Times

Understanding the relationship between nutrient loading and estuarine response requires knowing the potential influence of estuarine circulation and residence times. Estuarine circulation is driven primarily by tidal exchange and freshwater inflow and results in the transport of water quality constituents (e.g., salinity, nutrients, DO) within the system. The passes connecting Sarasota Bay to the Gulf of Mexico provide avenues for tidal exchange, with the resulting circulation within the estuarine system depending on the locations and sizes of these passes. This section briefly summarizes circulation within the system, including the results of a Sarasota County project that simulated the effects of the opening of Midnight Pass.

The northern region of the Sarasota Bay system connects to Tampa Bay through Anna Maria Sound. South of Anna Maria Sound, Longboat Pass connects the north end of Sarasota Bay to the Gulf, with New Pass connecting to the Gulf near the southern end of Sarasota Bay. Big Sarasota Pass provides the largest connection to the Gulf, between Sarasota Bay and Roberts Bay, and Venice Inlet is south of Little Sarasota Bay. Midnight Pass provided a connection to the Gulf near the middle of Little Sarasota Bay until 1983, when the pass was closed (ATM and ECE, 2004). South of Venice Inlet the Intracoastal Waterway (ICW) connects the Sarasota Bay system to Lemon Bay, which is tidally influenced by the Gulf through Stump Pass in the southern third of Lemon Bay.

The strongest currents in the system are found in the passes during incoming and outgoing tides, with the areas between the passes generally experiencing much weaker currents (Sheng, 1992). A three-dimensional model of tidal circulation in the Sarasota Bay system developed by Sheng and Peene (1991) showed that the areas between the passes, where the tidal signals entering from adjacent passes meet, are areas of very small current velocities. Consequently, these areas have relatively poor flushing rates. Modeling efforts identified Palma Sola Bay, Middle Sarasota Bay, and Middle Little Sarasota Bay as having the lowest flushing rates in the Sarasota Bay system (Sheng, 1992).

Additional modeling has been conducted to examine the effects of reopening Midnight Pass (Sheng, 1992; ATM and ECE, 2004). Flushing rates in different sections of the Sarasota Bay system were compared based on model output for the current system configuration and with Midnight Pass opened. Sheng (1992) found that opening Midnight Pass resulted in more rapid flushing of the middle portion of Little Sarasota Bay but reduced flushing to the north in Roberts Bay and to the south in Blackburn Bay. The reduced flushing in Roberts Bay would result from northward movement of the low circulation zone, where the tidal influences through Big Sarasota Pass and the reopened Midnight Pass would meet.

The more recent modeling effort examining the effects of reopening Midnight Pass (ATM and ECE, 2004) found that the *tidal prism*, defined as that volume of water passing through a given cross-section during a tidal cycle, would be decreased slightly at Big Sarasota Pass (~5%) and Venice Inlet (~5 to 10%) by reopening Midnight Pass. The study also found that the flushing rate would increase by over 100% in Little Sarasota Bay due to reopening of the pass. As in the previous study (Sheng, 1992), the more recent study also found that the area of low circulation in the middle portion of Little Sarasota Bay under existing conditions would be replaced by two areas of low circulation to the north and south of a reopened Midnight Pass, but that the overall flushing rate of Little Sarasota Bay would be improved.

As described above, flushing rates in Roberts Bay between Little Sarasota Bay to the south and Sarasota Bay to the north would likely be reduced slightly by the reopening of Midnight Pass. This is because the zone of low circulation, where the tidal signals from adjacent inlets meet, would be moved northward from the middle of Little Sarasota Bay by the reopening of Midnight Pass. Hydrodynamic model experiments that traced inputs from Phillippi Creek showed existing flushing rates in Roberts Bay North did not differ significantly from those predicted after Midnight Pass reopening (Peene, 2009).

Using a hydrodynamic model based on that used for the Midnight Pass evaluation (ATM and ECE, 2004), Janicki Environmental investigated fluxes into and out of Roberts Bay for existing conditions. These fluxes were across the boundary between Roberts Bay and Sarasota Bay to the north and across the boundary between Roberts Bay and Little Sarasota Bay to the south. Over a 2-year simulation period, water mass transfers between Roberts Bay and Sarasota Bay to the north were about twice as large as those between Roberts Bay and Little Sarasota Bay to the south. This is as expected since Big Sarasota Pass is adjacent to the northern end of Roberts Bay and the influence of the tidal signal is strongest across the northern boundary of the bay, resulting in more water movement across this boundary.

4.4.3 Nutrient Loading and its Impact on Estuaries

Tides and rivers offer a constant flow of water and nutrients that provide a beneficial environment for primary producers that form the base of the maritime food web. Watershed-driven nitrogen inputs from watersheds adjacent to coastal and estuarine waters can have significant impact on estuarine function. High rates of nutrient inputs from the land often stimulate very high rates of primary productivity. Due to high primary productivity, estuaries provide breeding and nursery grounds for many species of fish and shellfish. Hundreds of marine organisms, including commercially viable fish and shellfish such as shrimp, crabs, and trout, depend on estuaries during different stages of their lifecycles to provide valuable habitat (EPA, 1999).

In estuarine systems functioning without large anthropogenic disturbances, dissolved nutrients in river discharge constitute the primary nutrient source for many estuaries that receive significant freshwater input. Since the 1970s many scientists and managers have been studying the

deterioration of estuarine ecosystems via increases in nutrient loads and accompanying eutrophication (Paerl et al., 2006; Bricker et al., 2008; Fisher et al., 2006). The targeting of nutrient inputs from other points sources such as sewage outfalls and industrial effluent was met with much success, yielding improved water quality following implementation of advanced waste water treatment (Greening and Janicki, 2006). Unfortunately, population growth and the growing need for agricultural output have led to an increase in non-point-source pollution. It is estimated that human activity has increased the total rate of formation of reactive nitrogen globally by 33 to 55% through increases in agriculture via synthetic fertilizer (Howarth, 2008). Increases in reactive nitrogen have also resulted from increases in the encouragement of biological nitrogen fixation associated with agriculture and the inadvertent creation of reactive nitrogen through reaction with oxygen as fossil fuels are burned (Howarth, 2008; Paerl et al., 2006).

Excess nitrogen in estuarine ecosystems has led to increased rates of primary production, termed *eutrophication* (Nixon, 1995). Understanding the impacts of eutrophication and how anthropogenic impacts affect the structure and function of estuaries continues to be a research goal for scientists and managers worldwide (Paerl et al., 2006). Eutrophication has resulted in documented cases of reduced biodiversity, habitat degradation, and food web alterations (Nixon, 1995; Rabalais and Turner, 2001; Paerl et al., 2006; Bricker et al., 2008). Large-scale drivers of estuarine productivity include non-point and point source inputs from the watershed, riverine flow, and atmospheric deposition.

Symptoms of water quality decline are typically chlorophyll *a* and microalgae, low DO, loss of submerged aquatic vegetation, and occurrences of Harmful Algal Blooms (HABs) (Bricker et al., 2008) Chlorophyll *a*, a pigment used in photosynthesis, serves as a measure of biomass (abundance) of phytoplankton in estuaries. Planktonic algae provide a food source of filter-feeding bivalves (oysters, mussels, scallops, clams) and zooplankton (including the larvae of crustaceans and finfish). Chlorophyll *a* concentrations can also be used as a measure of overall ecosystem health. High amounts of chlorophyll *a* in estuarine waters are a primary indicator of nutrient pollution because excess nutrients fuel the growth of algae. High chlorophyll *a* values can have adverse impacts on aquatic life and human recreation.

DO is a very important limiting factor impacting estuarine systems. DO can be used as an indicator of the health of the ecosystem. Cultural eutrophication (nutrient excess leading to overproduction of microalgae and associate trophic imbalances) is common in estuaries near human population centers. Under conditions of eutrophication, DO can exhibit extreme diel cycles. Photosynthesis via algae elevates DO levels in the water during the day, but at night when respiration is high the DO can drop dangerously low. Eutrophication can lead to periodic or long -term hypoxia (water column oxygen concentrations less than 2 mg O_2/L) and anoxia in estuarine ecosystems. Fishes, crabs, and shrimp will attempt to move away from hypoxic conditions and few marine animals survive in prolonged exposure to it. DO levels are often quite variable in estuarine system due to fluctuations in temperature, salinity, basin morphology, and overall productivity.

Seagrasses serve significant functions. They help maintain water clarity by trapping fine sediments and particles with their leaves, and they stabilize the estuarine sediments with their roots. Seagrasses are very effective at removing dissolved nutrients from water that can enter from land runoff. The removal of sediment and nutrients improves water clarity, thereby improving overall ecosystem health. Seagrasses offer habitats for fish, crustaceans, and shellfish, providing a nursery ground for many recreationally and commercially valuable species. They are also food for organisms that inhabit them and marine mammals such as manatees and waterfowl such as ducks. Human activities can harm seagrasses by degrading estuarine water quality and promoting physical disturbances and algal blooms. Reductions in light availability associated with nutrient inputs and sediments can damage or eliminate seagrass habitat.

How any particular estuary will respond to excess nitrogen loading depends on numerous factors including freshwater inflow, residence time, and clarity or light attenuation (Howarth and Marino, 2006). Estuarine nutrient concentrations depend on freshwater inflow because freshwater is a source of nutrients. The rate of freshwater inflow can influence hydraulic residence time and hence the time available for nutrients to react in the estuary (Bricker et al., 2008). Flow may affect chlorophyll by increasing chlorophyll abundance via enhanced nutrient supply, changing the location of peak chlorophyll abundance or decreasing chlorophyll abundance and residence time. During times of low freshwater inflow, the chlorophyll maximum is typically located farther upstream than during times of high flow. Low flow also allows a longer residence time for chlorophyll and other nutrients. Longer residence times tend to promote slower-growing taxa, which include dinoflagellates, cyanobacteria, and HABs. Increased nutrient loading is associated with higher flows and is typically followed by increased algal biomass. During high flow conditions, flushing is more rapid and residence time in the river is reduced. These conditions tend to favor fast-growing phytoplankton such as chlorophytes (green algae) and various flagellates (Pinckney et al., 1999) At times, depending on the morphology of the river, high flows can be excessive. Very high flows may not result in higher chlorophyll abundance due to the relationship between the residence time of water in the system and uptake and growth rates of the phytoplankton community. Reductions in flow can also impact community composition with less-desirable species such as HABs occurring during times of low flow and longer residence times (Bricker et al., 2008).

Water clarity is a measure of the amount of sunlight that can penetrate the water. Water clarity is measured with a device called a *Secchi disk*. The measurement, named the *Secchi depth*, is the measure of water clarity and the depth at which sunlight is able to penetrate the water. Clear waters indicate a healthy estuary, although many factors impact water clarity. Excess suspended sediments from runoff and rainfall can negatively impact water clarity. Nutrients, mainly nitrogen and phosphorus, can fuel the growth of photosynthesizing algae. High chlorophyll *a* concentrations associated with high algal biomass can decrease light penetration, decreasing water clarity. Decreased water clarity can negatively impact the estuary in many ways. Reduced light transmission can decrease seagrass abundance, which can affect the entire food web. Decreases in seagrass reduce habitat to the hundreds of species that depend on the seagrass.

The successful management of coastal ecosystems requires long-term monitoring and accurate quantitative tools for managers, scientists, and the public at the local and regional levels to easily understand and apply basic principles of ecosystem management. Wide-scale nutrient reduction aimed at controlling ecosystem scale eutrophication needs to span freshwater and marine ecosystems. Additionally, managers must recognize that primary productivity and growth responses could take longer times (years to decades) for improved water quality but that implementing these reductions is imperative.

4.4.4 Response in Roberts Bay North to Variation in Nutrient Loading

The nexus between understanding the relationship between nutrient loading and response in the estuary and effective resources management is the ability to develop a tool that quantitatively links loading and response. The approaches that have been taken to develop such tools have ranged from complex, mechanistic models to empirical models (Boynton et al., 1995; Boynton et al., 1996; Brush et al., 2002). Empirical modeling approaches have been used for several Florida estuaries, including Tampa Bay (Janicki and Wade, 1996), Sarasota Bay (Tomasko et al., 1996), Lemon Bay (Tomasko et al., 2001) and Indian River Lagoon (Steward and Green, 2007).

We have used an empirical approach to quantify the relationship between nutrient (nitrogen) loading and chlorophyll *a* in Roberts Bay North. The data used to develop this empirical model have been examined earlier in this chapter. These include the loading data provided by the SIMPLE model for 1998 to 2007 and ambient water quality data provided by the County's monthly monitoring program.

Initially, a series of potential loading variables were calculated:

- Current month loading.
- Lagged monthly loading (e.g., last month's load).
- Cumulative monthly loading (e.g., the sum of the last months' loads).

The variation in these potential explanatory variables was compared to the variation in mean monthly chlorophyll concentrations. We found that the relationship between this month's mean chlorophyll and the cumulative load from this month and the previous month provided the best fit model. Monthly-specific intercept terms were then added to the model to account for the effect of seasonal variation in water temperature and incident light on chlorophyll *a*. Given the same monthly TN loads, we expect that chlorophyll *a* concentrations should be highest during the summer months when water temperature and incident light are greatest.

A plot of the relationship between the natural log transformed chlorophyll *a* and 2-month cumulative TN loads is given in Figure 4-46. A multiple regression technique was applied to these data. The slope of the overall model was significantly greater than 0 ($p < 0.0001$) and the coefficient of determination (R^2) was 0.59. Therefore, the variation in TN loads from the Roberts

Bay North watershed accounted for nearly 60% of the variation in chlorophyll *a* concentrations in the estuary. Figure 4-47 presents a plot of the observed chlorophyll *a* concentrations from Roberts Bay North and those predicted by the regression on TN loads.

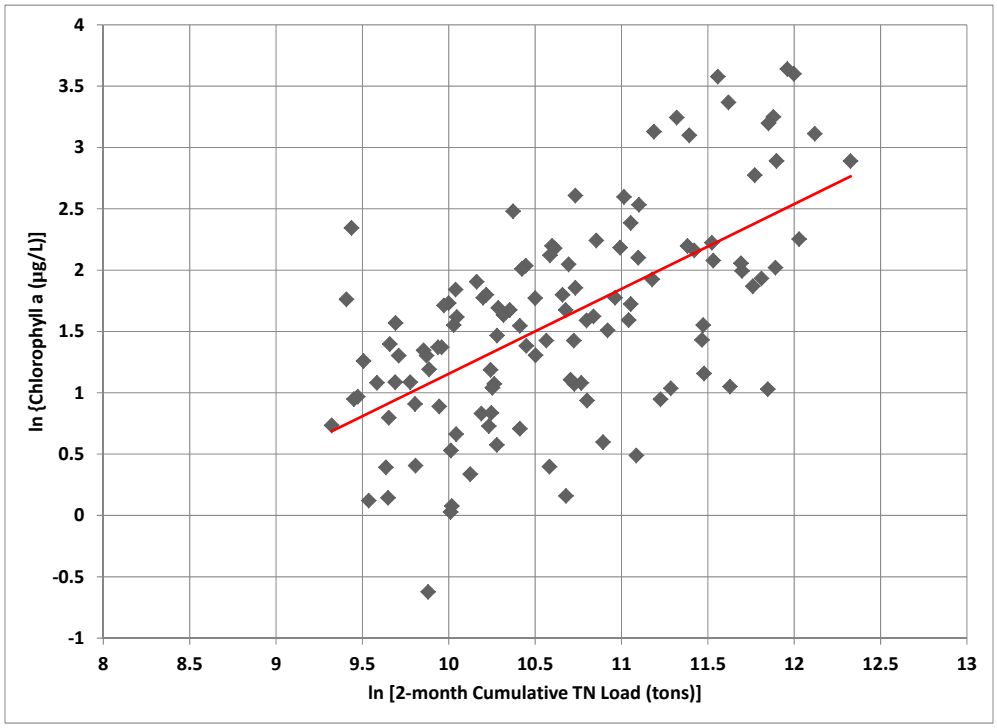

Figure 4-46 Relationship Between In-Transformed Chlorophyll *a* and 2-Month Cumulative TN Loads Data from Roberts Bay North (1998–2007)

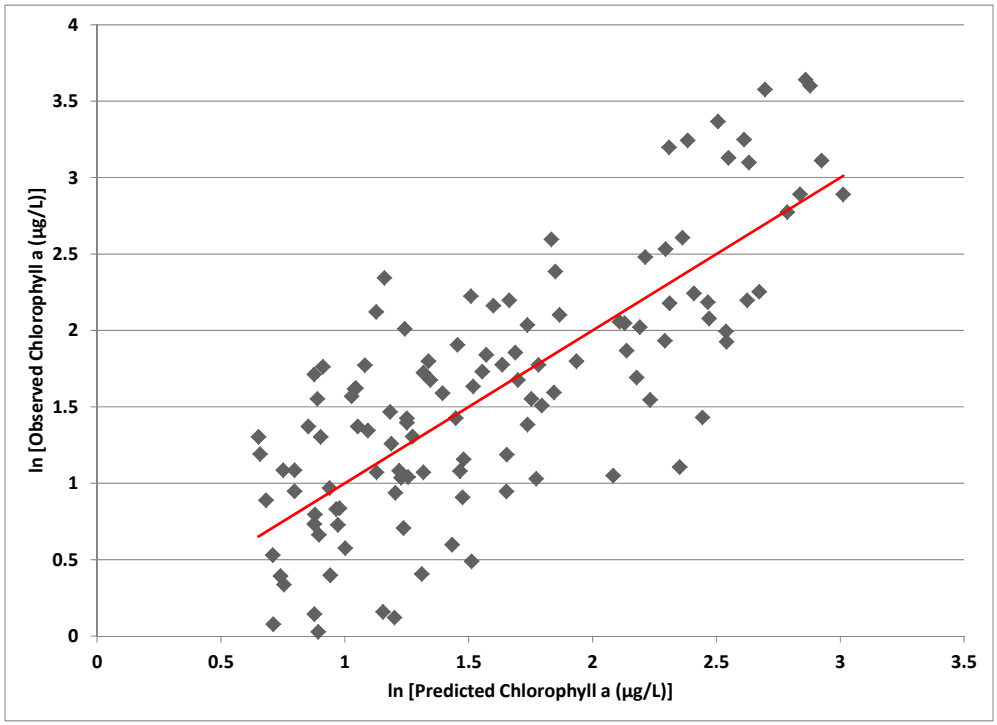

Figure 4-47 Comparison of Predicted and Observed Chlorophyll *a* Concentrations from Roberts Bay North (1998–2007)

Further analysis of the chlorophyll-TN load relationship included an examination of the residuals (the differences between the predicted and observed chlorophyll concentrations). We examined plots of the residuals against potential confounding variables to identify any apparent patterns. If there is no relationship between the residuals and any confounding variable, the plot will show more or less equal probability of either under- or over-predictions across the range of values of the confounding variable. This diagnostic tool can identify whether inclusion of any of these variables may improve the model predictions. In this case the plot of the model residuals with the mean monthly turbidity in Roberts Bay North shows a clear pattern (Figure 4-48). The probability of an over-prediction increased with increasing turbidity.

Given these results, the model was reformulated to include the effect of turbidity. As before, the slope of the overall model was significantly greater than 0 ($p < 0.0001$) and the R^2 increased to 0.76. Therefore, the new model accounts for nearly 80% of the variation in chlorophyll *a* concentrations in the estuary. Figure 4-49 presents a plot of the observed chlorophyll *a* concentrations from Roberts Bay North and those predicted by the regression on TN loads.

The results of the empirical modeling approach indicate that the management of nitrogen loading from the Roberts Bay North watershed will be essential if future changes in the watershed lead to potential increases in loads.

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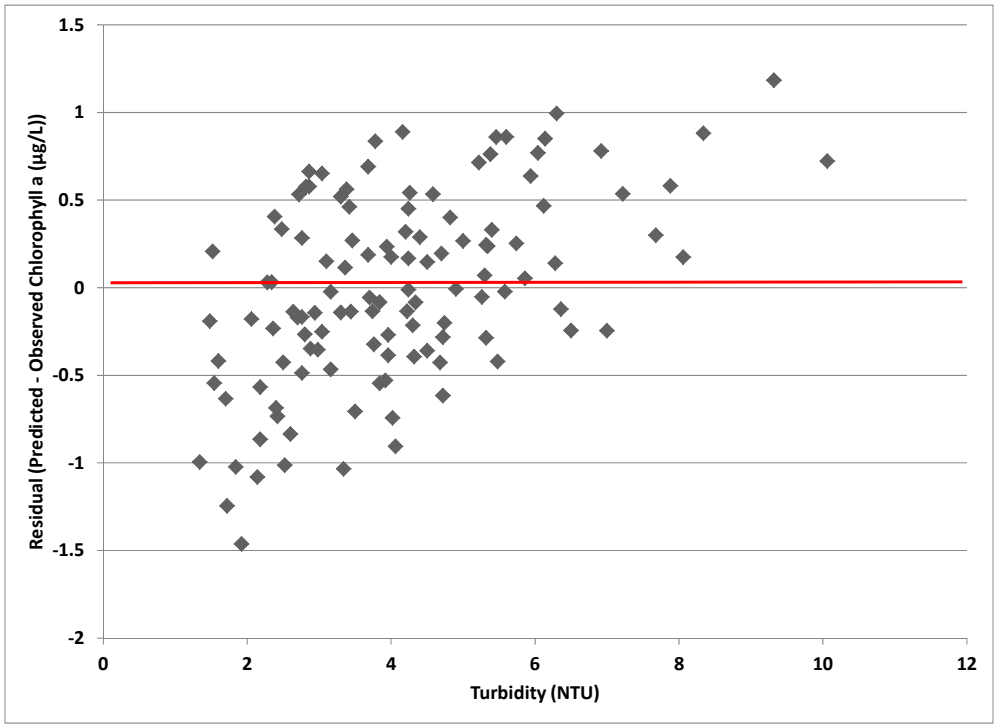

Figure 4-48 Comparison of Residuals From the Chlorophyll-TN Load Model for Roberts Bay North to Mean Monthly Turbidity Concentrations

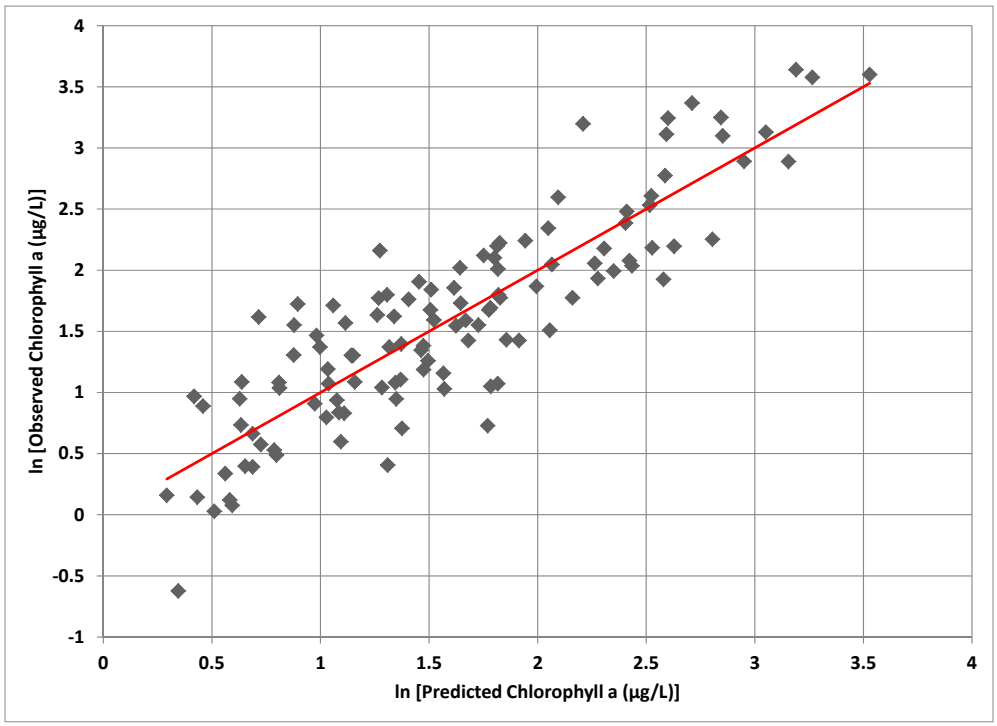

Figure 4-49 Comparison of Observed Chlorophyll *a* Concentrations from Roberts Bay North to the Predicted Concentrations from the Model including Mean Monthly Turbidity

4.4.5 Relationship Between Water Quality Conditions in Roberts Bay North Tributaries to Variation in Pollutant Loading

As discussed in Section 4.2, there are several impaired WBIDs within the Roberts Bay North watershed. WBIDs within both Phillippi Creek and Matheny Creek have recently been deemed impaired due to DO and fecal coliform exceedances. To examine the relationships between water quality in these creeks to loadings, we followed a similar approach to that described above for chlorophyll and TN loads.

Examination of the DO data obtained from Phillippi Creek shows the frequency of DO exceedances observed from 1994–2007 (Figure 4-50). These data were linked to the freshwater input and pollutant loadings provided by the SIMPLE model. Figure 4-51 presents plots of DO and these potential explanatory variables. Clearly, there is no apparent relationship between DO and any of the four variables examined (freshwater input, TN load, TSS load, BOD load [Figures 4-52 through 4-54]). The only clear pattern is that low DO conditions are more likely to be observed when each of the four variables examined are relatively high. Similarly, the highest DO conditions also are more likely to occur when each of the four variables examined are relatively low.

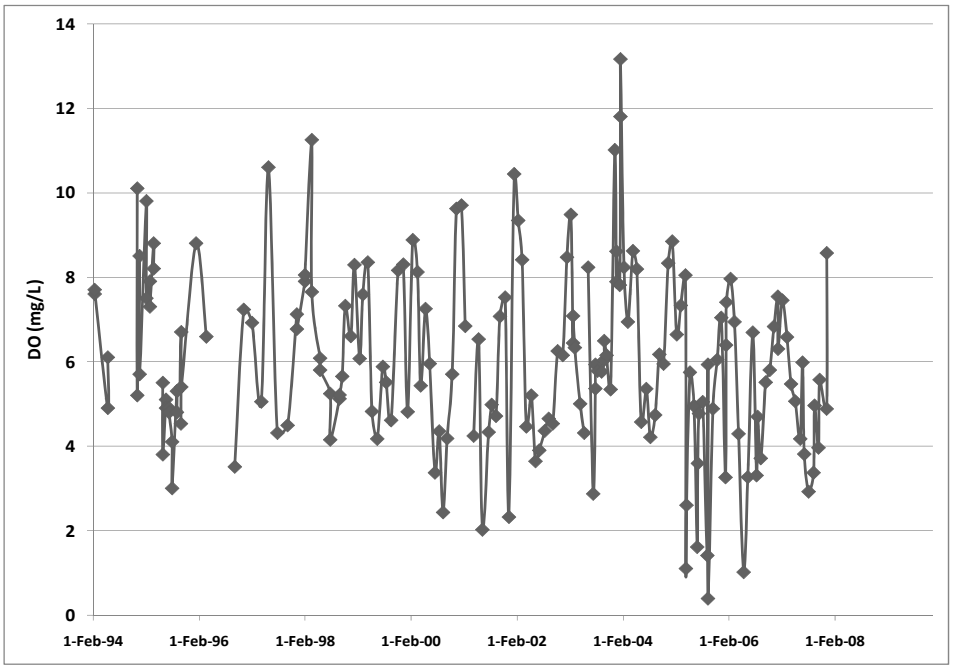

Figure 4-50 Time Series of DO Concentrations from Phillippi Creek

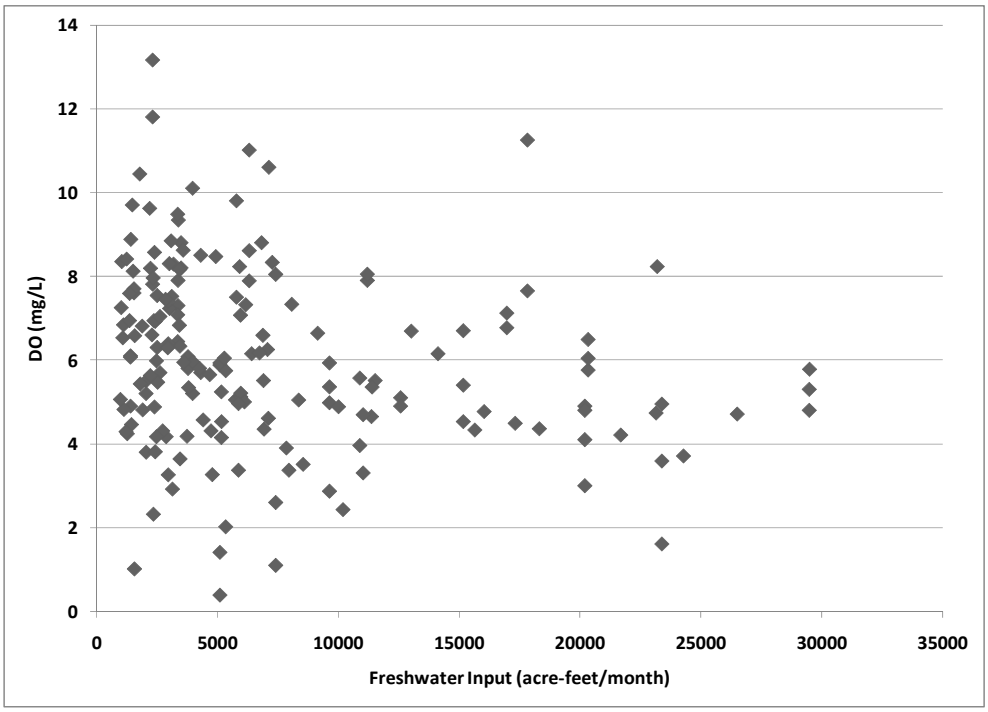

Figure 4-51 Relationship Between DO in Phillippi Creek and Freshwater Input,

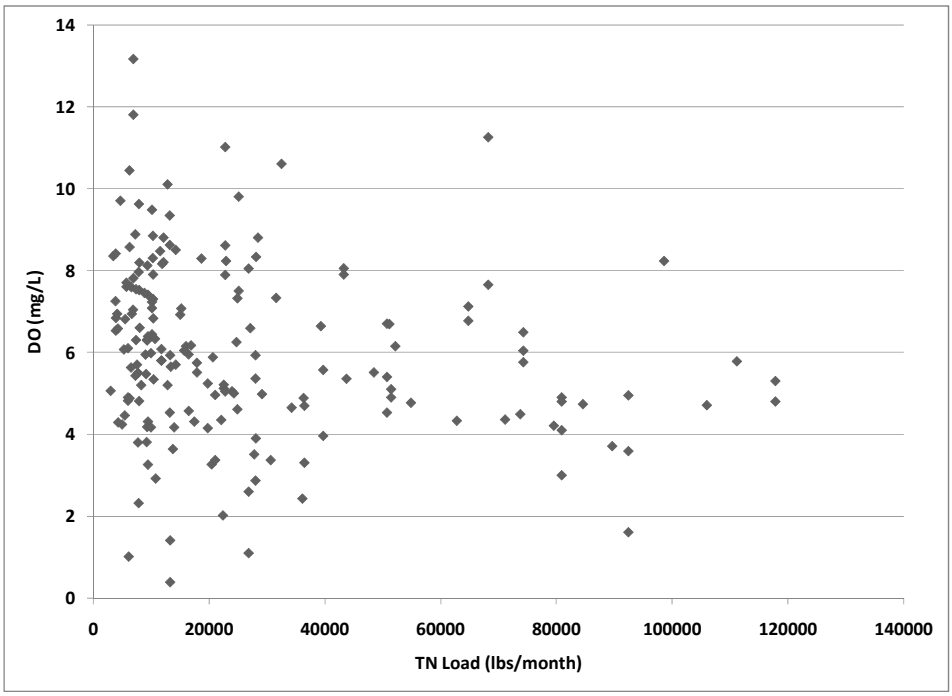

Figure 4-52 Relationship Between DO in Phillippi Creek and BOD loads from the Phillippi Creek Watershed

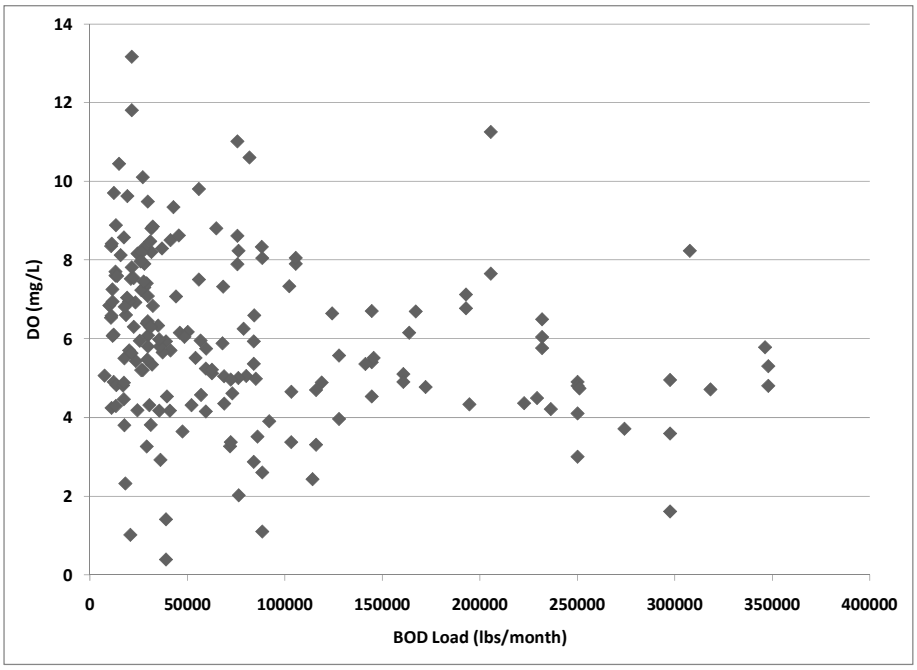

Figure 4-53 Relationship Between DO in Phillippi Creek and TN loads from the Phillippi Creek Watershed

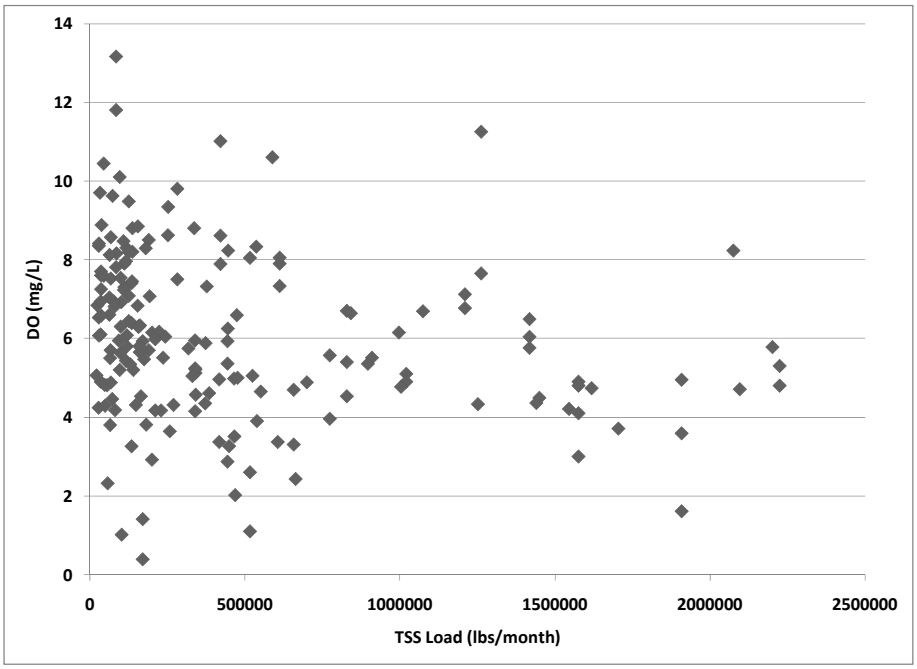

Figure 4-54 Relationship Between DO in Phillippi Creek and TSS loads from the Phillippi Creek Watershed

4.4.6 Freshwater and Pollutant-Load Targets and Reduction Goals for Roberts Bay North

As discussed in Section 4.2, comparing the historical and current water budgets shows that wet season flows are greater under current conditions. This increase in flow results in a modest change in salinity in Roberts Bay North. Projects or programs that can contribute to a reduction of wet season flows (i.e., during May through October) should be considered.

Also as discussed in Section 4.2, the recent seagrass coverage in Roberts Bay North meets or exceeds that estimated for the historical period (ca. 1950). The chlorophyll and K_d targets and thresholds reflect the conditions during 2001 to 2005, generally coinciding with the period during which the seagrass targets have been set and representing the recent wide range in rainfall in this region.

The chlorophyll *a* target for Roberts Bay North is 8.2 µg/L. The among-year standard deviation is 2.8 µg/L. Therefore, the threshold chlorophyll *a* concentration for Roberts Bay North is 11 µg/L. The K_d levels are the mean conditions during the 2001–2005 period, coinciding with the period during which the seagrass targets have been set and representing the recent wide range in rainfall in this region. The K_d target for Roberts Bay North is 1.0 m^{-1}. The among-year standard deviation is 0.09 m^{-1}. Therefore, the K_d threshold is 1.09 m^{-1}.

The analyses presented above indicate that meeting the chlorophyll *a* target for Roberts Bay North will depend on managing nitrogen loading to the bay. It logically follows that if the current water quality conditions have been adequate to maintain seagrass coverage at desired levels, the nitrogen loading is also at levels adequate to maintain the chlorophyll *a* concentrations at or near their desired levels. Therefore, the proposed nitrogen loading target is 213 tons/year, which is the average TN load for the period 2001–2005.

The proposed nitrogen loading target is 213 tons/year.

4.4.7 Comparison of the Proposed Nitrogen Loading Target to Future Nitrogen Loading to Roberts Bay North

Future loading estimates were developed following the methodology presented in Chapter 2. The SIMPLE model was used to develop estimates for what is essentially a “built-out” scenario. This entailed applying a land-use coverage that reflected build-out conditions where all “developable” polygons in the 2006 land-use coverage not classified as an environmentally sensitive land was converted to medium-density residential. Current BMPs, septic tank removal, and irrigation practices were also applied to the future load scenario. Future changes in atmospheric deposition follow methods used recently to estimate future atmospheric deposition loads to Tampa Bay (Janicki Environmental, 2008). Finally, the same rainfall record used to estimate the current loadings was used to drive the model. While many potential stormwater control rules/policies are currently under discussion and review, none of these has been applied to this “built-out”

scenario. Therefore, if any of these rules/policies are implemented, it can be expected that future loads will be less than those used in our analyses.

Figure 4-55 compares the current and future TN loads to Roberts Bay North. The "built-out" scenario loads are predicted to be consistently higher than the current loads. Clearly there are years when the 199 tons/year target is exceeded under both scenarios. This is not unexpected as year-to-year rainfall variation strongly influences the temporal variability in nitrogen loading. It was shown previously that the interannual variation in chlorophyll *a* concentrations reflects the variation in rainfall. An important observation is that while there are years when rainfall and nitrogen loads are relatively high and there is a concomitant increase in chlorophyll *a*, the bay responds (i.e., chlorophyll *a concentrations* drop) when the rainfall and nitrogen loads recede.

There are two critical considerations when evaluating these estimates. First, as discussed above, this is a build-out condition that if it is to occur will be in the distant future. Second, there will be years when the target is exceeded. Examining the monitoring data collected by the County will help in understanding why an exceedance has occurred and whether the bay is trending in an unwanted manner.

Figure 4-55 Current and Future TN loads to Roberts Bay North

4.5 CONCLUSIONS AND RECOMMENDATIONS

Conclusions and recommendations regarding water quality in Roberts Bay North include the following:

- Overall, the water quality in Roberts Bay North is good as evidenced by the chlorophyll *a* concentrations, water clarity, and resulting seagrass coverage.
- There is a clear relationship between chlorophyll *a* concentrations and TN loadings in Roberts Bay North. The nitrogen loads to Roberts Bay North do not exceed the target load. The focus of nitrogen load management will be on precluding significant load increases with any future land-use changes.
- Comparison of the historical and current hydrologic regimes for Roberts Bay North shows higher volumes under current conditions. This has apparently resulted in somewhat lower current salinities. Despite the observation that salinities were different between historical and current conditions and that those differences appeared to be largest during the summer, the current salinities in Roberts Bay North remained in the polyhaline to euryhaline range with summertime median and average salinities above 25 ppt. The historical hydrologic regime is recommended as the target water budget for Roberts Bay North.

4.5.1 Recommended Water Quality Improvement Programs

4.5.1.1 Septic Replacement Program

Septic systems have the potential to contribute significant pollutant loads to the primary receiving waters in the Roberts Bay North watershed. In April 1997, the Sarasota County Board of County Commissioners (BOCC) found that septic systems and small package wastewater treatment plants were factors contributing to documented pollution problems in Phillippi Creek. The BOCC directed staff to initiate a program (Phillippi Creek Septic Tank Replacement Program) to replace or upgrade septic systems in this area. This program will result in approximately 14,000 new wastewater connections implemented over an eight to ten year period. The BOCC further recognized the need for septic system replacement in other areas, and expanded the program to include the southern portions of the County. This project is known as the South County Wastewater Improvement Program and includes the area west of I-75 extending from Clark Road south to the County's southern perimeter in Englewood, excluding the Englewood Water District service area (Figure 1-1).

The South County Wastewater Improvement Program (SCWIP) evaluated whether existing wastewater treatment practices affect water quality in the project area and recommended that Sarasota County provide central sewers for those sub-areas with average acreage sizes less than 0.5 acres (Hazen and Sawyer, 2004).

The SCWIP recommendation to replace septic systems in certain areas is based on their analysis of compliance with Ordinance No. 83-83, which relates to the design, construction, installation, utilization, operation, maintenance, and repair of septic tank systems. The SCWIP found that only 24% of all developed parcels (3,052 out of 12,653) have been permitted post-1983 and thus meet current code separation requirements. SCWIP also determined that the majority of the soil types found in the project area are severely limited for use of conventional septic system drainfields due to high groundwater.

We further recommend the continuation of the Septic Replacement Program for portions of Roberts Bay North based on the SCWIP evaluation and recent fecal coliform TMDLs (see Section 4.1.2.2). Fecal coliforms may pose a special health risk for infants, young children, and people with severely compromised immune systems (epa.gov). Septic systems that are not properly installed or maintained can increase fecal coliform counts in Roberts Bay North and its tributaries.

4.5.1.2 Street Sweeping Recommendations

Street sweeping is a proven, effective practice to improve water quality. The effectiveness of street sweeping and its value as a County maintenance practice is discussed in detail in Chapter 7. Projects RBWQ12 and RBWQ13 highlight how street sweeping can be implemented to improve water quality. Street sweeping for water quality improvement should be evaluated further and should take into account County funding for maintenance practices, local and state jurisdictions related to streets and highways, and the implementation recommendations presented in Chapter 7, such as sweeping frequency related to season. Program recommendations are not ranked with the other project recommendations in this chapter but are further evaluated in Chapter 8.

We recommend street sweeping in two areas in the Roberts Bay North Watershed—the Matheny Creek Basin and areas near US41 and SR72. While street sweeping in general is beneficial, these two areas have been identified as hot spots for TSS, TP, and TN in the watershed, and bi-monthly street sweeping in these areas will improve water quality, habitat, and flood control conditions by removing sediments and their associated pollutants from streets before they enter the stream systems.

A. RBWQ12 (RBS13) – PC: General Street Sweeping

The area shows the second and fourth highest TSS loads and first and third highest TP and TN loads in lb/ac/yr in the watershed (Figure 4-56 and Table 4-4).

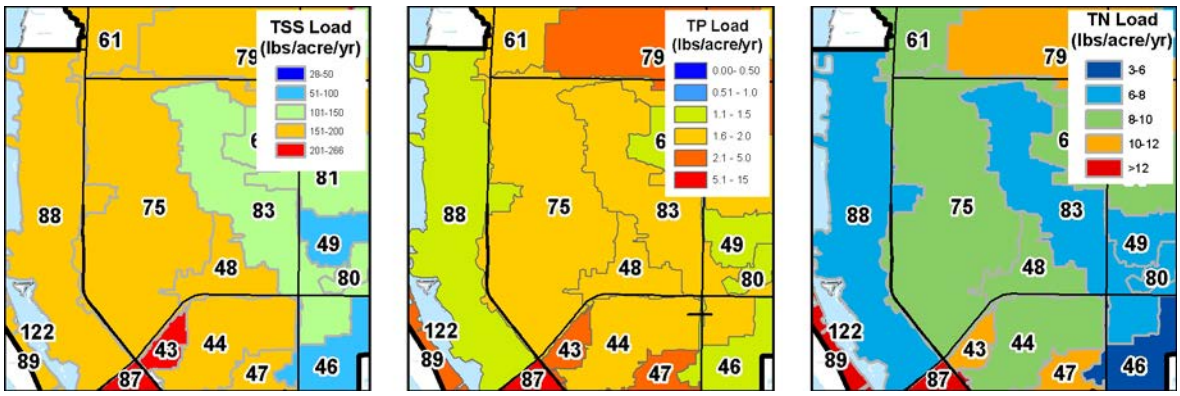

Figure 4-56 TSS, TP, and TN loads near US41 and SR72

B. RBWQ13 (RBS20) – RBC: General Street Sweeping

The limited space available in this highly urbanized area will not readily accommodate traditional stormwater BMPs. Sand from the roadways is a large contributor to the TSS pollutant load. Metals and toxic organic chemicals from vehicle usage that are attached to sediment particles can also be removed by street sweeping. Subbasins in the Roberts Bay Coastal Basin ranked 1 and 3 for TP and TN loads and 4 for TSS (lb/ac/yr). See Figure 4-57 and Table 4-4 for pollutant-load values from the SIMPLE model.

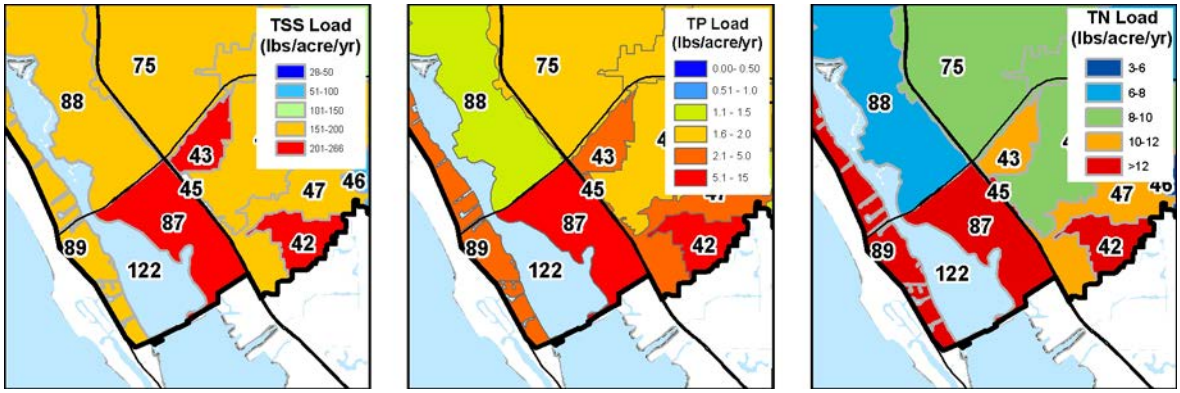

Figure 4-57 Roberts Bay North Coastal TSS, TP, and TN loads

4.5.2 Recommended Water Quality Improvement Projects

4.5.2.1 Introduction

Jones Edmunds identified potential water quality improvement opportunities in the Roberts Bay North watershed with a focus on improving the watershed's water quality functions. Eight potential sites were identified by Jones Edmunds based on a GIS desktop assessment using available digital datasets. Two sites were initially identified as part of the Sediment Management Plan but were reclassified to water quality projects during the analysis. A total of ten sites are reviewed and analyzed in this section. Detailed information for the reclassified sites can be found in Appendix C. However, a brief summary, cost estimate, and ranking is provided in this section.

4.5.2.2 Methods

A. Data Compilation and Analysis

Jones Edmunds used GIS to compile and review data developed from the Pollutant Loading Model results together with aerials and other base data obtained from the Sarasota County GIS library and SWFWMD. Specifically, these datasets included the following:

- Jones Edmunds pollutant-load results (TSS, TP, and TN)
- 1948 USDA aerials
- 2007 SWFWMD aerial imagery
- Public- and Agency-owned lands
  - SWFWMD
  - Airport Authority
  - Hospital
  - School Board
  - Federal
  - State
  - City

B. Field Investigations

Jones Edmunds conducted site visits to the water quality improvement sites in October 2008 to characterize the project areas and to identify and determine potential water quality treatment options. Site investigations for the reclassified sediment projects are detailed in Appendix C.

C. Quantifying Pollutant-Load Removal

The results of the SIMPLE model were used to calculate pollutant-loading rates in pounds per acre per year by catchment area. To calculate the range of pollutant removal by BMP, the loading rates normalized results by catchment from the SIMPLE model were multiplied by the

contributing area to create a pounds-per-year value. The pounds-per-year values were multiplied by the minimum and maximum reported efficiencies for the BMP to give a range of potential pounds per year of pollutant removed from stormwater runoff.

D. Opinions of Probable Cost

Cost of treatment was an important evaluation criterion for each site. Once the type of treatment method was determined, Jones Edmunds calculated the cost to implement the specific type of treatment activity. Some sites were determined to benefit large acreages with minimal cost for treatment, whereas other sites would require more costly treatment methods for a small amount of water quality improvement.

E. Site Ranking

Sites with a low cost to implement and high pollutant removal estimate were ranked higher than sites with a high cost and low pollutant removal estimates. Sites were ranked 1 through 11, with 1 being the highest ranked. To develop the ranking, Jones Edmunds divided the project cost by the high and low value in the range of pollutant removal estimates for each project to get a high and low cost per pound of pollutant removal. The high and low costs per pound of removal were averaged. The average cost per pound of removal is the value that was used to rank the sites.

4.5.2.3 Recommended Projects

Nine potential water quality improvement sites were identified during the initial GIS desktop assessment. Two projects were identified during the Sediment Management Plan analysis. The locations of these projects are shown in Figure 4-58. As a result, 11 potential water quality improvement projects were identified and assessed within Roberts Bay North watershed. The following sections describe site evaluations, proposed elements, and benefits for each project. Pollutant removal estimates, conceptual level opinion of probable costs, and ranking for each site are summarized in Tables 4-4, 4-5, and 4-6. The project names include the water quality conceptual project id (RBWQXX), the sediment project ID (RBSXX), if applicable, the basin initials (e.g., PC represents Phillippi Creek), and the site name.

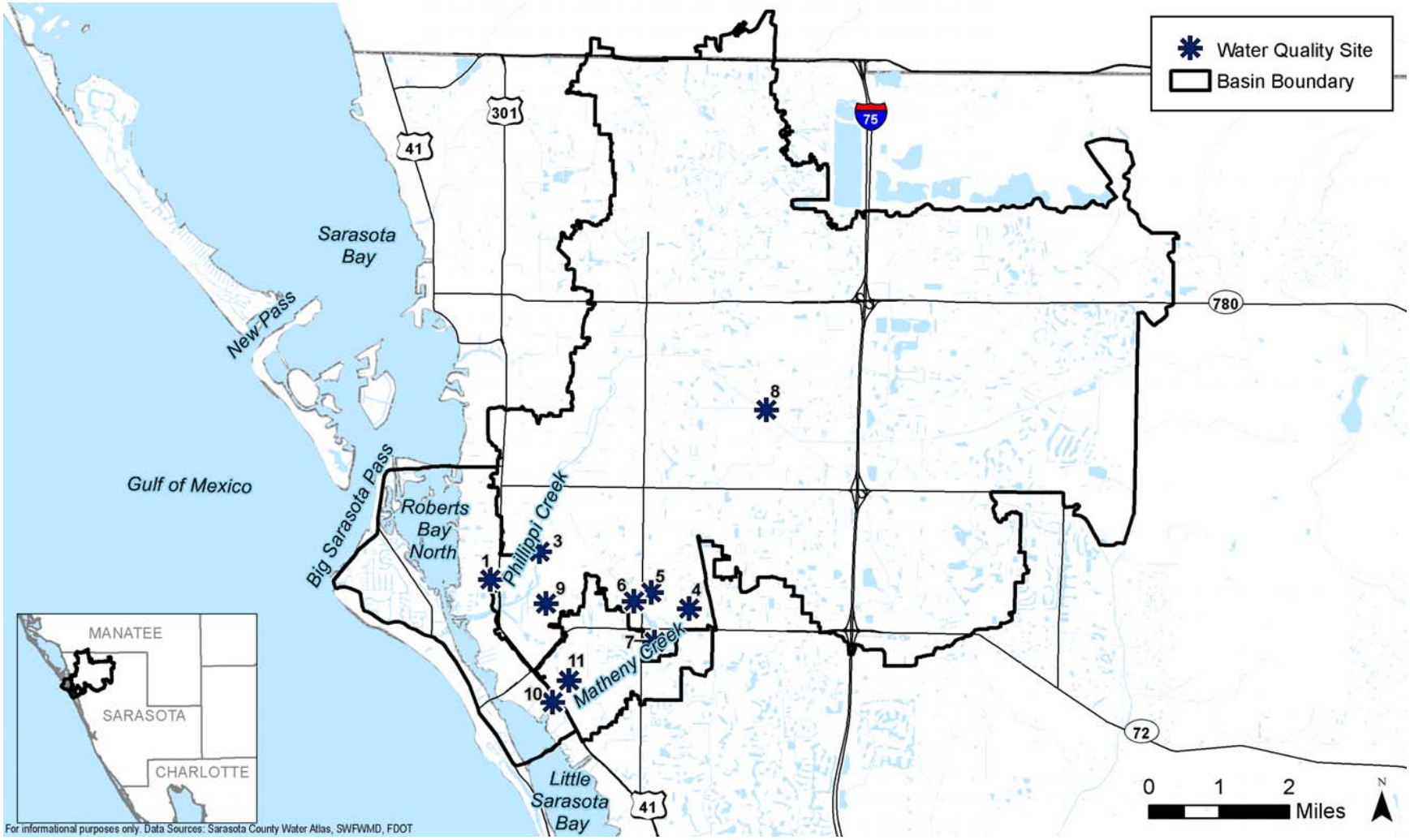

Figure 4-58 Roberts Bay North Watershed Water Quality Improvement Site Locations

A. RBWQ01 – RBC: The Landings to Phillippi Shores Park

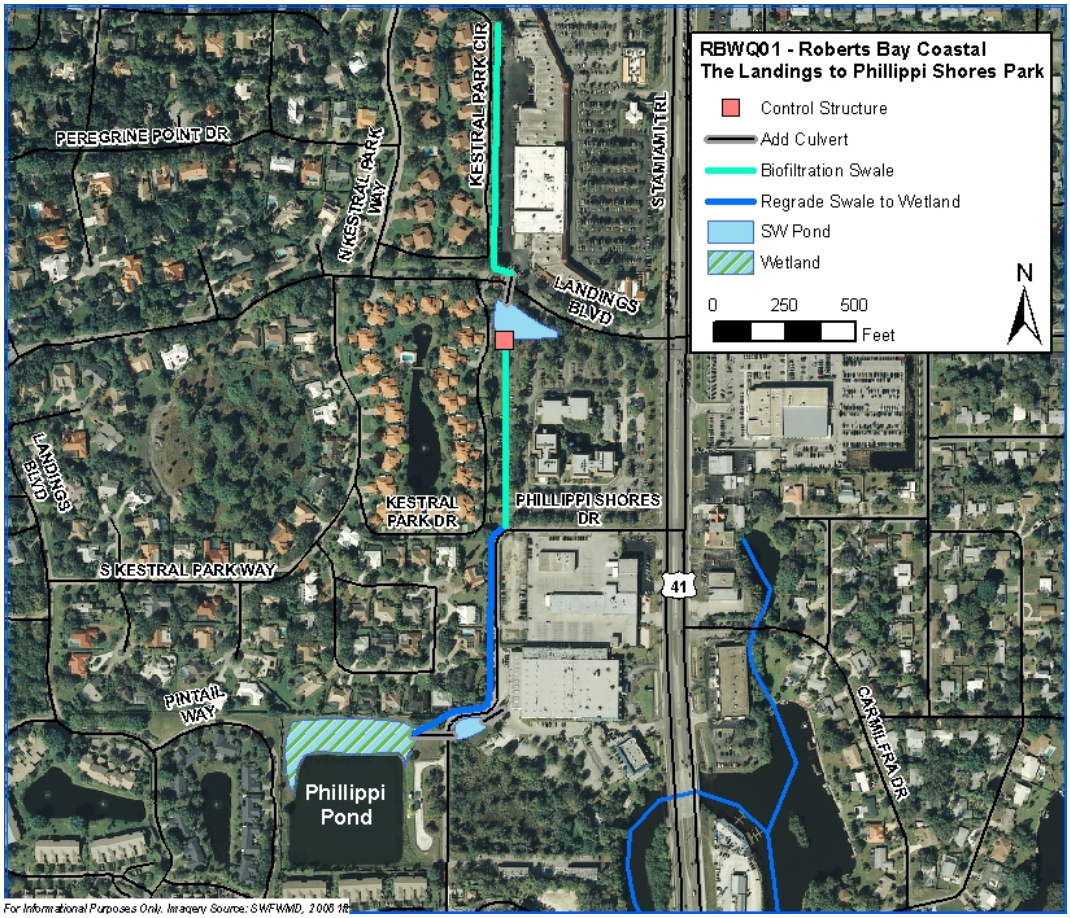

Figure 4-59 The Landings to Phillippi Shores Park Aerial Map

1. Site Evaluation

The area available for stormwater treatment is limited throughout this highly urbanized basin area, and the shopping center receives minimal stormwater treatment. TSS averages 163 lb/ac/yr and TN averages 12 lb/ac/yr. Phillippi Pond is a historical mining pit. A control structure on the southeast corner discharges to a series of swales and culverts leading to Phillippi Creek. Very limited runoff reaches the pond. This site is also being considered as an alternative water supply project in Chapter 5 of the WMP.

2. Proposed Project Elements

- Creating a 900-foot long biofiltration swale at the rear of the strip center and grading to flow south.
- Constructing a stormwater Pond at the north end of the parking lot on the school district property.
- Installing supporting infrastructure.

- Creating a small treatment wetland at the north end of the existing pond.
- Constructing a small dry retention pond.

3. Project Benefits

The biofiltration swale will provide pollutant removal for approximately 4 acres of impervious area and will decrease surface runoff. Approximately 6 acres of minimally treated impervious area can be easily diverted into the system for pollutant removal, thereby improving the water quality discharging to Phillippi Creek.

B. RBWQ03 – PC: Proctor Rd at Phillippi Creek

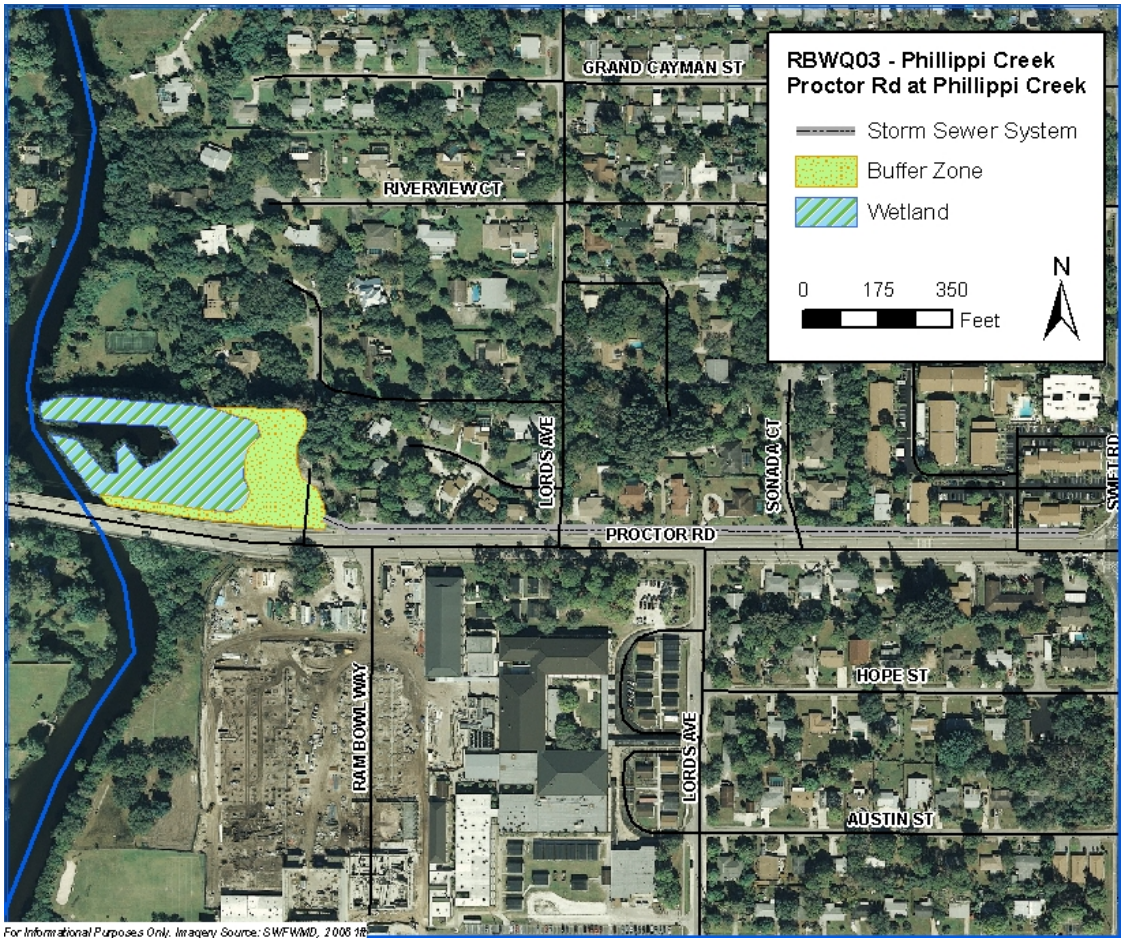

Figure 4-60 Proctor Rd at Phillippi Creek Aerial Map

1. Site Evaluation

Stormwater runoff from the catchment receives minimal treatment before entering the storm sewer system, which discharges into Phillippi Creek.

2. Proposed Project Elements

- Creating a wide riparian buffer zone and wetland.

3. Project Benefits

The improved natural system will provide treatment of runoff from approximately 8.5 acres directed to the area through a 1700-ft storm sewer system parallel to Proctor Rd.

C. RBWQ04 – PC: Pinkney Catchment

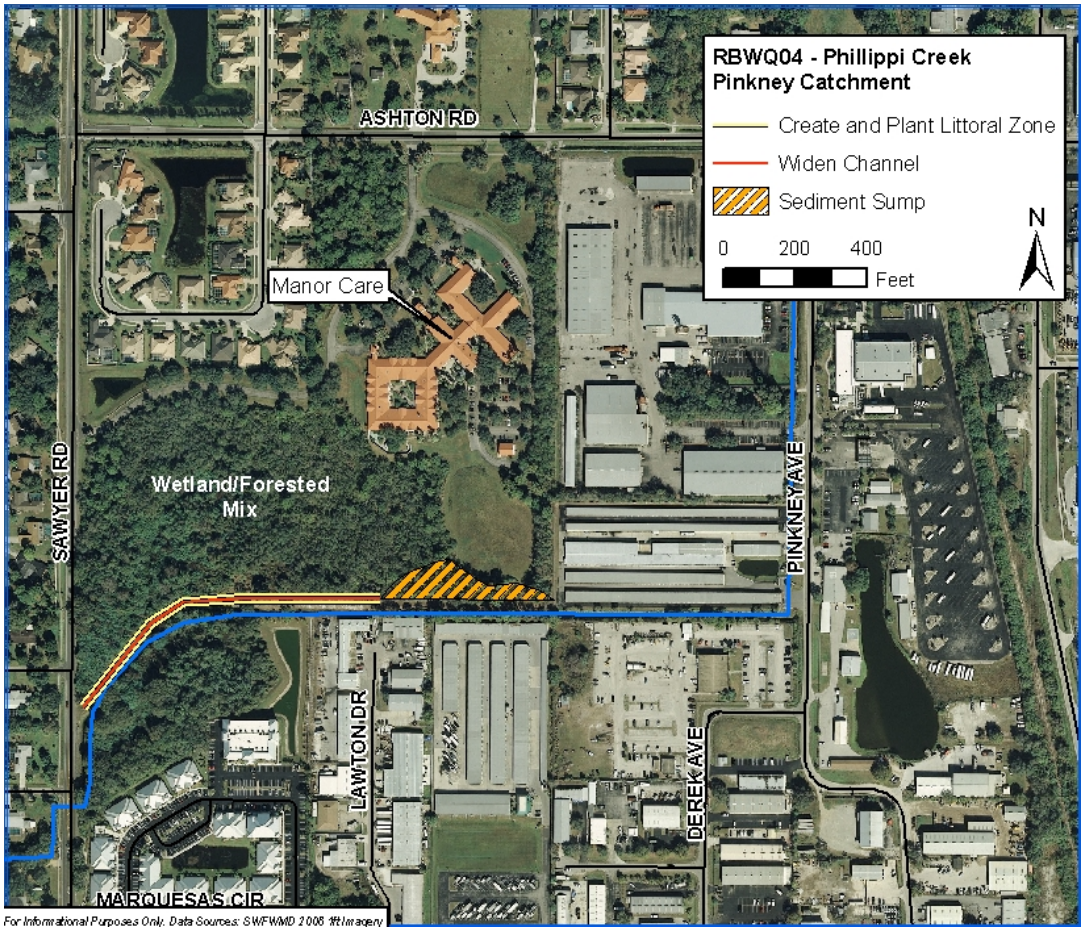

Figure 4-61 Pinkney Catchment Aerial Map

1. Site Evaluation

East of Sawyer Road is a wetland area at the south end of the Manor Care facility. The existing wetland is at the downstream end of a large industrial area. The existing channel is linear and at the southern boundary of the wetland. The current flowpath of the channel does not allow the treatment value of the wetland to be maximized. The TSS in stormwater runoff averages 160 lb/ac/yr and the TN averages 6.8 lb/ac/yr in the basin.

2. Proposed Project Elements

- Widening existing channel.
- Creating and planting littoral zone on both sides of the channel.
- Constructing a sediment sump.

3. Project Benefits

A sediment sump will encourage settling of suspended solids. Widening the channel will provide additional storage and establishing littoral zones will increase pollutant removal, particularly for low flow events.

D. RBWQ05 – PC: Ashton Catchment

Figure 4-62 Ashton Catchment Aerial Map

1. Site Evaluation

This 50-acre drainage area contains a wetland surrounded by a medium-density residential property. A linear channel conveying flow from the area is at the southern end of the wetland. The basin runoff averages 80 lb/ac/yr of TSS and 6 lb/ac/yr of TN.

2. Proposed Project Elements

- Adding a low-flow weir from the linear channel to the wetland to provide treatment.
- Widening existing channel.
- Creating and planting littoral zone on both sides of the channel.

3. Project Benefits

Adding a low flow weir from the channel to the wetland will increase residence times to allow increased uptake of nutrients by wetland vegetation. Widening the linear channel and planting littoral zones will facilitate pollutant removal.

E. RBWQ06 – PC: West Clark Lake

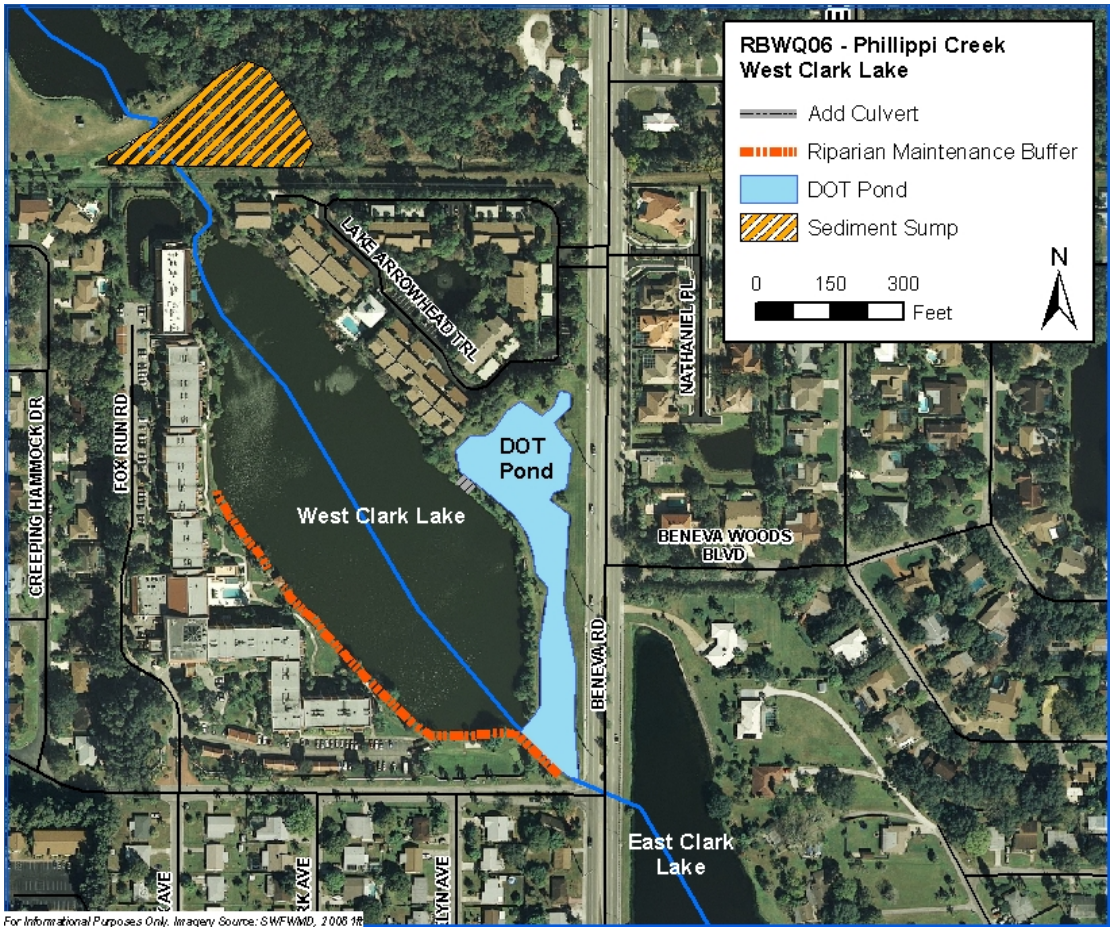

Figure 4-63 West Clark Lake Aerial Map

1. Site Evaluation

West Clark Lake was placed on the Verified List of impaired water bodies for nutrients in June 2005. FDEP and PBS&J jointly published an evaluation of restoration alternatives for the three lakes and Red Bug slough in December 2007.

2. Proposed Project Elements

- Altering flow path of discharge from East Clark Lake through the large Florida Department of Transportation (FDOT) pond adjacent to West Clark Lake by berming the inflow throat at the south end.
- Adding control structures/culverts from FDOT pond to West Clark Lake.
- Constructing a 1000-LF maintenance buffer on the southwest corner of the lake.
- Constructing a sediment sump.

3. Project Benefits

Altering the flowpath of the inflow from East Clark Lake by berming the inflow throat at the south and adding culverts on the northwest end to increase flow length will circulate the stormwater in the FDOT pond into the lake and eventually flow to Red Bug Slough. The FDOT pond is shallow and aerated. The maintenance buffer on the southwest side of the lake will reduce landscaping debris and fertilizer from reaching the lake. The sump area will provide a settling area for suspended solids that is easily accessible for maintenance and cleanout.

F. RBWQ07 – PC: Mirror Lake

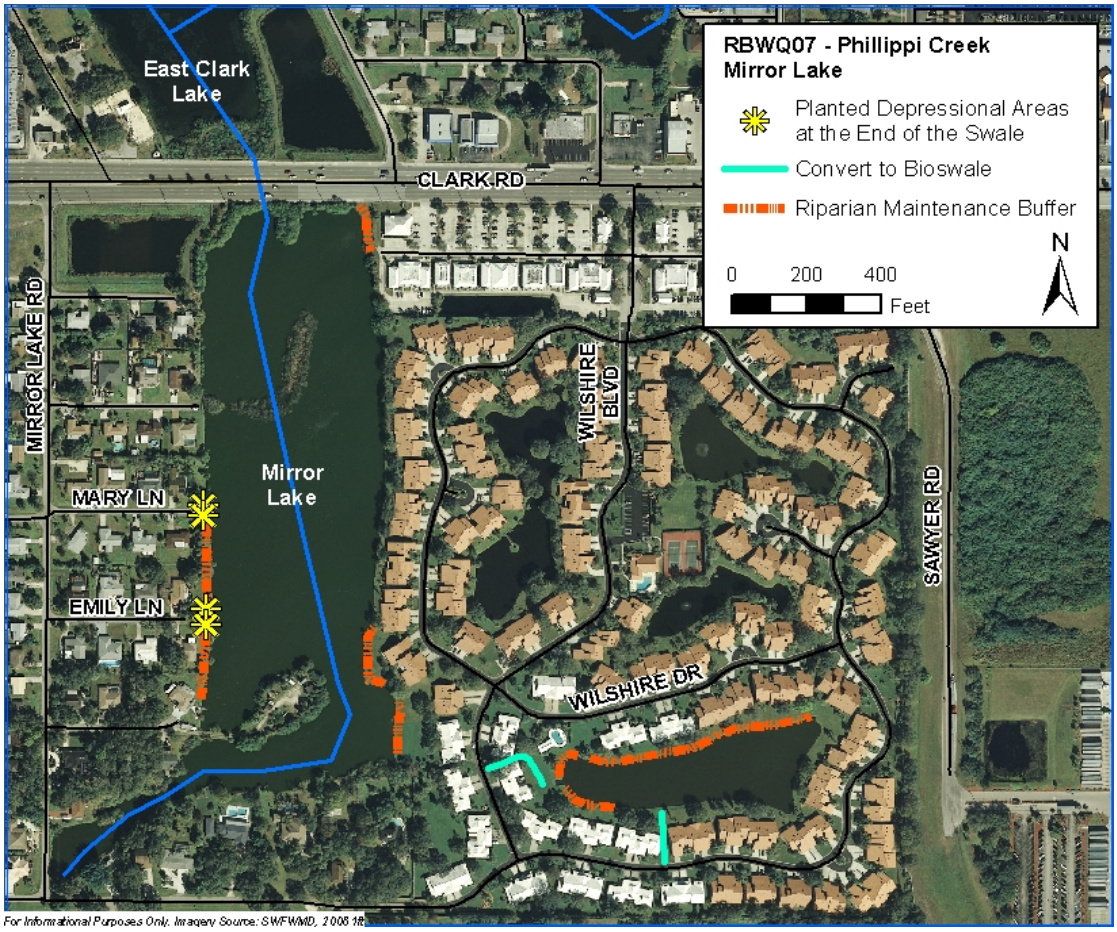

Figure 4-64 Mirror Lake Aerial Map

1. Site Evaluation

Mirror Lake is an upstream waterbody to West Clark Lake (an impaired waterbody). The contributing area to the lake is approximately 125 acres of primarily medium- and high-density residential areas. The area surrounding the lake is mowed and fertilized to the edge of the water.

2. Proposed Project Elements

- Adding a 1700-ft riparian buffer zone to reduce landscape debris and fertilizer from reaching the water body.
- Constructing two bioswales to treat runoff from Wilshire Court.
- Adding small depressional areas at the end of the roadside swales on Emily Lane and Mary Lane to capture and treat runoff from low flow events.

3. Project Benefits

The proposed project elements improve water quality in the lake by provided a buffer to treat stormwater runoff before it reaches the lake.

G. RBWQ08 – PC: Atlantic WWTP

Figure 4-65 Atlantic WWTP Aerial Map

1. Site Evaluation

On the southwest side of the decommissioned Atlantic WWTP lies an older neighborhood lacking stormwater treatment facilities. This medium-density residential area is approximately 235 acres. Runoff from this area travels through a swale and culverts system to collector channels in the Linwood area, discharging to Phillippi Creek.

2. Proposed Project Elements

- Enlarging existing pond on WWTP site.
- Adding two bubbler structures to carry flow under channel segment.
- Creating a 2500-ft sinuous channel.
- Adding control structures from the wetland to the channel.

3. Project Benefits

To provide treatment for this area, two bubbler boxes would convey flow from the Linwood system under the creek to the reclaimed water pond on the WWTP site. The pond would need to be enlarged and re-graded with the addition of a discharge structure to the north side.

H. RBWQ09 – PC: Riverbluff Parkway LID Retrofit

Figure 4-66 Riverbluff Parkway LID Retrofit Aerial Map

1. Site Evaluation

Strathmore Riverside Villas is a high-density condominium community that was constructed in 1970. Land available for traditional stormwater BMPs is not available in this location.

2. Proposed Project Elements

- Constructing biofiltration swales.
- Adding cisterns.

3. Project Benefits

The proposed biofiltration swales between the buildings will collect and direct runoff to larger biofiltration areas designed to capture and promote infiltration instead of the direct discharge to the channel in existing conditions. The cisterns will facilitate irrigation and maintenance of the biofiltration vegetation

I. RBWQ10 (RBS19) – MC: Cedar Cove

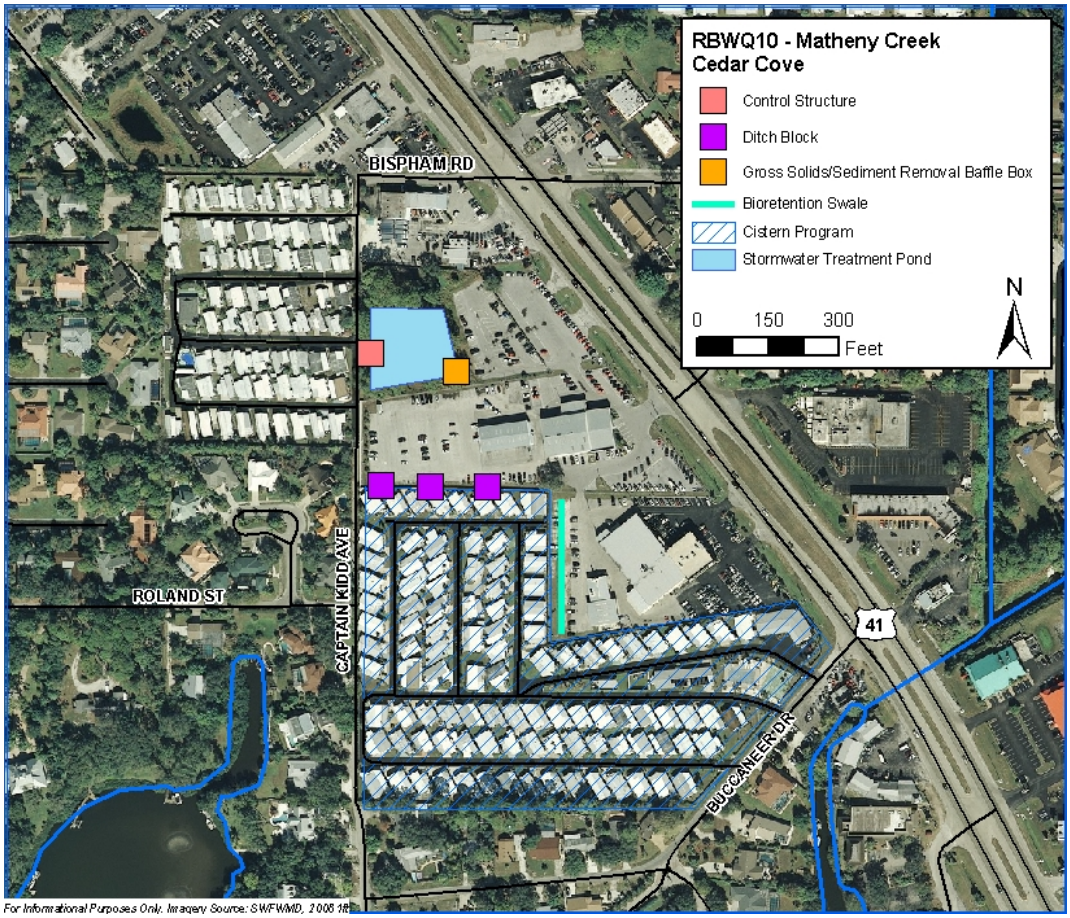

Figure 4-67 Cedar Cove Aerial Map

1. Site Evaluation

The area for proposed improvements drains to Cedar Cove with little treatment. Residents report sediments flow out of a culvert into the cove. The highly urbanized area limits the use of traditional stormwater BMPs. The project is composed of multiple elements and will need cooperative effort and partnership between the County and private land owners to be successful.

2. Proposed Project Elements

- Facilitating cistern use in the mobile home community.
- Developing a stormwater treatment system for a nearby car dealership—increasing the width of the existing swales, installing ditch blocks, altering the control elevation on the discharge structure.
- Constructing a stormwater treatment pond with a sediment-removal structure.

3. Project Benefits

Treatment BMPs improve water quality, habitat, channel, and flood control conditions by removing sediment and pollutants from the system. Cistern use will reduce the surface-water runoff from areas that do not currently receive stormwater treatment.

J. RBWQ11 (RBS18) – MC: Denham Lateral

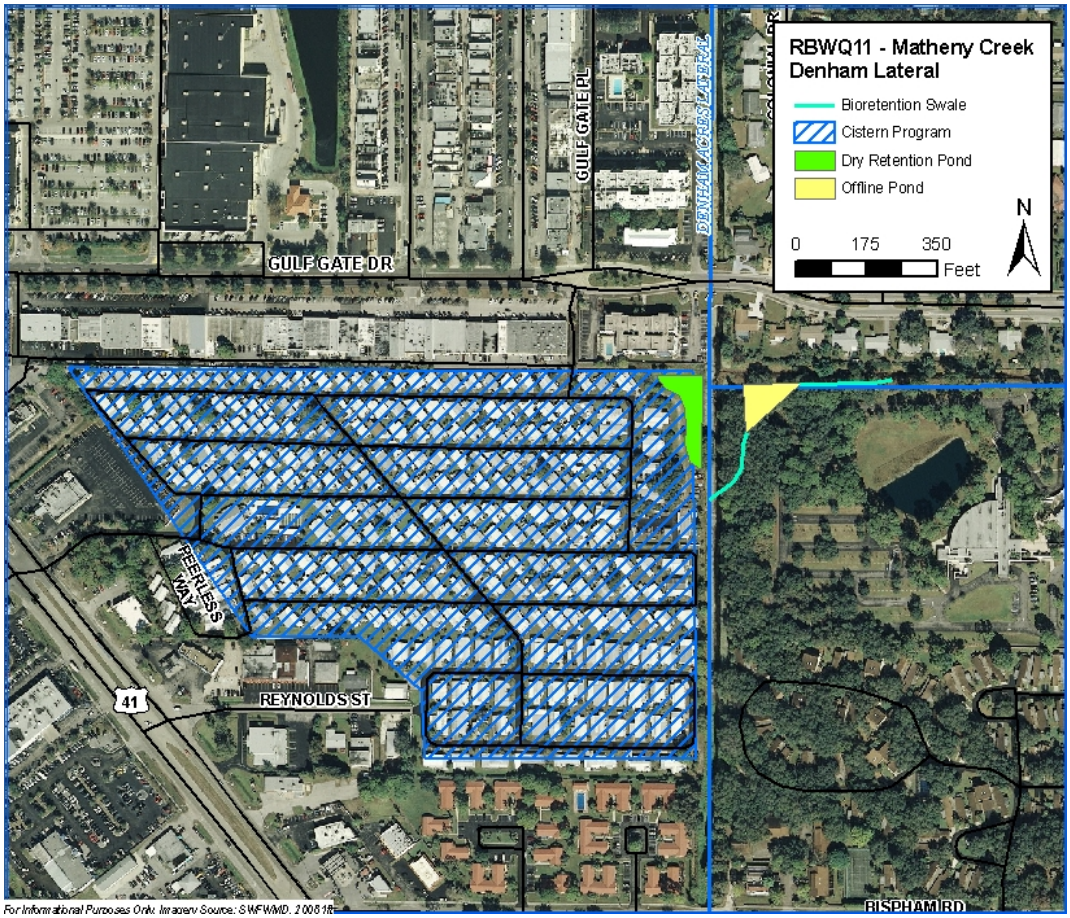

Figure 4-68 Denham Lateral Avenue Aerial Map

1. Site Evaluation

This site is between Gulf Gate Drive and Bispham Road, east of US 41. The northwest corner of St Thomas More Catholic Church is adjacent to a hardened tributary that flows into Denham Acres Lateral (DHL) channel. This site has undeveloped area available for an offline treatment system to treat flow originating to the east. A mobile home community occupies approximately 24 acres west of the DHL channel. The community does not have any stormwater treatment facilities. The project is composed of multiple elements and will need partnerships between the County and private land owners to be successful.

2. Proposed Project Elements

- Constructing an offline wet detention pond to treat runoff from the channel.
- Facilitating cistern use in the mobile home community.
- Constructing a bioretention swale to direct runoff to the offline pond.

- Adding a dry retention pond in the area adjacent to the channel that is used for parking boats and RVs and redirecting a portion of the untreated stormwater runoff from the storm sewer system to the pond.

3. Project Benefits

Stormwater treatment BMPs improve water quality, habitat, channel, and flood control conditions by removing sediment and pollutants from the system. Cistern use will reduce the surface-water runoff from areas that do not currently receive stormwater treatment.

4.5.2.4 Results and Discussion

A. Pollutant-Load Removal Estimates

Jones Edmunds reviewed the spatial results of the SIMPLE model to determine hot spots for TN, TP, and TSS in the watershed. The hot spots were part of the GIS desktop analysis used to identify potential projects. Table 4-4 summarizes the average annual loading in each subbasin. Figures 4-69, 4-70, and 4-71 show the conceptual project sites in relation to the spatial results of the average annual loads by subbasin for TN, TP, and TSS.

The results of the SIMPLE model were used to calculate loading rates in pounds per acre per year value by catchment area. To calculate the range of pollutant removal by BMP, the normalized results by catchment from the SIMPLE model were multiplied by the contributing area to create a pounds-per-year value. The pounds-per-year values were multiplied by the minimum and maximum reported efficiencies for the BMP to give a range of potential pounds per year of pollutant removed from stormwater runoff. Table 4-5 shows the estimated range of pounds per year of pollutant removed by the proposed BMP

Table 4-4 Annual Average Pollutant Loads (lb/ac/yr) and Rank

Subbasin ID	Basin Name	ICPR Group	Area (ac)	TSS (lb/ac/yr)	TSS Rank	TP (lb/ac/yr)	TP Rank	TN (lb/ac/yr)	TN Rank
42	MATHENY CREEK	MC-BREAK	114.3	251.54	3	7.46	2	33.59	2
43	MATHENY CREEK	MC-CORAL	93.3	260.49	2	2.01	10	11.02	6
44	MATHENY CREEK	MC-DENHM	517.0	169.70	12	1.65	21	8.28	14
45	MATHENY CREEK	MC-LOWER	9.1	310.72	1	2.18	7	12.74	4
46	MATHENY CREEK	MC-UPPER	455.7	98.95	42	1.09	45	5.92	47
47	MATHENY CREEK	MC-LOWER	275.4	161.96	16	2.13	8	10.23	7
48	MATHENY CREEK	MC-CLARK	258.6	150.69	19	1.69	18	8.20	16
49	PHILLIPPI CREEK	ASHTON	199.6	74.62	45	1.25	44	6.24	44
50	PHILLIPPI CREEK	BAY-ST	19.3	119.12	34	1.47	35	7.63	27
51	PHILLIPPI CREEK	BRAN-AC	1350.4	111.37	37	1.76	17	7.15	36
52	PHILLIPPI CREEK	BRAN-AE	1571.8	142.16	22	1.77	16	7.97	21
53	PHILLIPPI CREEK	BRANCHBA	1790.1	116.33	35	1.51	32	7.62	29
54	PHILLIPPI CREEK	BRANCHBB	1437.7	197.70	8	1.95	12	8.37	13
55	PHILLIPPI CREEK	BRANCHBC	599.3	110.84	39	1.31	42	6.67	42
56	PHILLIPPI CREEK	BRANCHC	1030.8	125.32	32	1.38	38	7.63	28
57	PHILLIPPI CREEK	BRANCHCB	1106.0	167.64	14	1.52	31	7.36	34
58	PHILLIPPI CREEK	BRNCHAA	3193.0	133.36	26	1.46	36	7.07	39
59	PHILLIPPI CREEK	CELERY	2387.0	72.11	47	0.98	48	5.35	48
60	PHILLIPPI CREEK	GERHARDT	17.1	74.33	46	1.06	46	6.13	46
61	PHILLIPPI CREEK	HYDEPAR	584.9	168.85	13	1.83	14	8.87	9
62	PHILLIPPI CREEK	JAMAICA	171.5	137.03	24	1.37	39	7.97	20
63	PHILLIPPI CREEK	LAT-AH	372.0	95.41	43	2.29	5	7.09	38
64	PHILLIPPI CREEK	LAT_AB	1180.5	133.08	27	1.63	22	7.95	22
65	PHILLIPPI CREEK	LAT_ABA	153.6	87.61	44	1.40	37	6.86	41
66	PHILLIPPI CREEK	LAT_ABC	820.0	123.43	33	1.61	23	7.66	26
67	PHILLIPPI CREEK	LATBB	553.2	171.96	11	1.55	29	8.18	17
68	PHILLIPPI CREEK	LATBC	242.2	156.99	17	1.81	15	8.50	10
69	PHILLIPPI CREEK	LATBCA	239.5	147.98	20	1.98	11	7.98	18
70	PHILLIPPI CREEK	LATBD	412.1	141.13	23	1.91	13	7.98	19
71	PHILLIPPI CREEK	LATCA	463.2	136.25	25	1.58	26	7.48	33
72	PHILLIPPI CREEK	LATCAZ	238.0	221.93	6	1.67	20	8.50	11
73	PHILLIPPI CREEK	LATCE	1483.7	64.46	48	1.01	47	6.22	45
74	PHILLIPPI CREEK	LINWOOD	617.5	111.36	38	1.55	28	7.58	31
75	PHILLIPPI CREEK	LWRPHIL	1593.3	179.35	10	1.61	25	8.26	15
76	PHILLIPPI CREEK	MAIN-A	4129.8	132.99	28	2.05	9	7.91	23
77	PHILLIPPI CREEK	MAINB	2071.3	145.69	21	1.56	27	7.51	32
78	PHILLIPPI CREEK	MAINC	639.1	227.39	5	1.36	40	7.16	35
79	PHILLIPPI CREEK	MIDPHILL	2067.9	185.60	9	2.46	4	11.54	5
80	PHILLIPPI CREEK	PINKNEY	176.4	132.98	29	1.27	43	6.55	43

Table 4-4 Annual Average Pollutant Loads (lb/ac/yr) and Rank

Subbasin ID	Basin Name	ICPR Group	Area (ac)	TSS (lb/ac/yr)	TSS Rank	TP (lb/ac/yr)	TP Rank	TN (lb/ac/yr)	TN Rank
81	PHILLIPPI CREEK	PROCTOR	451.3	130.84	30	1.69	19	8.48	12
82	PHILLIPPI CREEK	PUMPSTA	54.7	116.08	36	1.61	24	7.61	30
83	PHILLIPPI CREEK	REDBUG	969.1	126.22	31	1.53	30	7.87	24
84	PHILLIPPI CREEK	SEAVIEW	281.9	207.87	7	2.25	6	9.14	8
85	PHILLIPPI CREEK	TWOLAKES	55.3	106.63	40	1.48	34	7.14	37
86	PHILLIPPI CREEK	UPPER	1104.5	103.04	41	1.33	41	6.94	40
87	COASTAL	COASTAL	303.2	234.15	4	11.37	1	47.18	1
88	COASTAL	COASTAL	1241.6	156.87	18	1.50	33	7.67	25
89	COASTAL	COASTAL	1144.5	163.13	15	4.42	3	18.35	3

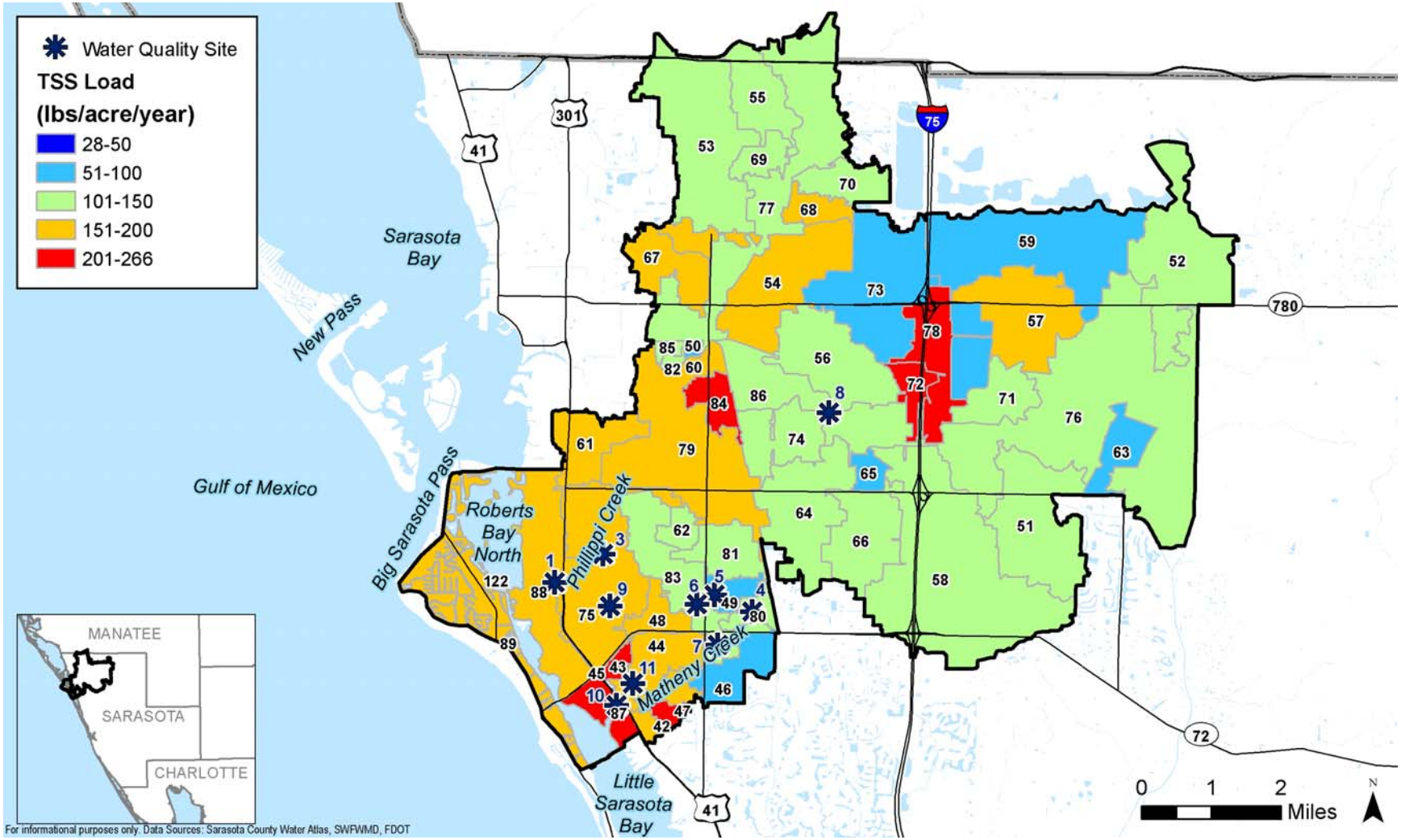

Figure 4-69 Roberts Bay North Watershed Water Quality Conceptual Site Locations Overlaid on the Average Annual TSS Load Per Unit Area Results

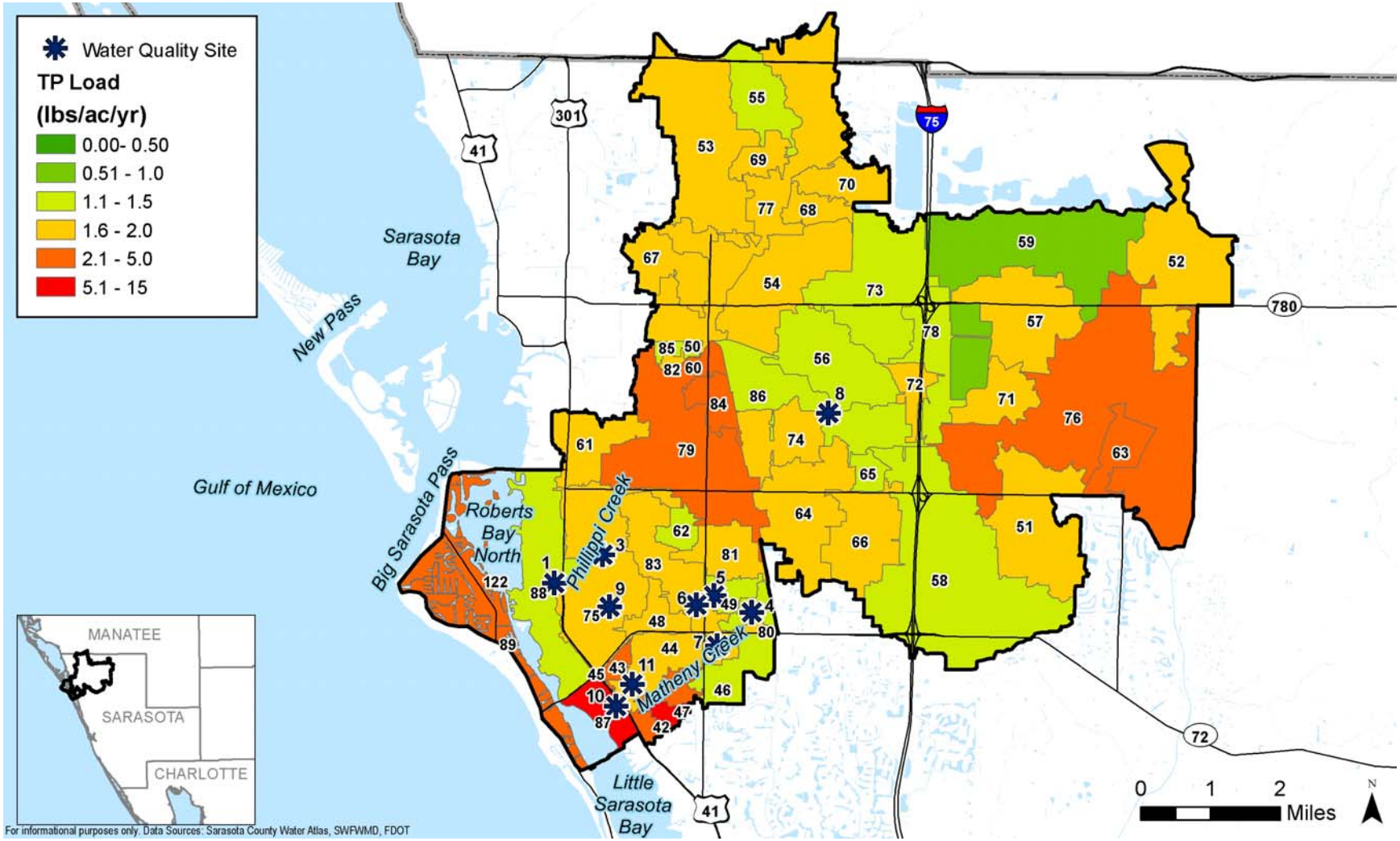

Figure 4-70 Roberts Bay North Watershed Water Quality Conceptual Site Locations Overlaid on the Average Annual TP Load Per Unit Area Results

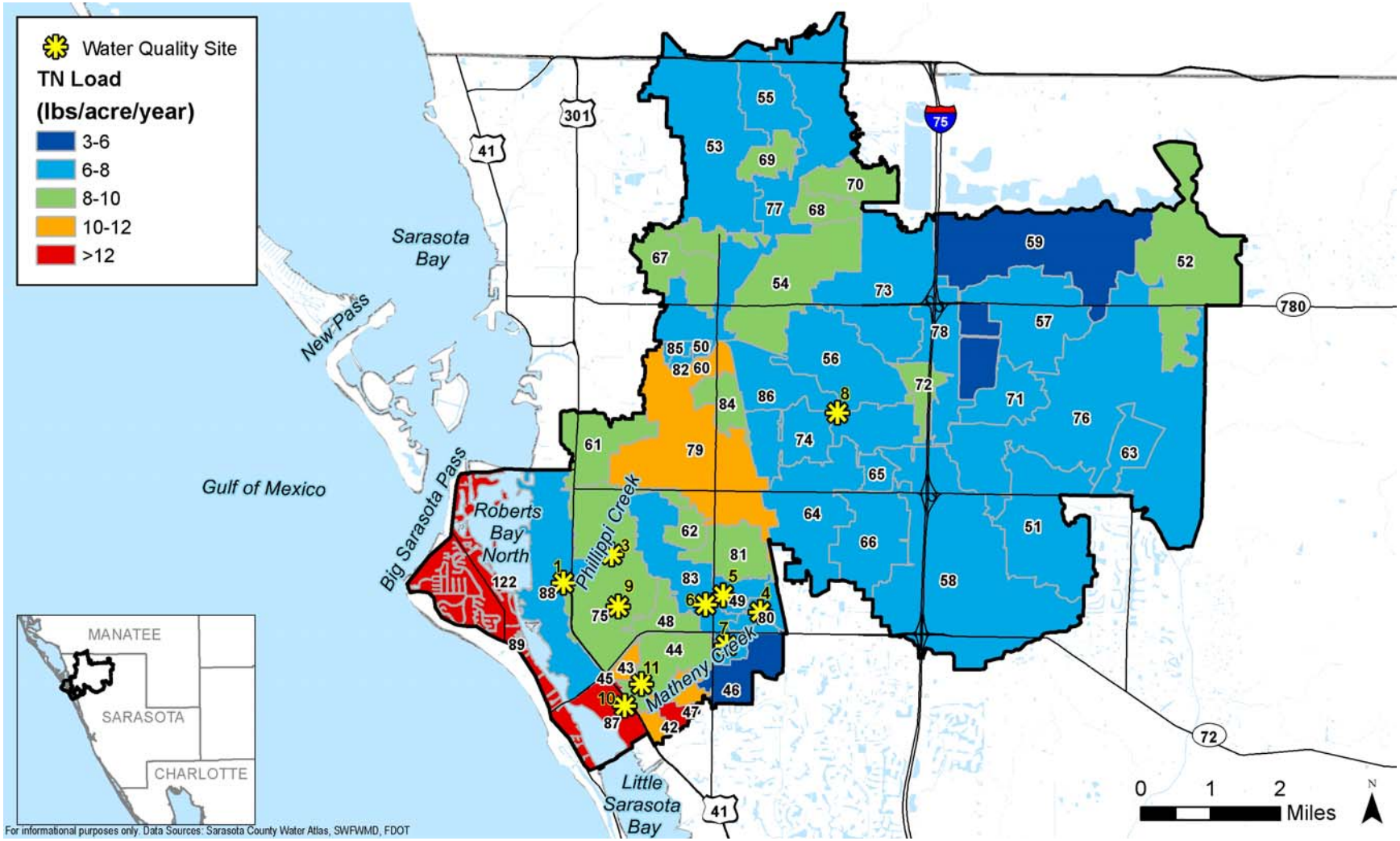

Figure 4-71 Roberts Bay North Watershed Water Quality Conceptual Site locations Overlaid on the Average Annual TN Load Per Unit Area Results

Table 4-5 Estimated Pollutant-Load Removal by Proposed BMP

Project ID	Basin	Project Name	BMP Type	Estimated Drainage Area (Ac.)	Estimated Pollutant Removal (lb/yr)		
					Total Suspended Solids	Total Phosphorus	Total Nitrogen
RBWQ1	RBC	The Landings to Phillippi Shores	SW Pond	4	400 - 600	0 - 10	0 - 30
			Wetland	6	800 - 1000	0 - 10	10 - 40
			Dry Pond	1.2	200 - 200	0 - 10	10 - 10
			Bioswales	4	100 - 500	0 - 10	20 - 30
			Total		**1400 - 2300**	**0 - 40**	**50 - 110**
RBWQ3	PC	Proctor Rd	Wetland	8.5	1200 - 1500	0 - 10	10 - 40
			Riparian Maintenance Buffer	8.5	600 - 1300	0 - 10	50 - 70
			Total		**1800 - 2800**	**0 - 20**	**60 - 100**
RBWQ4	PC	Pinkney Catchment	Sediment Sump	20	800 - 2200	0 - 10	0 - 0
			Littoral Zone	20	600 - 2500	0 - 10	50 - 80
			Total		**1400 - 4800**	**0 - 20**	**50 - 80**
RBWQ5	PC	Ashton Catchment	Littoral Zone	25	400 - 1600	0 - 10	60 - 90
			Total		**400 - 1600**	**0 – 10**	**60 - 90**
RBWQ6	PC	W Clark Lake	Riparian Maintenance Buffer	0.5	0 - 100	0 - 10	0 - 0
			Sediment Sump	12	400 - 1000	0 - 10	0 - 0
			Total		**400 - 1100**	**0 – 20**	**0 - 0**

Table 4-5 Estimated Pollutant-Load Removal by Proposed BMP

Project ID	Basin	Project Name	BMP Type	Estimated Drainage Area (Ac.)	Estimated Pollutant Removal (lb/yr)		
					Total Suspended Solids	Total Phosphorus	Total Nitrogen
RBWQ7	PC	Mirror Lake	Riparian Maintenance Buffer	0.75	0 - 100	0 - 10	0 - 0
			Bioswales	0.5	0 – 100	0 - 10	0 - 0
			Total		**0 - 200**	**0 - 20**	**0 - 0**
RBWQ8	PC	Atlantic WWTP	SW Pond	235	14600 - 25000	0 - 10	190 - 2960
			Total		**14600 - 25000**	**0 - 10**	**190 - 2960**
RBWQ9	PC	Riverbluff Blvd	Cisterns	7	400 - 800	0 - 10	10 - 30
			Bioretention Swales	7	300 - 1000	0 - 10	30 - 40
			Total		**600 - 1800**	**0 - 20**	**40 - 70**
RBWQ10 (RBS19)	MC	Cedar Cove	Cisterns	3.3	200 - 300	0	0 - 20
			Wet Detention Pond	9	800 - 1400	0 - 20	0 - 70
			Bioretention Swale	2.3	100 - 300	0 - 10	0 - 20
			Total		**1100 - 2000**	**0 - 30**	**0 - 110**
RBWQ11 (RBS18)	MC	Denham Lateral	Wet Detention Pond	4	400 - 600	0 - 10	0 - 20
			Dry Retention Pond	5.6	800 - 900	0 - 10	30 - 50
			Cisterns	7.5	400 - 800	0 - 10	15 - 30
			Total		**1600 - 2300**	**0 - 30**	**85 - 100**

B. Conceptual Level Cost Estimates

Table 4-6 Conceptual Level Estimates of Probable Cost

Project ID	Basin	Project Name	Total Project Cost+	Construction Costs*	Engineering Services Costs	Maintenance Costs
RBWQ1	RBC	The Landings to Phillippi Shores	$508,000	$414,000	$94,000	$2500
RBWQ3	PC	Proctor Road at Phillippi Creek	$601,000	$490,000	$111,000	$200
RBWQ4	PC	Pinkney Catchment	$802,000	$653,000	$149,000	$2,100
RBWQ5	PC	Ashton Catchment	$113,000	$82,000	$31,000	$200
RBWQ6	PC	West Clark Lake	$134,000	$107,000	$27,000	$2,100
RBWQ7	PC	Mirror Lake	$91,000	$61,000	$30,000	$1,500
RBWQ8	PC	Atlantic WWTP	$816,000	$665,000	$151,000	$1,100
RBWQ9	PC	Riverbluff Blvd	$177,000	$141,000	$36,000	$1,500
RBWQ10	MC	Cedar Cove	$337,000	$275,000	$62,000	$5,700
RBWQ11	MC	Denham Lateral	$577,000	$470,000	$107,000	$3,500

+ Total Project Cost includes Mobilization and Contingency costs along with Construction Costs and Engineering Design Services

* Design Services include Survey, Geotechnical Investigation, Engineering Design, and Permitting

Ranking of Potential Projects

Table 4-7 Ranking of Potential Projects

Project ID	Basin	Project Name	Pollutant Removal Estimate (lb/yr)								Total Project Cost	Cost per lb/yr range		Average $/lb	Rank
			TSS		TP		TN		Total						
RBWQ08	PC	Atlantic WWTP	14,600	25,000	0	10	190	2,960	14,790	27,970	$816,000	$55	$29	$42	1
RBWQ05	PC	Ashton Catchment	400	1,600	0	10	60	90	460	1,700	$113,000	$246	$66	$156	2
RBWQ09	PC	Riverbluff Blvd	600	1,800	0	20	40	70	640	1,890	$177,000	$277	$94	$185	3
RBWQ06	PC	West Clark Lake	400	1,100	0	20	0	0	400	1,120	$134,000	$335	$120	$227	4
RBWQ10	MC	Cedar Cove	1,100	2,000	0	30	0	110	1,100	2,140	$337,000	$306	$157	$232	5
RBWQ03	PC	Proctor Road at Phillippi Creek	1,800	2,800	0	20	60	100	1,860	2,920	$601,000	$323	$206	$264	6
RBWQ01	RBC	The Landings to Phillippi Shores	1,400	2,300	0	40	50	110	1,450	2,450	$508,000	$350	$207	$279	7
RBWQ11	MC	Denham Lateral	1,600	2,300	0	30	85	100	1,685	2,430	$577,000	$342	$237	$290	8
RBWQ04	PC	Pinkney Catchment	1,400	4,800	0	20	50	80	1,450	4,900	$802,000	$553	$164	$358	9
RBWQ07	PC	Mirror Lake	0	200	0	20	0	0	0	220	$91,000	$91,000	$414	$45,707	10