BREVARD COUNTY GROUNDWATER MONITORING AND MODELING REPORT

Final Report for the Groundwater Pollution, Engaging the Community in Solutions (FDEP Contract #LP05112) and Save Our Indian River Lagoon Project Plan (SOIRLPP) Groundwater Quality Monitoring (Task Order #271010-14-003)

PREPARED FOR:

PREPARED BY:

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Revised v.2

Executive Summary

The goal of this project was to better understand the extent that wastewater contaminated groundwater is impacting the Indian River Lagoon by examining groundwater nutrient concentrations in different regions of Brevard County. The comprehensive study included groundwater and soil sampling, spatial analyses, groundwater modeling, a representative septic tank behavioral survey, and investigating a septic tank additive designed to reduce nutrient leachate. This report provides the final analysis of groundwater and soil data and the final Groundwater Modeling Report, satisfying deliverables for FDEP Contract #LP05112 Tasks 2 & 5 and Brevard County Task Order #271010-14-003.

The project examined groundwater nutrient concentrations in residential communities that had different wastewater treatment types: 1) septic tanks; 2) municipal sewer systems; or 3) municipal sewer systems with reclaimed irrigation. Monitoring wells were installed in 13 residential neighborhoods and 3 natural areas located in five regions of Brevard County including the mainland and barrier islands. Regions and treatment types were compared with each other and with natural areas to see differences in the extent of nutrient pollution. Isotopes were used to clarify sources and denitrification dynamics. Field data were used to replace the uniform TN and TP concentrations in the watershed loading model that estimated baseflow nutrient loadings into the Indian River Lagoon. The data collected can guide wastewater retrofit project selection and evaluation as well as verify and update nutrient pollutant load estimates to the Indian River Lagoon.

As part of the legislatively funded project, homeowners were engaged in an inexpensive intervention strategy that could potentially reduce the pollutants leaching from their septic tanks. To address this goal, an in-situ septic treatment product called BiOWiSH was distributed to nearly every resident located in the Turkey Creek septic tank community. Surveyed participants found the project easy to use, they believed it was having a positive impact and they were willing to pay to continue using the product. Based on preliminary data, no consistent, across the board reduction in TN and/or TP concentrations are apparent after the BiOWiSH product is delivered.In this study, reclaimed communities had as much total nitrogen pollution in groundwater as septic communities. There was no significant difference in groundwater Total Nitrogen (TN) concentrations in septic and reclaimed communities. Although the differences were not statistically significant, the TN concentration in the reclaimed communities (4.2 mg/L) was higher than the septic community (2.6 mg/L). The sewer communities TN concentration was 1.2 mg/L and natural areas was 0.35 mg/L.

Total Nitrogen (TN) is made up of both organic and inorganic forms of nitrogen. We found differences in which form of nitrogen was present in the two most polluting treatment types. Septic communities' groundwater nitrogen was dominated by organic forms of nitrogen (TKN and ammonia, NH_3). Reclaimed communities were dominated by inorganic nitrogen (nitrate/nitrite, NO_X). This is interesting considering both sources of nitrogen are organic forms derived from human waste. The differences between them must be associated with nitrification and denitrification processes.

Septic communities' groundwater phosphorus concentration was nearly five times higher than the others. Both Total Phosphorus (TP) and ortho-phosphate (PO_4^{3}) were significantly higher in septic communities than in sewer and reclaimed communities and natural areas. The median septic TP concentration was 0.57 mg/L, with concentrations that ranged from 0.97 in Turkey Creek to 0.46 mg/L in Suntree. Reclaimed communities also had high TP concentrations, with the Beaches reclaimed community (0.72 mg/L) significantly higher than the reclaimed communities in Suntree (0.036 mg/L), Titusville (0.110 mg/L) and Turkey Creek (0.012 mg/L).

Septic contaminant plume maps showed that nutrient plumes are extending to receiving waters and that denitrification processes are working to reduce concentrations along the way. Isotopes indicated that there were different denitrification rates occurring in different sites. Future research may clarify conditions where denitrification is more effectively reducing groundwater nutrient concentrations prior to reaching the Lagoon.

Septic and reclaimed communities were equally polluting in this study, but sewer communities also had high groundwater nutrient concentrations although these varied tremendously. Some sewer communities had nutrient concentrations similar to those in natural areas and others had groundwater concentrations higher than those in septic communities. Of the five sewer communities investigated as part of this study, Turkey Creek had the highest nutrient concentrations.

The Turkey Creek sewer community had twice as much organic nitrogen (3.7 mg/L TKN) as the other sewer communities in the study and three times more reactive phosphorus (0.49 mg/L PO_4^{3-}). In fact, in Turkey Creek, the sewer community had higher organic nitrogen concentrations than the septic community in the same region (1.40 mg/L TKN). Isotopic signatures revealed numerous nitrogen sources in the Turkey Creek sewer community. One source appeared to be synthetic fertilizer. The other source appeared to be highly enriched wastewater, possibly the result of leaking sewage lines. Further investigation showed groundwater nitrogen increased in the Turkey Creek sewer community in September and October 2017, when Hurricane Irma deluged the area with rain.

This and the 3-day data used in the Principal Component Analyses demonstrate that rainfall is an important driver of nutrient concentrations that is site and region specific. In some cases, 3-day rainfall (72-hours of cumulative rainfall) was positively related to nutrient concentrations and in other cases, negatively related. When and how rainfall impacts groundwater nutrients is key to understanding nutrient fate and transport. In our study, a better understanding of groundwater elevation is needed to better estimate changes in groundwater flow direction and velocity; to refine loading estimates; and to better delineate contaminant plumes. Existing monitoring wells should be surveyed so groundwater elevations can be compared.

The original SWIL loading model that calculated stormwater and baseload contributions of TN and TP to the Indian River Lagoon used a uniform TN concentration of 0.86 mg/l to estimate baseflow loads. This value is lower than the TN concentrations measured in this study, with the largest discrepancies in the septic (2.55 mg/L) and reclaimed (4.50 mg/L) communities. The uniform TP concentration that was used for baseflow loading calculations in the SWIL model (0.112 mg/L) were closer to those found in this study, with the exception of the septic communities, which had substantially higher TP concentrations (0.6 mg/L). Replacing the uniform groundwater nutrient concentrations with actual values in the SWIL model and running the model for a small (5,627 acre) subset of the watershed, increased the estimated TN baseflow nutrient loadings by 84% or an additional 22,016 lbs./yr and TP by 13% or another 458 lbs./yr.

Across every region and treatment there were potential sources of variation that may originate with actions that the homeowner takes. The most evident of these actions are the use of lawn fertilizers and reclaimed water. A representative survey of homeowners in Brevard County could be conducted to better understand the timing, types, and amount of fertilizer being applied to residential lawns as well as irrigation practices. In addition, the residents living in the homes where the monitoring wells are installed could be surveyed to understand seasonal occupancy, visitors, fertilizer use and septic maintenance practices that can help explain variability in groundwater concentrations.

Data collected from this study can be used to update several modeling efforts previously completed to guide the septic moratorium, prioritize septic upgrades and septic to sewer conversion projects, and to refine the baseflow component of the previously developed watershed loading model (SWIL). Since the data collection has been extended until December 2020, these updates will likely only take place with the entire period of record dataset in the spring/summer of 2021.

This study should also be linked to other surface water studies to better understand the link between groundwater and receiving surface waters. Turkey Creek provides an opportunity to work with FIT scientists who are already examining surface water nutrients. Seepage meters could be installed in Turkey Creek to measure the volume, concentration, and form of nitrogen entering the lagoon through seepage. This would provide a critical missing link between load estimates and actual conditions.

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Introduction

In June 2017, the Marine Resources Council (MRC) and Applied Ecology, Inc. (AEI) initiated a pilot project in Turkey Creek with legislative funding provided to the Florida Institute of Technology. The pilot project included the installation and sampling of 11 monitoring wells for ten months to understand groundwater nutrient concentrations in various land uses.

A year later, additional legislative funding was provided to Marine Resources Council to expand the study. The project titled "Groundwater Pollution, Engaging the Community in Solutions" funded the installation of 20 additional groundwater monitoring wells and the continuous monitoring of 30 wells for 18 months. One dry well had to be re-installed. The study also included spatial data analyses, groundwater modeling, a representative behavioral survey, and investigating a septic tank additive designed to reduce nutrient leachate.

An additional 15 groundwater wells were installed and monitored with funding from the Brevard County Save Our Indian River Lagoon Project Plan (SOIRLPP) Fund. Funding was also provided to continue monitoring Turkey Creek until the legislatively funded "Groundwater Pollution" funding was received. In addition to examining potential groundwater pollution differences regionally and between treatments, the SOIRLPP Groundwater Quality Monitoring project evaluates the effectiveness of septic tank retrofit projects by providing performance measures before and after retrofit completion.

This expansive and comprehensive project includes the data collected as part of the monthly sampling of 45 monitoring wells for eighteen months and an additional 12 months of monthly data in Turkey Creek collected as part of the initial pilot project. This report provides the final monitoring report for the legislatively-funded project titled Groundwater Pollution, Engaging the Community in Solutions, FDEP Contract #LP05112; and the final report for the Brevard County funded project titled Save Our Indian River Lagoon Project Plan (SOIRLPP) Groundwater Quality Monitoring Task Order #271010-14-003.

The goal of this project is to better understand the extent that wastewater contaminated groundwater is impacting the Indian River Lagoon by examining groundwater contamination in different regions of Brevard County within communities that are receiving three different wastewater treatments. Septic tanks communities, sewer communities, and sewer communities that also received reclaimed wastewater for irrigation were selected for the study. Homeowners within those communities were recruited for participation, and property access agreements were executed for well installation and sampling. If a natural area was located nearby, monitoring wells were installed there as an indication of background groundwater nutrient conditions.

Study Areas

There are five regional study areas being investigated in this project, each with a different grouping of septic, sewer, and reclaimed communities and natural areas (Figure 1). Here we describe the communities and ecological characteristics of each region as an introductory background to the study areas. Site selection details are included in the Methods section.

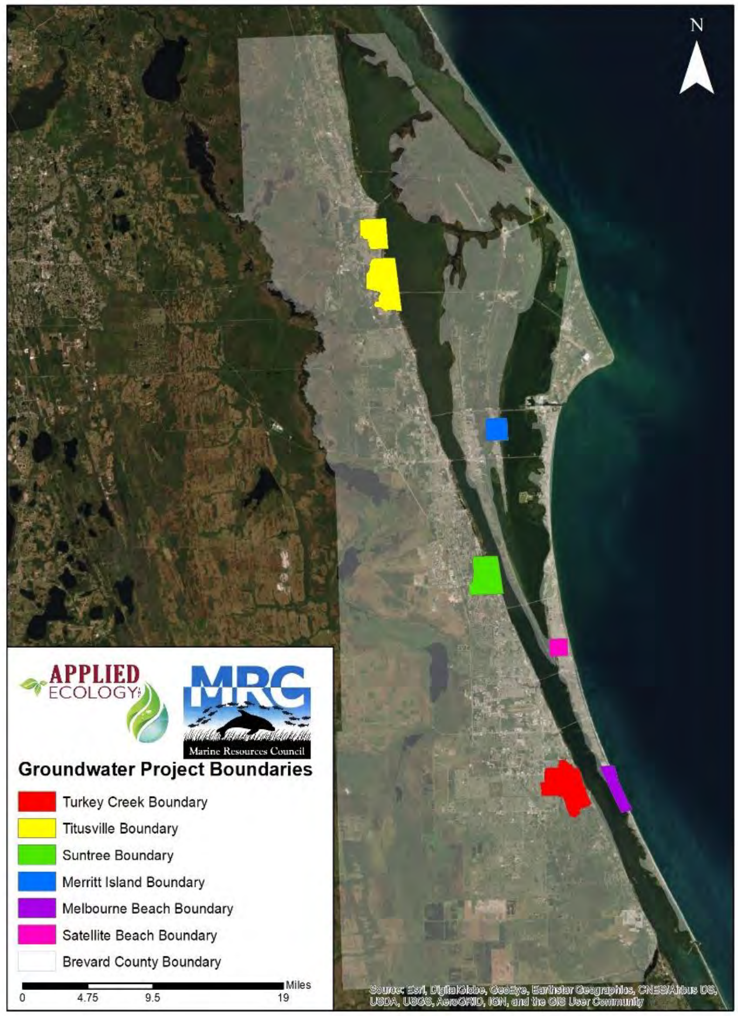

Figure 1: Study site map. The Melbourne Beach and Satellite Beach regions combined and are referred to as Beaches in this report.

Turkey Creek

The Turkey Creek study area consisted of three wells in a septic community, three wells in a sewer community, three wells in a reclaimed community, and two wells in a natural area. This and the Beaches regions are the only study areas where all four treatments are available for comparison. The beaches region includes Melbourne Beach and Satellite Beach communities. The Turkey Creek region is located south in Brevard County on the mainland. Turkey Creek is a main tributary that is fed by a large drainage canal that drains a large watershed that includes the City of Palm Bay. The Turkey Creek reclaimed community is served by the City of Palm Bay WWTF, which during the study, discharged reclaimed water for irrigation that had an annual average TN of 29.40 mg/L and TP of 1.40 mg/L.

The communities in Turkey Creek are located proximal to each other either directly on Turkey Creek or on drainage channels that meet Turkey Creek. Turkey Creek runs through a sand pine habitat with connecting wetlands. Sand pine communities are characterized by higher elevations, sandy soils, and sand pine trees. Elevations in the area change dramatically from well above sea level to at (or below) sea level as the region matrix transitions from sand pine to wetland. Soils in the Turkey Creek study area have the lowest average organic content of the study regions and high variability ranging from 0.36% to 2.9% organic content.

Beaches

The Beaches region included two wells in a natural area, three wells in a reclaimed community, and three wells in a septic community in Melbourne Beach and three wells in a sewer community in Satellite Beach. The Beaches reclaimed community received irrigation water from the South Beaches WWTF, which contained 9.30 mg/L of TN and 1.27 mg/L of TP on average during the study period.

The Beaches region communities are located on a sand bar that separates the Atlantic Ocean from the Indian River Lagoon estuary. As such, the regional elevation transcends from higher at the dune line along the Atlantic coastline on the east to what was once a mangrove fringe on the lagoon side. We expect soils to transition from primarily sand to more organic content as sites approach the lagoon. Our soil sampled confirms that the organic content is pretty consistent in these communities, with organic matter ranging from 2.3% to 3.9%.

Suntree

In the Suntree region, there are three wells in a sewer community, three wells in a septic community, and three wells in a reclaimed community. The reclaimed community is served by the South Central Regional WWTF, which during this study discharged reclaimed water with annual nutrient concentrations of 6.70 mg/L TN and 0.88 mg/L TP. Suntree is located on the mainland in an area that was once a pine flatwoods/wetland matrix leading to what would have likely been a mangrove fringe along the lagoon. The Suntree septic community is located directly on the Lagoon and is canaled to allow drainage and boat transportation. The other two communities were located west of Highway 1 on the downside of what was once the coastal ridge. Suntree soils had the highest average organic content of 5.5%, but the three communities differed in their soil organic content. The septic community average organic content was 2.6%, the reuse community 4%, and the sewer community had average organic content of 9.8%. These differences must be considered when examining denitrification potential.

Merritt Island

The Merritt Island region included two canal communities immediately on the Indian River Lagoon. Three wells were installed in one community that had sewer service, and three wells were installed in an adjacent septic community where plans are in place to connect to the sewer system. Merritt Island is an independent barrier island peninsula formed in the middle of the Indian River Lagoon. It is relatively low in elevation overall with wetlands and mangroves throughout. The two study communities are located on the east side of Merritt Island, where the lagoon was dredged to create canals and fingers of land for canal-front homes. The Merritt Island soils had the second-highest average organic content (4.75% organic matter).

Titusville

The Titusville study area consisted of a sewer community and a reclaimed community with three wells installed in each and a natural area located a short distance away from where two wells were installed. No septic communities were located in this region. The reclaimed community in Titusville received irrigation water from the Osprey North WWTF, which had reported annual concentrations of 17.90 mg/L of TN and 0.78 mg/L TP during the study period.

Titusville is located on the mainland and is the farthest north study area, located along the margin of a different climate zone. The two Titusville communities are located just west of the historic coastal ridge, and the natural area is located in a preserve that has a matrix of habitats ranging from sand pine to pine flatwoods to wetlands. The natural area wells are situated between the pine flatwoods and the sand pine. The soil organic content in this study area was pretty consistent across the three sites, with an average of 3.4% organic content.

Well and Sample Numbers

This project was initiated in Turkey Creek in 2017, and in some instances, all of the Turkey Creek data are used during specific analyses (30 months). However, for the aggregate data analysis and comparison with other regions, only 19 months of Turkey Creek data are used. In all other communities, 18 months of monthly sampling data was used for analysis. Table 1 summarizes the regions and the monitoring wells installed in each treatment type with the funding agency depicted by superscripts.

Table 1: Total number of wells within each region and treatment type. Each well was sampled monthly for all parameters, with the exception of total phosphorus which was sampled bimonthly.

Region	Total Wells	Septic Wells	Sewer Wells	Reclaimed Wells	Natural Wells	Sampling Events	Samples Collected
Melbourne Beach*	8	3	-	3	2	18	144
Merritt Island*^	6	3	3	-	-	18	108
Satellite Beach^	3	-	3	-	-	18	54
Suntree*^	9	3	3	3	-	18	162
Titusville*^	8	-	3	3	2	18	144
Turkey Creek*^	11	3	3	3	2	19/30	209/330
Totals*^†	45	12	15	12	6	-	942

*Funding source identified by the following superscripts: *Brevard County Legislative ^Save Our Indian River Lagoon Project Plan Respond Fund †FIT Legislative*

Methods

In each community, wells were installed to intercept the natural groundwater flow path between a majority of homes in the community and the receiving surface water body. An initial uncalibrated ArcNLET model was run for each study area to obtain a better understanding of groundwater movement and preliminary well locations were mapped. Thereafter, field visits were conducted to recruit homeowners, identify easements, and determine accessibility. Homeowner property access agreements were secured to allow access on private property for well installation and sampling.

A hydraulic geoprobe was used for well installation. Initially, soil cores were collected to characterize soil types and estimate groundwater depths. Thereafter, a hollow core was pushed to the total well depth and a 10’ long 1.5” diameter pre-screened and sand packed, slotted well casing was inserted to a depth that would fully encompass the top of the water table. A solid riser was added to connect the top of the well screen to the surface and sand (20/30) was used to back-fill the bore hole. The well was developed and flush-finished with a locking well cap and a 12” concrete pad. Well depths were dependent on depth to water and varied between 12’ and 30’ total depth. Well permits and completion logs for all wells were provided in previous quarterly reports.

Sampling

Groundwater Wells

A Quality Assurance Project Plan (QAPP) was submitted to and approved by FDEP before sampling initiated. All of the wells were sampled monthly and Total Phosphorus (TP) was sampled bimonthly at each well. Laboratory and field datasheets were provided throughout the project.

All monitoring wells were sampled in compliance with FDEP-SOP-001/01; FS2200 Groundwater Sampling. During sampling, field data were collected that included temperature, depth to water (DTW), pH, conductivity, dissolved oxygen, and turbidity and one blank and/or duplicate samples was collected for every 20 analyzed samples (5%) as required by the approved QAPP. For each well, a 250-mL aliquot was collected in a preserved sampling bottle that contained sulfuric acid to bring pH < 2 SU in preparation for analyses of ammonia (NH_3-N), Total Kjeldahl nitrogen (TKN), and nitrate/nitrite (NO_X-N). An aliquot of 120-mL was collected for fecal coliform analysis and an aliquot of 250-mL was collected for orthophosphate (ortho-p, PO_4^{3-}) analysis. Every other month, an additional 250-mL sample was collected in an acid-preserved bottle for total phosphorus (TP) analysis. All samples were placed on ice and taken directly to Pace Analytical Laboratory in Ormond Beach to meet the fecal coliform 6-hour hold time.

For analysis of δ15N and δ18O isotopes in NOx, an additional 30-mL aliquot was collected and filtered through a 0.1-micron filter and frozen in preparation for shipping. Samples that had a minimum nitrate concentration of 0.12 mg/L NOx were packed in dry ice and mailed to the University of California-Davis isotope lab for δ15N and δ18O in nitrate isotope analysis.

Push Points

Push points are a non-invasive, fast, and relatively inexpensive method to collect surficial groundwater samples. In this study, we use push points to better define and understand septic plume dynamics in yards where monitoring wells were installed. An additional objective was to evaluate the accuracy of push point data by comparing it to a permanent monitoring well sample taken at the same time.

Ninety-eight push point samples were collected (plus blanks and duplicates) in October 2019 and analyzed for the same groundwater parameters as the monitoring wells. Additionally, a soil sample was collected from each push point and sent to the Florida Institute of Technology (FIT) for sieve testing and analysis of organic content by Loss on Ignition. Push point groundwater nutrient data were then sorted into three bins of low, medium and high relative concentrations using Jenks and delineated on maps to create contours.

The push point apparatus initially developed by FIT includes a 6’ long solid steel push point borer that creates a ½ inch diameter hole and a hollow steel rod that is slotted the last 10” or so from the tip. The solid borer is pushed into the ground and removed to create a hole and then the hollow, slotted rod is dropped into the hole (Figure 2). Tubing is attached to the top of the hollow rod to draw groundwater using a peristaltic pump. The push points were purged until physical parameters stabilized and then sampled in accordance with the approved QAPP.

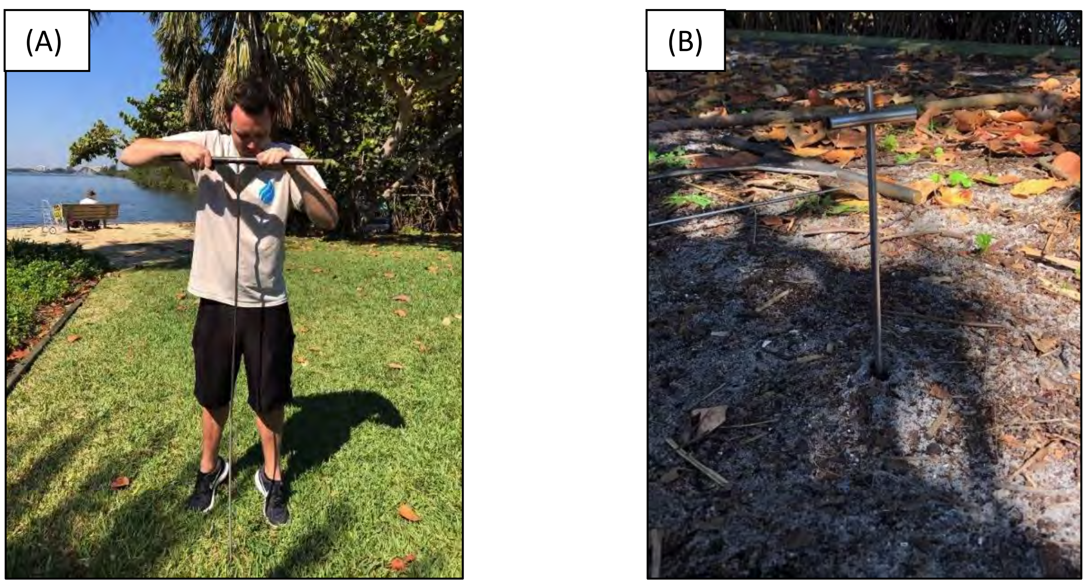

Figure 2: (A) Technician using a boring rod to create a hole for the push point sampling rod. (B) The push point sampling rod inserted into the bored hole ready for tubing and pump connection.

Septic sites were selected for the push point monitoring effort if depth to water was shallow enough to be reached (<6’ DTW). Five septic wells with particularly high TN and/or TP were selected for push point sampling: one in Merritt Island, two in Suntree, two in Turkey Creek, and a large region of the Melbourne Beach septic community. Fifteen (15) to twenty-six (26) push point samples were collected at each septic tank site and thirty-six push points samples were collected throughout the Melbourne Beach septic community. Sampling design was customized for each location based on location of the drain field, modeled plume flow direction and practical constraints (access, depth to water, etc.).

Soil

Soils were collected during monitoring well installation and push point sampling. When the hydraulic geoprobe drilled the wells, cores were produced and a composite sample from each well core was collected for soil characterization. During push point sampling, a 16” field auger was pushed to a total depth of about 12” to collect a composite soil sample at each of the push point locations. The soil samples bags were sent to FIT for soil characterization using a sieve method and % organic through Loss on Ignition.

Soil data were quality assured and averaged over the regions to better understand differences that can explain nutrient biogeochemical processes.

Data Analysis

Analysis of the data was performed using the Microsoft Excel extension XLStat (2020.1.1), Minitab 17 and Statistica 13.3. The normality of data was tested using the Shapiro-Wilk test, and homogeneity of variances using Levene’s and Brown-Forsythe methods. The Box-Cox transformation and other traditional data transformations types were applied but all were unable to transform the data into normal distributions for parametric analyses. Time series analysis such as autocorrelation function, plots, and the Box-Ljung statistic were also implemented by monitoring well and parameter to determine if significant autocorrelation was present between monthly sampling events.

Multivariate Analysis

Principle Component Analysis (PCA) was used to reduce the number of variables in the larger dataset to principle components. This is particularly useful when variables are highly correlated or redundant to understand which groups of variables most explain variations in the data. PCA results guide further analysis to focus on the variables that are most important and help organize interpretation. The PCA is a powerful multivariate analysis commonly used in water quality analysis as it is resistant to the collinearity and spatiotemporal influences typically present. The PCA transforms the original correlated variables into a set of uncorrelated factors referred to as Principal Components (PCs).

PCA was applied to the water quality dataset to identify which water quality variables were responsible for the greatest variation in the data. In addition, rainfall data was obtained from the nearest weather station maintained by the SJRWMD, NOAA, or USGS. The rainfall that occurred 24 hours, 3 days, and 30 days prior to the sampling day were summed and evaluated in an exploratory PCA analysis. As all three were found to be highly related, the 3-days of rainfall was selected for use in further analysis due to its stronger explanatory power. Thereafter, data in the PCA graphs were coded to see if regional and treatment type differences were apparent. The results from the PCA were used to guide additional analysis on the identified PCs and inform interpretation.

Non-Parametric and Descriptive Statistics

Statistical differences in water quality parameters between treatment types and regions were determined using the non-parametric Kruskal-Wallis comparison of independent samples and Steel-Dwass-Critchlow-Fligner (SDCF) multiple pairwise comparison tests. In cases, where only two groups were being compared (e.g. septic vs. sewer within Merritt Island) a Mann-Whitney U test, a non-parametric equivalent of the t-test for independent samples, was used.

Data interpretation and visualization tools like line graphs, bar graphs, maps and contours were used to clarify results and define plumes. Boxplots of the data distribution were created for the entire dataset and per treatment type and region to be used in the identification of trends and clusters between groups.

Intervention Strategy

An in-situ septic treatment product called BiOWiSH was distributed to nearly every resident located in the Turkey Creek septic tank community as part of the study's intervention strategy. BiOWiSH is described as an advanced enzyme technology that rapidly breaks down waste materials and reduces odor-causing compounds. It is a readily available, inexpensive, and easy-to-use product that is flushed down the toilet by homeowners quarterly. BiOWiSH advertises that it can reduce total nitrogen by 52.9%, chemical oxygen demand (COD) by 76.6%, and suspended solids by 89.2%. The Turkey Creek septic community was selected to receive the BiOWiSH because there was sufficient monthly data to constitute a pre- BiOWiSH condition. Turkey Creek sampling initiated in June 2017 as a pilot project to test the research methodology before implementing the County-wide project. The BiOWiSH intervention was initiated in the second quarter of this study and continued until the end of the sampling program (November 2019). A total of 76 homeowners (96%) within the community of interest agreed to actively participate in this intervention study and apply the product to their toilet every three months. The BiOWiSH product was delivered to homeowners quarterly for five quarters on October 22-23, 2018, January 23, 2019, April 25, 2019, July 24, 2019, and October 25, 2019. Post-interventional changes in the sampled parameters of the three Turkey Creek septic community wells, particularly nitrogen constituents, were examined for any potential changes in concentrations. More details on this study is included as Appendix A.Groundwater Modeling

As part of the Legislative funded study, "Groundwater Pollution, Engaging the Community in Solutions" project, hereafter called Groundwater Pollution Study, significant modeling effort took place. This included fate and transport of nitrate and ammonia, uncertainty modeling using Monte Carlo Simulations and a refinement of the baseflow component of a watershed loading model. A description of the rationale for model selection, methodology undertaken for each of these efforts, and a synthesis of these results are detailed in Appendix B. This section described a synthesis of the methodology used to implement the modeling component of the study.

ArcNLET Modeling Effort

Based on the literature review, it was decided that the ArcNLET model would be used to assess the potential contribution of OSTDS to the overall nitrate and ammonium loading of the study area. ArcNLET was selected based on the following rationale: 1) it is a relatively simple model that required limited input data but still incorporates key hydrogeological processes of groundwater flow and nutrient transport as well as spatial variability, 2) it is the model currently accepted by the FDEP to receive BMAP credit for removing or retrofitting septic tanks within a watershed with a Total Maximum Daily Load (TMDL), and 3) it can be calibrated with in-situ measured data for hydraulic head and nitrate and ammonium concentrations which are key to providing realistic results..

The ArcGIS-based Nitrogen Load Estimation Toolkit (ArcNLET) model was developed by the Florida Department of Environmental Protection (FDEP) and Florida State University (FSU) to model the fate and transport of nitrate and ammonia in surficial groundwater, originating from septic tanks (Rios, Ye, Wand, and Lee, 2011; Rios, Ye, Wang, Lee, Davis, and Hicks, 2013). The GIS-environment allows the import of various layers of information and the output concentrations of each nitrogen plume are mapped onto a raster layer, allowing representation of multiple plumes on a map.

A limitation of this model is that it assumes the concentration reaching the water table is the same as the initial concentration, which can result in over or underestimations of the mass loading from the system. Additional limitations of this model include: 1) treating the water table as a subdued replica of topography and representing groundwater flow in 2-D and a steady-state and 2) the need for an empirical or calibrated value for the decay coefficient. More detailed information about model uses, limitations, as well as inputs and outputs from this study can be found in Appendix B.

During the initial task of the Groundwater Pollution Study, a series of preliminary groundwater model runs based on a modification of ArcNLET. The preliminary model runs were basin specific and only used the historically available data (i.e., water levels and surficial water quality data) for calibration. These initial outputs were also used to guide well installation efforts to enhance the value of the collected groundwater quality data to represent the community of interest and calibrate these initial ArcNLET model runs. Once site-specific data for the 18 sampling events were collected, median concentration data of nitrate and ammonia were incorporated, along with more precise septic tank quantity and location data, into ArcNLET to refine and individually calibrate nitrate and ammonium loads for the Merritt Island, Suntree, Melbourne Beach, and Turkey Creek study areas.

Uncertainty Modeling

Critical driving factors of the nitrate transport were evaluated as part of the Uncertainty Quantification effort to better understand the magnitude of uncertainty inherent to nitrate load estimates developed for management and planning purposes. To accomplish the quantification the Monte Carlo (MC) Simulation for Uncertainty Quantification function of

ArcNLET was used. The following single parameters were explored using the MC Simulation: Smoothing Factor, Hydraulic conductivity, Porosity, and septic tank source nitrate concentration. The simulation was applied to two study areas of interest, one representative of Barrier Island conditions (Melbourne Beach) and another mainland conditions (Suntree). Results were synthetized to describe the variability of the estimated loadings based on randomized runs of parameters of interest, highlighting the impact of each the environmental variables on the predicting nitrate from ArcNLET based on Brevard County conditions. For more details on methods and results from the uncertainty modeling, please review Appendix B.

Refinement of the Spatial Watershed Iterative Loading Model

As part of the Groundwater Pollution Study, we explored the potential impact of the field collected groundwater quality data as input information for the baseflow component of a regional watershed loading model, the Spatial Watershed Loading Model (SWIL).

SWIL is a custom ESRI ArcGIS toolset, originally designed to provide a continuous monthly simulation of runoff (surface and baseflows) over a 20-year period, yielding a more robust representation of pollutant loadings and freshwater volumes in the IRL. The goal of the SWIL model development was to provide a GIS-based model that can be adaptive to changes in input and can batch complex processes through several months or years on demand. SWIL aims to provide both spatially and temporally fine-scale volumes and loads (TP and TN), allowing input data to be related to water quality parameters. A limitation to the model is that the TN and TP concentrations used in the baseflow component of the SWIL model are uniform (independent of land use) and static (not variable through time) due to the limited availability of groundwater concentration data during the SWIL's development in 2015. The SOIRL Groundwater Study along with Brevard County's Legislative Study has provided, for the first time, water quality collected at a groundwater monitoring network of 45 wells located throughout Brevard County. Median water quality data from this network of wells was synthetized from the first 18 months of collection and used in the SWIL baseflow component of the model in lieu of the original one size fits all concentration values. Comparisons of results between the original model and refined model using recent baseflow concentration data are described Appendix B.

Results

Data from 18 months of continuous sampling in Merritt Island, Suntree, Titusville, and the Beaches and nineteen months of sampling in Turkey Creek were analyzed to examine differences in treatment types and regions. The total dataset of 30 months of sampling in Turkey Creek was included in the isotope analysis to inform sources.

Soils

Soil characterization is provided and organized by region to provide a glimpse into differences in soils that may influence the nitrogen cycle. There is more efficient mineralization of ammonia in soils that have higher organic content. In the absence of carbon, ammonia is more likely to stay in solution or volatilize. Nitrification of ammonia (NH_3) to nitrate/nitrite (NO_X) requires oxygen, which is slightly higher in courser sediments that have larger pore spaces between them. Denitrification occurs in the absence of oxygen and tightly packed soils can become a sink for denitrification, with highly enriched NO_X trapped between particles. Results are raw soil data collected during this study are included in Appendix C.

Of our five study regions, Turkey Creek clearly differs from the others because it has more course soil containing less organic content (1.4%) and carbonate (0.93%) (Figure 3). Suntree has the highest organic content (5.5%), followed closely by Merritt Island (4.75%), Titusville (3.41%), and Melbourne Beach (3.3%).

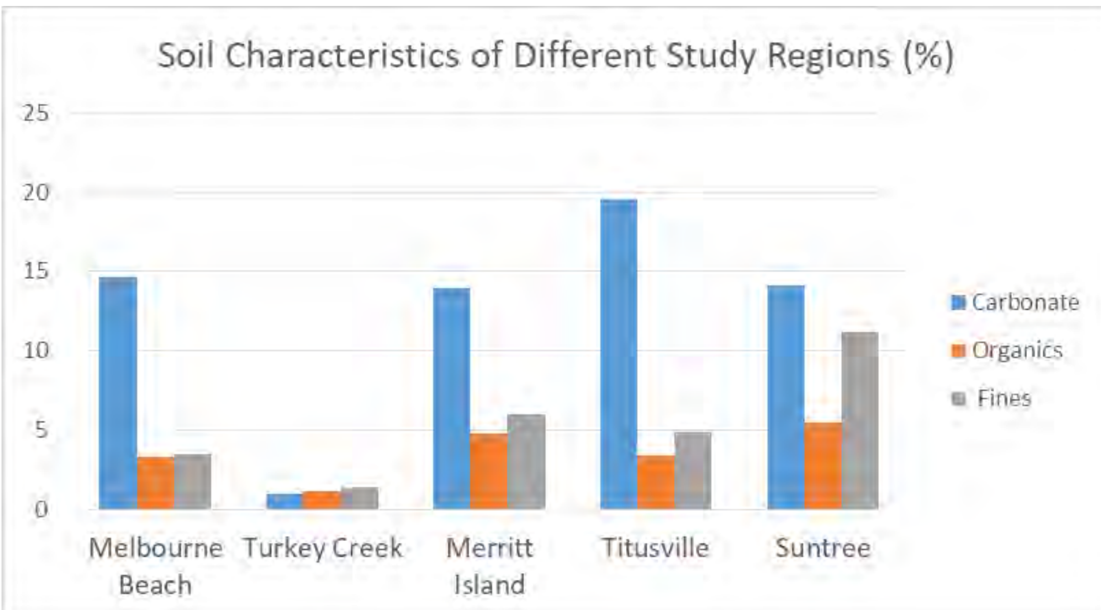

Figure 3: Soil characteristics in the five study regions

Intervention Strategy Results

With only five deliveries in one community and uncertainties related to seasonal variabilities, it is impossible to draw any definite conclusions about the effectiveness of the proprietary product in improving groundwater quality. Based on preliminary data, no consistent, across the board reduction in TN and/or TP concentrations are apparent after the BiOWiSH product was delivered. Additional results are described in Appendix A.

Comparing Treatment Types across the County

The complete groundwater dataset was examined using descriptive statistics, non-parametric comparison tests, and principal component analysis (PCA). Median values were used as the measure of central tendency for analysis due to the heavily skewed, non-normal data distribution. Median concentrations of all analytical parameters were compared using the Kruskal-Wallis or Mann-Whitney U test, depending on number of treatment types. A PCA was applied to identify which analytes were representative of water quality variation between treatments and locations. The PCA also identified outliers and spatiotemporal influences.

Complete Countywide descriptive statistics (mean, median, percentiles, *etc.*) for each treatment type are provided in Appendix D.

Summary of Findings

Principal Components Analysis found two major factors that explain most of the variance in groundwater nutrient concentrations. Bivariate analyses found significant differences in nutrient concentrations between septic, sewer, reclaimed and natural area wells. Most of the samples were ordinated in space by differences in concentrations in either organic nitrogen (TKN and NH_3) and ortho-p (PO_4^{3-}) or inorganic nitrogen (NO_X). The first principal component suggests that high organic nitrogen and PO_4^{3-}concentrations are key drivers in differentiating the data and typically vary together. Along the second principal component inorganic nitrogen appears to be the major driver of the variability found between samples in terms of groundwater contamination. Total phosphorus was a redundant variable to $PO_4^{3-}$and was removed from the PCA due its lack of additional explanatory power when examining differences between samples.

The septic communities had significantly higher organic nitrogen (NH_3 and TKN) and phosphorus concentrations (TP and PO_4^{3-}). Reclaimed communities had significantly higher inorganic nitrogen concentrations (NO_X). However, there was no significant difference in total nitrogen (TN) between septic and reclaimed communities although both were significantly higher than sewer and natural areas. It is important to note that TN is never the key variable with greatest explanatory power in any of the first three principal components, since several of the different sample types do have overlapping TN concentrations.

Fecal coliform concentrations between treatment types were found to be significantly different (Kruskal-Wallis p<0.00001), even though this data set is highly variable. All three communities had significantly higher fecal coliforms than the natural area. There were no significant differences in fecal coliform concentrations between the three treatments (septic, sewer, and reclaimed), although the difference between septic and sewer communities was marginally significant.

Principal Components Analysis (PCA)

The PCA of the measured nutrient concentrations and the 3-day rainfall data identified four major components that explained 98% of the variation in groundwater quality (Table 2). The first two factors that explain 71% of the variance are almost equally split with organic N (NH_3 and TKN) and ortho-P concentrations loading on Principal Component One (PC1, 38%) and inorganic N (NOx) loading on Principal Component Two (PC2, 33%). This suggests that there are two scenarios that should be explored for further data analysis. One where we expect to see high organic nitrogen and PO_4^{3-} and another scenario where we expect inorganic nitrogen to be the major contributor to groundwater nutrient contamination. The equal loading of TN on both the inorganic and organic N components (PC1 and PC2) suggests there is no difference in TN in these two scenarios.

Table 2: Loadings of six water quality variables on the first four PCs for county-wide groundwater samples.

Analyte	PC1	PC2	PC3	PC4
NH_3	0.77	-0.59	0.06	-0.16
TKN	0.73	-0.62	0.12	-0.19
NO_X	0.45	0.88	0.04	-0.14
TN	0.69	0.69	0.08	-0.20
PO_4^{3-}	0.67	0.06	-0.15	0.72
3 Day Rainfall Sum	-0.11	0.03	0.98	0.17
Variability (%)	37.7	33.3	16.8	11.3
Cumulative %	37.7	71.0	87.8	99.0

Looking at the plotted graphs of the loading PCs discriminated by treatment type (Figure 4) and by study region (Figure 5), there are some interesting loading differences in treatment types, but not much difference in study region. In other words, while clustering of groups can be seen for some treatment types, samples from all regions are well mixed in space and difficult to discern. This suggests that treatment types differ, but that if you don't take treatment type into effect, the regions do not differ in how the data are distributed across the components.

The sewer and natural community samples tend to load more on the organic component (PC1), and the reclaimed community samples tends to load on the inorganic component (PC2). The septic samples appear to be scattered across all components and represent a broader variability. This variability within the septic dataset will be further explored when examining regional differenced within treatment types (section below). It is interesting that component 3 (PC3) is an independent factor influenced almost entirely by 3-day rainfall data, the only independent variable used in this analysis. The fact that this variable loaded on a component that only explains 17% of variance suggests some minor effect of rainfall on the water quality differences between samples examined in this study. In the aggregate dataset, this influence is positive for nitrogen and negative for PO_4^{3-}. This means higher 3-day rainfall amounts would result in increased nitrogen (especially organic nitrogen) and reduced PO_4^{3-}.

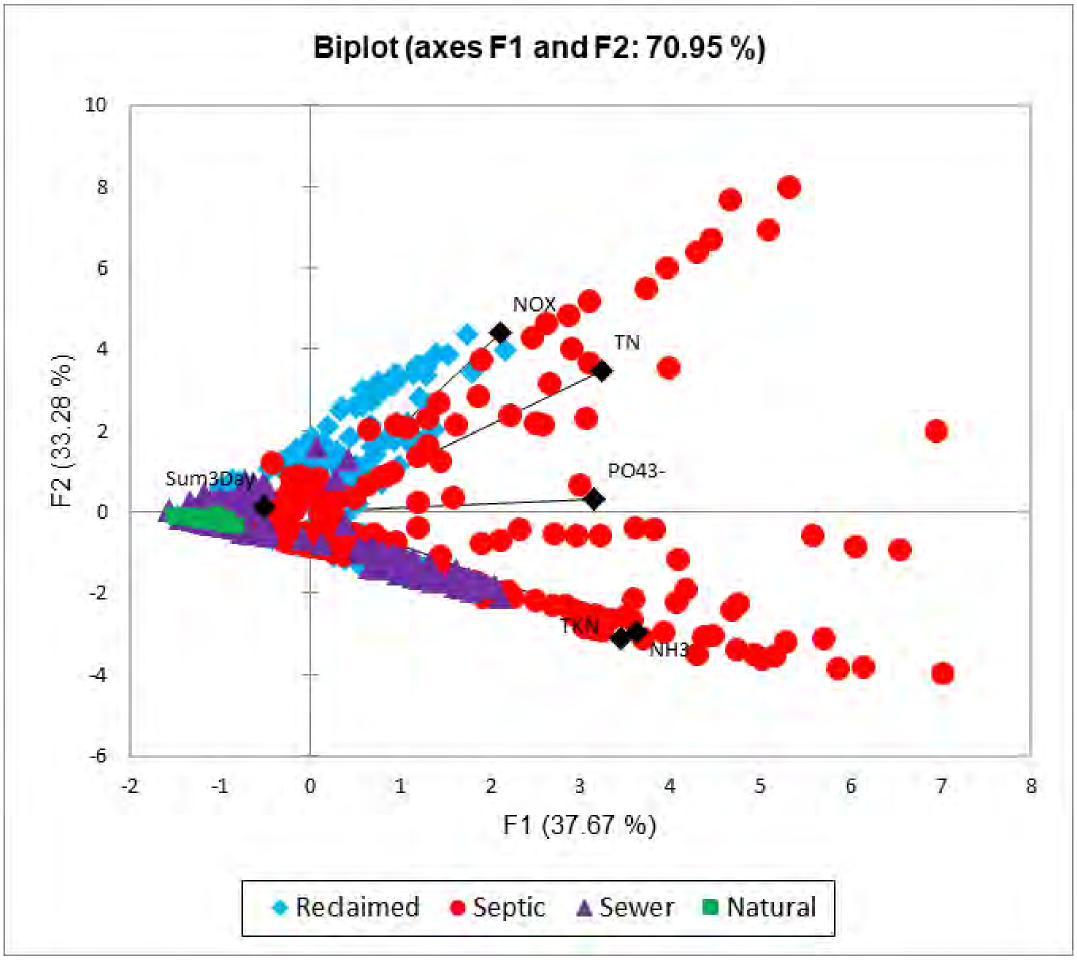

Figure 4: Coordinates of the PCs based on the treatment type. The color of the dots denotes its classification as shown in the figure legend.

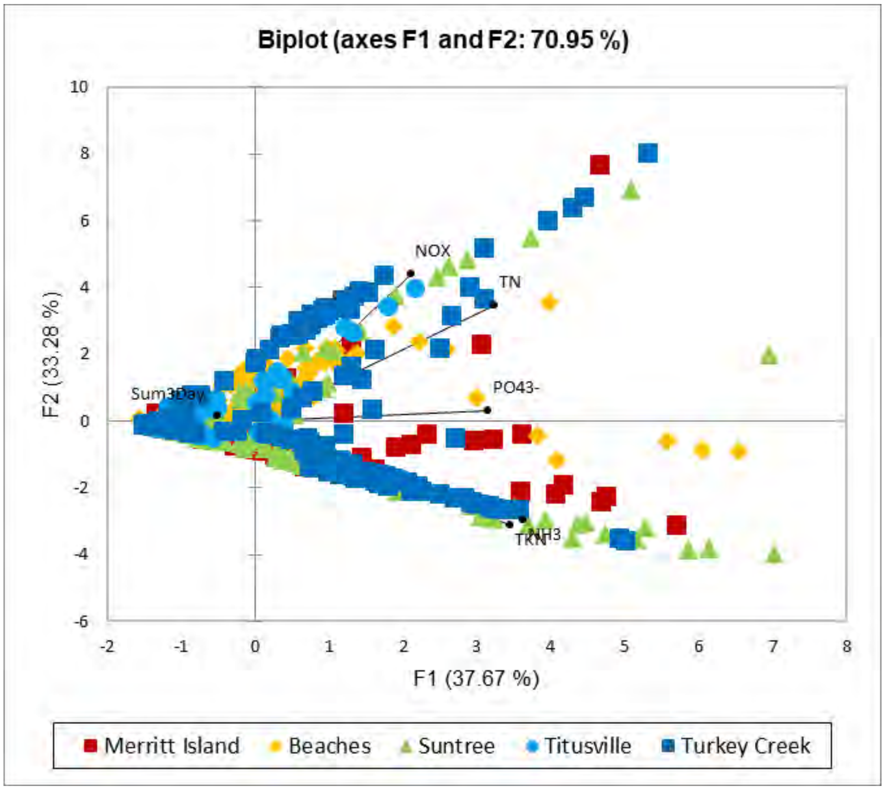

Figure 5: Coordinates of the PCs based on the study region. The color of the dots denotes its classification as shown in the figure legend.

Non-Parametric Bivariate and Descriptive Analysis

Summary test results are included for all analyzed parameters (nutrient and bacteriological analytes) and comparison among treatments. Table 3 summarizes the findings with subscripts indicating statistically significant differences. The same letter subscript indicates values were not significantly different. Different letters indicate statistically significant differences.

Table 3: Differences in nutrient and bacteria median concentrations between treatment types.

Analyte	Septic	Sewer	Reclaimed	Natural
***NH_3 (mg/L)**	**0.560**^{a}	0.160^b	0.035^c	0.065^c
***NO_X (mg/L)**	0.200^a	0.042^b	**2.800**^{c}	0.028^d
***TKN (mg/L)**	**1.000**^{a}	0.720^b	0.620^c	0.305^d
***TN (mg/L)**	2.600^a	1.200^b	**4.200**^{a}	0.350^c
***PO_4^{3-} (mg/L)**	**0.480**^{a}	0.089^b	0.052^c	0.073^c
***TP (mg/L)**	**0.570**^{a}	0.110^b	0.089^b	0.120^b
***Fecal (CFUs/100mL)**	1.000^{a,b}	1.000^b	1.000^a	1.000^c

**Significantly different median with p<0.00001 using Kruskal-Wallis*
Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate row median significant differences at p<0.05. If significant differences were found, the highest value is in bold.

Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate row median significant differences at p<0.05. If significant differences were found, the highest value is in bold.

NH_3 concentrations are significantly higher in the septic communities than in the sewer, reclaimed, and natural areas (p<0.00001) and significantly higher (p<0.00001) in the sewer communities than reclaimed and natural areas. The reclaimed communities had the lowest NH_3 concentration, although it did not differ significantly from the natural area.

The NO_X concentration in reclaimed communities was significantly higher than in the septic, sewer, and natural treatment types (p<0.00001). In fact, the NO_X median was more than10 times higher in the reclaimed communities than the other treatments. The NO_X concentration in the other three treatment types significantly differed with each other with septic higher than the sewer, which was higher than natural (p<0.00001).

The septic treatment had significantly higher TKN concentrations than the other treatments (p<0.00001). The sewer communities had significantly higher TKN concentrations than the reclaimed and natural areas (p<0.00001). The reclaimed treatment TKN concentrations were significantly higher than the natural areas (p=0.001).

Although organic nitrogen concentrations were highest in the septic community wells, there was no significant difference in calculated TN in the septic and reclaimed communities. Both were significantly higher than the sewer communities, which was significantly higher than the natural areas.

The septic treatment had significantly higher phosphorus concentrations (PO_4^{3-} and TP) than the other treatment types (p<0.00001). There was not a significant difference in TP concentrations among sewer, reclaimed, and natural areas, but sewer communities had significantly higher ortho-p (PO_4^{3-}) than the reclaimed and natural communities. There was no significant difference in ortho-p between reclaimed and natural areas.

Fecal coliform concentrations between treatment types were found to be significantly different (Kruskal-Wallis p<0.00001), even though this data set is highly variable. All three communities (septic, sewer, reclaimed) had significantly higher fecal coliforms than the natural area. However, there were no significant differences in fecal coliform concentrations between the three treatments (septic, sewer, and reclaimed), although the difference between septic and sewer communities was marginally significant.

Because the median used for statistical analysis is not very descriptive, means are provided below for general comparisons (Table 4). Septic communities had the highest fecal coliform counts (μ = 13.47 CFU/100 ml), followed by reclaimed communities (μ =12.16 CFU/100 ml) sewer communities (μ = 12.08 CFU/100 ml) and natural areas (μ=2.69 CFU/100 ml).

Table 4: Mean fecal coliform counts by treatment type.

Treatment Type	Mean fecal coliform count (CFU/100 ml)
Natural/Control	2.69
Reclaimed	12.16
Septic	**13.47**
Sewer	12.08

Total Nitrogen (TN)

Total Nitrogen (TN) is a calculation of organic and inorganic nitrogen that is used for loading models and general comparisons. In our dataset, TN is calculated by adding TKN and NO_X reported in mg/L. Summary statistics for the aggregate dataset are provided as boxplots (Figure 6), as a graph of mean concentrations over time by treatment type (Figure 7), and in Appendix D.

In Figure 6, we see that the mean and median TN concentration values are highest for the reclaimed treatment, closely followed by those for the septic treatment. Mean TN values for the sewer treatment type are typically less than half of those measured for both the reclaimed and septic communities, while natural TN mean concentrations for the control wells are consistently below the means for any of the other treatments (mean of 0.38 mg/L versus means of 1.92 mg/L– 6.05 mg/L). Turkey Creek region had highest median TN, followed by Suntree, Merritt Island, Beaches, and Titusville.

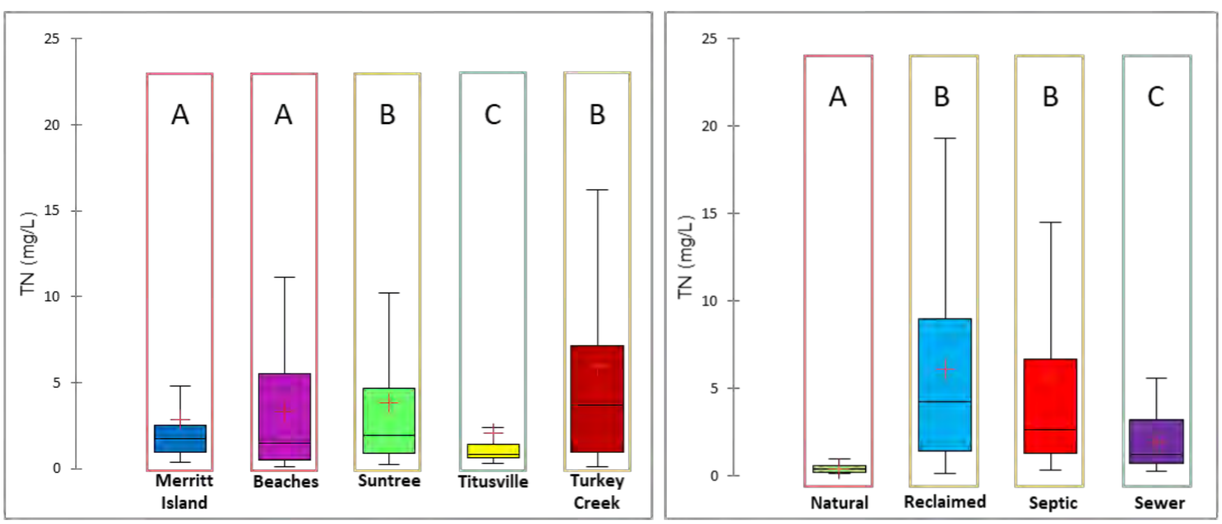

Figure 6: Boxplot of total nitrogen (TN) by region and treatment type with whiskers, 1st to 3rd, and median. The red cross represents the mean concentration. Significant differences between treatments indicated by different colors.

Figure 7 shows the sewer and natural treatments have been relatively stable throughout the monitoring period, with a slight increase in sewer concentrations during the dry season (March 2019 to June 2019) followed by an increase during the wet season (June 2019 to August 2019). In contrast, the septic and reclaimed treatments show some distinct seasonal patterns. In the first and second quarters, both the reclaimed and septic TN concentrations present a similar decreasing trend. Between November 2018 and February 2019, the patterns of the TN concentrations for septic and sewer diverge dramatically.

While the mean TN concentrations for reclaimed treatment sharply increased in December and January, the opposite trend is noticeable for the septic treatment. In February 2019, a reversal of these trends was apparent with sharp increases in TN concentrations for the septic wells, and a decrease in TN concentrations for the reclaimed wells. In March 2019, both the reclaimed and septic communities decrease in mean TN concentration at a similar rate, diverge in April 2019 with the reclaimed communities stabilizing the concentrations, while the septic communities continue to exhibit decreasing TN concentrations. By May 2019, both septic and reclaimed have similar mean TN concentrations around 5 mg/L (< 0.09 mg/L difference between the two means).

After the initial increase in concentration at the beginning of the 2019 wet season (May/June 2019), there was an overall decrease in the sewer, septic, and reclaimed treatments from June to August 2019. In reclaimed and sewer treatments, TN concentrations remained stable from September – November 2019, but the TN in the septic communities increased dramatically (> 2 mg/L). This resulted in septic TN concentrations that were higher than any other treatment type that quarter. Groundwater total nitrogen concentrations in the natural areas varied little and are well below any of the other treatments' measured concentration data.

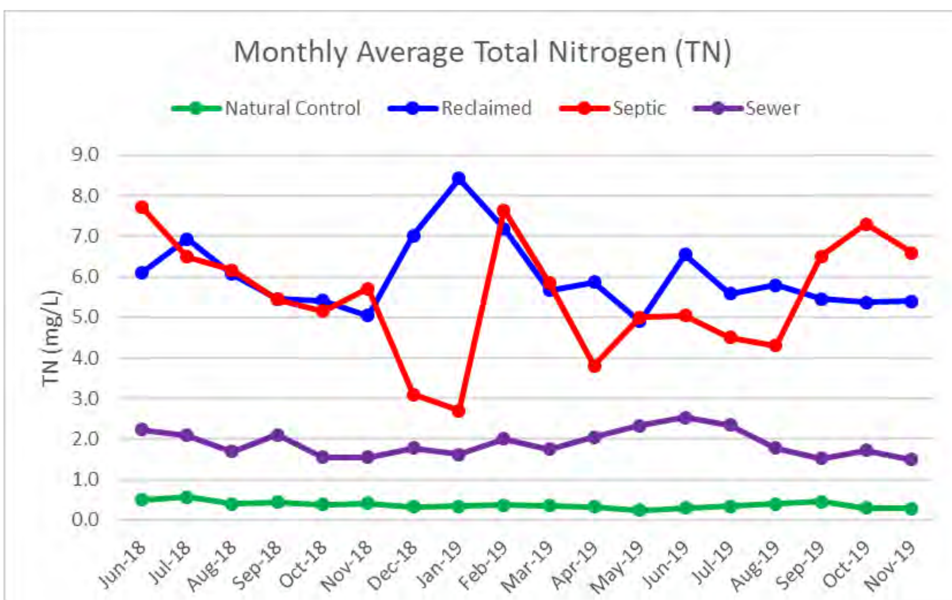

Figure 7: Average total nitrogen (TN) in each treatment over time.

As TN is a calculation of all forms of nitrogen measured, it is useful to determine the extent to which TN is dominated by inorganic or organic nitrogen (Table 5). Total Nitrogen is dominated by inorganic nitrogen in the reclaimed treatment and by organic nitrogen for the remaining three treatments.

Table 5: Mean percent contributions of nitrogen constituents for TN by treatment type. Values in bold indicate the predominant nitrogen constituent for each treatment type.

Treatment Type	Mean NO_X%	Mean NH_3 %	Mean TKN %
Natural/Control	16	27	**92**
Reclaimed	**58**	12	43
Septic	37	43	**64**
Sewer	25	34	**77**

Nitrate/Nitrite (NO_X)

NO_X summary statistics for the 18 months of sampling are provided as boxplots (Figure 8), a graph of mean concentrations by treatment type (Figure 9), and in Appendix D. The highest mean NO_X concentration was observed for the reclaimed treatment (5.11 mg/L), followed by the mean for the septic communities (3.48 mg/L). Both the sewer and control treatments have means that represent a small fraction of those described for the reclaimed and septic, with lowest values (0.05 mg/L) reported for the natural wells. Median values for NO_X in the sewer and natural communities are similar to each other (0.042 and 0.028 mg/L, respectively) and near the laboratory detection limit (0.025 mg/L).

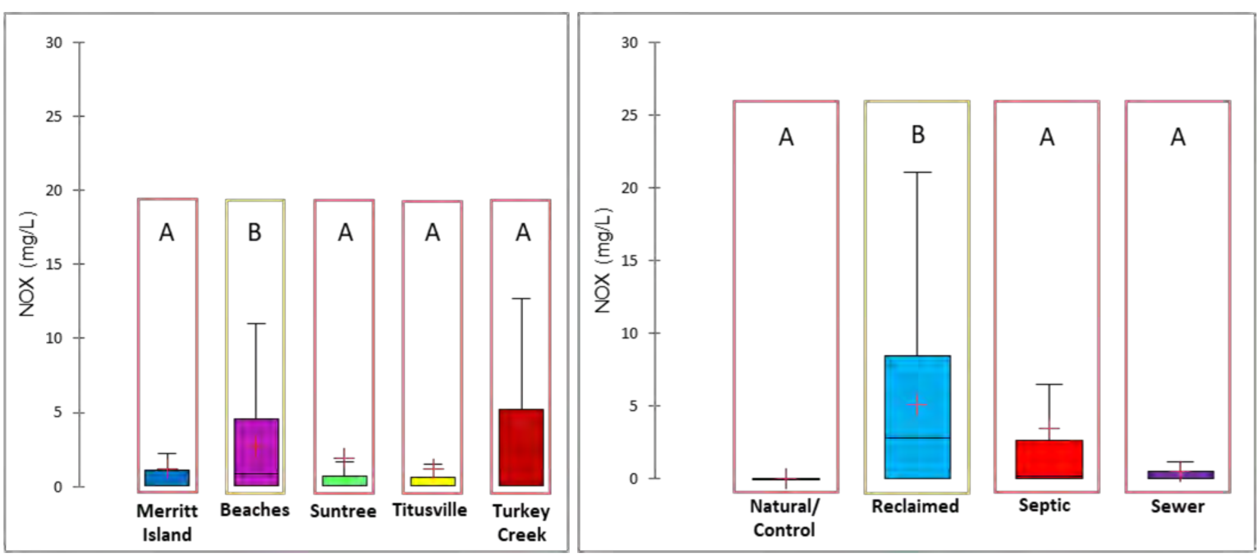

Figure 8: Boxplot of nitrate and nitrite (NO_X) with whiskers, 1st to 3rd, and median by region and treatment type. The red cross represents the mean concentration. SDCF pairwise groupings are shown by color, any categories that share a color are not significantly different from one another.

While monthly mean NO_X concentrations are relatively stable in the natural areas and fluctuate slightly sewer areas, the variability in the septic and reclaimed areas is dramatic (Figure 9). Throughout the wet season from June to September 2018, NO_X concentrations gradually declined across the board. However, as the dry season progressed, the mean concentration data for septic and reclaimed areas began to diverge. Mean septic NO_X concentrations decreased substantially from November 2018 through January 2019, followed by a rapid increase of more than 4 mg/L in February to the highest concentrations reported in the monitoring period (6.15 mg/L) for this treatment. Previously, NO_X concentrations within the septic treatment had been demonstrating a decreasing trend, aside from a brief increase in concentration from April 2019 to May 2019. However, another sharp increase in concentrations (> 2 mg/L) during the transition from the fifth to sixth quarter, making concentrations more comparable to those measured within the reclaimed treatment.

In contrast, mean NO_X concentrations within the reclaimed treatment increased drastically from November 2018 through January 2019, followed by a steep decrease in February and March 2019. After March 2019, mean NO_X concentrations for the reclaimed wells have fluctuated around 5.0 mg/L with no discernable unidirectional trend. While the behavior of the NO_X concentrations is different for the septic and reclaimed communities, inter-monthly trends were similar from July 2018 to October 2018 and mean concentration values were very similar in February 2019, May 2019, and September 2019. While the reclaimed treatment appears to have a similar pattern in the summer 2018 and 2019 (peaks in June/July with the first high rainfall event of the wet season, subsequent decreases), the septic treatment appears to have very different trends between the two monitored wet seasons.

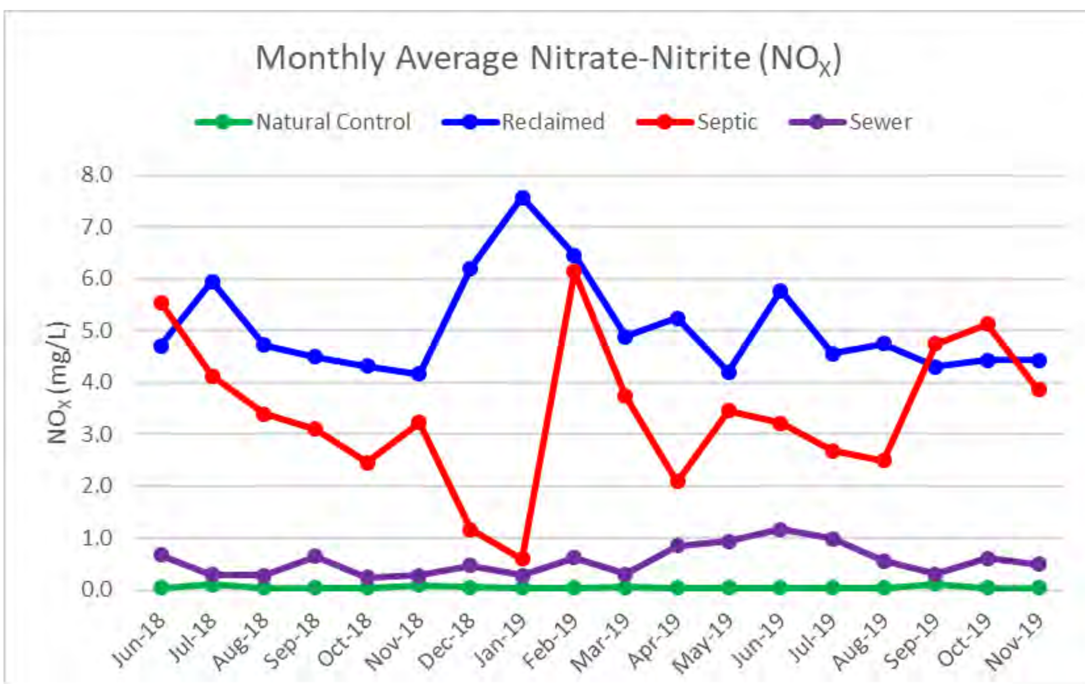

Figure 9: Average monthly NO_X concentrations by treatment type.

Total Kjeldahl Nitrogen (TKN)

TKN is a measure of the concentration of organic nitrogen and NH_3. Organic nitrogen is found in the cells of all living things and is a component of peptides, proteins, and amino acids. During mineralization, bacteria convert organic nitrogen to NH_3, which can then be nitrified to NO_X by specific nitrifying bacteria. Differences in the relative proportions of NH_3 to organic nitrogen can help better understand nitrogen dynamics and sources. Summary statistics for sampling are provided as boxplots (Figure 10), a graph of mean concentrations by treatment type (Figure 11), and in Appendix D. The mean TKN concentration in the septic treatment (2.11 mg/L) is consistently higher than the means for the other treatment types (0.35-1.39 mg/L,). Lowest TKN means were observed for the natural treatment (0.35 mg/L), followed by those of reclaimed and sewer treatments (0.97-1.39 mg/L, respectively).

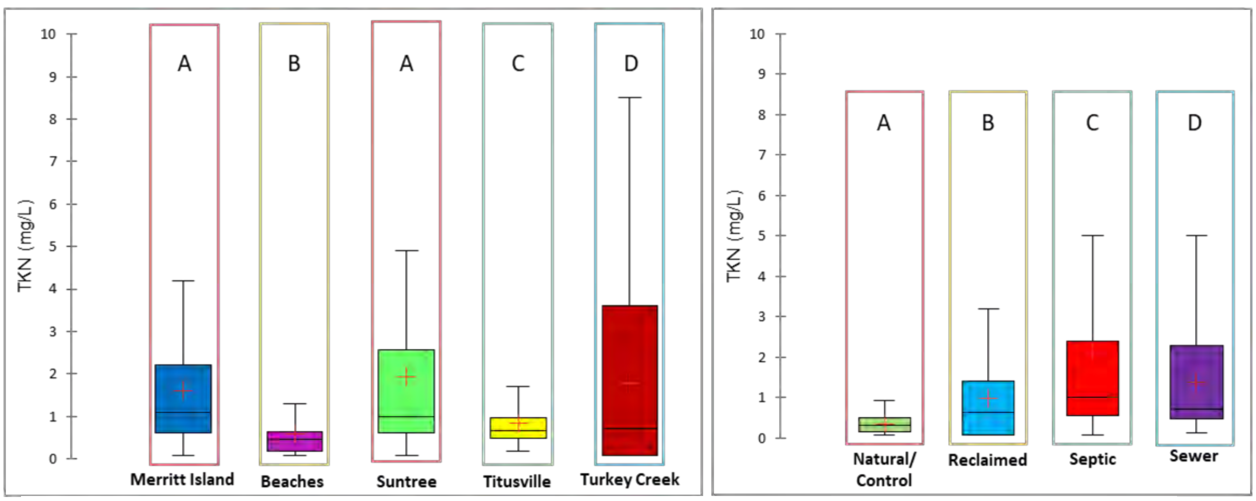

Figure 10: Boxplot of total Kjeldahl nitrogen (TKN) with whiskers, 1st to 3rd, and median by region and treatment type. The red cross represents the mean concentration. SDCF pairwise groupings are shown by color, any categories that share a color are not significantly different from one another.

Figure 11 demonstrates inter-monthly variability in measured TKN concentrations was highest for the septic treatment (data range >1.4 mg/L). Contrastingly, a more subdued, decreasing trend is observed for the sewer and reclaimed treatment types (data range < 0.7 mg/L) and barely noticeable for the control sites (data range < 0.25 mg/L). This indicates that the responses to external factors that might impact the TKN concentration data (e.g., preceding rainfall) are likely differ depending on the treatment. Overall, regardless of the inter-monthly variability and slope of change, all four treatments appear to show a decreasing concentration of TKN in the first 12 months of sampling. However, during the fifth and sixth quarters, varying trends in concentrations are observed across the treatments with the progression of the wet season. Following an initial increase in mean TKN from May 2019 to June 2019, the septic communities show relatively stable concentrations through the end of the fifth quarter (August 2019). Similar to the trend in NH_3, there is a slight decrease in concentrations for the sewer communities and slight increases in concentrations observed in the reclaimed and natural communities. During the sixth quarter, slight decreasing trends were observed at the sewer, reclaimed, and natural communities, while a steep increase in concentrations have been observed in the septic treatment (~ 1 mg/L increase from September to November 2019). This increase in TKN for the septic communities is clearly driven by the measured increase in ammonia described above.

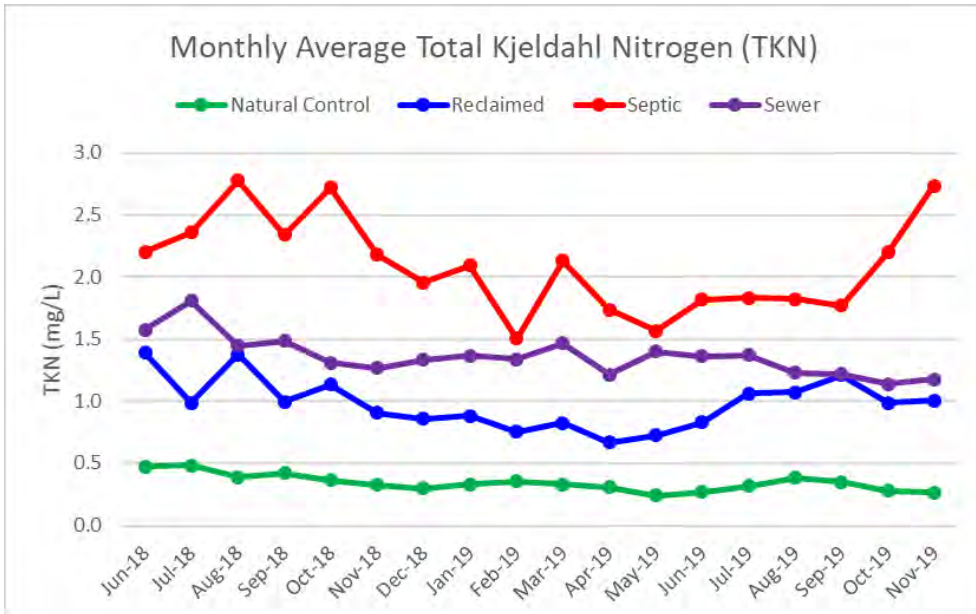

Figure 11: Average monthly TKN concentrations by treatment type.

Differences in the amount of ammonia in the TKN samples are evident (Table 6). The septic community had the largest percentage of ammonia making up TKN, (58%). Since the majority of the TKN in this treatment type is composed of NH_3, it is not surprising that the overall TKN trends observed in the septic treatment are similar to those observed for the NH_3 (Figure 13). The reclaimed, sewer, and natural treatments had a majority of TKN composed of organic nitrogen, with NH_3 only making up 29% of the reclaimed TKN concentration and 36% of the sewer treatment TKN. Similar to the reclaimed treatment, the natural area proportion of ammonia was 29%.

Table 6: Mean percent contribution of NH_3 for TKN by treatment type. Bold values indicate if the composition is greater than 50%.

Treatment Type	Mean % NH_3 Contribution
Natural/Control	29%
Reclaimed	29%
Septic	**58%**
Sewer	36%

Ammonia (NH_3)

NH_3 summary statistics are provided as boxplots (Figure 12), a time-series graph of mean concentrations by treatment type (Figure 13), and in Appendix D. Average NH_3 concentrations varied among the four treatment types, with the highest concentrations consistently observed for the septic treatment, followed by the sewer, the reclaimed, and lowest for the natural wells. Once the data from all 12 septic wells were aggregated, the mean NH_3 concentration was more than twice the mean from the sewer treatment wells. The septic treatment still presents the highest variability throughout the monitoring period, with mean concentrations fluctuating from 1.16 and 2.57 mg/L and individual well concentrations ranging between non-detect to 9.7 mg/L. Sewer mean concentrations also demonstrated some variability throughout the eighteen months of sampling (ranging between 0.48 and 1.01 mg/L), although this is a result of the unusually low mean concentration measured in the February 2019 sampling event. Mean NH_3 values are similar for both the reclaimed communities and natural sites, with slightly higher concentrations at the reclaimed treatment.

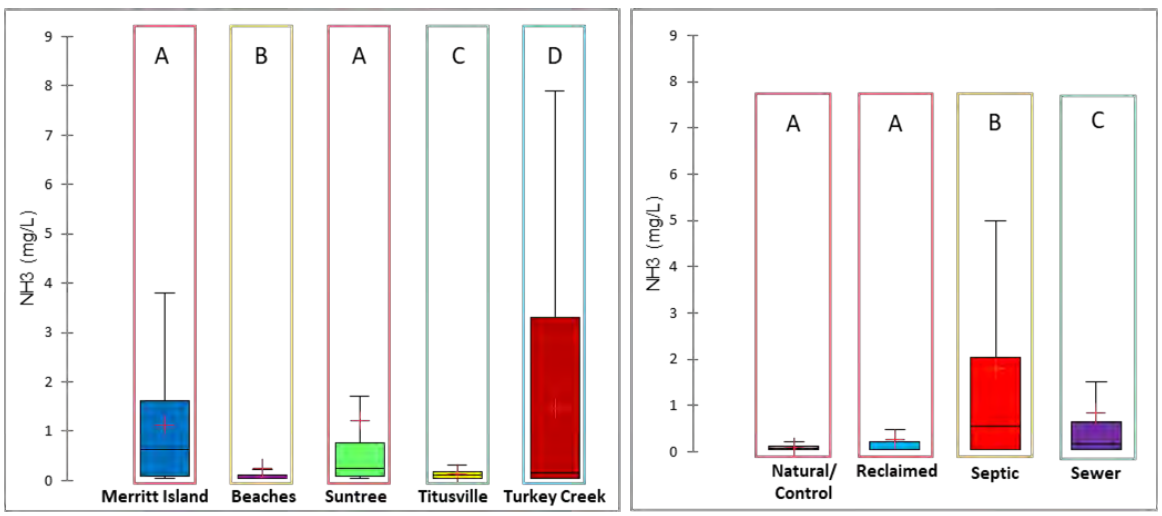

Figure 12: Boxplot of ammonia (NH_3) with whiskers, 1st to 3rd, and median by region and treatment type. The red cross represents the mean concentration. SDCF pairwise groupings are shown by color, any categories that share a color are not significantly different from one another.

NH_3 concentrations varied greatly in the reclaimed treatment wells demonstrated monthly variability until January 2019 with subsequent relatively low ammonia concentrations values (at or near MDL) for the fourth quarter, and slight increases of ammonia concentration data above the MDL are visible in the fifth and sixth quarters (Figure 13). As expected, aggregated means for the natural wells were once again stable and consistently near or below minimum detection limits (MDL). Decreases in concentrations of ammonia are visible in February 2019 for all treatment types with detectable ammonia (septic, sewer, and reclaimed wells), whereas the following month (March 2019) both sewer and septic wells increased in concentration and reclaimed remained relatively low. The septic wells have had consistently higher mean NH_3

concentrations than all other treatment types throughout the study period, with the sixth quarter demonstrating a steep increasing trend that resulted in the highest mean NH_3 concentration of the entire study in November 2019 (2.57 mg/L).

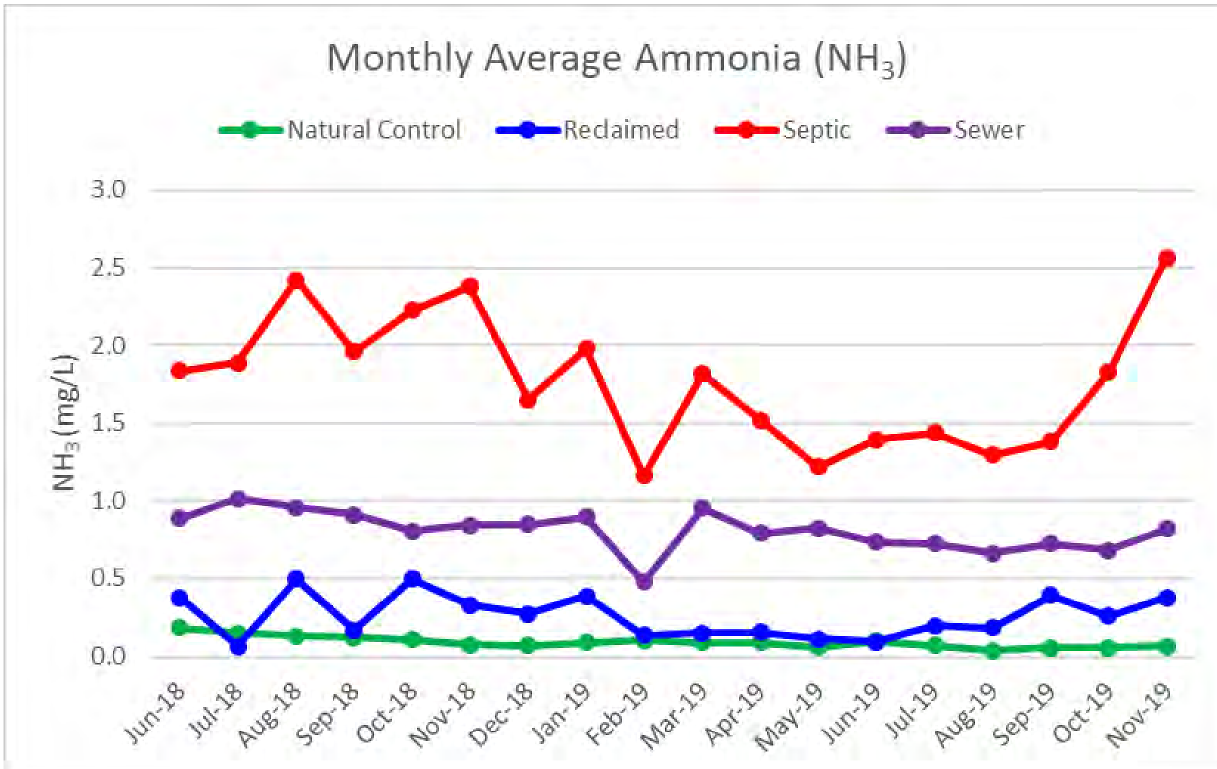

Figure 13: Average monthly NH_3 concentrations by treatment type.

Total Phosphorus (TP)

Unlike the other parameters, TP was monitored every other month rather than a monthly interval. During the first quarter, Turkey Creek communities were sampled in odd months (May, July, and September), while all others only during even months (June and August). In the second quarter, the monitoring schedule for this parameter was adjusted to sample all the wells for TP during the same bi-monthly schedule, allowing a better comparison of treatment types through time. To synchronize the sampling effort, Turkey Creek was sampled in both September and October, while all other areas continued to be sampled on even months. Summary statistics for TP of the sampling events during the 18 months are provided as boxplots (Figure 14), a time-series graph of mean concentrations by treatment type(Figure 15 and Figure 16), and in Appendix D. Generalized trends that were observed for PO_4^{3-} were also observed for TP, even though sampling was selective for TP: the highest average TP concentration was observed for the septic treatment (0.75 mg/L), followed by the reclaimed (0.23 mg/L), sewer (0.18 mg/L), and natural (0.13 mg/L) treatments.

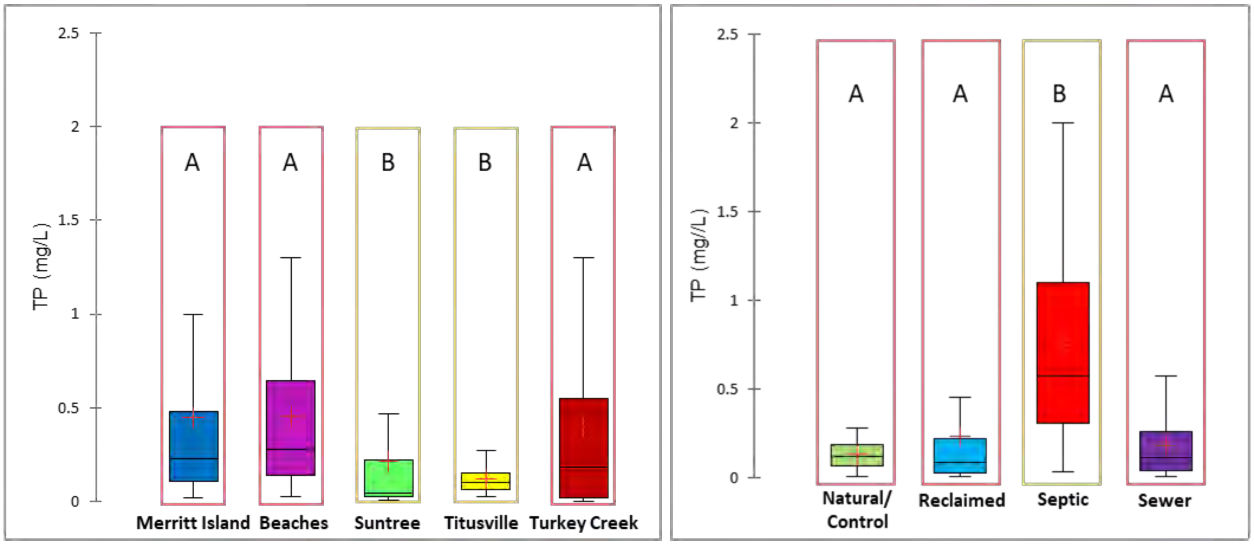

Figure 14: Boxplot of total phosphorus (TP) with whiskers, 1^{st} to 3^{rd}, and median by region and treatment type. The red cross represents the mean concentration. SDCF pairwise groupings are shown by color, any categories that share a color are not significantly different from one another.

Monthly mean concentrations for the septic treatment were consistently more than double than the concentrations observed in the other treatment types. In the first five months of sampling, June and August were the only months to represent concentrations for most (33) of the wells; May, July, and September means are indicative exclusively of Turkey Creek community data (Figure 15). October was the first month in which synchronized sampling commenced across all wells. The data for October, December, February, and April (all wells sampled) shows that the reclaimed, sewer, and natural wells monthly mean TP concentrations were stable with none exceeding 0.30 mg/L (Figure 16). Mean TP concentrations for the septic treatment were higher at the beginning of the sampling period, lowest at the February 2019 monthly event, and then slightly elevated from February to April 2019, a similar pattern to the one described for the PO_4^{3-} concentrations. From June and August 2019, the concentrations for all treatment types were stable with very little change in mean concentration with no treatment exceeding a ≤0.043 mg/L difference. In the sixth quarter, minimal changes were observed among all treatments, with the exception of the septic treatment; the septic areas saw a steep increase in TP concentrations from August to October 2019 (>0.30 mg/L), identical to the pattern observed for PO_4^{3}.

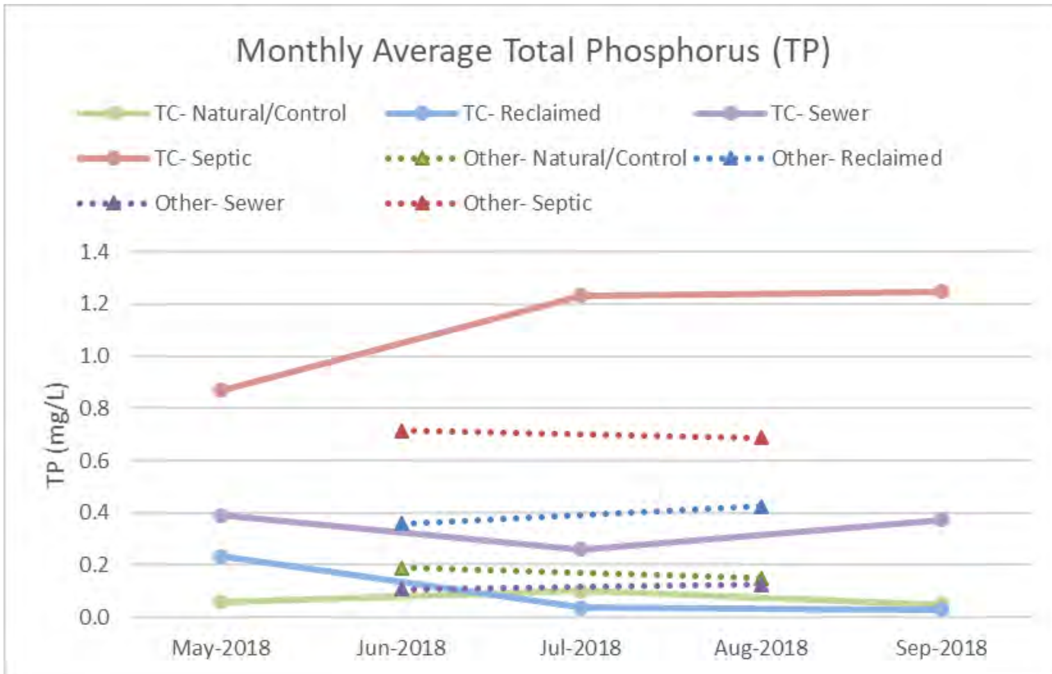

Figure 15: Average monthly TP⁻ concentrations by treatment type. The circle data points differentiate the sampling events which is only representative of the Turkey Creek region and the triangle points differentiate the sample events which all wells except Turkey Creek are represented.

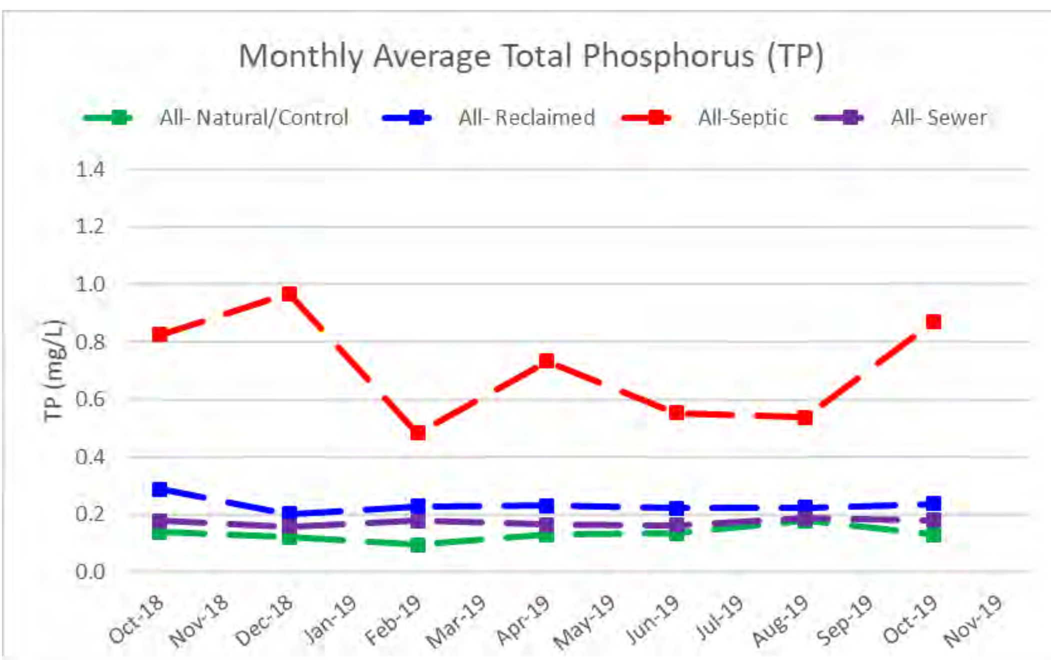

Figure 16: Average monthly TP concentrations by treatment type. The square points differentiate the sample events which all wells are represented, as compared to the previous graph.

As TP provides a measure of all forms of phosphorus, it is useful to determine the dominating percentage of a specific phosphorus constituent. Specifically, we are interested in determining if PO_4^{3-} was a large enough component of the TP value to use as a surrogate for TP. To better understand this, the percentage of PO_4^{3-} in the TP was calculated where data were available for both PO_4^{3-} and TP (Table 7). In the septic communities, the PO_4^{3-} contribution consistently exceeded 75% of the TP concentration and may be considered a surrogate for TP, eliminating the need for both analyses. In natural treatments, TP values presented a greater degree of variability and had lower PO_4^{3-} concentrations, indicating PO_4^{3-} would not be a good surrogate to test for TP.

There were little (<1%) to no changes in PO_4^{3-} contributions from fifth to sixth quarters at the natural, reclaimed treatments, or septic treatments, while there was a slight increase observed in the sewer treatment (4%). It should be noted that during the fifth quarter, the percentage of samples with PO_4^{3-} compositions >80% decreased among all treatments aside from the septic treatment. This could indicate that we are beginning to see seasonal differences in the phosphorus compositions of TP, and further supports the idea that PO_4^{3-} might not be used as an appropriate surrogate for TP, particularly for the reclaimed and natural sites.

Table 7: Percent of samples with greater than 80% contribution of PO_4^{3-} for TP by treatment type. It should be noted only sampling events with both TP and PO_4^{3-} were used in this calculation.

Treatment Type	Percent of Samples with PO_4^{3-} Percent Contribution > 80% of TP
Natural/Control	38%
Reclaimed	51%
Septic	77%
Sewer	66%

Orthophosphate (PO_4^{3-})

Summary statistics for PO_4^{3-} collected during the eighteen months of sampling are provided as boxplots (Figure 17), a time-series graph of mean concentrations by treatment type (Figure 18). As PO_4^{3-} significantly correlates ($r^2 = 0.926$, $p<0.0001$) with TP and as PO_4^{3-} was measured more frequently, it will be used to represent TP in the remainder of the report.

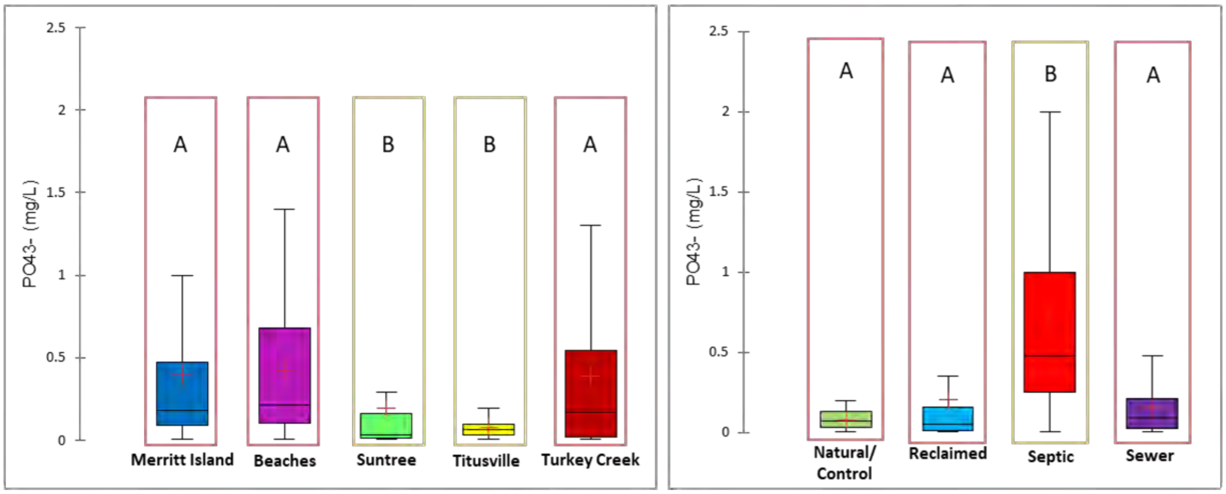

Figure 17: Boxplot of orthophosphate (PO_4^{3-}) with whiskers, 1st to 3rd, and median by region and treatment type. The red cross represents the mean concentration. SDCF pairwise groupings are shown by color, any categories that share a color are not significantly different from one another.

The highest mean PO_4^{3-} concentration was observed in the septic treatment (0.70 mg/L), followed by the reclaimed, sewer, and natural treatments (0.21, 0.16, and 0.08 mg/L, respectively). Mean and median concentrations for the septic treatment were between four to nine times higher than those for any other treatment types. Minimal variability was observed in the PO_4^{3-} concentrations for the sewer, reclaimed, and natural treatment types throughout the monitoring period (June 2018 - November 2019). However, the septic treatment shows clear inter-monthly variations in the PO_4^{3-} concentrations: an increasing trend is apparent from June 2018 to December 2018, followed by a decreasing trend from December 2018 to June 2019, some inter-monthly fluctuations occurred between June and August 2019, before finally another steep increase in mean concentration from August to November 2019 (Figure 18).

The initial increase during the June to July 2019 transition was possibly caused by the start of the wet season with increased precipitation, accompanied by a decrease in mean concentrations during the relatively dry month of August. The continued rise in concentrations could also be driven by rainfall, as in some areas experienced roughly 2” of rain prior to the sample event (i.e., Turkey Creek sampling in October took place 24 hours after a 1.95” rainfall event). With one year and three months of data, seasonality in the orthophosphate concentrations appears to be present for the septic communities only, with higher concentrations in the wet season and lower concentrations during the dry season.

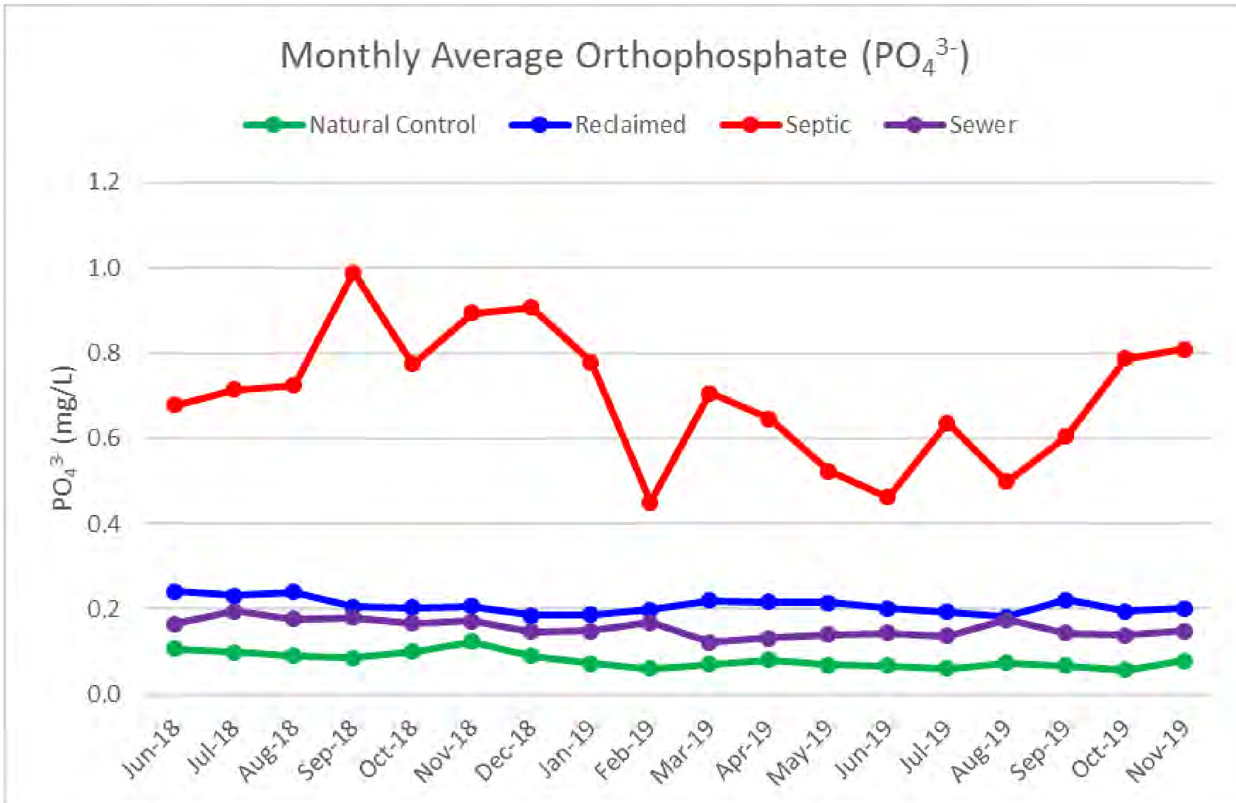

Figure 18: Average monthly PO_4^{3-} concentrations by treatment type.

Fecal Coliform

Fecal coliform summary statistics for the eighteen months of sampling are provided in a time-series graph of mean concentrations by treatment area is included in Figure 19 and Appendix D. Highest mean concentrations for fecal coliform were observed in the septic treatment (13 CFU/100 mL), closely followed by those in the sewer and reclaimed treatments (12 CFU/100 mL), and finally the natural treatment (3 CFU/100 mL) . However, geometric means, a measure of central tendency typically used to describe bacterial counts, is highest for the reclaimed treatment type (2.04 CFU/100 mL), but closely followed by the septic (1.86 CFU/100 mL), sewer, and once again lowest for the natural treatment.

There were a total of 11 "Too Numerous to Count" (TNTC) events throughout the monitoring period. Suntree region had a total of 5 TNTC events, with two events each in the reclaimed and sewer treatments and one event in the septic treatment, followed by the Merritt Island septic and Satellite Beach sewer treatments with each having three events. TNTC results indicate there were too many fecal colony forming units (CFUs) to allow individual colony count, even after attempted dilutions. In some cases, confluent results, even after multiple dilutions did not yield an estimated colony count. In cases where the samples were flagged as "TNTC," 500 CFUs/100mL value was assumed for statistical analyses and reporting. It is important to note that these spikes of contamination are sporadic and not consistent throughout the monitoring effort at one single community/well, as indicated by the geometric means that range between 1.25 and 2.04 CFU/100mL.

While there have been periods of stability and extreme fluctuations (driven by sporadic TNTC events) throughout the monitoring period across all treatments, the sixth quarter demonstrated stable trends with mean concentrations at or around the MDL across the sewer, reclaimed, and natural communities (Figure 19). It is important to note that although the septic experienced a drastic increase during the October 2019 sampling event, this was driven by the previously discussed TNTC result.

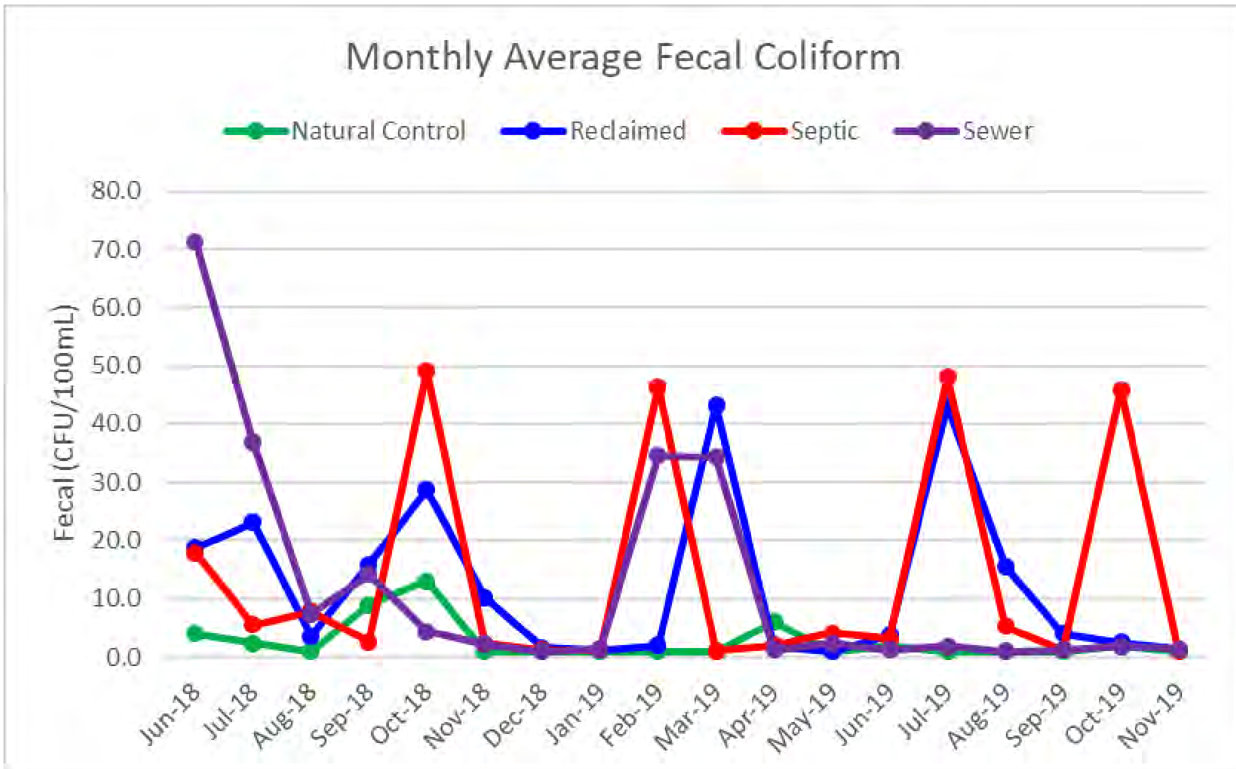

Figure 19: Average monthly fecal coliform- concentrations by treatment type.

Fecal coliforms are regulated by EPA through three different target criteria; one of these criteria sets a limit to the number of samples collected that exceed 31 CFUs/100mL to less than or equal to 10% of the total number of samples. The percentage of samples that exceed the EPA fecal coliform standard is presented in Table 8. The Merritt Island septic community (13%), Titusville reclaimed community (11%), and the Suntree reclaimed community (13%) all exceeded the 10% exceedance above 31 CFUs/100mL. There were no natural or sewer communities that exceeded the 10%, with the natural community having the lowest percentages (0-3%).

Table 8: Percentage of samples that exceed the EPA standard of 31 CFUs/100mL for fecal coliform for all 18 sampling events for each region per treatment type. Percentages that exceeded the 10% of the total number of samples are bolded.

Treatment Type	Region	Samples > 31 CFUs/100mL	Total Number of Samples	% Occurrence over 31 CFUs/100mL
Natural	Turkey Creek	1	36	2.78
	Beaches	0	36	0.00
	Titusville	1	36	2.78
Sewer	Turkey Creek	0	54	0.00
	Beaches	3	54	5.56
	Merritt Island	1	54	1.85
	Suntree	5	54	9.26
	Titusville	0	54	0.00
Septic	Turkey Creek	0	54	0.00
	Beaches	2	54	3.70
	Merritt Island	7	54	**12.96**
	Suntree	1	54	1.85
Reclaimed	Turkey Creek	2	54	3.70
	Beaches	1	54	1.85
	Suntree	7	54	**12.96**
	Titusville	6	54	**11.11**

Comparing Treatment Types within Regions

In this section, we zoom into each study region to examine the effect of the different treatment types. The regions include Melbourne and Satellite Beach (combined as "Beaches"), Merritt Island, Suntree, Titusville, and Turkey Creek.

Summary tests are included for all analyzed parameters (nutrient and bacteriological analytes) and comparison among treatments are tested using appropriate statistical tests.

Complete descriptive statistics (mean, median, percentiles, *etc.*) for each treatment type by region are provided in Appendix E.

Summary of Findings

Across all regions, the septic and reclaimed treatments had significantly higher concentrations of TN, TP, and PO_4^{3-} in comparison to sewer and natural. Only in Titusville, which did not have a septic treatment, was TP not significantly different from the other treatments.

Within each region there are also one or more wells that had a much wider range of nutrient concentrations in comparison to all other wells in that region. When present in a region, these outlier wells were in septic communities except in Titusville, where the reclaimed community wells varied. This suggests that there are local drivers influencing variability in nutrient concentrations, particularly in septic communities.

A key observation across regions in the PCA, is that the reclaimed wells tend to cluster at higher nutrient concentrations, particularly those in the form of NOx, than other treatment wells. In Turkey Creek, the Beaches, and Titusville the reclaimed water treatment wells were observed to have the most significantly high concentrations of TN (14.10, 6.45, and 1.30 mg/L) across all treatments. In Suntree, it was significantly higher than the sewer treatment with 2.55 mg/L TN. Turkey Creek and the Beaches had correspondingly significantly high median NO_X, while in Titusville it had the highest TKN. Turkey Creek and Titusville are served by WWTF that produce reclaimed irrigation water with the highest TN concentrations, while the Beaches and Suntree regions the lowest. This suggests that the use of reclaimed water irrigation has an influence on the variation in groundwater nitrogen, and the characteristic of this variability can differ by the facility, usage patterns, or other geophysical factors.

Turkey Creek

A PCA and bivariate analyses were conducted on the Turkey Creek wells to understand the differences between the treatments in this region (Figure 20). Like the aggregate data PCA, the Turkey Creek PCA had two major loading factors. One factor was primarily inorganic nitrogen and the other organic nitrogen and PO_4^{3-}. The reclaimed treatment aligned on the inorganic component, the sewer and natural aligned on the organic axis, and the septic wells were divided between the two. Unlike the aggregate data, in the Turkey Creek PCA, the inorganic factor explained more variance than the organic nitrogen/ PO_4^{3-} factor.

Figure 20: Turkey Creek Site Map of the four treatment areas and relative location of wells.

The inorganic nitrogen in Turkey Creek is being driven by the reclaimed community, which had significantly higher NO_X concentrations than the other treatments, in fact, the NO_X median in the reclaimed community was nearly 400 times higher than the next highest concentration. The highest organic nitrogen concentrations (NH_3 and TKN) were in the sewer community, both significantly higher than the other treatments. TP and PO_4^{3-} were significantly higher in the septic community.

Non-Parametric Analysis

Statistically significant differences in medians were detected among treatment types for every parameter analyzed (Table 9). Summary statistics for Turkey Creek wells are presented in Appendix E. The natural treatment has the lowest median and mean values for all of the analyzed parameters: TN, TKN, PO_4^{3-}, and TP. Median concentrations of NH_3 and TKN are highest for the sewer treatment, NO_X and TN are highest for the reclaimed community, and highest PO_4^{3-} and TP are in the septic community. All treatments fell below the fecal coliform target exceedance of 10%.

Table 9: Differences in nutrient median concentrations between treatment types in Turkey Creek. Highest mean and median values are in bold.

Analyte	Septic	Sewer	Reclaimed	Natural
***NH_3 (mg/L)**	0.930^a	$\mathbf{3.400^b}$	0.035^c	0.035^d
***TKN (mg/L)**	1.400^a	$\mathbf{3.700^b}$	0.086^c	0.215^d
***NO_X (mg/L)**	0.037^a	0.025^b	$\mathbf{14.10^c}$	0.025^d
***TN (mg/L)**	4.800^a	3.700^a	$\mathbf{14.10^b}$	0.235^c
***PO_4^{3-} (mg/L)**	$\mathbf{0.970^a}$	0.490^b	0.014^c	0.035^c
***TP (mg/L)**	$\mathbf{1.200^a}$	0.510^b	0.098^c	0.168^c

**Significantly different median at p<0.001 using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at p<0.05 within rows. If significant differences were found, the highest value is in bold.*

The sewer community in Turkey Creek had significantly higher organic nitrogen (NH_3 and TKN) than the other treatments (Table 9).

The reclaimed community had significantly higher inorganic nitrogen (NO_X) and TN than the other treatments. There was a significant difference in median NO_X concentrations between septic and sewer treatments (p=0.042). There was no significant difference in NO_X concentrations in the natural area and the sewer or septic communities.

The septic community in Turkey Creek had significantly higher and PO_4^{3-} than the other treatments and the second highest NH_3, which was significantly higher than reclaimed and natural treatments.

PCA

Table 10 below presents the loadings for the first four PCs of the Turkey Creek groundwater samples. The first four PCs account for 99.6% of the variability in Turkey Creek, Figure 21 displays a plot of the first two Turkey Creek PCs with the points colored to represent treatment type. The first two PCs account for 68.1% of total variability and can present a general view of the dominant forces driving differences in concentrations in Turkey Creek.

Table 10: PCA loadings of six water quality variables on the first four PCs for the Turkey Creek groundwater samples.

Analyte	PC1	PC2	PC3	PC4
NH_3	-0.82	0.50	0.12	-0.23
TKN	-0.83	0.51	0.12	-0.16
NOX	0.86	0.49	0.07	-0.12
TN	0.70	0.69	0.12	-0.17
PO_4^{3-}	-0.13	0.48	-0.19	0.84
3 Day Rainfall Sum	0.02	-0.15	0.96	0.25
Variability (%)	43.4	24.6	16.7	14.9
Cumulative %	43.4	68.1	84.7	99.6

As with the aggregate data PCA, we see separate factors emerging with inorganic nitrogen as one component and organic nitrogen (NH_3 and TKN) and PO_4^{3-} another. In Turkey Creek, however, the inorganic factor (PC1) explains more of the variance in water quality than the organic nitrogen/ PO_4^{3-} component (PC2). The 3-day rainfall data explanatory power was almost exactly that of the aggregate data PCA.

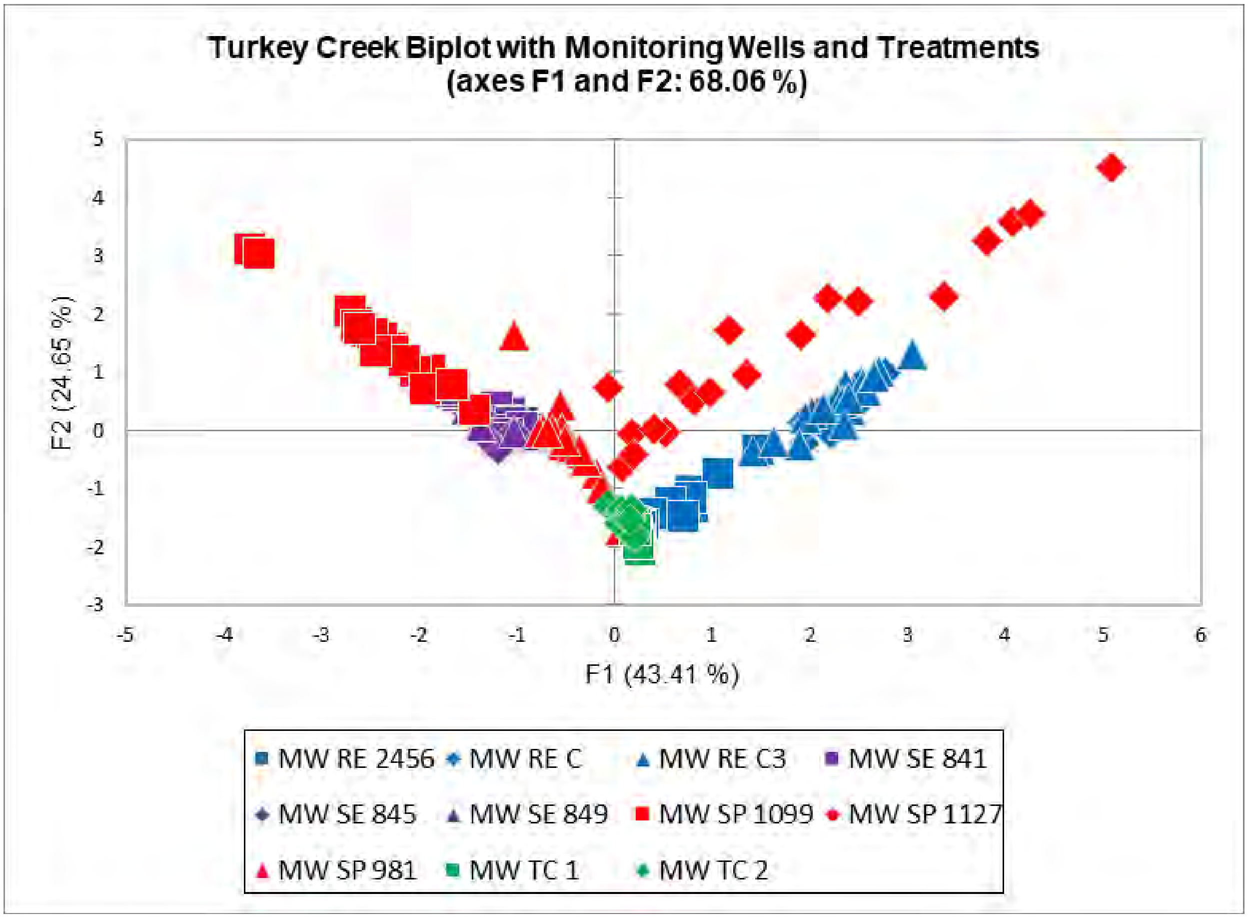

Figure 21: PCA of Turkey Creek monitoring well data with treatment types indicated by color and well indicated by shape. (Blue=reclaimed, Red=septic, Purple=sewer, Green=natural).

Plotting the two components by treatment type and well, we can see variability and grouping of the groundwater samples within Turkey Creek (Figure 21). The treatment types tend to cluster together, with the reclaimed wells clustered on the inorganic nitrogen factor and the sewer and natural wells clustered along the organic nitrogen/ PO_4^{3-}factor. The three septic wells, however, appear to diverge, with samples from monitoring wells SP 981 and SP 1099 appearing along the organic nitrogen/ PO_4^{3-} factor trajectory and monitoring well SP 1127 aligning with inorganic nitrogen. This suggests that there are differences in organic and inorganic nitrogen concentrations that may be explained by differences in the nitrification and denitrification processes in the groundwater monitored by these wells.

Beaches

A PCA and Bivariate analyses were conducted in the two regions of the barrier island combined as the Beaches region (Figure 22). The PCA plot shows one outlier septic well that appears to have samples with a very large varaibility of nitrogen constituents through the sampling period. Some of the samples appear to be driven by high inorganic nitrogen, while others by organic sources of nitrogen; this can indicate mixing of nitrogen sources or nitrification and denitrification processes that change through time. It appears that PO_4^{3-} has a stronger role driving the differences between treatment types within the beaches community.

Figure 22: Beaches community locations with well locations.

In the Beaches region, the reclaimed community once again had the highest total nitrogen concentration, with the primary constituent being inorganic nitrogen (NO_X). The reclaimed community NO_X concentration was seven times higher than the Beaches septic community NO_X concentration, but half as high as the Turkey Creek reclaimed community NO_X concentration. On the beachside, there were no significant differences in the organic nitrogen concentrations (NH_3 and TKN) between septic and sewer communities. Total Phosphorus and PO_4^{3-}were not significantly different between septic and sewer communities.

Non-Parametric Analysis

In the Beaches region, statistically significant differences in medians were detected among treatment types for every parameter analyzed (Table 11). Summary statistics for the Beaches are presented in Appendix E. Statistical comparisons between treatments in the Beaches region are presented in Table 11.

Higher organic nitrogen concentrations (NH_3 and TKN) were found in the septic and sewer communities, which did not significantly differ. Significantly higher inorganic nitrogen (NO_X) and phosphorus (TP and PO_4^{3-}) were found in the Beaches reclaimed community. TN was also significantly higher in the reclaimed community, driven by high NO_X.

Table 11: Differences in nutrient median concentrations between treatment types in Melbourne and Beaches. Highest mean and median values are in bold.

Analyte	Septic	Sewer	Reclaimed	Natural
***NH_3 (mg/L)**	0.074^a	**0.100^a**	0.035^b	0.049^c
***NO_X (mg/L)**	0.855^a	0.405^b	**6.250^c**	0.200^d
***TKN (mg/L)**	0.600^a	**0.610^a**	0.086^b	0.230^b
***TN (mg/L)**	1.550^a	1.050^b	**6.450^c**	0.230^d
***PO_4^{3-} (mg/L)**	0.410^a	0.083^b	**0.750^a**	0.120^c
***TP (mg/L)**	0.460^a	0.098^b	**0.720^a**	0.215^c

**Significantly different median at p<0.001 using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at p<0.05 within rows. If significant differences were found, the highest value is in bold.*

PCA

The Beaches PCA looks very different from the aggregate and Turkey Creek PCAs. We do not see the clear split between organic and inorganic nitrogen and in fact, it appears that phosphorus plays a more powerful role driving the differences between samples on the beachside. Table 12 below presents the loadings for the first four PCs of the Beaches groundwater samples, which make up 99.0% of the variability. Principal Component 1 (PC1) includes organic nitrogen, inorganic nitrogen, TN, and PO_4^{3-}explaining 53% of the variance. PC2 includes the organic nitrogen (NH_3 and TKN), and together these two components account for 75.6% of the variance in groundwater nutrient concentrations.

Table 12: PCA loadings of six water quality variables on the first four PCs for the Beaches groundwater samples.

	PC1	PC2	PC3	PC4
NH_3	0.77	0.60	0.07	-0.10
TKN	0.72	0.66	0.11	-0.08
NOX	0.77	-0.62	0.01	-0.18
TN	0.88	-0.43	0.04	-0.18
PO_4^{3-}	0.82	-0.11	-0.16	0.54
3 Day Rainfall Sum	-0.05	-0.11	0.99	0.11
Variability (%)	52.6	23.1	16.9	6.4
Cumulative %	52.6	75.6	92.6	99.0

The Beaches PC1 accounts for 52.6% of all variability with all nutrients loading strongly, including both organic and inorganic nitrogen and ortho-p. PC2 explains 23.1% of the variability with the organic species of nitrogen having strong, positive loadings with opposing inorganic nitrogen loading. PC3 represents 16.9% of variability and is dominated by the 3-day rainfall sum with weak or no loading from the nutrients, suggesting that rainfall has little influence on water quality.

In Figure 23, the biplot of PC1 and PC2 highlights both the variability and clustering of the groundwater samples within the Beaches region by treatment. The septic treatment well SP 250 had several measurements that extreme outliers to all other wells in the Beaches. The remainder of the data aligned primarily along the inorganic nitrogen and PO_4^{3-} axis, with the reclaimed wells all clustering separate from the other treatments.

SP 250 has significantly elevated nutrient concentrations compared to the neighboring septic wells, and the other regional wells and its alignment with inorganic and organic nitrogen and PO_4^{3-}may suggest a continued source that is denitrifying over time or mixing of sources through the sampling period.

The cluster and alignment of the reclaimed treatments with inorganic nitrogen and PO_4^{3-} suggest that there may be nutrient enrichment from the application of reclaimed water irrigation. As all three reclaimed wells clustered similarly, this suggests that they are representative of the same application patter of reclaimed water.

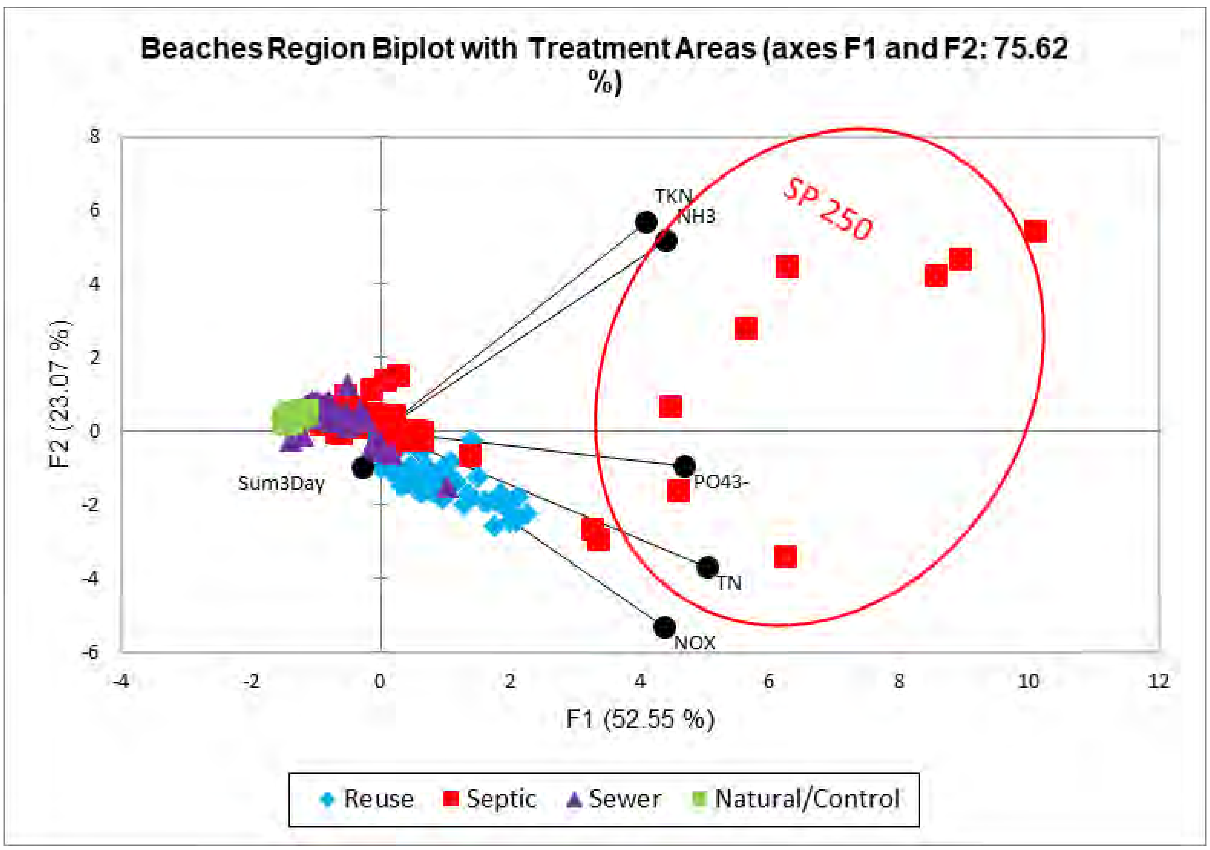

Figure 23: Coordinates of the Beaches PCs based on the treatment type. Notable wells have been labeled and identified.

Suntree

PCA and bivariate analyses were conducted of the Suntree region wells (Figure 24). The PCA components appear to be driven by two extreme septic wells, each loading along a different factor. As with the Beaches, inorganic nutrients load on one factor, and organic nitrogen loads on the other.

Unlike the other two regions, in Suntree, inorganic nitrogen (NO_X) was significantly higher in the septic community than the sewer and reclaimed communities. Ammonia (NH_3) concentrations were higher in the reclaimed community, but the concentrations did not significantly differ from those in the septic community. TKN was significantly higher in the reclaimed community than the other two treatments. Consistent with the other regions, Total Phosphorus and PO_4^{3-} were significantly higher in the septic community.

Figure 24: Suntree site map identifying the three treatment areas.

Non-Parametric Analysis

In Suntree, there were statistically significant differences in nutrient concentrations among treatment types (Table 13). Summary statistics for Suntree are presented in Appendix E.

The Suntree septic community had significantly higher concentrations of inorganic nitrogen (NO_X), TN, and phosphorus (PO_4^{3-} and TP) than the other treatments.

The reclaimed community had significantly higher TKN concentrations than the other treatments but did not have a significantly different NH_3 concentrations than the septic community. Interestingly, the highest average fecal coliforms were also in the reclaimed community.

Table 13: Differences in nutrient median concentrations between treatment types in Suntree.

Analyte	Septic	Sewer	Reclaimed
***NH_3 (mg/L)**	0.455[a]	0.170[b]	**0.575[a]**
***NO_X (mg/L)**	**0.620[a]**	0.027[b]	0.051[b]
***TKN (mg/L)**	0.930[a]	0.027[b]	**2.100[c]**
***TN (mg/L)**	**6.050[a]**	0.785[b]	2.550[c]
***PO_4^{3-} (mg/L)**	**0.265[a]**	0.015[b]	0.027[c]
***TP (mg/L)**	**0.420[a]**	0.015[b]	0.036[c]

**Significantly different median at $p<0.001$ using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at $p<0.05$ within rows. If significant differences were found, the highest value is in bold.*

PCA

Table 14 below presents the loadings for the first four PCs of the Suntree groundwater samples. The first four PCs account for 99.0% of the variability, Figure 25 displays a plot of the first two PCs with the points colored to represent treatment type. The first two PCs account for 75.9% of total variability and can present a general view of the dominant nutrient forces driving the variation between the treatment types in Suntree.

Table 14: Loadings of six water quality variables on the first four PCs for the Suntree groundwater samples.

	PC1	PC2	PC3	PC4
NH_3	0.67	0.71	-0.01	-0.07
TKN	0.62	0.75	0.06	-0.14
NOX	0.64	-0.75	0.03	-0.17
TN	0.87	-0.43	0.05	-0.22
PO_4^{3-}	0.84	-0.11	-0.08	0.52
3-Day Rainfall Sum	-0.03	-0.01	1.00	0.07
Variability (%)	45.2	30.7	16.8	6.3
Cumulative %	45.2	75.9	92.7	99.0

The Suntree PC1 explains 45% of the variability and is loaded by all nutrients. The strong loadings of TN and PO_4^{3-} suggest that variability is driven by the same source. PC2 is responsible for 30.7% of the variability and is strongly loaded by the nitrogen constituents, with TN having a moderate loading.

PC3 explains 16.8% of the variability and is completely dominated by the 3-day rainfall sum, suggesting that rainfall has a limited influence on water quality. PC4 explains 6.3% of the variability and is moderately loaded by PO_4^{3-}.

The Suntree PCA biplot shown in Figure 25 highlights how the majority of variability and highest nutrient concentrations in the region were observed at the wells SP 6398 and SP 6215. The split between organic and inorganic nitrogen variability for these two wells, along with SP 6155's low concentration and variability, suggest differences in denitrification rates at these sites.

The RE FL1 and FL2 wells cluster together along the organic nitrogen axis than the RE FL3 well. This may suggest different patterns of reclaimed water use or the potential for nutrient inputs from reclaimed water.

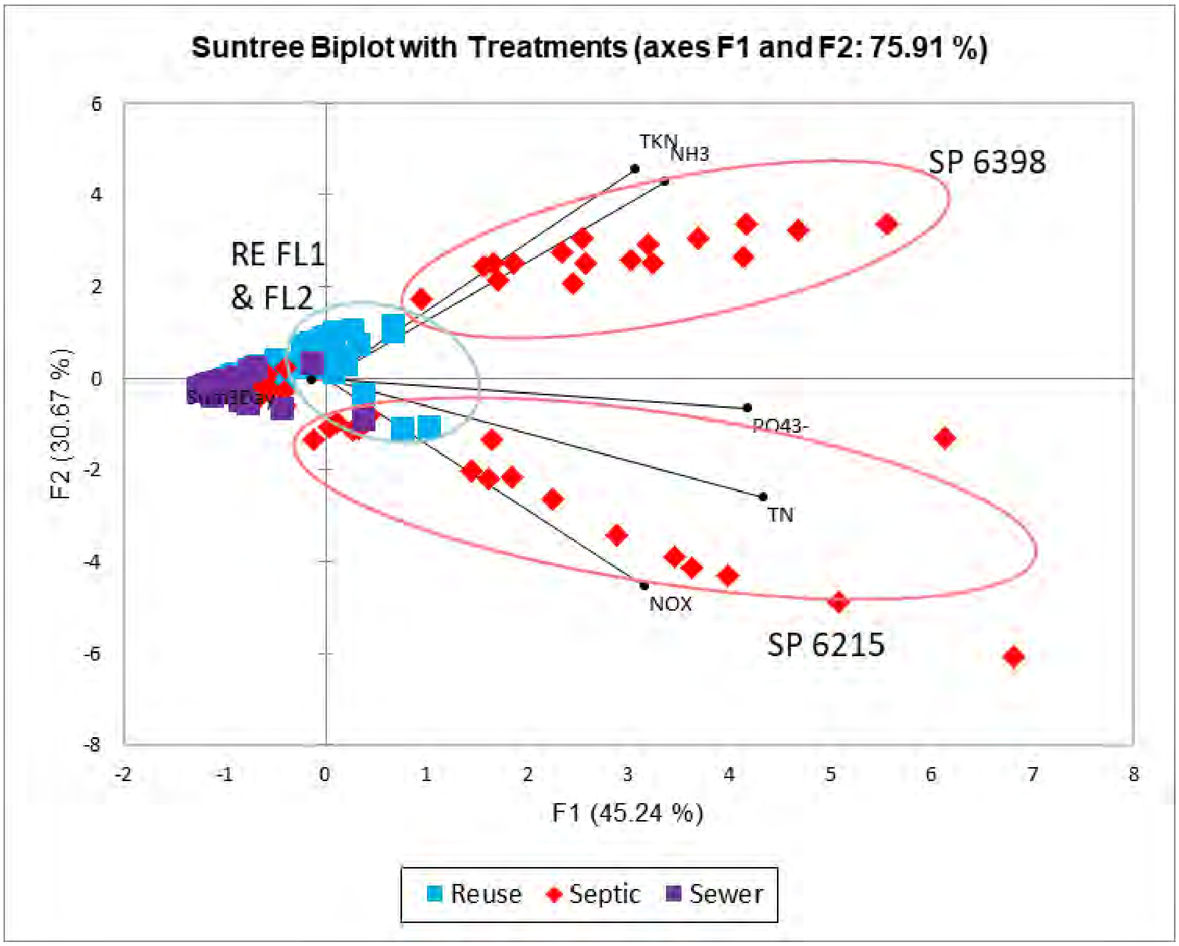

Figure 25: Coordinates of the Suntree PCs based on the treatment type. Notable wells have been labeled and identified.

Merritt Island

There are only septic and sewer treatments in Merritt Island (Figure 26), providing an opportunity to compare the two. Nutrient concentrations were relatively low compared to Turkey Creek. The septic community had significantly higher organic nitrogen (NH_3 and TKN) and phosphorus (TP and PO_4^{3-}). The sewer community had significantly higher inorganic nitrogen (NO_X). There was no significant difference in TN between the two treatments.

PCA analysis revealed similar results as the aggregate data and the Turkey Creek data. Organic nitrogen and PO_4^{3-}loaded on one factor and inorganic nitrogen and TN loaded on the other. However, in Merritt Island, there is a slightly greater influence of the 3-day rainfall data loading on PC3 with organic nitrogen.

Figure 26: Merritt Island site map identifying the two treatment areas.

Non-Parametric Analysis

Summary statistics for Merritt Island are presented in Appendix E. There was no significant difference in the TN concentrations between septic and sewer treatments ($p=0.296$) (Table 15). The NH_3 and TKN median concentrations were significantly higher in septic treatments ($p<0.001$), while NO_X was found to be significantly highest in the sewer treatments ($p<01$). PO_4^{3-} and TP were significantly higher for the septic treatment than the sewer treatment ($p<0.001$).

Table 15: Differences in nutrient concentrations between treatment types in Merritt Island.

Analyte	Septic	Sewer
***NH_3 (mg/L)**	**1.050**	0.088
****NO_X (mg/L)**	0.025	**0.120**
***TKN (mg/L)**	**1.600**	0.685
TN (mg/L)	1.700	1.750
***PO_4^{3-} (mg/L)**	**0.365**	0.093
***TP (mg/L)**	**0.430**	0.150

**Significantly different median at p<0.001 using Mann-Whitney. ** Significantly different median at p<0.01 using Mann-Whitney.*

PCA

Table 16 below presents the loadings for the first four PCs of the Merritt Island groundwater samples, which make up 99.3% of the variability in groundwater nutrient concentrations. Figure 27 displays a plot of the first two PCs with the points colored to represent the treatment type. The first two PCs account for 78.4% of total variability and can present a general view of the dominant nutrient forces driving the variation between the treatment types in Merritt Island.

Table 16: Loadings of six water quality variables on the first four PCs for the Merritt Island groundwater samples.

	PC1	PC2	PC3	PC4
NH_3	0.92	-0.31	0.07	-0.19
TKN	0.90	-0.31	0.13	-0.24
NOX	0.27	0.95	-0.17	0.00
TN	0.57	0.81	-0.12	-0.08
PO_4^{3-}	0.85	-0.11	0.13	0.50
Sum3Day	-0.19	0.35	0.92	-0.03
Variability (%)	47.1	31.3	15.2	5.7
Cumulative %	47.1	78.4	93.6	99.3

The Merritt Island PC1 explains 47.1% of the total variability with the organic nitrogen and PO43- dominating. PC2 explains 31.3% of the variability and is dominated by inorganic nitrogen and TN. PC3 explains 15.2% of variance and is dominated by the 3-day rainfall sum. PC4 explains 5.7% of variability and as PO43- had the highest loading at a moderate level.

In Figure 27 the biplot of PC1 and PC2 highlight both the variability of a couple of individual septic wells and clustering of the remaining groundwater samples within Merritt Island by treatment. The septic well SP 1739 is a clear outlier of all wells as it has the largest variability and highest concentrations of all nutrients. The sewer well SE 1735 is also an outlier with an alignment along the organic nitrogen and PO_4^{3-}axis.

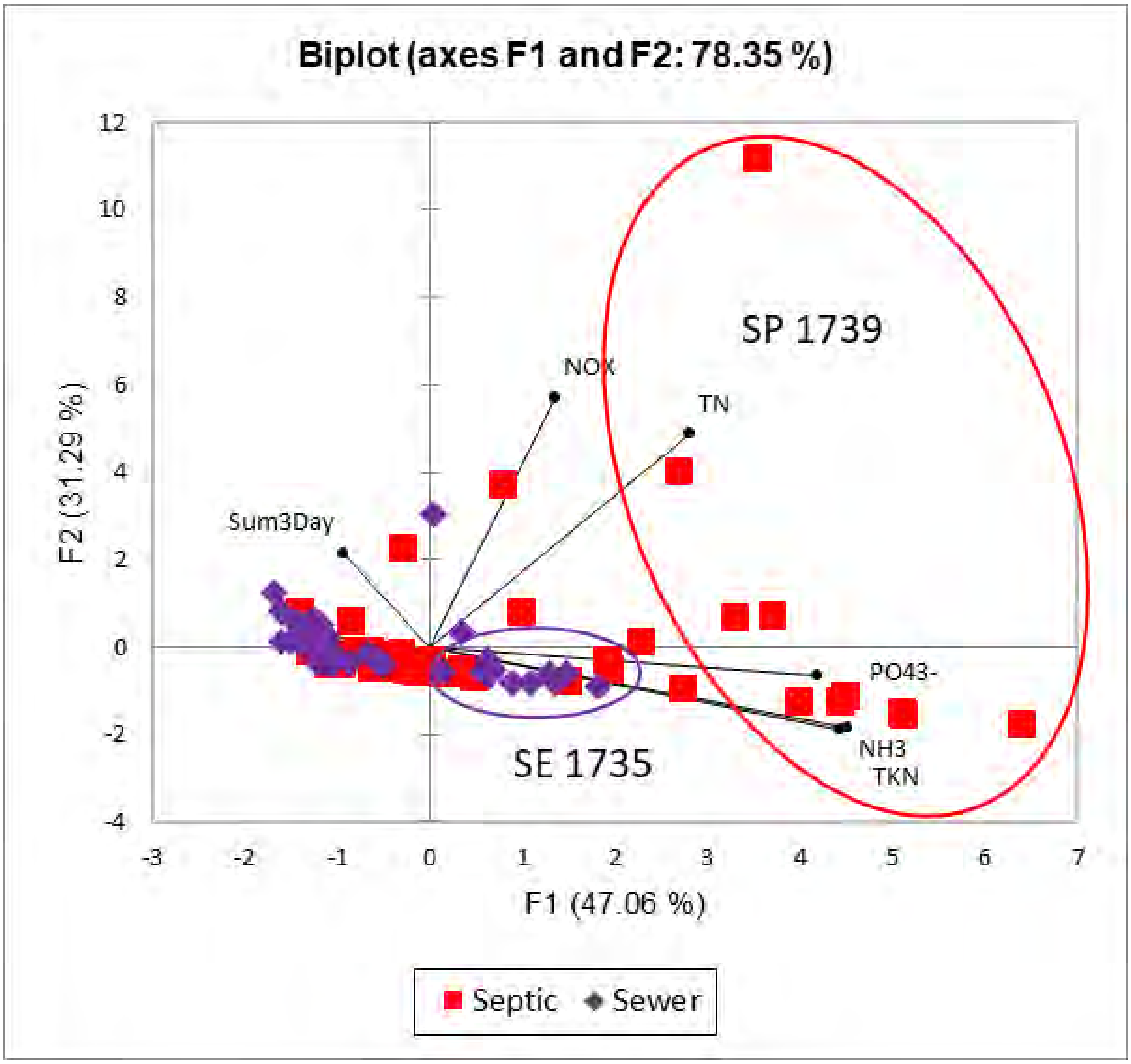

Figure 27: Coordinates of the Merritt Island PCs based on the treatment type. Notable wells have been labeled and identified.

Titusville

Sewer, reclaimed, and natural treatment were compared in the Titusville region (Figure 28). Because this area does not include septic, it presents the only nutrient dataset absent of the highly influential and extreme values found in septic communities. As a result, individual wells can have a dramatic effect.

Figure 28: Titusville site map identifying the three treatment types.

Inorganic nitrogen (NO_X) and PO_4^{3-} were significantly higher in the sewer community than the other two treatments. TKN and TN were significantly higher in the reclaimed community than the other two. There were no significant differences between the median NH_3 (p=0.815) and TP (p=0.267) concentrations.

The Titusville PCA is vastly different. The first two factors explain less variance than what we have seen in other regions, only 60%. This is likely due to the lack of data from a septic community in this region. For the first time, there is no clear delineation between inorganic and organic nitrogen constituents. Inorganic nitrogen (NOx) and organic nitrogen (TKN) load together on the first factor (PC1) and organic nitrogen (NH_3) and PO_4^{3-} load on the second factor (PC2). Rainfall loads strongly onto PC3, suggesting that its influence is easier to see in absence of the septic tank effects.

Non-Parametric Analyses

Significantly different medians were identified for NO_X, TKN, TN, and PO_4^{3-} (Table 17). Summary statistics for Titusville are listed in Appendix E. The sewer treatment wells had the significantly highest medians for NO_X (p<0.0001) and PO_4^{3-} (p=0.001). The reclaimed treatment wells, however, had the most significantly high medians for TKN (p<0.0001) and TN (p<0.0001).

There were marginally no significant differences between reclaimed and natural for NO_X (p<0.062). For TKN, sewer and natural were not significantly different (p=0.430). Though in median TN sewer was significantly higher than natural (p<0.0001). Lastly, there was no significant difference between sewer and natural for PO_4^{3-} (p=1.0).

There were no significant differences between the median NH_3 (p=0.815) and TP (p=0.267) concentration among sewer, reclaimed, and natural treatments.

Table 17: Differences in nutrient median concentrations between treatment types in Titusville.

Analyte	Sewer	Reclaimed	Natural
NH_3 (mg/L)	0.120	0.077	0.100
***NO_X (mg/L)**	**0.106^a**	0.029^b	0.025^b
***TKN (mg/L)**	0.480^a	**1.150^b**	0.560^a
***TN (mg/L)**	0.815^a	**1.300^b**	0.610^c
****PO_4^{3-} (mg/L)**	**0.078^a**	0.061^b	0.056^b
TP (mg/L)	0.110	0.110	0.090

**Significantly different median at p<0.001 using Kruskal-Wallis. **Significantly different median at p<0.01 using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at p<0.05 within rows. If significant differences were found, the highest value is in bold.*

PCA

Table 18 presents the loadings for the first four PCs of the Titusville groundwater samples, which account for 91.0% of the measured variability in groundwater concentrations. Figure 29 displays a plot of the first two PCs with the points colored to represent the treatment type. The first two PCs account for 60% of total variability and can present a general view of the dominant nutrient forces driving the variation in groundwater nutrients.

Table 18: Loadings of six water quality variables on the first four PCs for the Titusville groundwater samples.

	PC1	PC2	PC3	PC4
NH_3	-0.47	-0.67	-0.13	0.35
TKN	0.59	-0.42	0.16	0.54
NOX	0.94	-0.07	-0.12	-0.20
TN	0.97	-0.14	-0.08	-0.08
PO_4^{3-}	0.17	0.70	-0.33	0.59
3 Day Rainfall Sum	0.10	0.20	0.94	0.14
Variability (%)	40.3	19.7	17.5	13.8
Cumulative %	40.3	60.0	77.5	91.3

The Titusville PC1 is responsible for 40.3% of the variability and is dominated by TN and inorganic nitrogen, along with a moderate TKN loading. PC2 explains 19.7% of the variability and is dominated by a strong, positive PO_4^{3-} loading.

PC3 explains 17.5% of the variability and is dominated by the 3-day rainfall sum, suggesting rainfall has an influence on nutrient variability in this particular region. PC4 explains 13.8% of the variability with a moderate loading between the organic species of nitrogen and PO_4^{3-}.

The Titusville PC1 and PC2 biplot displayed in Figure 29 identified two clusters of large variability and high concentrations of nitrogen and phosphorus. The well RE 549 appears to account for much of the variability measured for both NO_x and TN in this region. This particular reclaimed well appears to be dominated by inorganic nitrate unlike the other monitored reclaimed wells in the region.

SE 540 is another notable outlier that clustered mostly along the PO_4^{3-} loading factor and accounts for a large portion of the measured variability in PO_4^{3-} in the region. As the well is also geographically separate from the other two sewer treatment wells, this may suggest a unique condition with the sewer condition, soil type, or other site-specific local geophysical difference.

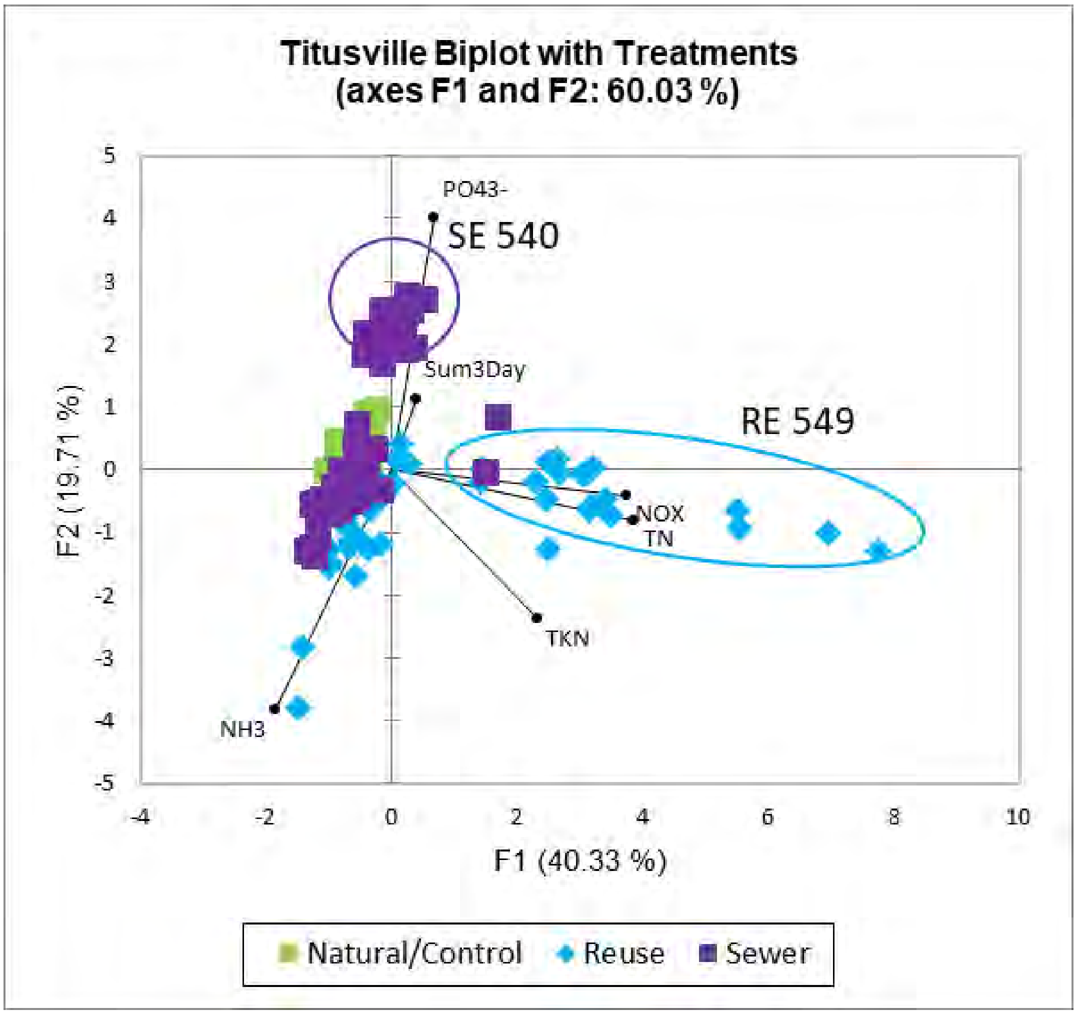

Figure 29: Coordinates of the Titusville PCs based on the treatment type. Notable wells have been labeled and identified.

Examining Regional Differences

Summary of Findings

Comparing each treatment across regions can help focus priorities and clarify regional drivers of nutrient concentrations. There were significant differences in the natural area well nutrient concentrations that may indicate background conditions that drive concentrations in treatment areas. For example, the Beaches have significantly higher phosphorus and NO_X in the natural area that may represent a higher background of these constituents and/or slow denitrification. The Titusville natural area had significantly higher organic nitrogen (NH_3, TKN), which may be an indication of high mineralization or slow nitrification processes. It is important to understand the differences in regional background concentrations. Complete descriptive statistics (mean, median, percentiles, *etc.*) for each treatment type by region are provided in Appendix F.

Natural Areas

In the aggregate dataset, the nutrient concentrations in the natural area were not always lower than the other treatments, suggesting there are background conditions influencing nutrient concentrations and dynamics. Natural area concentrations of ammonia (NH_3) and ortho-p (PO_4^{3-}) did not significantly differ from those in reclaimed communities, and in fact, median ammonia was higher in the natural areas (Table 19).

Non-parametric comparisons of the natural areas in the three regions, Beaches, Titusville, and Turkey Creek, provides an opportunity to further explore the possibility of background conditions that may confound results. We found that there were significant differences in the regional natural area nutrient concentrations. Titusville's natural area had significantly higher NH_3, TKN, and TN than Beaches or Turkey Creek. This may be an indication of slower or less reactive denitrification processes. The Beaches region has significantly higher TP and PO_4^{3-}, suggesting that natural conditions on the beachside may lead to high phosphorus in groundwater. This may be related to the historic marine sediments leaching calcium carbonate and phosphate from aragonite into groundwater, but this line of inquiry requires further investigation. Nitrate/nitrite (NO_X) was significantly higher in the Beaches natural area than in the Turkey Creek natural area, however, differences between them were minimal and could be attributed to denitrification rates. Further investigation of denitrification rates would help clarify.

Table 19: Statistical significance testing comparing the natural areas in different study regions.

Analyte	Beaches	Titusville	Turkey Creek
$^*NH_3$ (mg/L)	0.049^a	**0.100^b**	0.035^a
$^{**}NO_X$ (mg/L)	**0.037^a**	$0.025^{a,b}$	0.025^b
***TKN (mg/L)**	0.200^a	**0.560^b**	0.215^a
***TN (mg/L)**	0.230^a	**0.610^b**	0.235^a
$^*PO_4^{3-}$ (mg/L)	**0.120^a**	0.056^b	0.035^b
***TP (mg/L)**	**0.215^a**	0.091^b	0.057^b

**Significantly different median at p<0.001 using Kruskal-Wallis. **Significantly different at p<0.05. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate significant differences at p<0.05 within rows. If significant differences were found, the highest value is in bold.*

Sewer

The Turkey Creek sewer treatment has significantly higher organic nitrogen (NH_3, TKN), TN, and phosphorus (TP, PO_4^{3-}) than the other regions. Beaches had the highest inorganic nitrogen (NO_X) concentration, but it was not significantly higher than Merritt Island or Titusville (Table 20).

Table 20: Statistical significance testing comparing the sewer communities in different study regions.

Analyte	Merritt Island	Beaches	Suntree	Titusville	Turkey Creek
$^*NH_3$ (mg/L)	0.088^a	0.100^a	0.170^a	0.120^a	**3.400^b**
$^*NO_X$ (mg/L)	0.120^a	**0.405^a**	0.027^b	0.106^a	0.025^c
***TKN (mg/L)**	0.685^a	0.610^a	0.675^a	0.480^a	**3.700^c**
***TN (mg/L)**	1.750^a	$1.050^{a,b}$	0.785^b	0.815^b	**3.700^c**
$^*PO_4^{3-}$ (mg/L)	0.093^a	0.083^a	0.015^b	0.078^a	**0.490^c**
***TP (mg/L)**	0.150^a	0.098^a	0.015^b	0.110^a	**0.470^c**

**Significantly different median at p<0.001 using Kruskal-Wallis. **Significantly different median p<0.05. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at p<0.05 within rows. If significant differences were found, the highest value is in bold.*

Septic

In general, the septic communities in this study have significantly higher organic nitrogen (NH_3, TKN) and phosphorus (TP, and PO_4^{3-}) than the sewer and reclaimed communities. This section compares the septic communities in the Beaches, Merritt Island, Suntree, and Turkey Creek regions to see if there are significant differences between regions (Table 21). We also conducted a PCA analysis on the septic wells to better understand the variance.

The data show there are more similarities than differences in septic communities across regions, with no single region exhibiting higher nutrient concentrations across the board. The one exception is that the Turkey Creek septic community has significantly higher TP and PO_4^{3-} concentrations than the other regions and the second highest NH_3 and TKN median concentrations, although the differences weren't significant. Merritt Island has significantly higher organic nitrogen (NO_X and TKN) than Beaches, but not Suntree or Turkey Creek. In contrast, the Beaches had significantly higher inorganic nitrogen (NO_X) than Merritt Island and Turkey Creek, but not Suntree. Suntree had significantly higher TN than the others.

Table 21: Statistical significance testing comparing septic communities in different regions.

Analyte	Beaches	Merritt Island	Suntree	Turkey Creek
***NH_3 (mg/L)**	0.074^{a}	**1.050^{b}**	$0.455^{a,b}$	$0.930^{a,b}$
***NO_X (mg/L)**	**0.855^{a}**	0.025^{b}	$0.620^{a,c}$	$0.037^{b,c}$
***TKN (mg/L)**	0.600^{a}	**1.600^{b}**	$0.930^{a,b}$	1.400^{b}
***TN (mg/L)**	1.550^{a}	1.700^{a}	**6.050^{b}**	4.800^{b}
***PO_4^{3-} (mg/L)**	0.410^{a}	0.365^{b}	0.265^{b}	**0.970^{c}**
***TP (mg/L)**	0.460^{a}	0.430^{a}	0.420^{a}	**0.970^{b}**

** Significantly different median at $p \leq 0.005$ using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at $p<0.05$ within rows. If significant differences were found, the highest value is in bold.*

Septic PCA

The PCA for the septic tank communities exhibits the pattern that we see in the complete dataset. Inorganic and PO_4^{3-}loading on one component (PC1) and organic nitrogen loading on the other (PC2). Unlike the aggregate dataset PCA, TN loads strongly on PC1, indicating that inorganic nitrogen is the driver of TN in septic communities. The 3-day rainfall data strongly load on factor 3, positively correlated with nutrient concentrations, suggesting high rainfall increased nitrogen concentrations in septic communities and decreased PO_4^{3-}.

Table 22 presents the loadings for the first four PCs, which account for 99.8% of the variability in the septic wells. The first two PCs account for 70.59% of total variability and can present a general view of the dominant nutrient forces driving the variation between the treatment types in the septic treatment wells. Figure 30 displays a plot of the first two septic treatment PCs with the points colored to represent treatment type.

Table 22: Loadings of six water quality variables on the first four PCs for the septic community groundwater samples.

	PC1	PC2	PC3	PC4
NH_3	0.01	0.99	0.10	-0.09
TKN	-0.03	0.99	0.12	-0.08
NOX	0.95	-0.23	0.12	-0.18
TN	0.96	0.09	0.16	-0.21
PO_4^{3-}	0.56	0.26	-0.32	0.72
3 Day Rainfall Sum	-0.10	-0.12	0.93	0.33
Variability (%)	35.7	34.9	17.2	12.0
Cumulative %	35.7	70.6	87.8	99.8

The septic treatment PC1 and PC2 biplot displayed in Figure 30 identifies a series of clusters that drive the variability in the dataset. The wells SP 1099, 1127, and 1739 all cluster along the organic nitrogen axis. The wells SP 6215 and 6398 are primarily along the NO_X axis. The well SP 250 trends between organic nitrogen and PO_4^{3-}. These clusters suggest high variability within the septic wells in each region. The differences in factor loading between inorganic and organic nitrogen suggests denitrification activity than varies across wells and within regions.

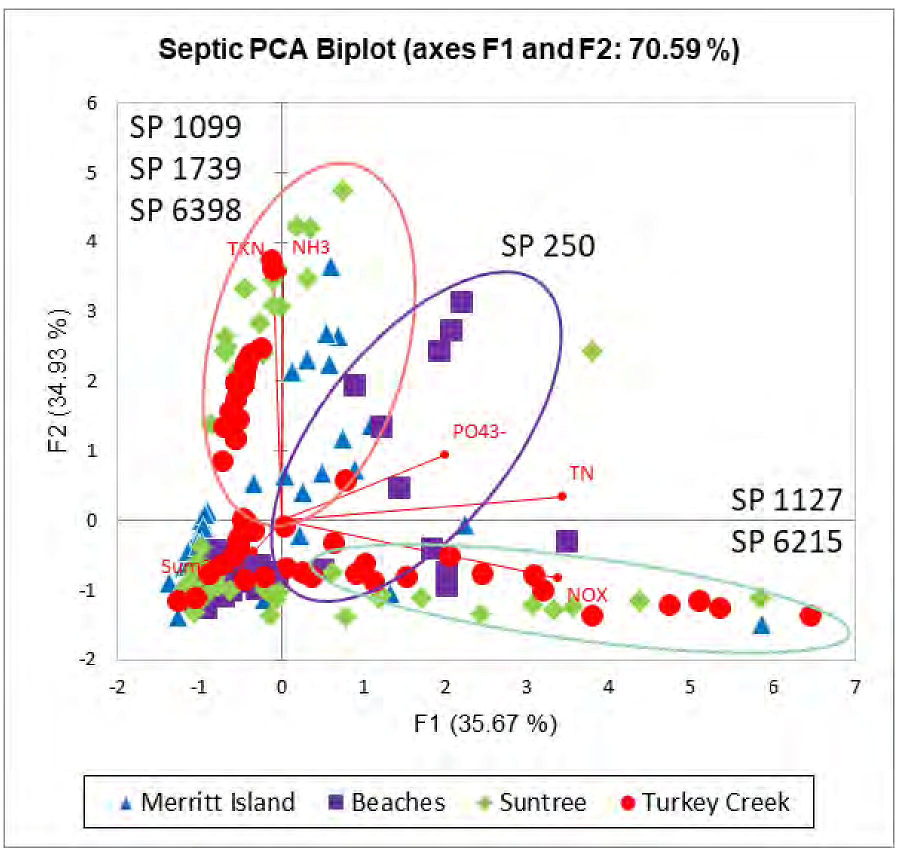

Figure 30: Coordinates of the septic areas PCs based on region type. Notable wells have been labeled and identified.

Reclaimed

The communities irrigated with reclaimed water receive a supply of nutrient rich water from wastewater treatment plants that are permitted to discharge different nutrient concentrations and volumes (Table 23) We see significant differences between reclaimed communities in this study that are likely related to plant discharge concentration and volume, although a direct correlation cannot be made without more specific area data.

Table 23: Reclaimed water source facilities and the annual TN and TP nutrient concentrations during the study period.

Facility	Region	Annual TN (mg/L)	Annual TP (mg/L)
South Central Regional WWTF	Suntree	6.7	0.88
South Beaches WWTF	Beaches	9.3	1.27
Osprey WWTF	Titusville	17.9	0.78
Palm Bay WWTF	Turkey Creek	29.4	1.4

One irrigation water sample was collected in the Turkey Creek and Titusville reclaimed communities, confirming that the majority of nitrogen coming out of the reclaimed is in the form of organic nitrogen (TKN & NH_3) (Table 24).

Table 24: Irrigation water samples nutrient concentrations

Location	NH_3	TN	TKN	NO_X	PO_4^{3-}
Titusville Irrigation	7.3	10.9	8.9	2	1.5
Turkey Creek Irrigation	28.6	28.9	27.1	1.8	NA

Countywide, in the aggregate dataset, the reclaimed treatment had significantly higher NO_X concentrations than septic and sewer (Table 25). Looking at the reclaimed communities in four regions: Beaches, Suntree, Titusville, and Turkey Creek, we see that the high NO_X concentration is driven by Turkey Creek and Beaches regions. This is interesting considering the irrigation water coming out of the two tested wastewater plants appears to be higher in organic nitrogen than NO_X. The explanation that the NO_X concentration is driven entirely by the irrigation water nutrient input only makes sense if high denitrification is occurring.

Turkey Creek's NO_X median concentration is significantly higher than the reclaimed communities in the other three regions and more than twice the next highest NO_X found in the Beaches. The NO_X in Turkey Creek is driving the TN value, which is also significantly higher than the other three treatments. We don't know what form of nitrogen is being discharged by the plant servicing Melbourne Beach.

In Suntree, we find significantly higher organic nitrogen concentrations (NH_3 and TKN, *$p<0.05$*). The South Central Regional WWTF that serves irrigation water to Suntree has the lowest TN discharge rate of the four treatment facilities serving our communities. Unfortunately, we were unable to collect an irrigation water sample in Suntree because the reclaimed water there was not available. Interviews with the management company confirmed that Suntree is not receiving adequate reclaimed water for irrigation, and thus, it is rarely used. The volume of reclaimed water being used in Suntree may be less than in other communities.

The Beaches region had significantly higher phosphorus (TP and PO_4^{3-}) concentrations than the other three regions, but you may recall that the Beaches natural area had significantly higher phosphorus concentrations than the natural areas located on the mainland. The South Beach WWTF that serves the reclaimed community in the Beaches also reported the second-highest TP discharge rate (1.27 mg/L).

Table 25: Statistical significance testing comparing the sewer communities in different study regions.

Analyte	Beaches	Suntree	Titusville	Turkey Creek
***NH_3 (mg/L)**	0.035^a	**0.575^b**	0.077^c	0.035^a
***NO_X (mg/L)**	6.250^a	0.051^b	0.029^b	**14.100^d**
***TKN (mg/L)**	0.086^a	**2.100^b**	1.150^c	0.086^d
***TN (mg/L)**	6.450^a	2.550^b	1.300^b	**14.100^c**
***PO_4^{3-} (mg/L)**	**0.750^a**	0.027^b	0.061^b	0.014^c
***TP (mg/L)**	**0.720^a**	0.036^b	0.110^b	0.012^c

** Significantly different median at $p \leq 0.005$ using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at $p<0.05$ within rows. If significant differences were found, the highest value is in bold.*

Reclaimed PCA

The PCA loading factors for the reclaimed communities show component PC1 has the inorganic nitrogen and PO_4^{3-} together with TN, indicating TN is driven mostly by NO_X in reclaimed communities. PC2 includes the organic nitrogen (TKN and NH_3). Interestingly, the 3-day rainfall data appears to have a negative relationship with nutrient concentrations, especially NH_3.

Table 26 below presents the loadings for the first four PCs, which account for 94.8% of the variability in the reclaimed wells. Figure 31 displays a plot of the first two components with the points colored to represent the treatment type.

Table 26: Loadings of six water quality variables on the first four PCs for the reclaimed community groundwater samples.

	PC1	PC2	PC3	PC4
NH_3	-0.67	0.53	-0.27	0.22
TKN	-0.74	0.46	-0.04	0.29
NO_X	0.90	0.42	-0.02	0.10
TN	0.82	0.54	-0.03	0.17
PO_4^{3-}	0.24	-0.54	-0.49	0.64
Sum3Day	-0.05	-0.08	0.89	0.44
Variability (%)	42.5	20.9	18.3	13.1
Cumulative %	42.5	63.5	81.8	94.8

In the PC1 and PC2 biplot, we can see that each region clusters in a different part of the plot. Turkey Creek and Beaches reclaimed communities line up along the inorganic factor, with the Beaches community also intersecting with the PO_4^{3-} line. As expected, the Suntree reclaimed community aligns with the organic nitrogen component. The Titusville reclaimed community wells are situated in the middle, with a majority of samples aligning with the organic nitrogen component and some outliers that show variability between inorganic and organic.

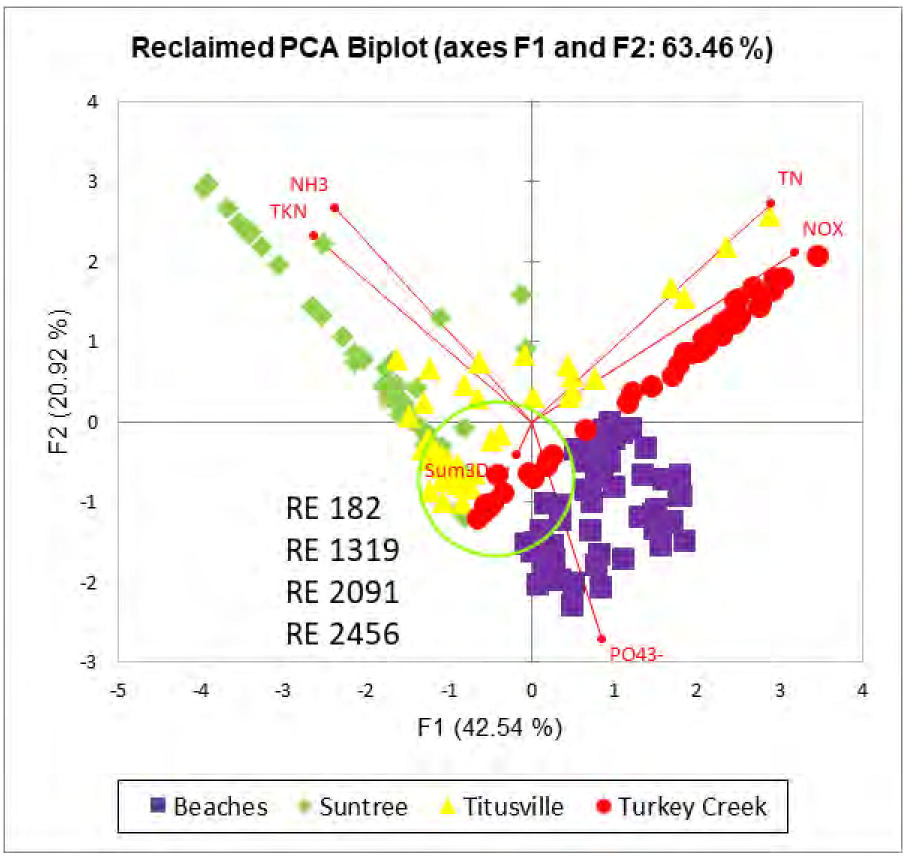

Figure 31: Coordinates of the reclaimed areas PCs based on region type. Notable wells have been identified.

Delineating Septic Plumes with Push Points

There were two objectives of the push point sampling:

1) Better understand septic tank effluent plume dynamics
2) Evaluate the accuracy of push points at measuring septic effluent relative to permanent monitoring wells

The data demonstrate that push points can be a useful tool for delineating the horizontal relative extent of septic plumes in a shallow aquifer system. Relative, as there were little similarities between push point and monitoring well data collected on the same day in the same location. Push points were consistently lower. This is likely due to the shallow portion of the aquifer that can be accessed with the push point method. The hand push method can only extend to a total depth of about 6', in some cases, barely reaching the top of the aquifer. Septic effluent is denser than groundwater and as it is released from the septic drain field, it will sink vertically through the aquifer as it moves horizontally. In Brevard County, where the surficial aquifer extends 50-100 feet before reaching an intermediate confining layer, it is hard to predict the extent of vertical migration.

Results of TN and TP for the push point sampling efforts are grouped by well and presented in the figures. Contours that delineate areas of similar nutrient concentration provide a picture of the plume extent. One push point sample was collected immediately adjacent to the permanent monitoring well at each site, as a means to compare the results. The push point groundwater samples have consistently lower concentrations than the monitoring well samples collected at the same time. This suggests that push points are a good screening tool and may be compared with each other if they reach the same portion of the aquifer, but they may not provide an accurate assessment of concentration at a site.

MW SP 1739

A total of 15 push point samples were collected in or around the Merritt Island septic monitoring well MW SP 1739. TN concentrations in the push points ranged from 0.45 mg/L to 55.80 mg/L, with ten (66%) of the push points having concentrations lower than those measured at the MW SP 1739 (Figure 32). The TN and TP concentrations in the push point sample located closest to the monitoring well (PP30) was roughly half that measured in the permanent well.

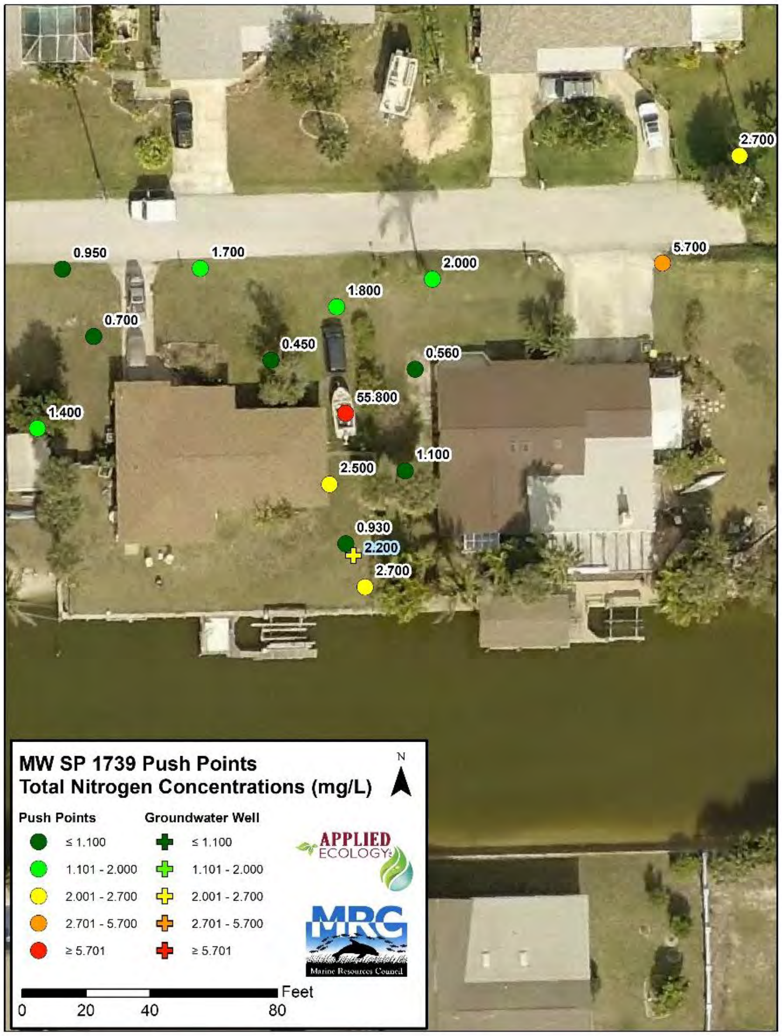

Figure 32: TN concentrations of the push point locations (circles) and Merritt Island septic monitoring well MW SP 1739.

Up gradient of the permanent well, one push point had extremely high TN (55.80 mg/L) along with high fecal coliform counts (between 200 CFUs/100 mL to too numerous to count). We suspect that this push point was situated closest to the septic drain field. Two additional push points near the monitoring well also had relatively high TN concentration data (2.50 mg/L for PP229 and 2.70 mg/L for PP31). Push points along the road easement had lower concentrations in general, with the exception of one in the neighbor's yard that is likely picking up another source.

TP concentrations in the push points ranged from 0.014 to 1.1 mg/L, which were considerably less than the measured TP in the permanent monitoring well (2.4 mg/L) (Figure 33). The high TN push point also had very high TP (8.40 mg/L).

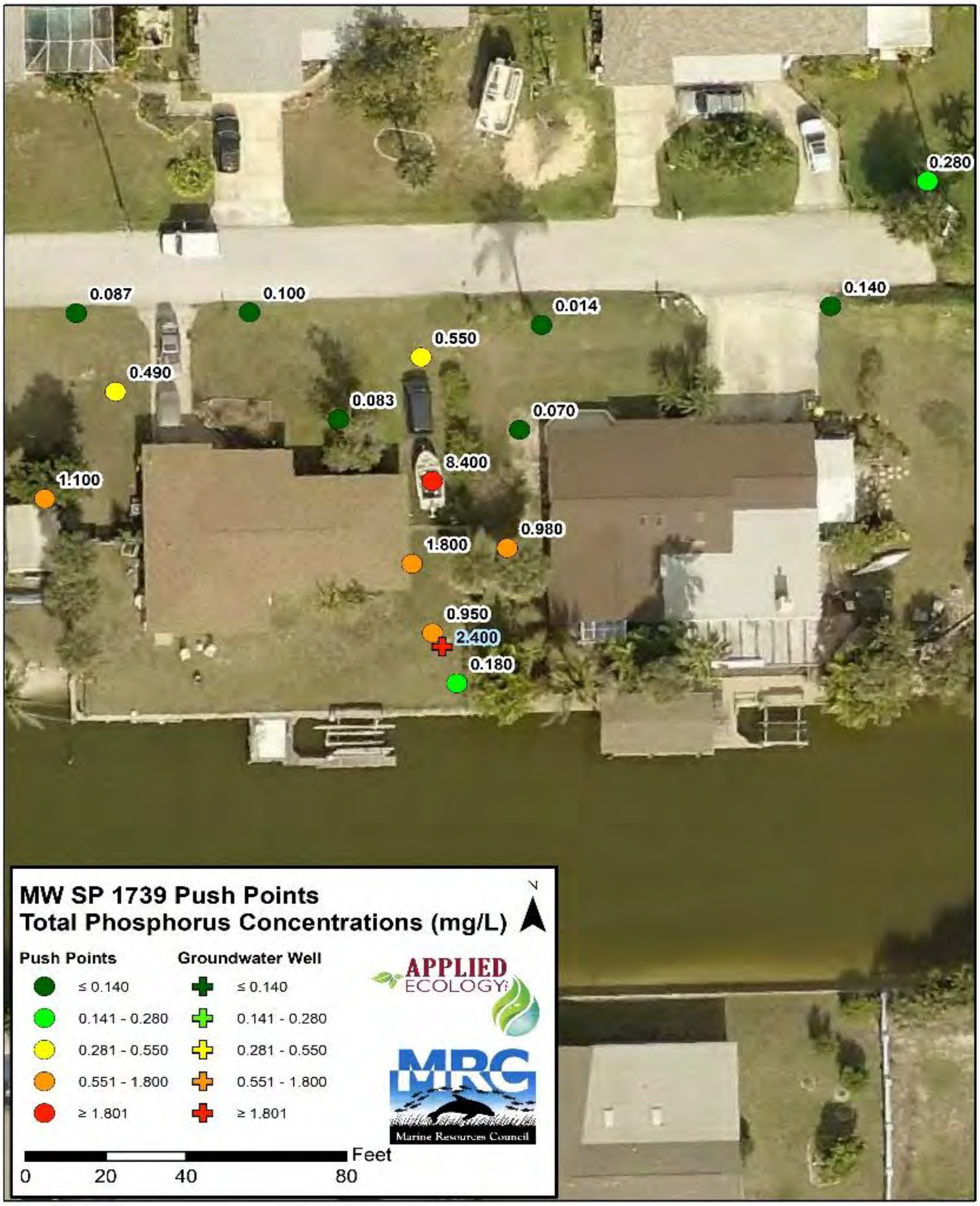

Figure 33: TP concentrations of the push point locations (circles) surrounding MW SP 1739. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

Although the push point data are not as accurate as the monitoring well data, these relatively inexpensive screening measures can be compared with each other to create contour maps of plume dynamics at the site. The following contour maps provide general plume contours for the different nitrogen organic and inorganic compounds and TP (Figure 34-Figure 37). In this septic community, two different nitrogen plumes appear, indicative of multiple sources.

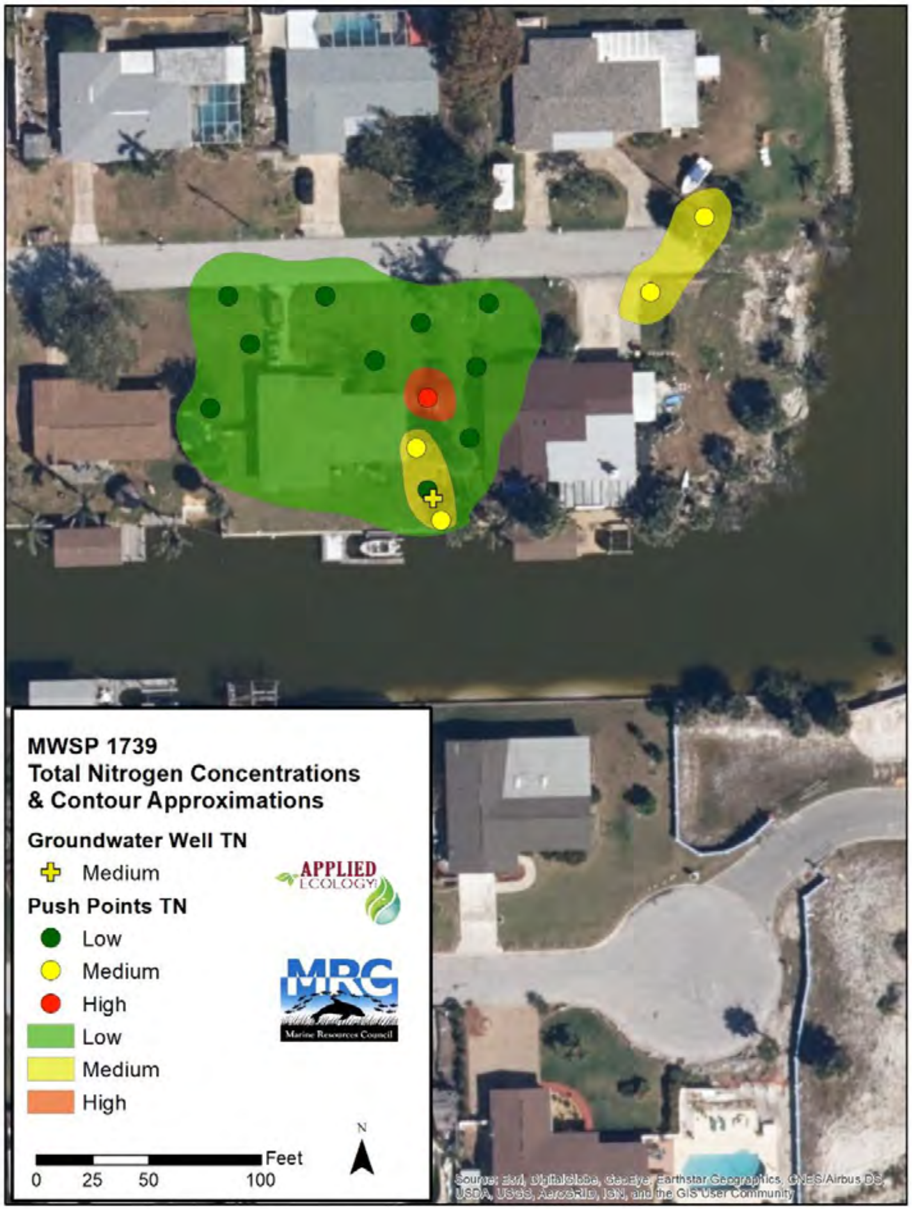

Figure 34: Total nitrogen (TN) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1739. Additionally, the TN concentration of the monitoring well itself is also mapped (plus sign).

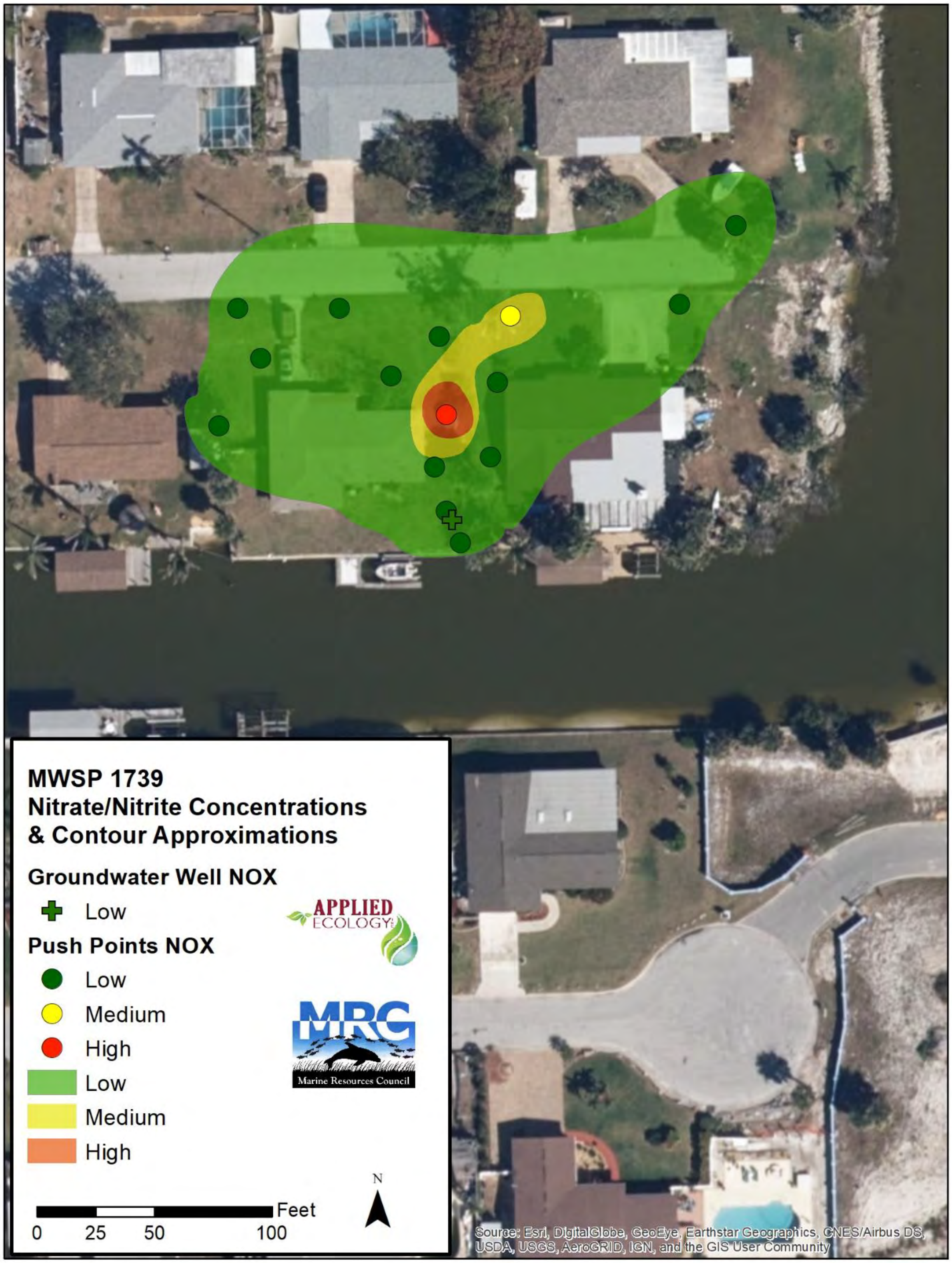

Figure 35: Nitrate/nitrite (NO_X) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1739. Additionally, the NO_X concentration of the monitoring well itself is also mapped (plus sign).

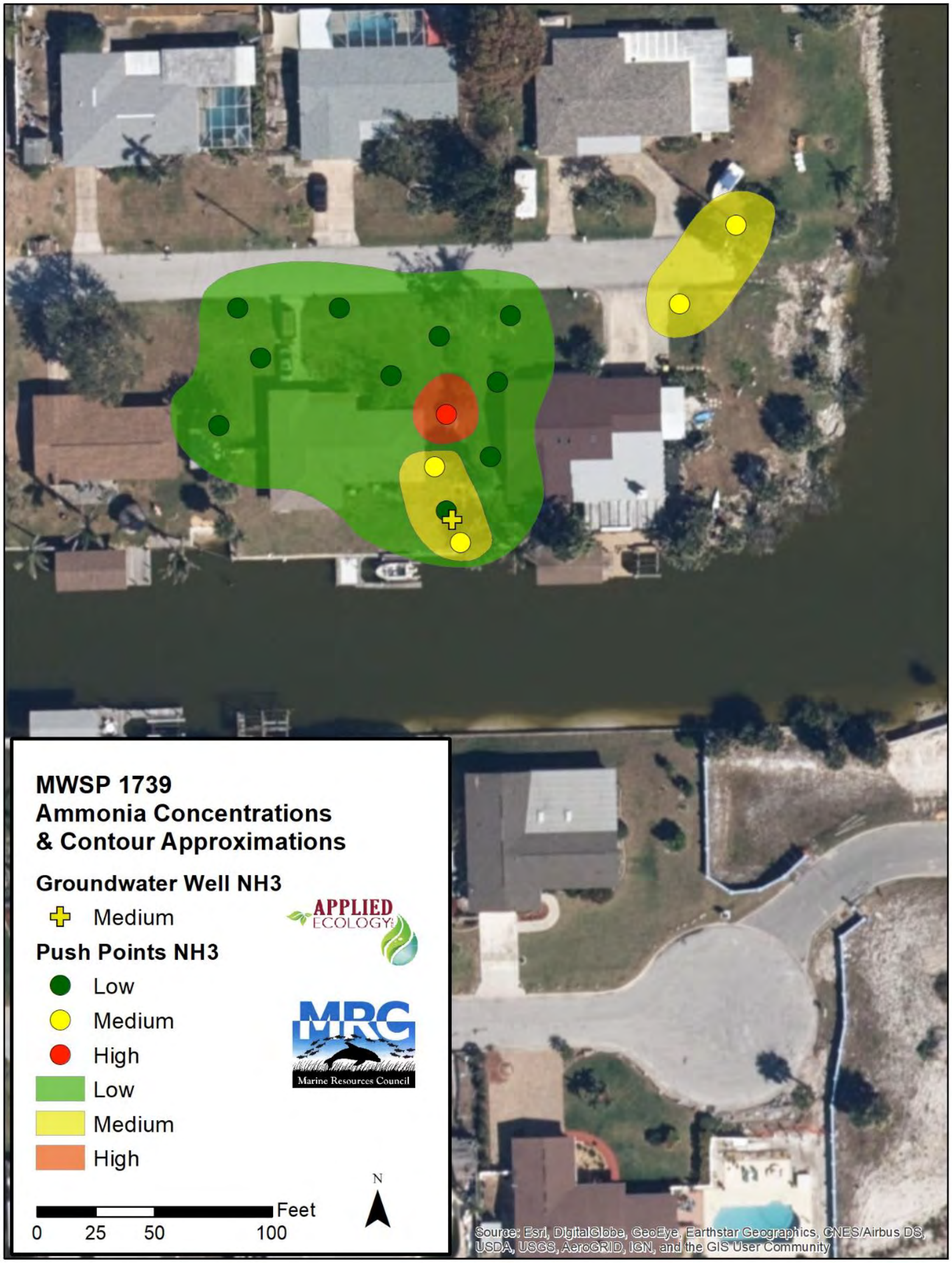

Figure 36: Ammonia (NH_3) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1739. Additionally, the NH_3 concentration of the monitoring well itself is also mapped (plus sign).

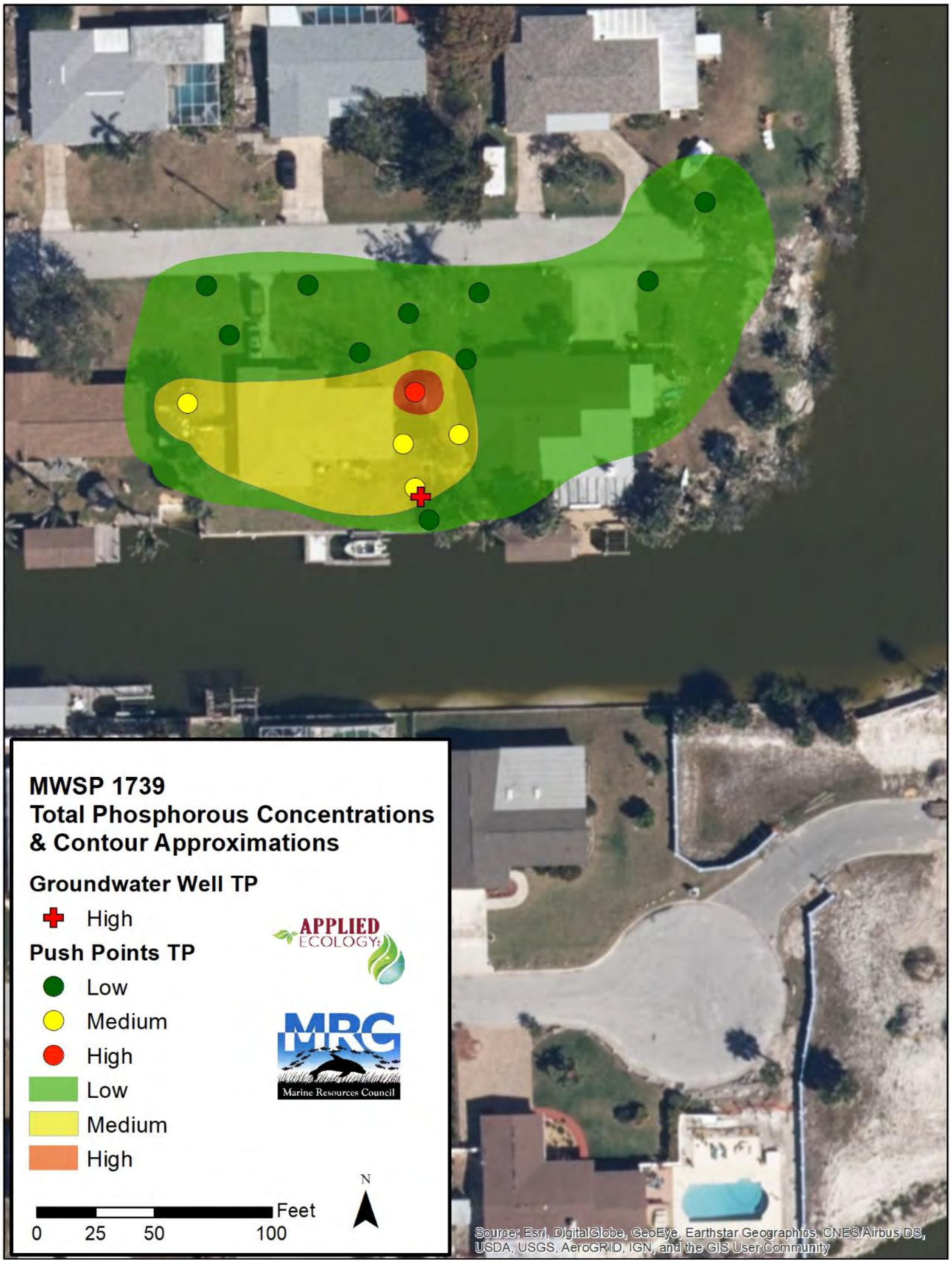

Figure 37: Total phosphorus (TP) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1739. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

MW SP 6398

A total of 15 push point samples collected near Suntree septic well MW SP 6398 (Figure 38). Push point TN concentrations ranged from 0.38 mg/L to 12.80 mg/L, considerably lower than the TN measured in the permanent well (30.3 mg/L) on the same date. Overall, higher TN data are spatially located near the well and downgradient along the swale between the two properties. Nitrogen contour maps show two different areas along the seawall where nitrate concentrations are relatively high.

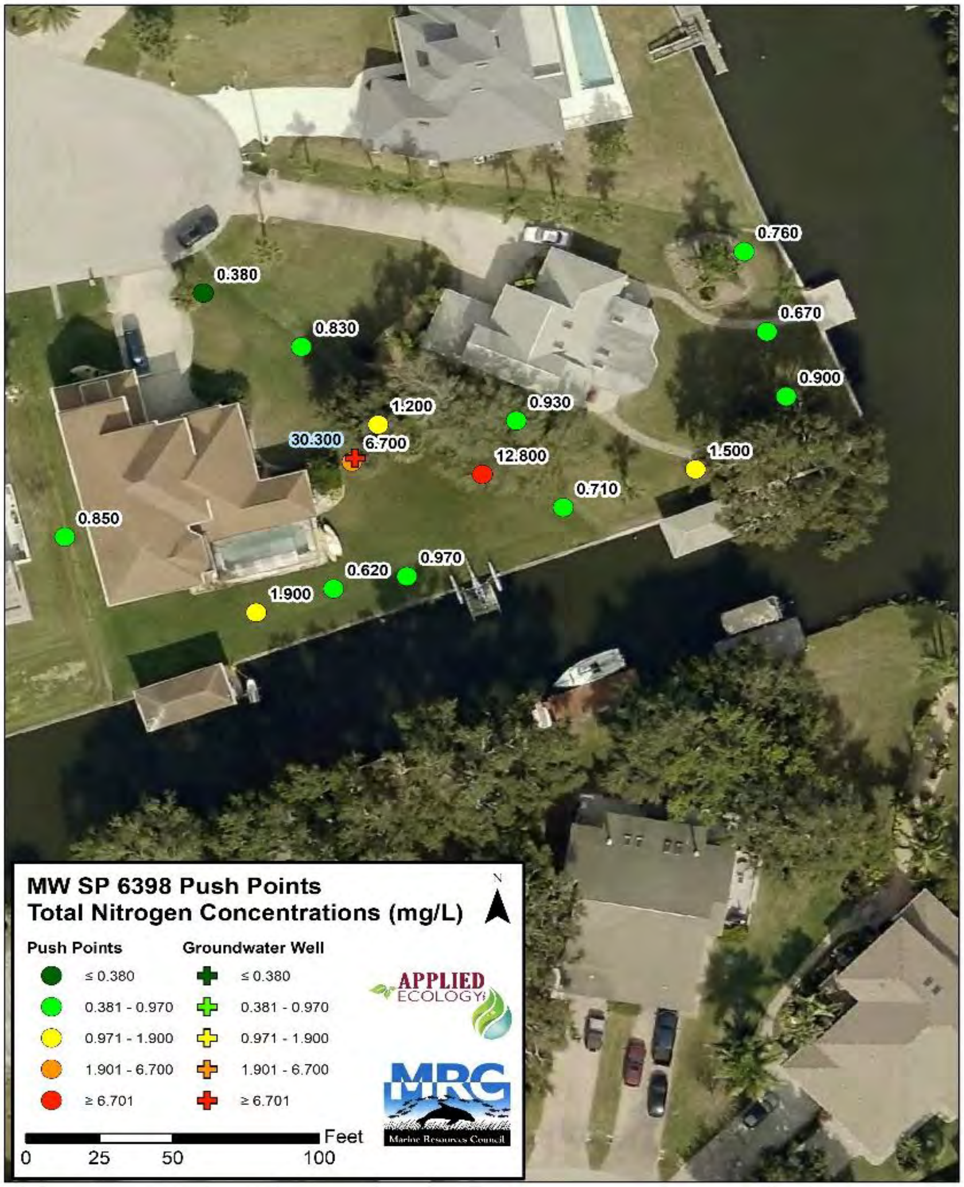

Figure 38. TN concentrations of the push points and Suntree septic monitoring well MW SP 6398.

Push point TP concentrations ranged from 0.38 to 2.3 mg/L (Figure 39), with the permanent well TP measuring 0.42 mg/L. In this case, almost half of the push points have a higher TP concentration than the monitoring well.

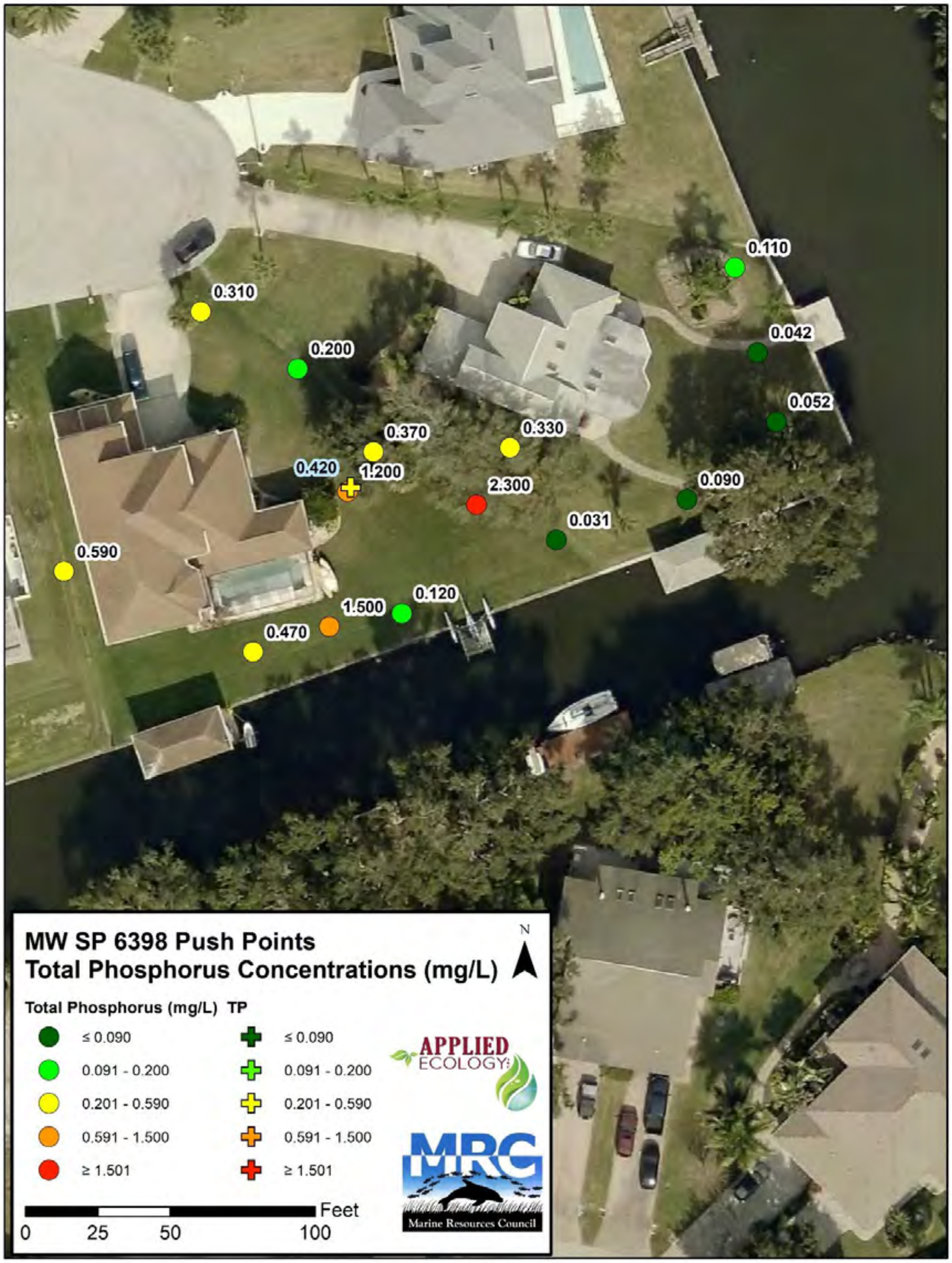

Figure 39. TP concentrations of the push points and Suntree septic monitoring well MW SP 6398.

Contour maps for inorganic and organic nitrogen compounds and TP are provided to show the relative area of contamination for each in Figure 40 - Figure 43.

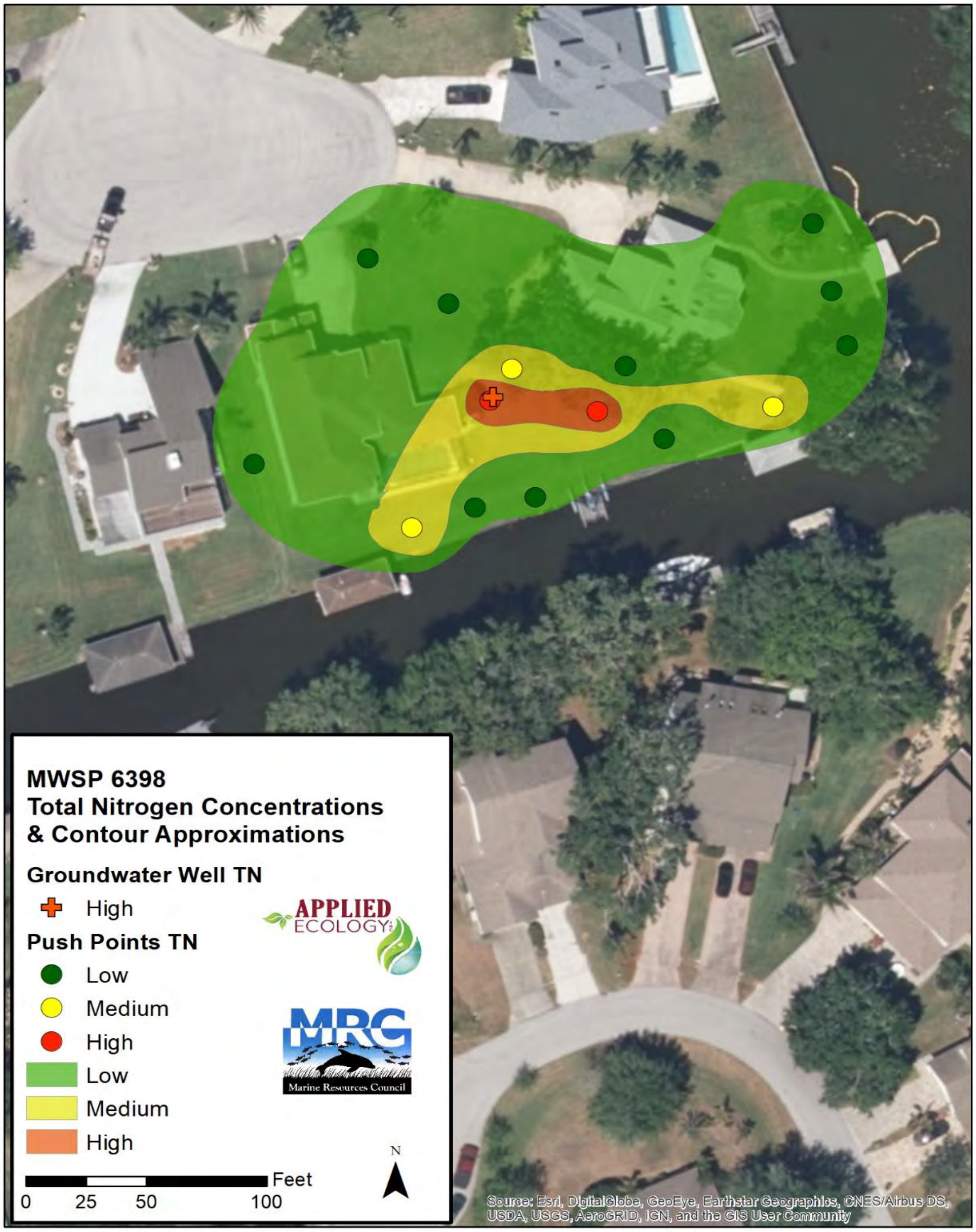

Figure 40: Total nitrogen (TN) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6398. Additionally, the TN concentration of the monitoring well itself is also mapped (plus sign).

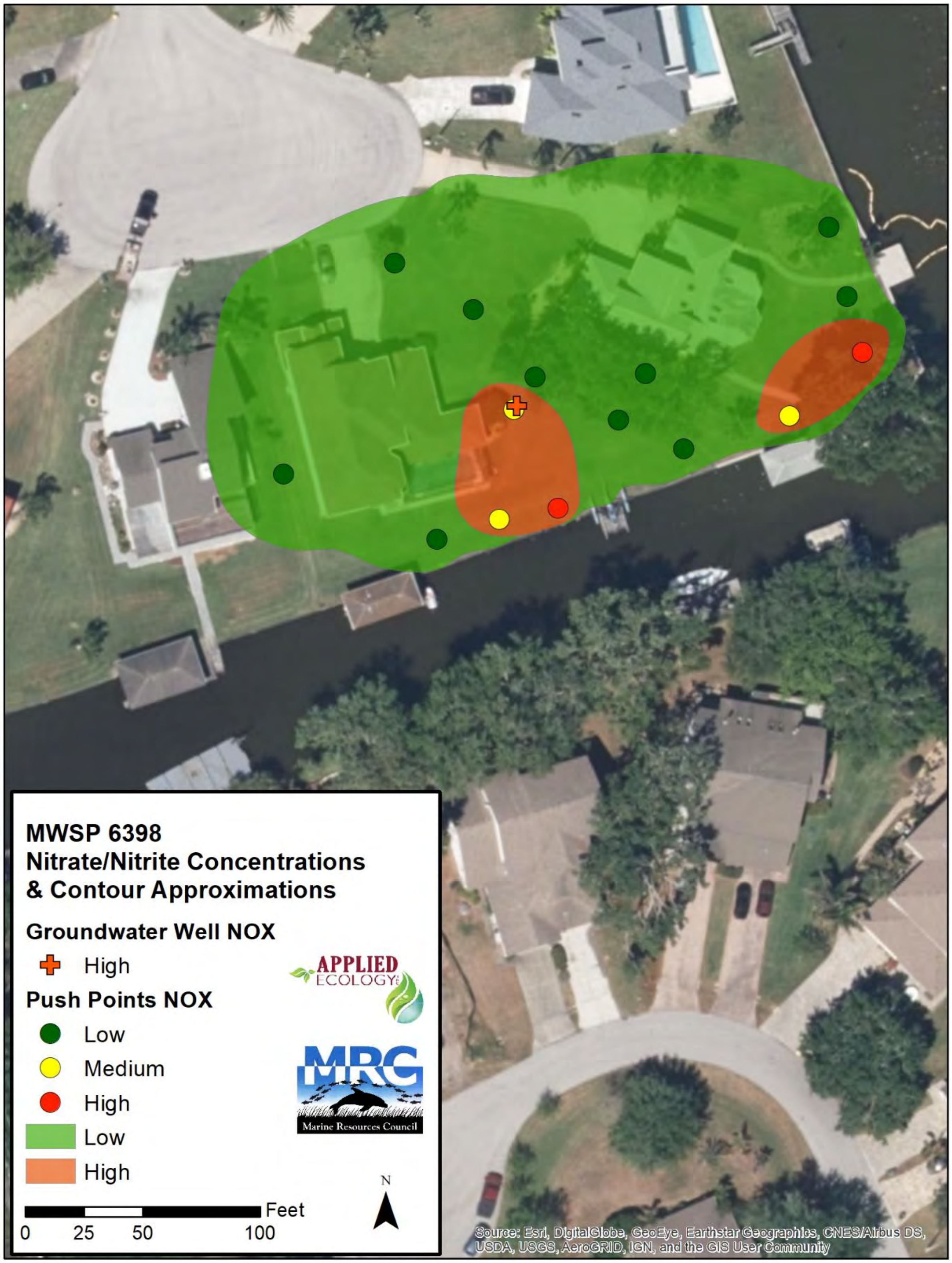

Figure 41: Nitrate/nitrite (NO_X) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6398. Additionally, the NO_X concentration of the monitoring well itself is also mapped (plus sign).

Figure 42: Ammonia (NH_3) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6398. Additionally, the NO_X concentration of the monitoring well itself is also mapped (plus sign).

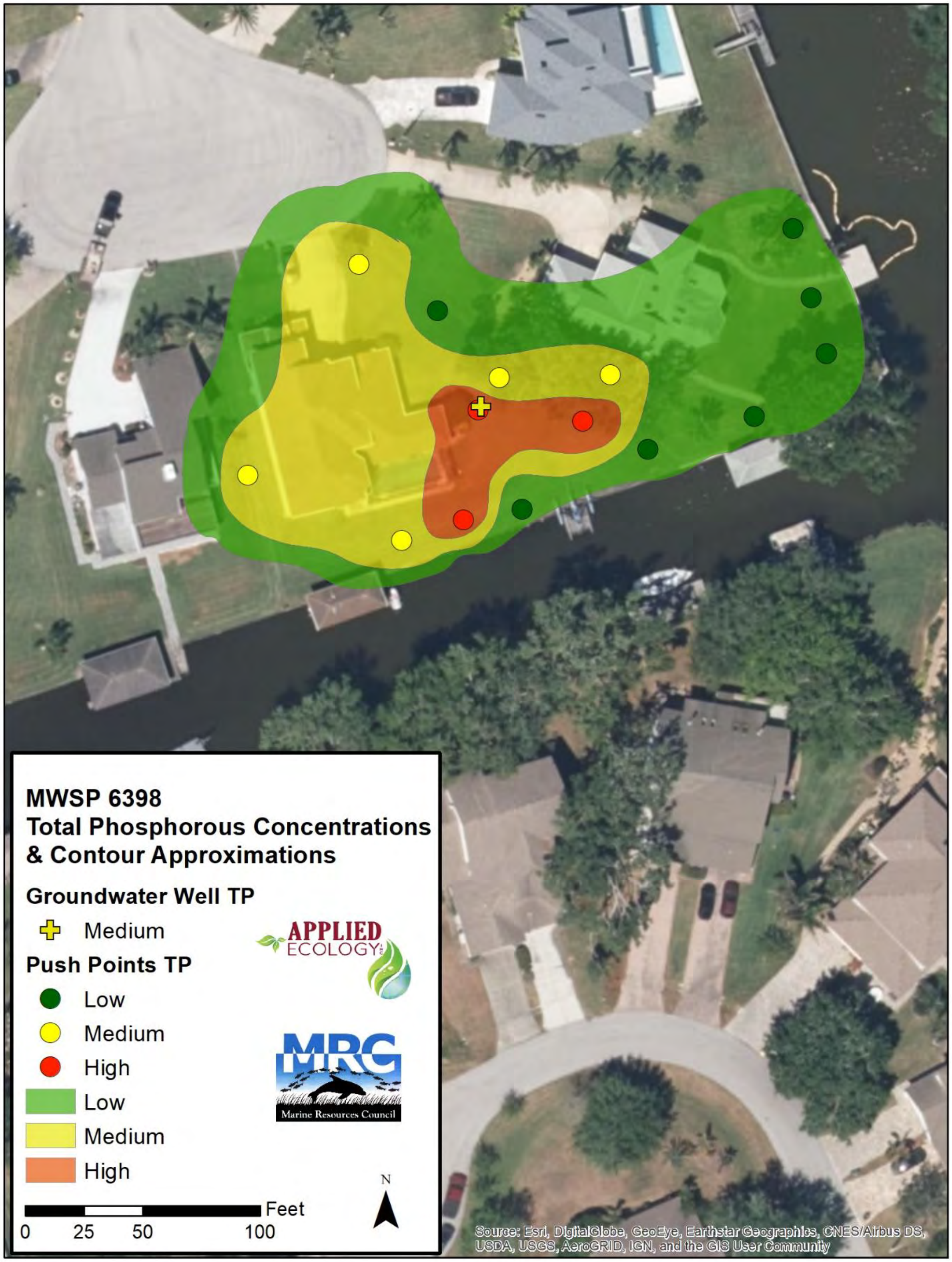

Figure 43: Total phosphorus (TP) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6398. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

MW SP 6215

There were 11 push point samples collected in or around the property Subtree septic monitoring well MW SP 6215. TN concentrations ranged from 0.42 mg/L to 25.8 mg/L in the push points and TN was 25.6 mg/L in the permanent monitoring well, providing the closest measurement between the permanent well and adjacent push point (Figure 44). Contour maps show the nitrogen and phosphorus plumes heading into the Lagoon (Figure 46-Figure 49).

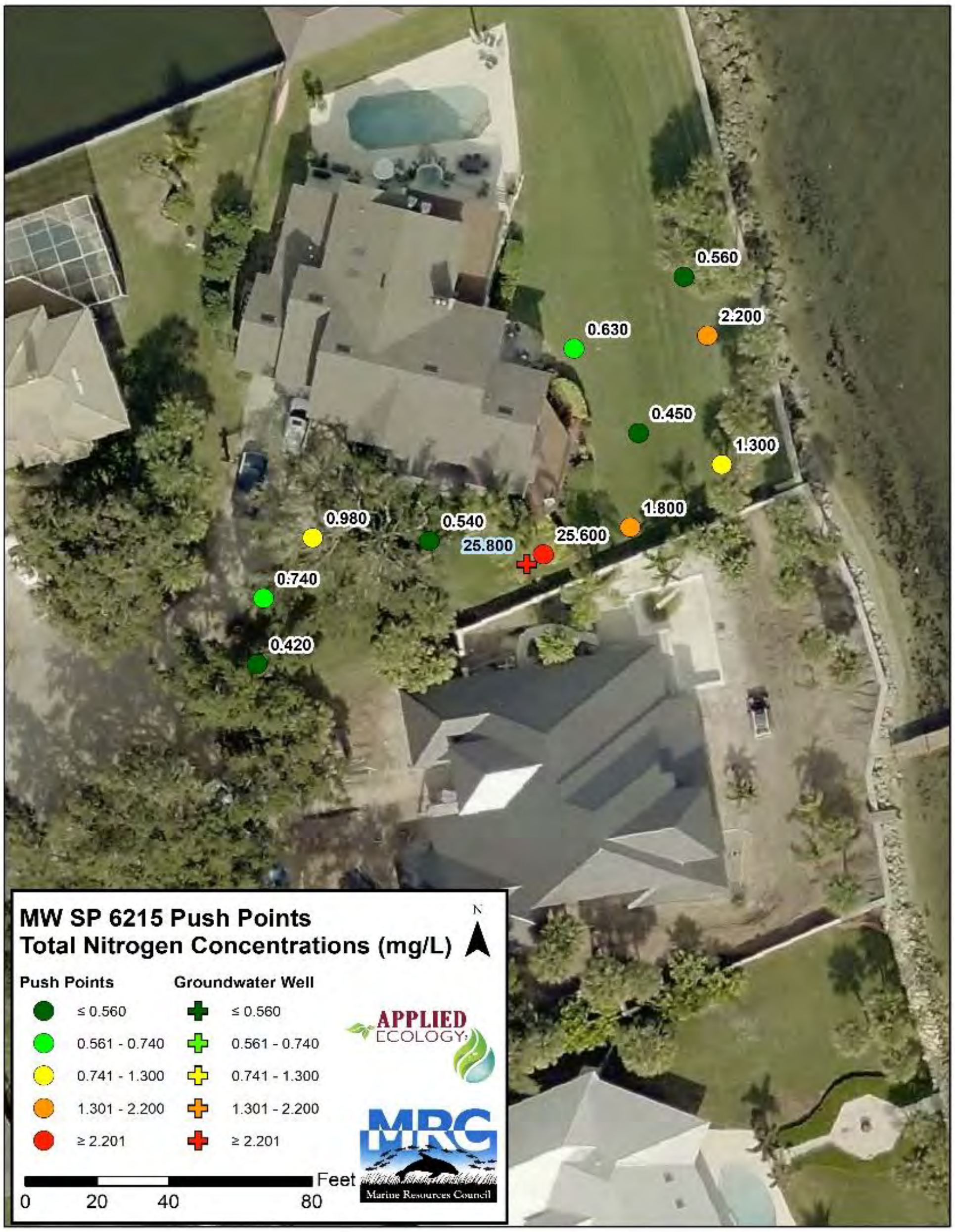

Figure 44. TN concentrations of the push point locations (circles) surrounding MW SP 6215. Additionally, the TN concentration of the monitoring well itself is also mapped (plus sign).

TP concentrations in the push points ranged between 0.012 to 0.380 mg/L and the monitoring well TP concentration was 1.3 mg/L demonstrating that the push points concentrations underrepresent the conditions (Figure 45).

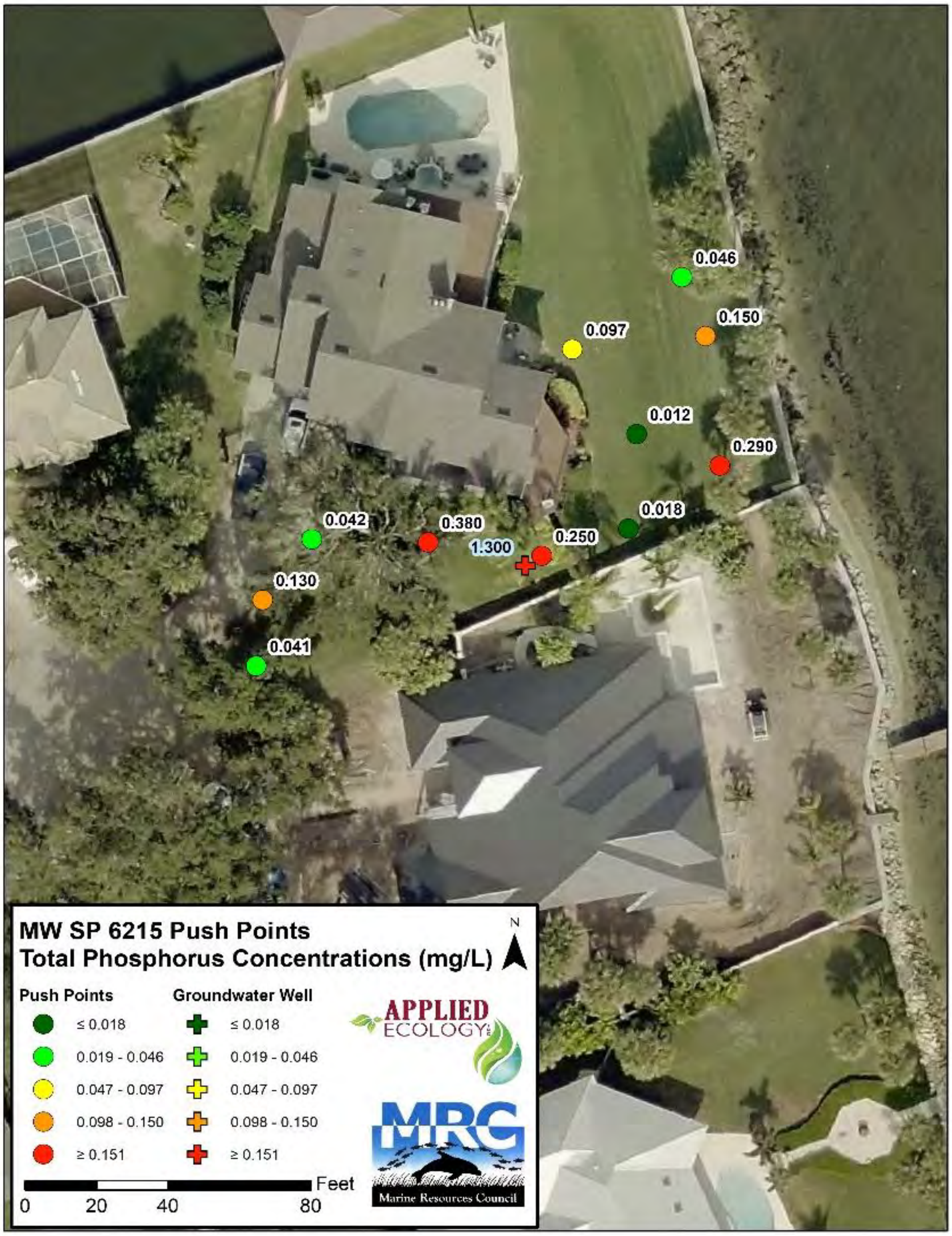

Figure 45. TP concentrations of the push point locations (circles) surrounding MW SP 6215. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

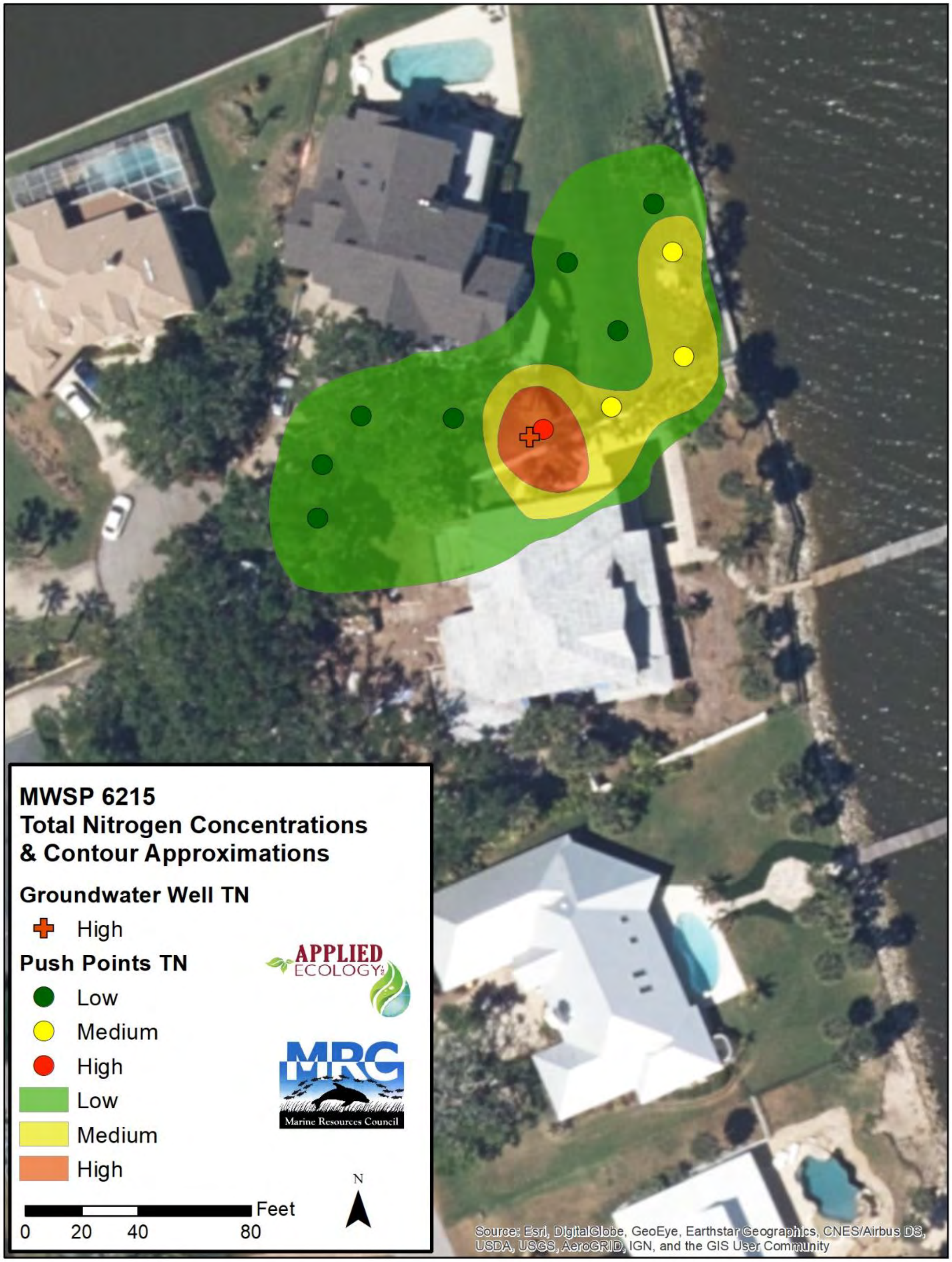

Figure 46: Total nitrogen (TN) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6215. Additionally, the TN concentration of the monitoring well itself is also mapped (plus sign).

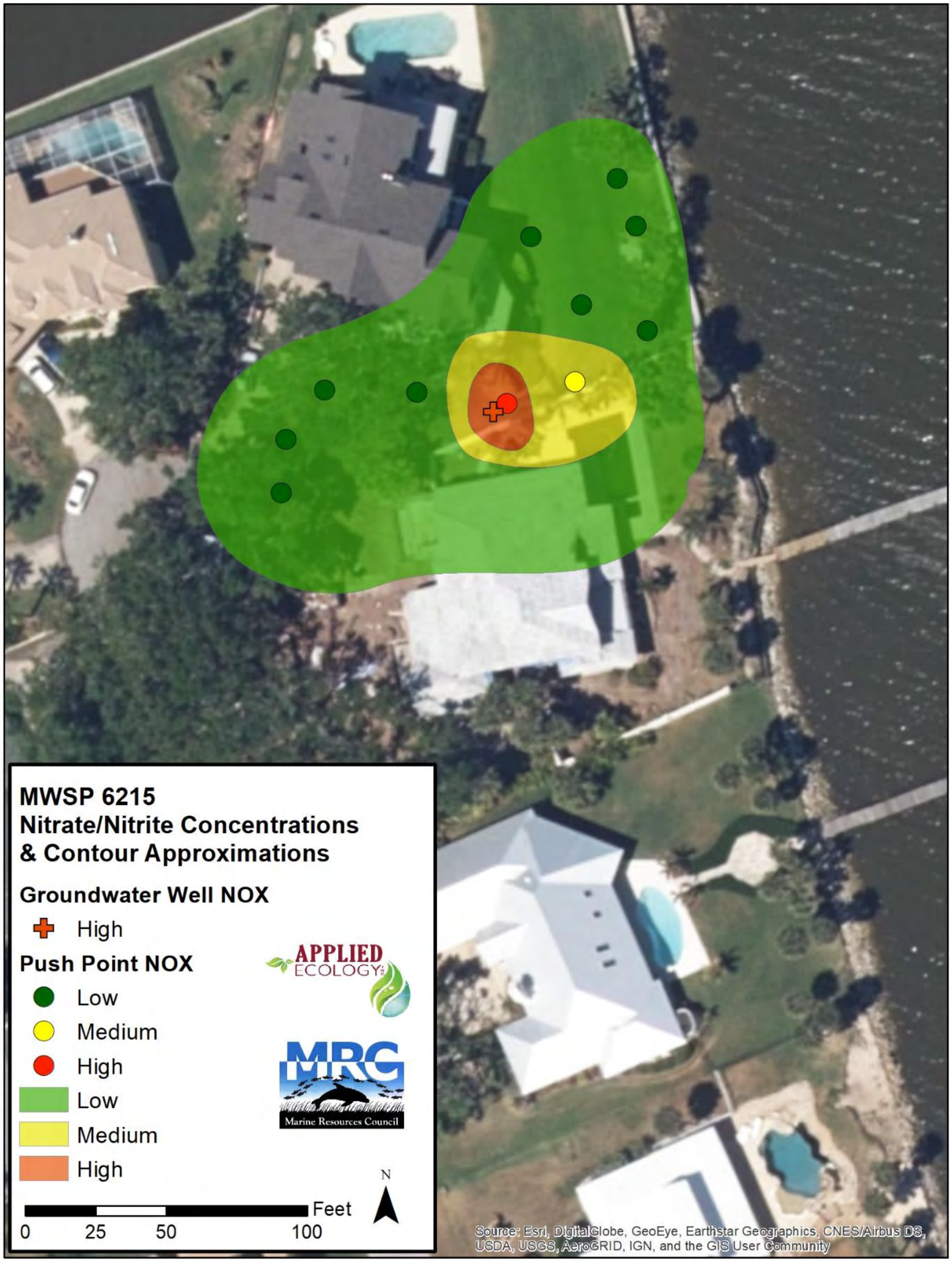

Figure 47: Nitrate/nitrite (NO_X) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6215. Additionally, the NO_X concentration of the monitoring well itself is also mapped (plus sign).

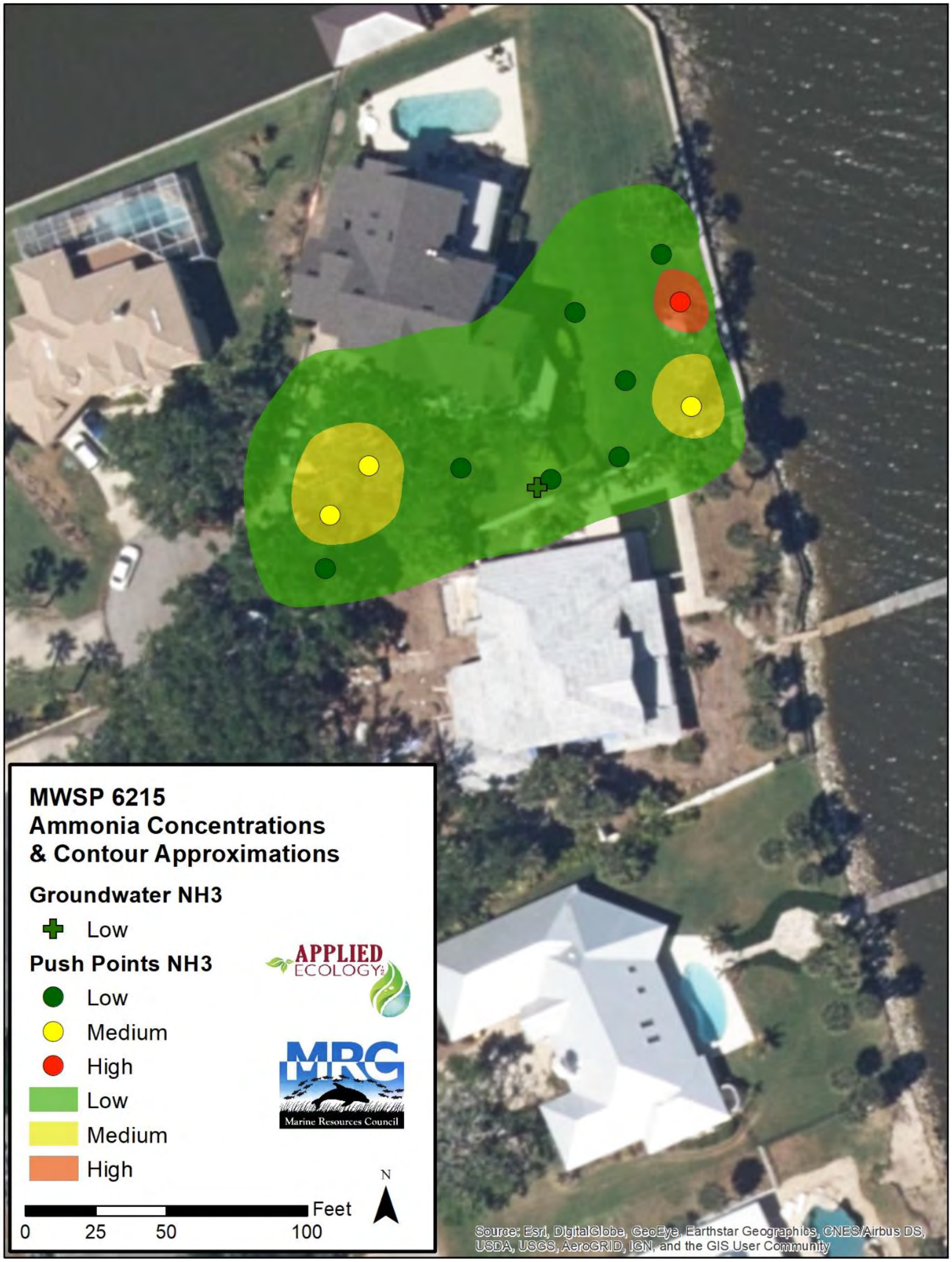

Figure 48: Ammonia (NH_3) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6215. Additionally, the NH_3 concentration of the monitoring well itself is also mapped (plus sign).

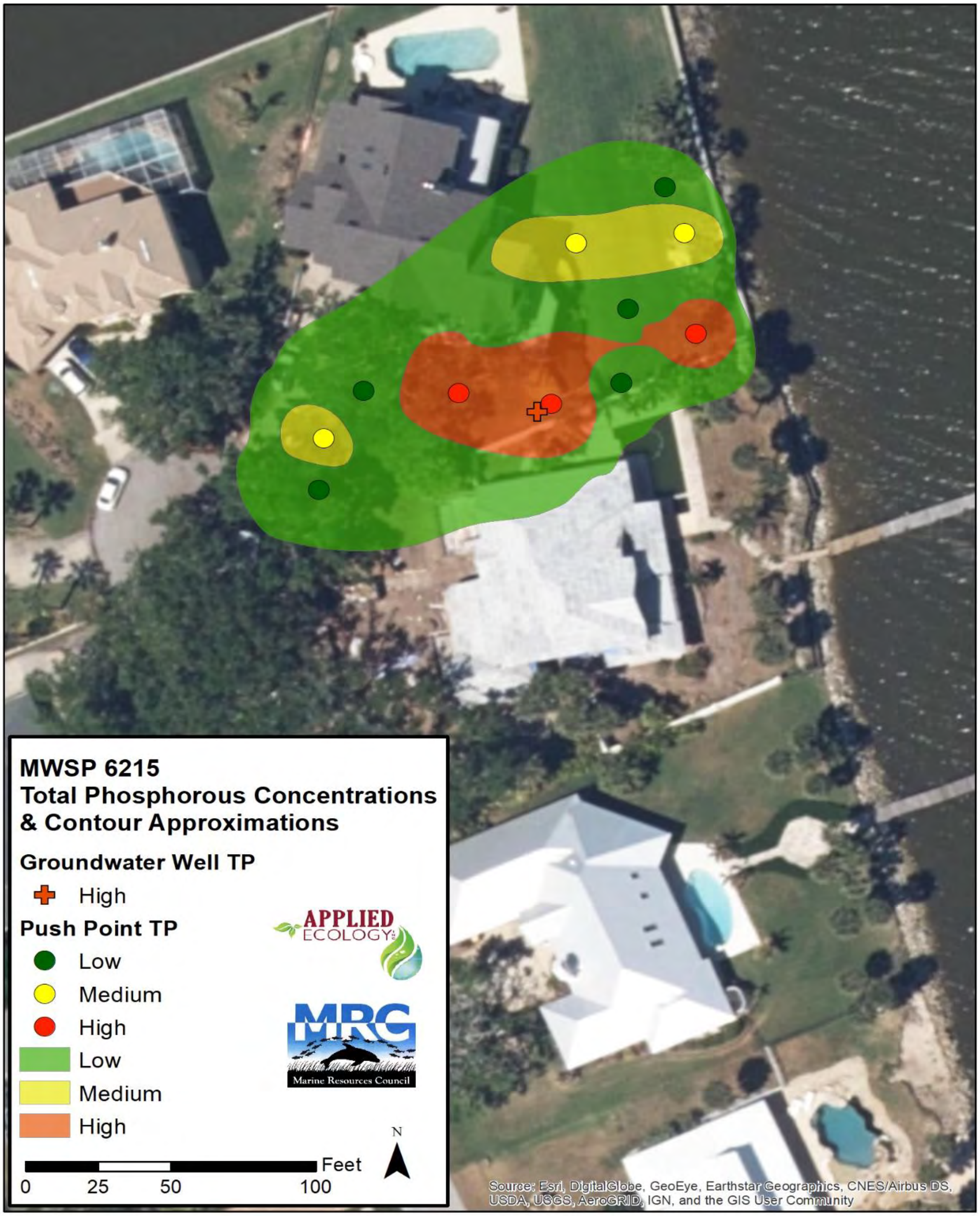

Figure 49: Total phosphorus (TP) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 6215. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

MW SP 1127

A total of 12 push point samples were collected around Turkey Creek monitoring well MW SP 1127 (Figure 50). Measured TN concentrations for the push points ranged from 0.2 to 2.8 mg/L and the TN concentration in the permanent well was 5.70 mg/L. The push point sample closest to the monitoring well had the lowest measured TN. Many of push points on the eastern side of the property had similar concentrations; this could confirm the plume directionality towards the finger canal located west/northwest from the drain field location.

Figure 50. TN concentrations of the push point and Turkey Creek monitoring well MW SP 1127.

TP push point concentrations ranged from 0.19 to 1.1 mg/L. and the TP concentration in the monitoring well was 1.3 mg/L (Figure 51) with the closest push point most similar to the well concentration. The distribution of measured TP data across the site appears to closely mimic the one described for the TN concentration data (Figure 51).

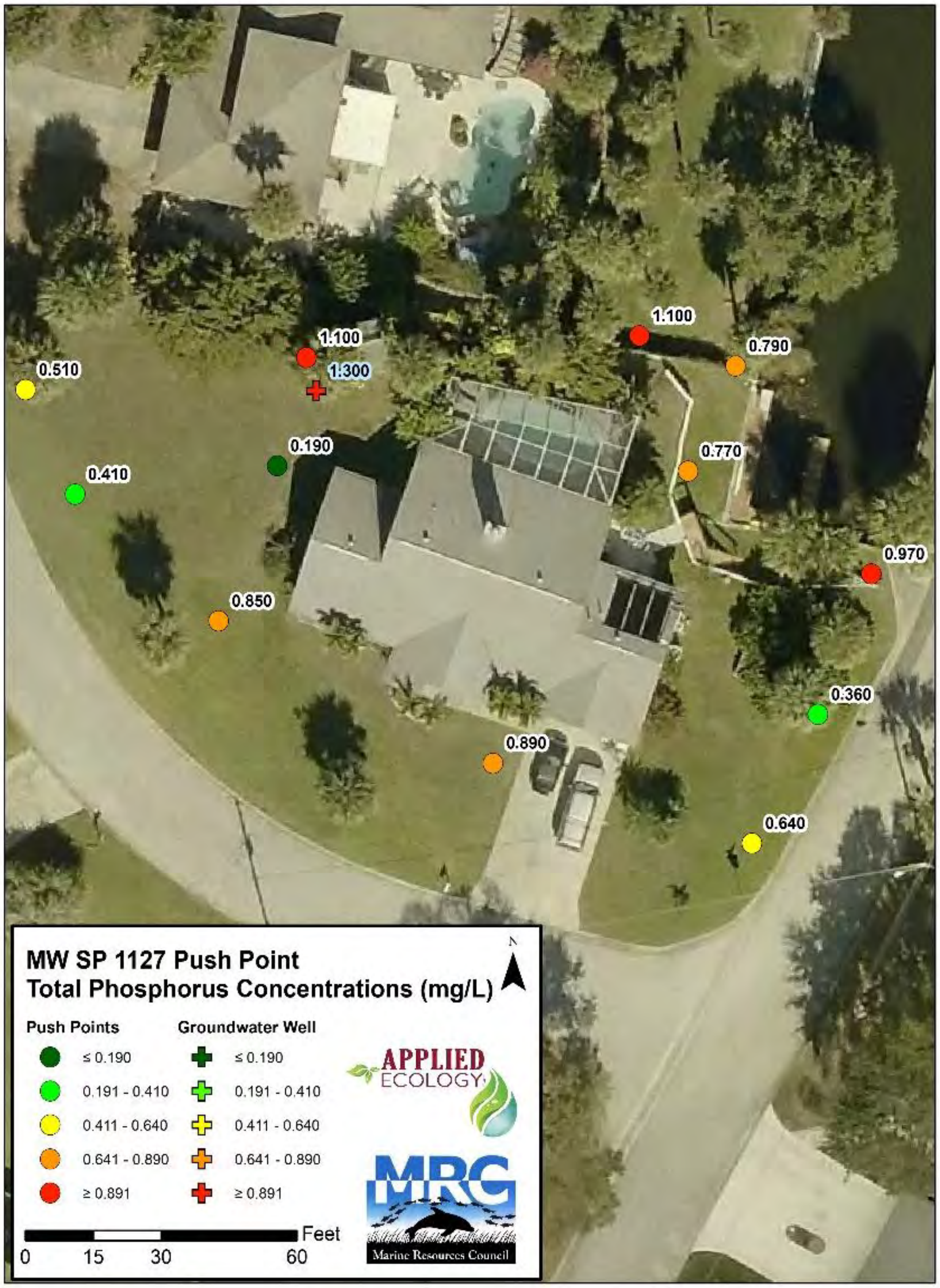

Figure 51. TP concentrations of the push point locations (circles) surrounding MW SP 1127. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

Contour maps show the nitrogen and phosphorus plumes for MW SP 1127 (Figure 52-Figure 55).

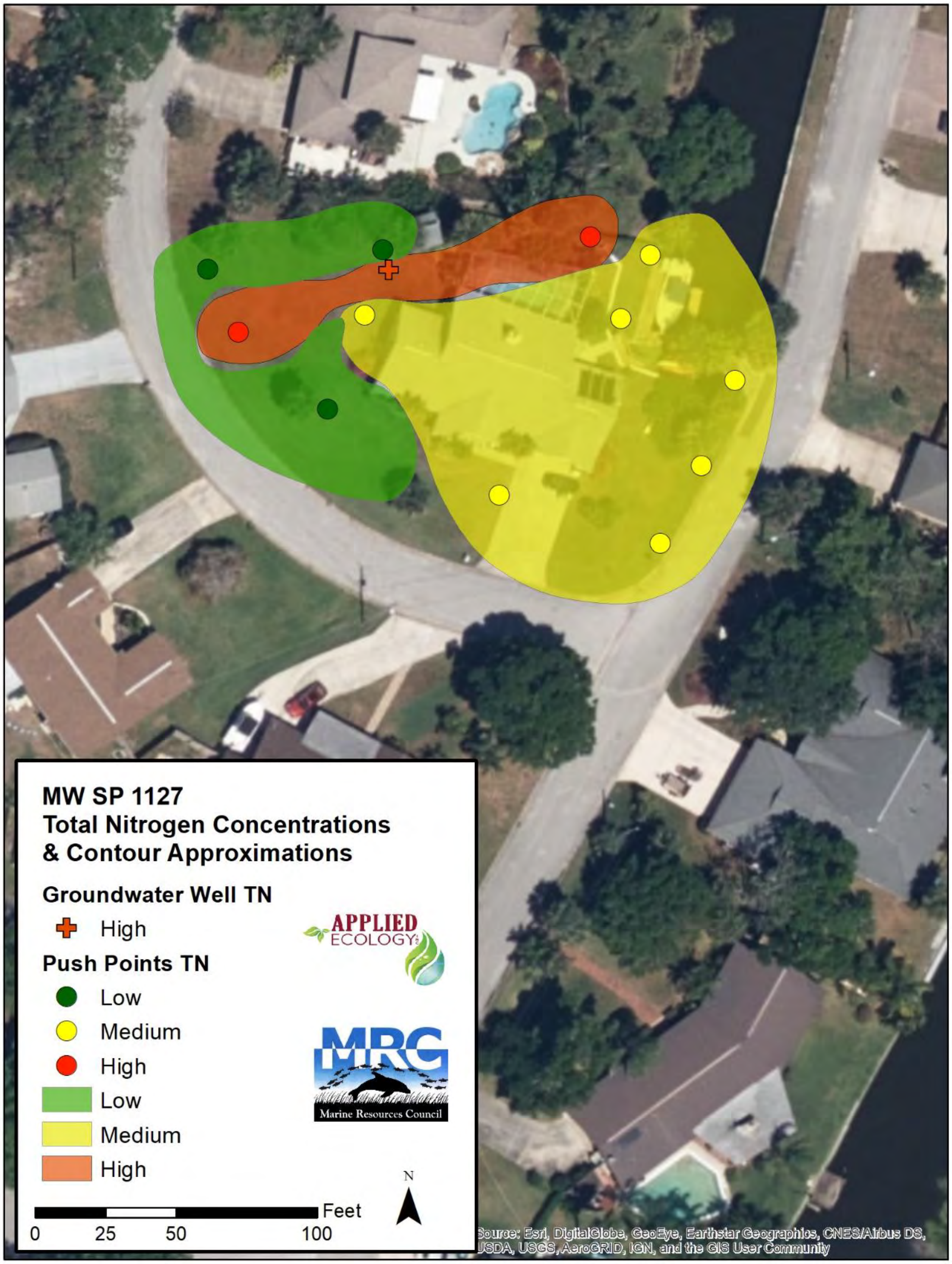

Figure 52: Total nitrogen (TN) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1127. Additionally, the TN concentration of the monitoring well itself is also mapped (plus sign).

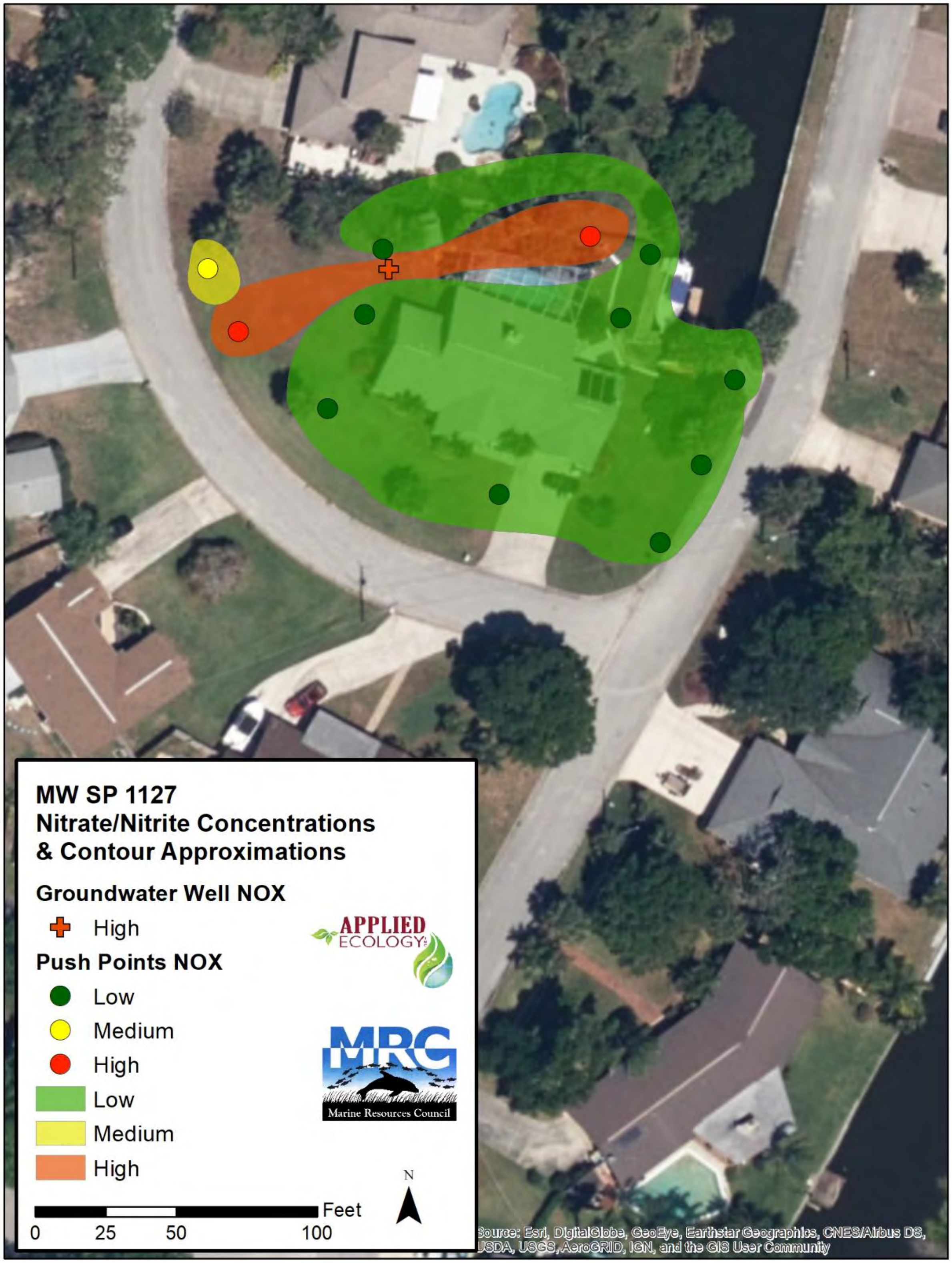

Figure 53: Nitrate/nitrite (NO_X) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1127. Additionally, the NO_X concentration of the monitoring well itself is also mapped (plus sign).

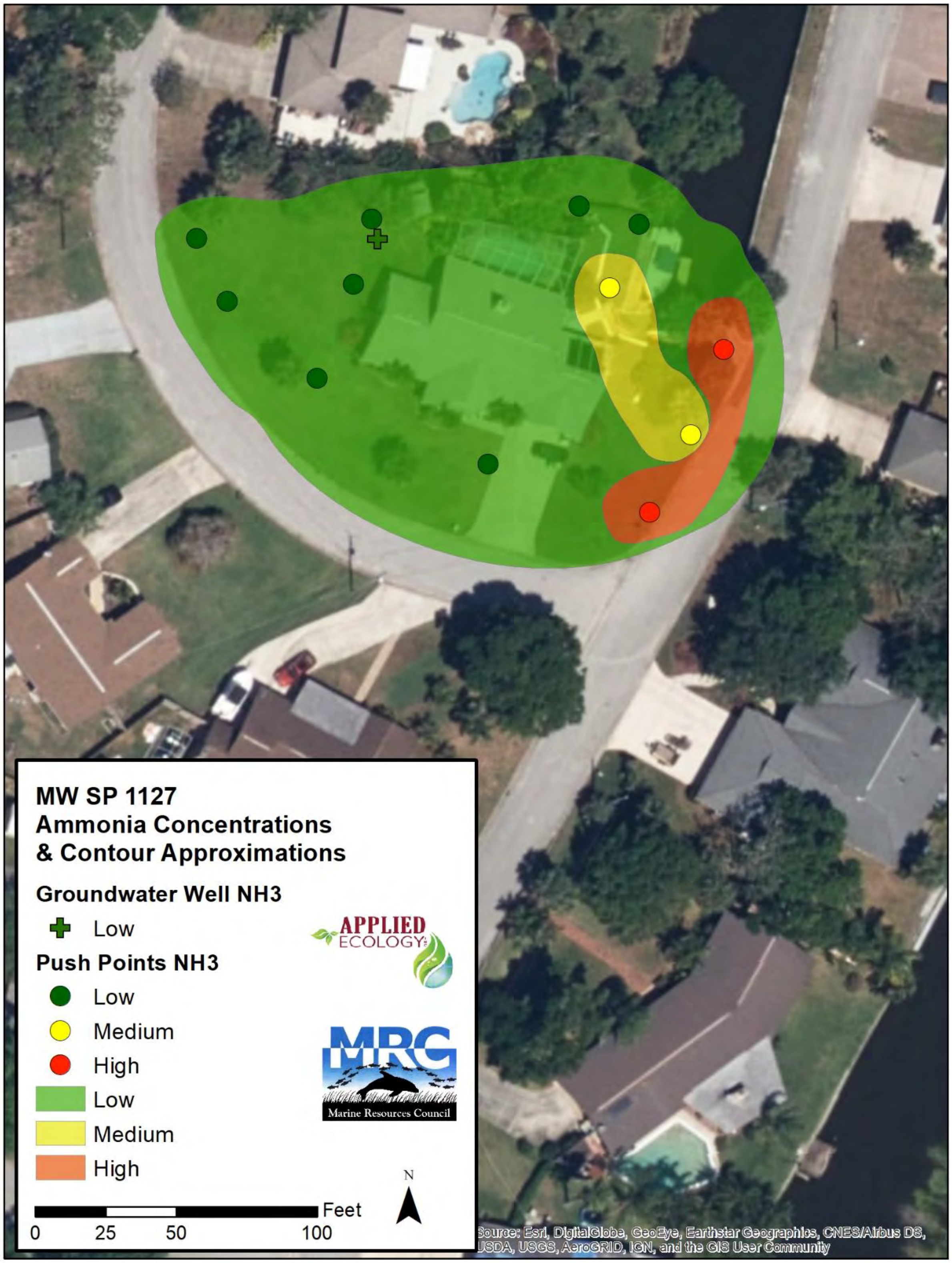

Figure 54: Ammonia (NH_3) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1127. Additionally, the NH_3 concentration of the monitoring well itself is also mapped (plus sign).

Figure 55: Total phosphorus (TP) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1127. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

MW SP 1099

Nine push points samples were collected around Turkey Creek septic well MW SP 1099. TN concentrations in the push points ranged from 0.63 mg/L to 54.6 mg/L, and the monitoring well TN was 5.7 mg/L (Figure 56). The push point with the highest TN concentration was located immediately adjacent to the septic tank drain field located in the front yard of the house In terms of spatial distribution, it appears that groundwater may flow directly east across the property towards the canal leading into Turkey Creek. A push point located upgradient of the drain field that measured 7.1 mg/L of TN may be showing an upgradient source of N and P that is impacting this property.

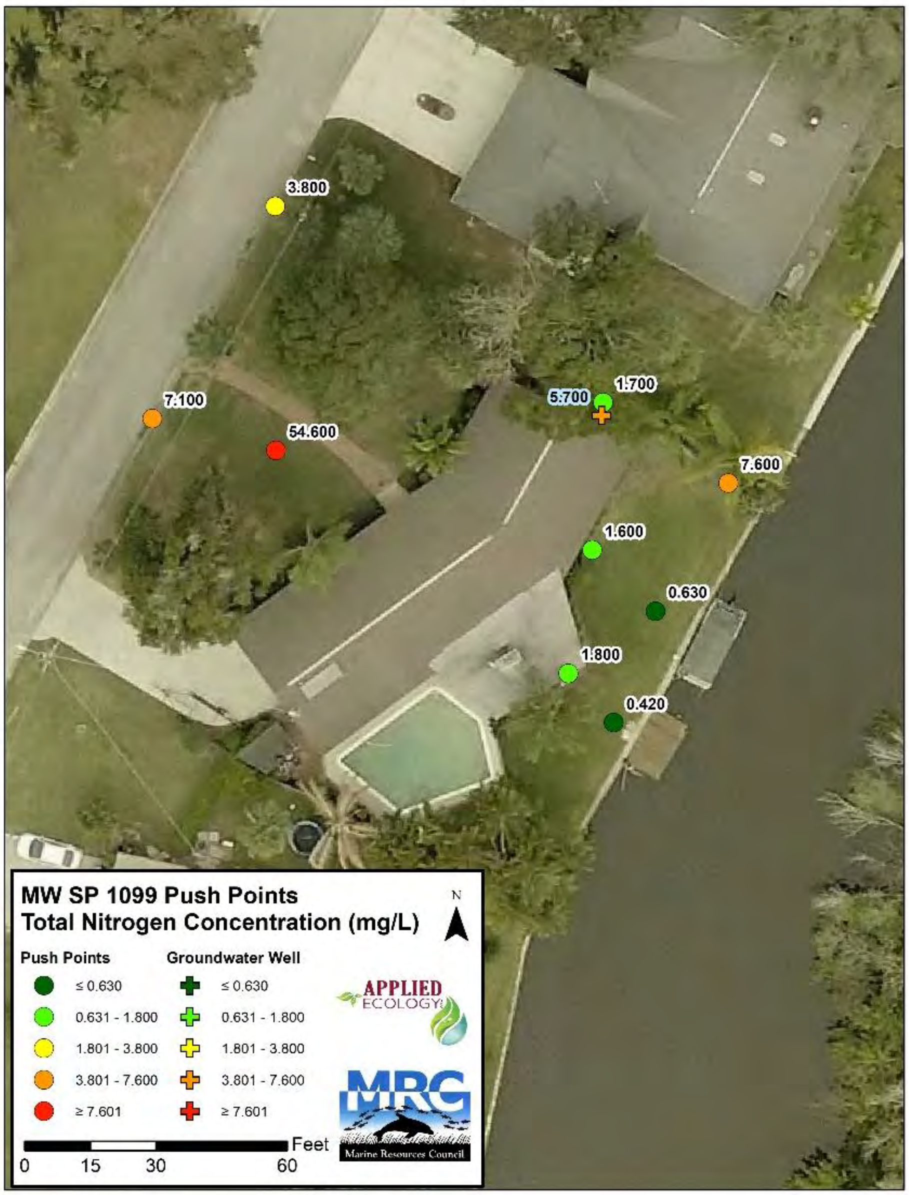

Figure 56. TN concentrations of the push point and Turkey Creek monitoring well MW SP 1099.

Five of the push point samples had TP concentrations lower than that measured TP concentration at the monitoring well, ranging from 0.041 to 5.5 mg/L, and the monitoring well TP concentration was 0.66 (Figure 57). The push point sample closest to the monitoring well (PP7) had a slightly lower TP concentration than that of monitoring well (0.18 mg/L). The TP concentrations appear to mimic TN, suggesting a general east-northeasterly flow into the water east of the house. Contour maps show the nitrogen and phosphorus plumes for MW SP 1099 (Figure 57-Figure 60).

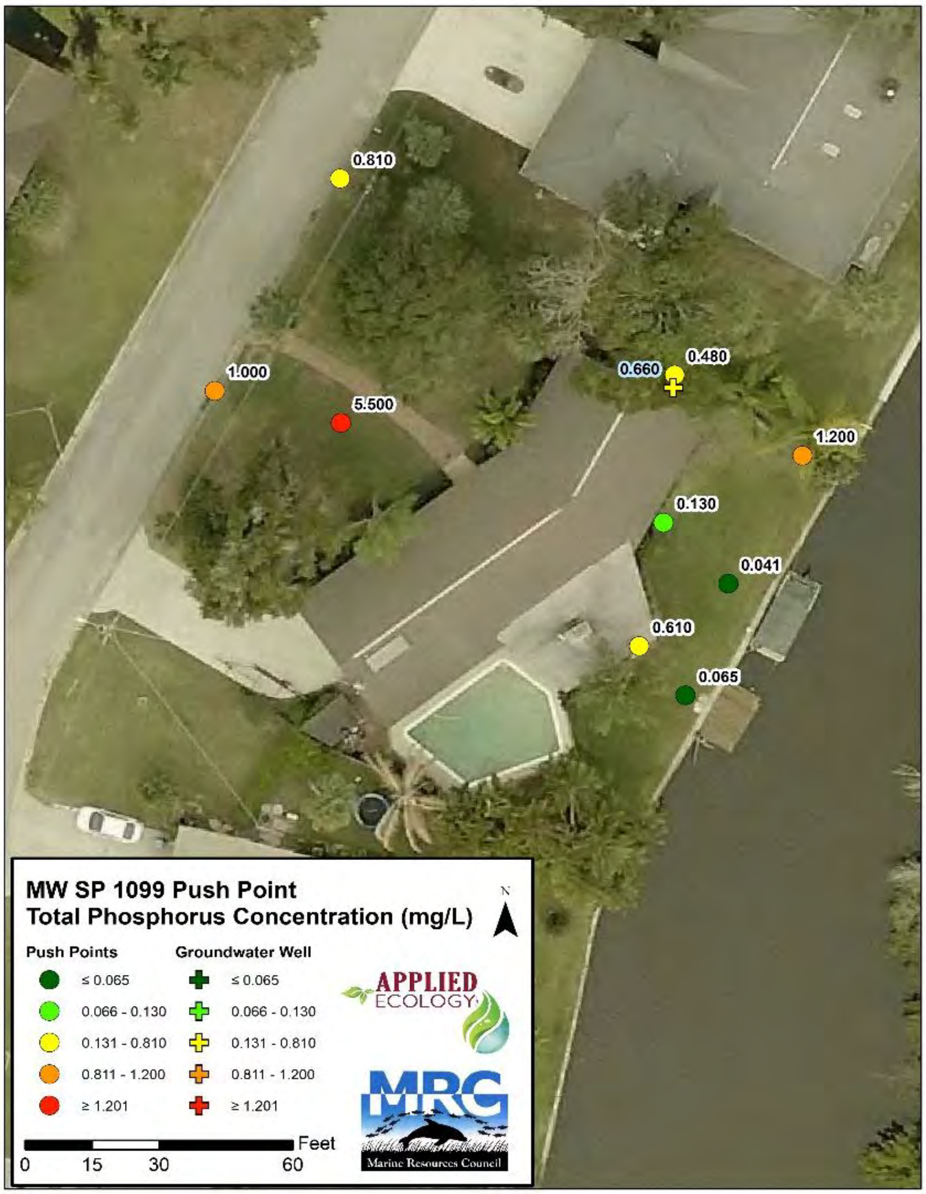

Figure 57. TP concentrations of the push points and Turkey Creek septic monitoring well MW SP 1099

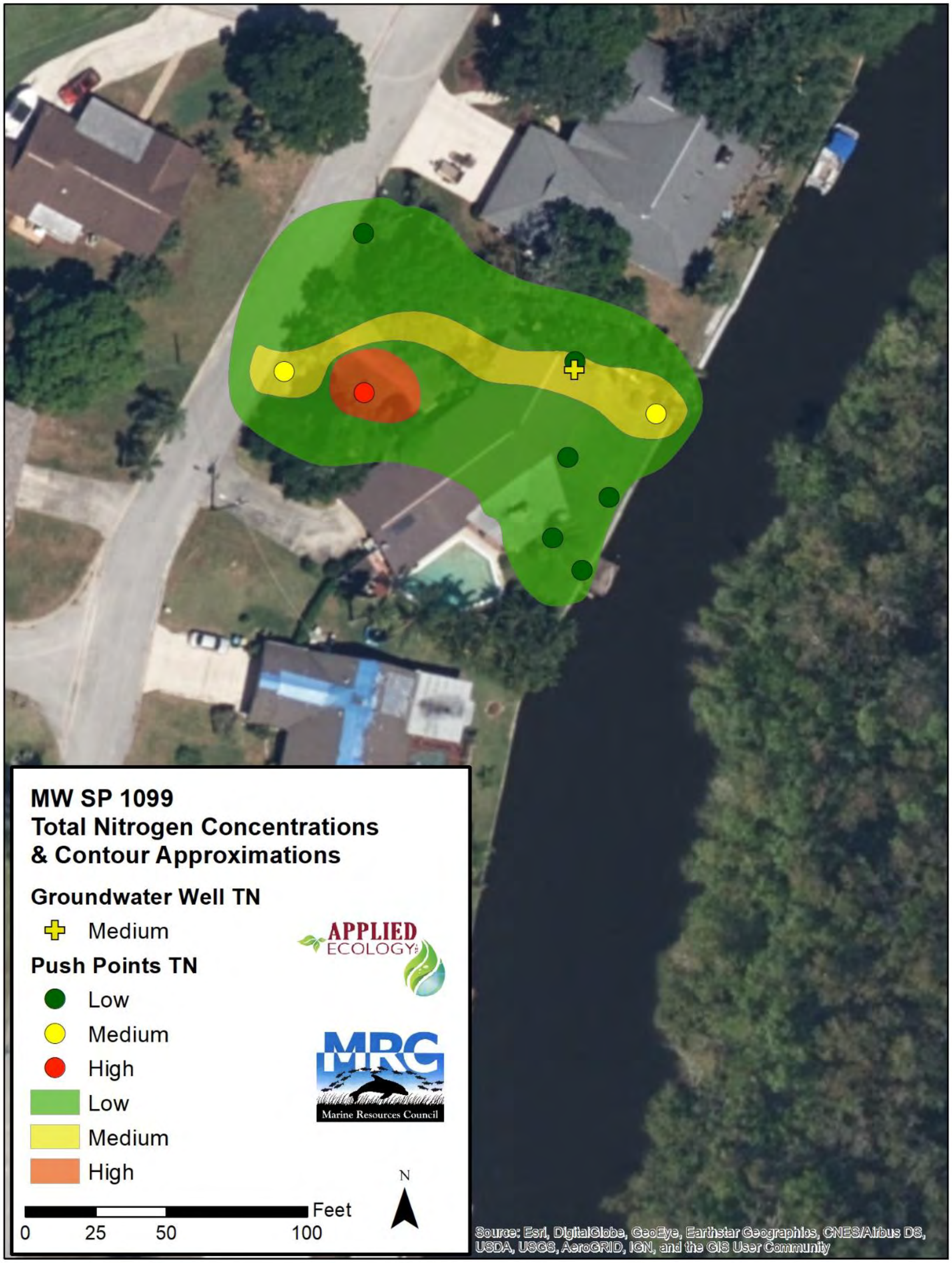

Figure 58: Total nitrogen (TN) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1099. Additionally, the TN concentration of the monitoring well itself is also mapped (plus sign).

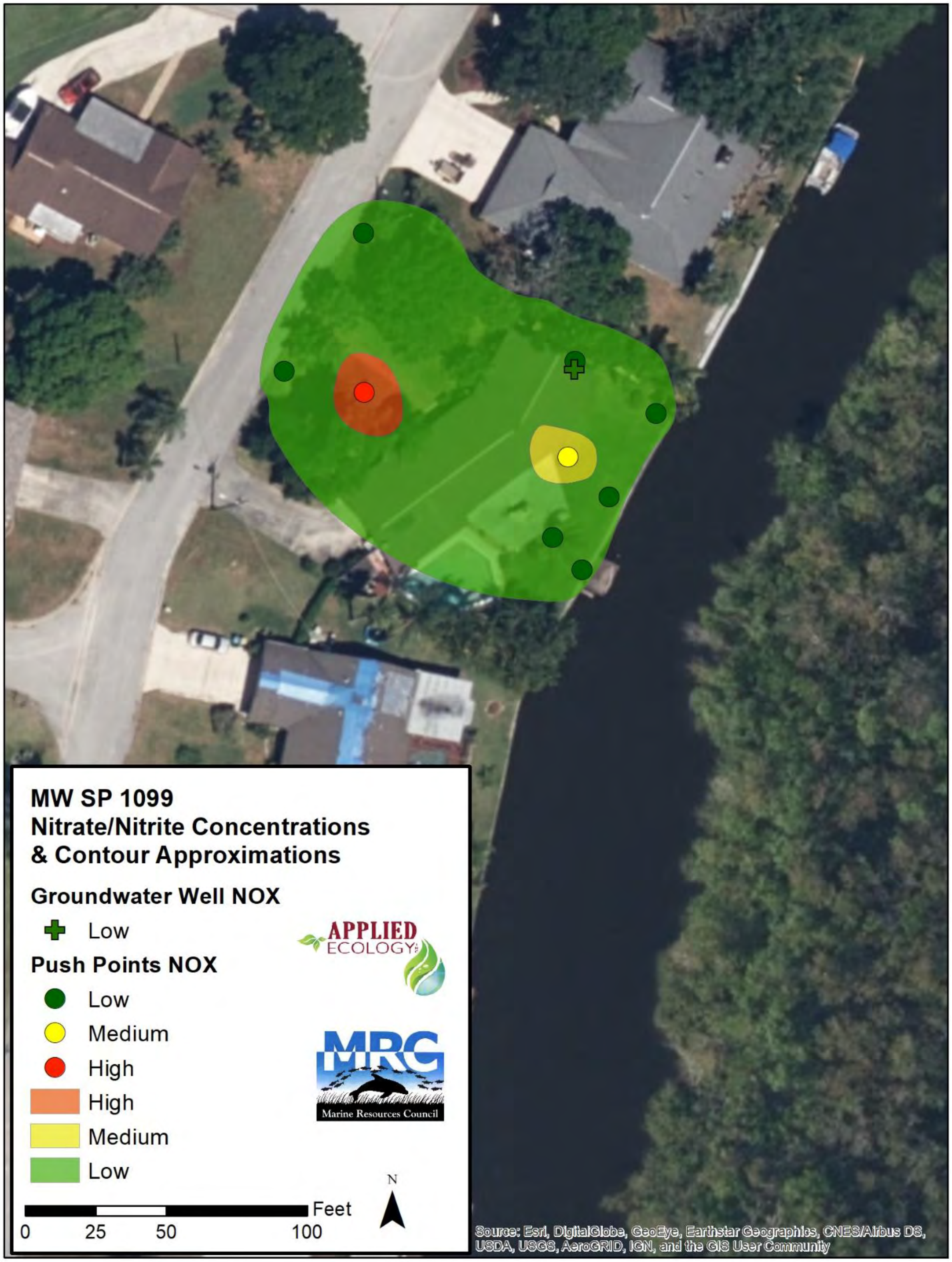

Figure 59: Nitrate/nitrite (NO_X) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1099. Additionally, the NO_X concentration of the monitoring well itself is also mapped (plus sign).

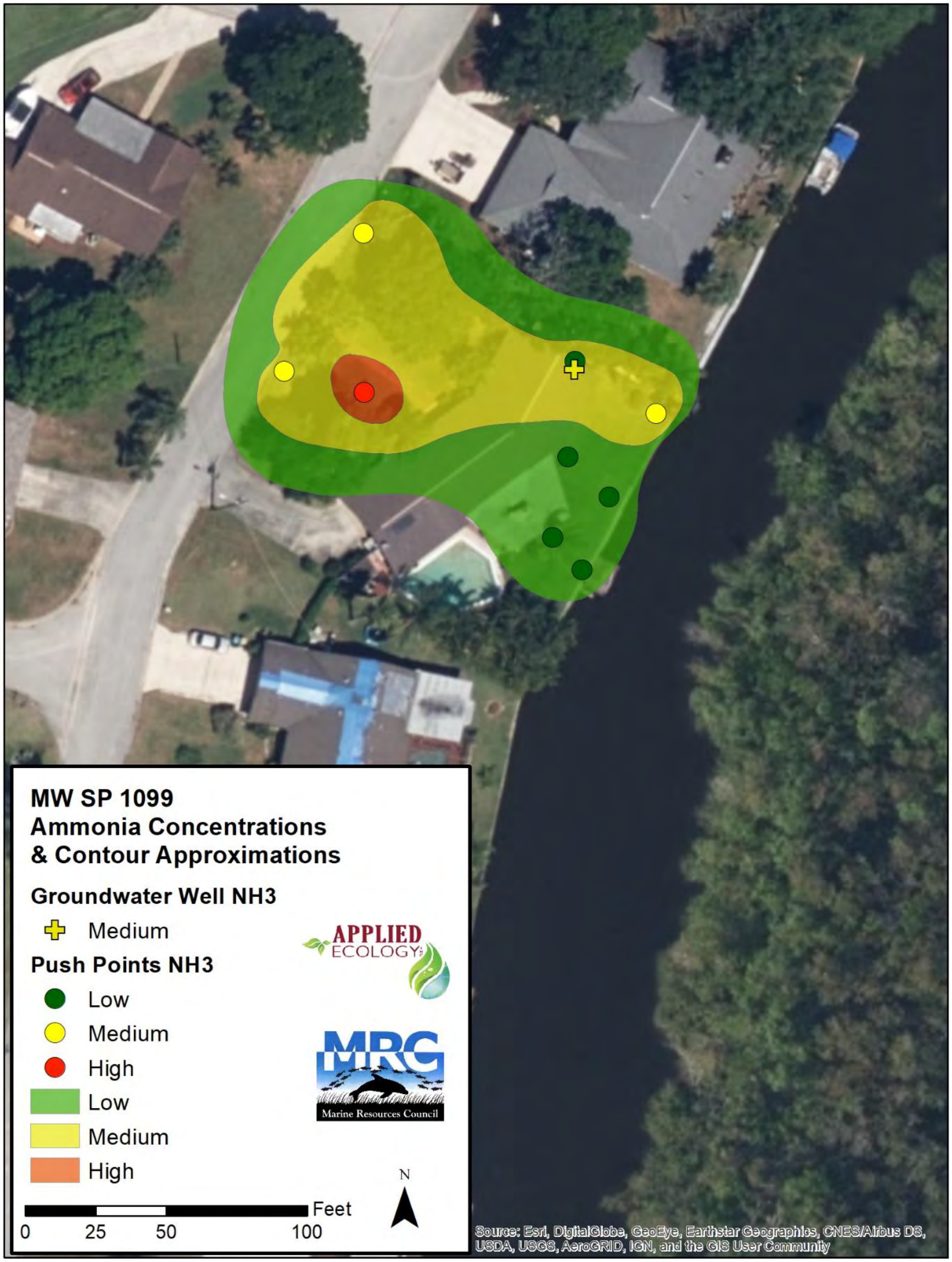

Figure 60: Ammonia (NH_3) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1099. Additionally, the NH_3 concentration of the monitoring well itself is also mapped (plus sign).

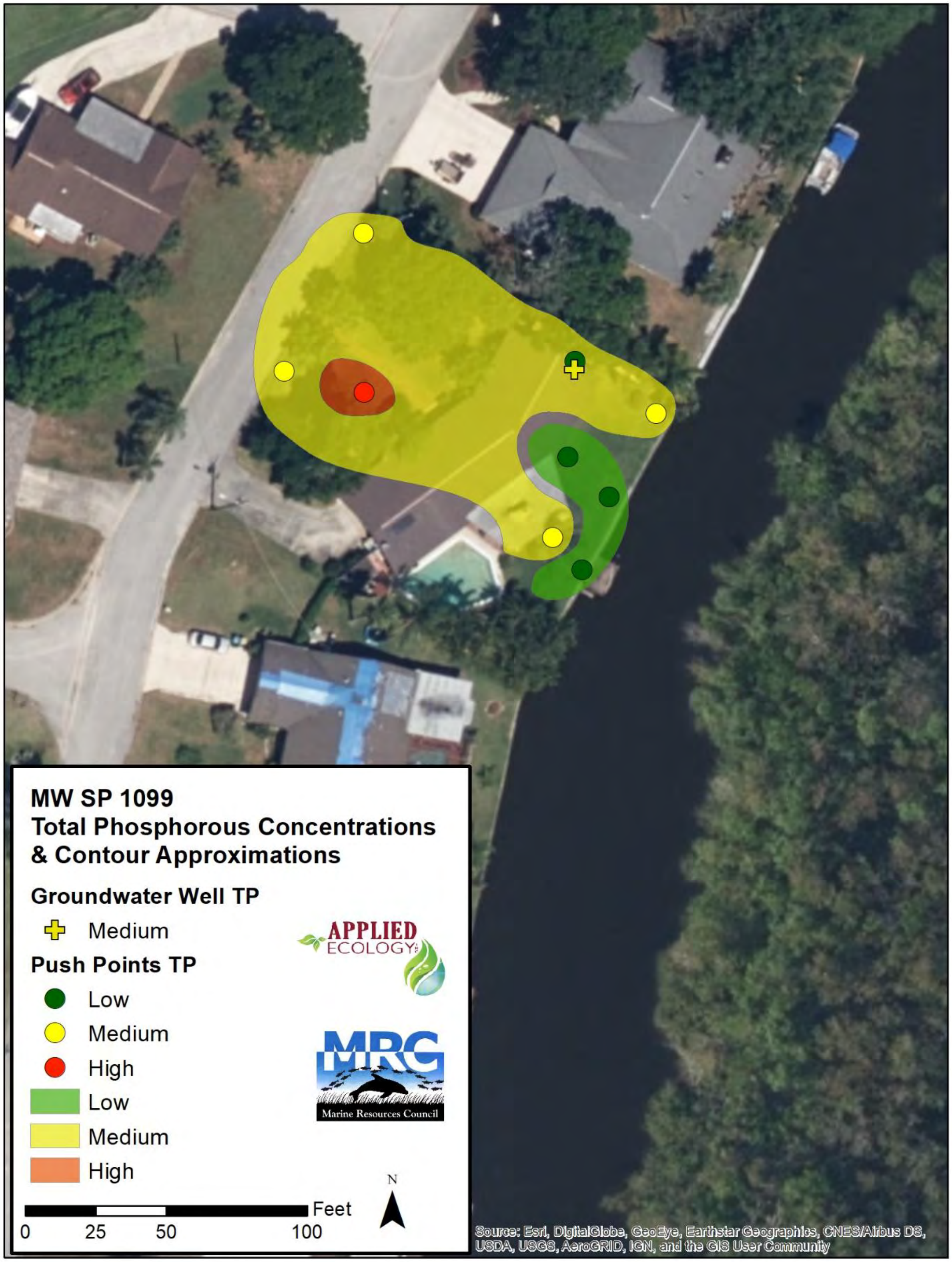

Figure 61: Total phosphorus (TP) concentrations and contour approximations of the push point locations (circles) surrounding MW SP 1099. Additionally, the TP concentration of the monitoring well itself is also mapped (plus sign).

Melbourne Beach Community

Unlike the other push point sites, in Melbourne Beach, we conducted a regional study to determine the feasibility that septic community related nutrient pollution was reaching the Indian River Lagoon. Multiple yards were accessed for this purpose, and limitations prevented access in some instances.

A total of 36 push points were performed in the Melbourne Beach septic community. General descriptive statistics for push point TN and TP concentrations are presented in Table 27. The range of TN concentrations in the push points was 0.17-8.0 mg/L while the TN concentrations in the three monitoring wells sampled the same day ranged from 1.3-8.9 mg/L.

Table 27: Descriptive statistics of TN and TP for the push points performed within the Melbourne Beach septic community.

Parameter	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
TN	1.601	0.805	0.360	1.650	0.170	8.000
TP	0.515	0.400	0.208	0.610	0.029	3.400

**Measured value below the Minimum Detection Level (MDL)*

According to the flow paths generated by ArcNLET, we expect the flow in this region to be generally from the northeast corner toward the southwest. Thus, it is interesting that the push point that we to be the upgradient of the community had one of the higher TN concentrations (2.1 mg/L). From the pattern in Melbourne Beach, there is a cluster of high TN concentration samples in the center of the community and lower concentrations along the lagoon, with the exception of one high TN push point near the lagoon that had a TN concentration of 5.2 mg/L.

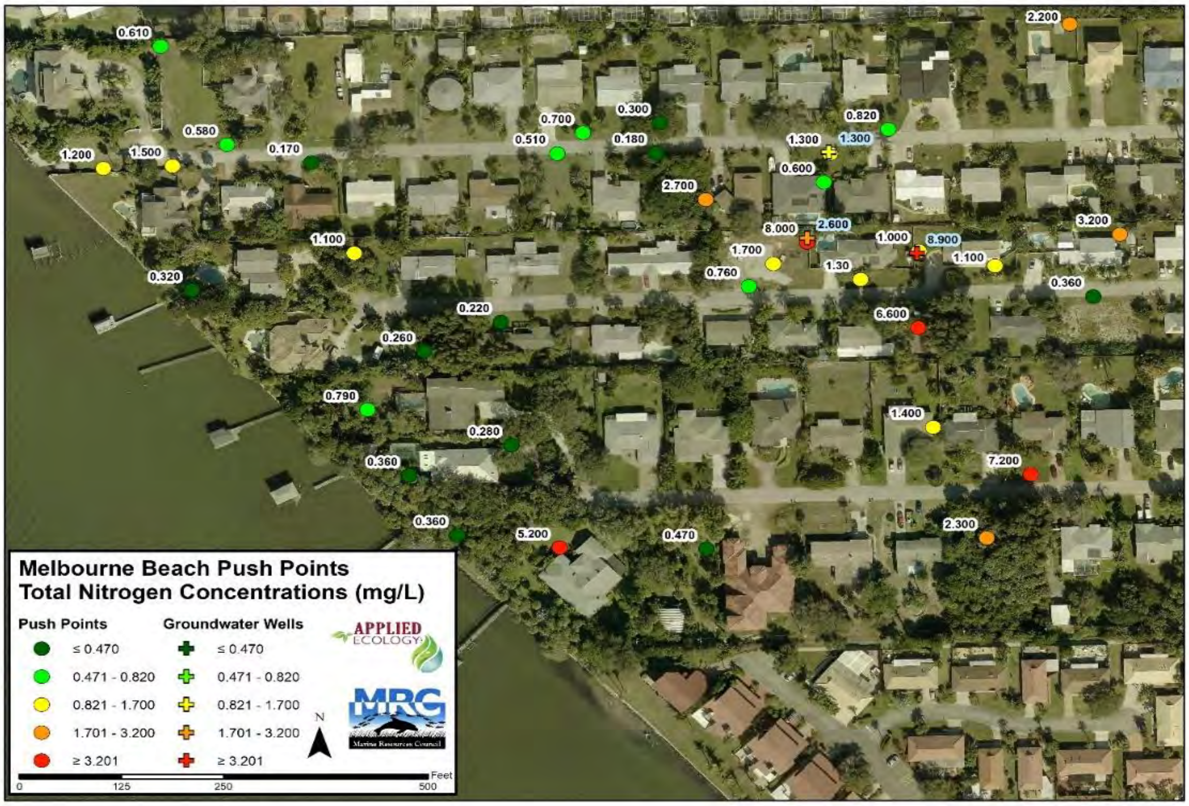

Figure 62: TN concentrations of the push point locations and three Melbourne Beach monitoring wells.

Push point TP concentrations ranged from 0.029-3.4 mg/L, and monitoring well concentrations ranged from 0.32 – 2.0 mg/L. The spatial distribution of TP concentrations is not clear with varying magnitudes of concentration data throughout the community (Figure 63). Unlike for TN, many of the push points located closer to the Lagoon had measured TP concentration data at or above the median values measured for the community. Most of the TP concentration hotspots, however, were located near the well data in the center of the community.

Figure 63: TP concentrations of the push point locations (circles) within the Melbourne Beach community. Additionally, the TP concentration of the monitoring wells are also mapped (plus sign).

Groundwater Modeling

The extensive modeling efforts performed as part of the Groundwater Pollution project and incorporated as Appendix B allowed several conclusions to be drawn. Some of these might be important for future data acquisition, analysis, and interpretation of the ArcNLET model for management of priority projects to restore the Lagoon. The conclusions are related to the three components of this memorandum report: the ArcNLET calibration effort, the uncertainty modelling, and the refinement of the Spatial Watershed Iterative Loading (SWIL) Model.

ArcNLET

- ArcNLET severely underestimates nitrogen loading potentials contributed by groundwater sources if not adequately calibrated with measured concentration data.
- Model calibration improved the accuracy of groundwater flow, direction, and plume intensity
- Even with calibration, ArcNLET appears to systematically underestimate total contributions of septic tanks, with nitrogen (nitrate + ammonia) values ranging from 3-5 g/day/septic tank
- To better dissect the factors that are driving the underestimation, the following data should be collected:
    - Input nitrate and ammonia concentration data from septic tanks based on water usage information (an indirect measure of number of residents per house)
    - Transect based groundwater quality with seepage information to follow nitrate and ammonia transport to the receiving waterbody (i.e., Lagoon)
    - Soil hydraulic conductivity values for representative soil types that make up the Lagoon’s watershed
    - Long-term groundwater quality data for nitrogen constituents

Uncertainty ArcNLET Monte Carlo Simulations

- Hydraulic conductivity is a key driver of nitrate transport from septic tanks into receiving water bodies according to the ArcNLET model
- There is a significant positive linear relationship between hydraulic conductivity and nitrate loads: higher hydraulic values will typically result in higher nitrate load predictions
- Porosity is the measure of the void spaces between the soil as a percentage between 0% and 100%. Permeability is a function of porosity, particle size, and the arrangement of these particles. Typically, surface soil horizons have large void spaces and higher porosity than deeper soils due to compaction over time. Soil porosity is inversely correlated to soil hydraulic conductivity: soils with high hydraulic conductivity usually have lower soil porosity

- Soil porosity also an important driving factor, albeit less significant, for nitrate loading, best described by an inverse linear relationship: lower soil porosity typically results in higher predicted nitrate loadings
- From the limited modeling effort, the relationship between hydraulic conductivity/soil porosity and nitrate loads varies spatially at several scales: regionally (beaches versus mainland) and locally (within communities)
- Within the Melbourne Beach community, soil parameters (hydraulic conductivity and porosity) have a greater explanatory power in the output nitrate loadings for plumes generated closer to the Lagoon; the same relationship is not as visible for the mainland community
- Smoothing factor presents a very different relationship to nitrate predictions than the soil parameters, often bidirectional and poorly described by any linear relationship
- Smoothing factor using to create the hydraulic head based on a replica of the Digital Elevation Model appears to have most significant impact in nitrate load prediction in the lower end of the range (0-30) with often steeper slopes and a different directionality than at higher ranges
- Calibration with field collected groundwater water level data is critical to ensure optimum smoothing factor is used, since small changes in the lower end range have yield dramatically different nitrate load outputs

SWIL Refinement

- The original uniform TN concentration used for baseflow loading calculations within the SWIL model (0.886 mg/L) is lower than the overall median TN concentration measured throughout the 18 months of sampling across all developed treatment types, with the largest discrepancies in the septic (2.55 mg/L) and reclaimed (4.50 mg/L) treatments; even within the same community (treatment type and relatively close proximity), groundwater concentrations might not be uniform and some attenuation from the measured median values might be applicable to certain watershed areas.
- The uniform TP concentration that was used for baseflow loading calculations in the SWIL model (0.112 mg/L) were closer to those found in this study, with the exception of the septic communities, which had substantially higher TP concentrations (0.6 mg/L).
- Replacing the uniform groundwater nutrient concentrations with measured values in the SWIL model and running the model for a small (5,627 acres) subset of the watershed, increased the estimated baseflow nutrient loadings by 84% or an additional 22,016 lbs./yr and TP by 13% or another 458 lbs./yr.

Understanding Nitrogen Sources and Denitrification Effects using Isotopes

Introduction

A better understanding of nitrogen (N) sources that are contaminating groundwater with NO_X, and processes that reduce NO_X, can be accomplished by examining naturally occurring stable N isotopes. The most common form of N is the stable isotope, ^{14}N. This is the form that makes up gaseous N found in the Earth's atmosphere. The less common form of nitrogen, ^{15}N, is a naturally occurring stable N isotope that has one more neutron than ^{14}N. The ratio of ^{15}N:^{14}N differs only slightly in N pools, typically falling within the range of -0.0040 to +0.0060. The differences in isotope ratios are described as isotopic signatures that are measured as delta values of the isotope ratio ($\delta^{15}N$) expressed in parts per thousand ($^{0}/_{00}$) or "mils".

Isotopes that have an extra neutron, like ^{15}N, are heavier and less reactive because they require more energy to react. As a result, heavier isotopes are left behind in chemical reactions causing an accumulation, or enrichment, in substrates and solutions. For example, NO_X left behind in the substrate during nitrification and denitrification processes is naturally enriched in ^{15}N. The gases that are released as part of the reaction (NO_2 and N_2) are depleted in ^{15}N. Organics and waste products tend to be enriched in the heavier isotope ^{15}N.

Atmospheric nitrogen is made up the lighter isotope, ^{14}N. Many fertilizers are produced by fixing atmospheric and therefore they are depleted in ^{15}N relative to organic biomass and waste products. Atmospherically derived chemicals and fertilizers are described as being "lighter" or "depleted" in ^{15}N, because they have a smaller amount of ^{15}N relative to ^{14}N.

The differences in isotopic signatures between organic and atmospheric nitrogen compounds can help distinguish nitrogen sources. In comparing $\delta^{15}N$ signatures, one study found that natural soil organics ranged from +4 to +7 $^{0}/_{00}$, commercial fertilizers were near 0 $^{0}/_{00}$ and septic waste ranged from +8 to +10 $^{0}/_{00}$, (Showers *et al*. 2007). When these potential sources enter the environment, nitrogen processes and chemical reactions will modify the original source $\delta^{15}N$ signature which complicates field interpretation. In this case, the use of two isotopes can help tease apart denitrification products that are enriched in ^{15}N from new sources.

In this study, we examine $\delta^{15}N$ and $\delta^{18}O$ isotopes in nitrate/nitrite (NO_X) to better understand the extent of denitrification and the signatures of new sources. The objectives of the isotope research were to understand denitrification processes that reduce nitrate in groundwater, to see if there are differences in isotopic signatures in treatment areas, and to test a method for teasing out new source inputs from naturally occurring enrichment. Due to laboratory limitations, only groundwater samples with a minimum NO_X concentration of 0.12 mg/L were analyzed for $\delta^{15}N$ and $\delta^{18}O$ isotopes. A total of 419 monitoring well samples, 22 push point samples and 2 reuse irrigation samples were analyzed for $\delta^{15}N$ and $\delta^{18}O$ isotopes.

As denitrification happens in groundwater, we expect $\delta^{15}N$ and $\delta^{18}O$ to enrich at a relative rate of between 2:1 to 1:1 indicated by a linear trend with a slope of 0.5 – 1.0 when plotted. The denitrification trend can be easily seen in the graph of plotted $\delta^{15}N$: $\delta^{18}O$ values for all of the isotope data collected in this study classified by treatment area (Figure 64). The scattered data points that can be seen below and above the line are likely new sources. From the figure, the natural area data tracks closely to the denitrification trend line, while the sewer and septic wells are more scattered.

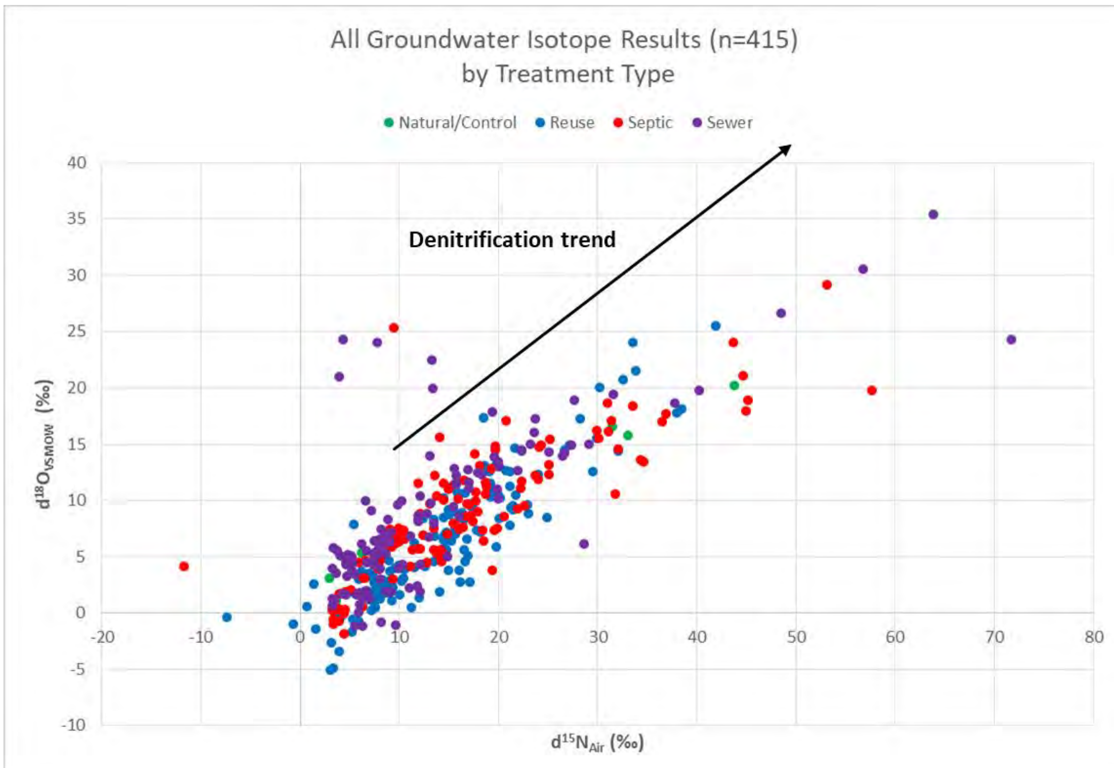

Figure 64: Denitrification Trend Line.

We also expect that as denitrification occurs, the concentration of nitrate/nitrite (NO_X) in groundwater will decrease and become more heavily enriched in $\delta^{15}N$. This can be seen in the plot of all of the $\delta^{15}N$ data plotted against the NO_X concentration (Figure 65). The cluster of data that climb vertically at about the +10 $^0/_{00}$ $\delta^{15}N$ mark are indicative of new, enriched sources like wastewater. The low NO_X concentration data with a low $\delta^{15}N$ signature (-5 $^0/_{00}$ to +5$^0/_{00}$) would be indicative of a new source that is atmospherically derived, like lawn fertilizer. If NO_X is reduced as a result of denitrification, the ^{15}N value would be high. Thus, it is easier to see atmospherically derived ^{15}N in the environment because as soon as denitrification occurs, enrichment occurs.

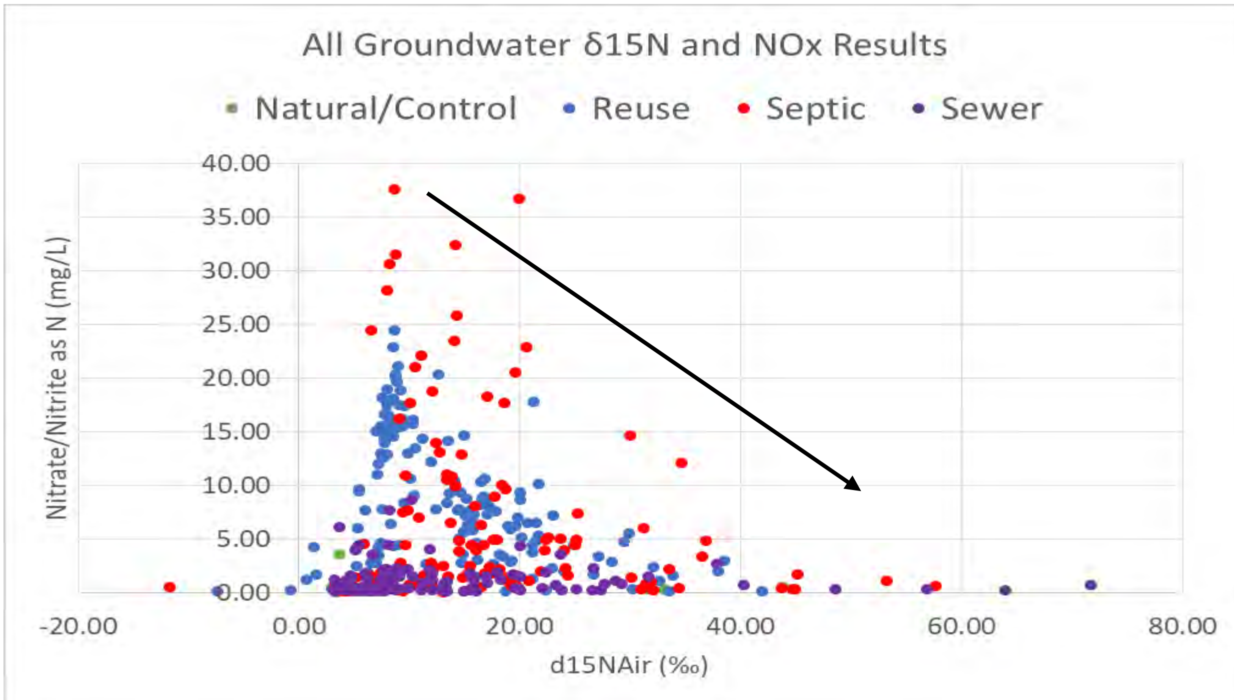

Figure 65: $\delta^{15}N$ and NO_X of all qualifying groundwater samples by treatment type.

Regional Differences

Graphing the complete set of isotopic data demonstrates the power of denitrification as the major driver of NO_X enrichment. To better understand source and denitrification dynamics, analysis was focused on regional and individual monitoring wells to tease out new contributions from denitrification trends. In this section, we examine the $\delta^{15}N$ and $\delta^{18}O$ ratios in each study region to see if differences emerge in source contributions and denitrification trends among treatments. The aggregate $\delta^{15}N$ and $\delta^{18}O$ data for each treatment are plotted in Turkey Creek, Beaches, Merritt Island, Suntree & Titusville.

Turkey Creek

The Turkey Creek region has all three treatments represented (septic, sewer, reclaimed) and a natural area which acts as a control. From the $\delta^{15}N$ and $\delta^{18}O$ plots, distinctly different groups are evident clustered around different potential source inputs (Figure 66).

The reclaimed (reuse) community NO_X isotopic data are less enriched relative to the other treatments in this region. The trend line also appears to source in the soil and mineralized fertilizer area, indicating that soil denitrification is a major contributor of NO_X $\delta^{15}N$ and $\delta^{18}O$. This makes sense in a community where NH_3 is being applied in irrigation water. In aerobic soils, this ammonia would nitrify to NO_X, resulting in a lighter product (low $\delta^{15}N$-NO_X). In the anaerobic pore space of the groundwater vadose zone, denitrification would reduce the NO_X to

NO_2 and N_2 gas, leaving behind more enriched NO_X. The lighter isotopic data that have negative values for $\delta^{15}N$ and $\delta^{18}O$ suggest that there are other depleted sources of nitrate entering the system that are atmospherically derived or industrially depleted. These could be fertilizers or other atmospherically derived chemicals. It is curious that the data are so clustered and short, appearing as a short, steep line.

Trending upward along the denitrification line is the septic tank data, which also appears to represent a mixture of sources. The denitrification line initiates somewhere between the manure & septic box and the mineralized fertilizer box and then proceeds along a wide denitrification band. This is an indication of source contributions of different enrichments proceeding along different denitrification paths. Only one of the septic points is depleted in $\delta^{15}N$ (-10 $^{o}/_{oo}$), which is clearly atmospherically derived and not a wastewater source. In the Turkey Creek septic wells, we see mixing of sources, along with strong denitrification occurring.

The sewer data stand out the most. These data points initiate in the synthetic NO_X fertilizer, but instead of following the expected 2:1, $\delta^{15}N$:$\delta^{18}O$, denitrification trend line, they appear to continue horizontally and maintain a stable $\delta^{18}O$ value while enriching in $\delta^{15}N$. The source appears to be fertilizer related, but the incredible enrichment of some of the points is hard to explain. Typically, a flat denitrification line is indicative of new inputs, but these high $\delta^{15}N$ and $\delta^{18}O$ values are not explained in the literature. Could this highly enriched NO_X be an indication of leaking sewer pipes?

The natural area wells proceed from soil along a denitrification line as expected. The $\delta^{15}N$ in the natural area approached +50 $^{o}/_{oo}$. If taken out of context of denitrification, this looks like a site heavily influenced by a waste source, but this could also be the residual NO_X resulting from years of denitrification.

The Turkey Creek Community is further investigated later in the report in a comparison analysis of representative wells in each of the treatment areas.

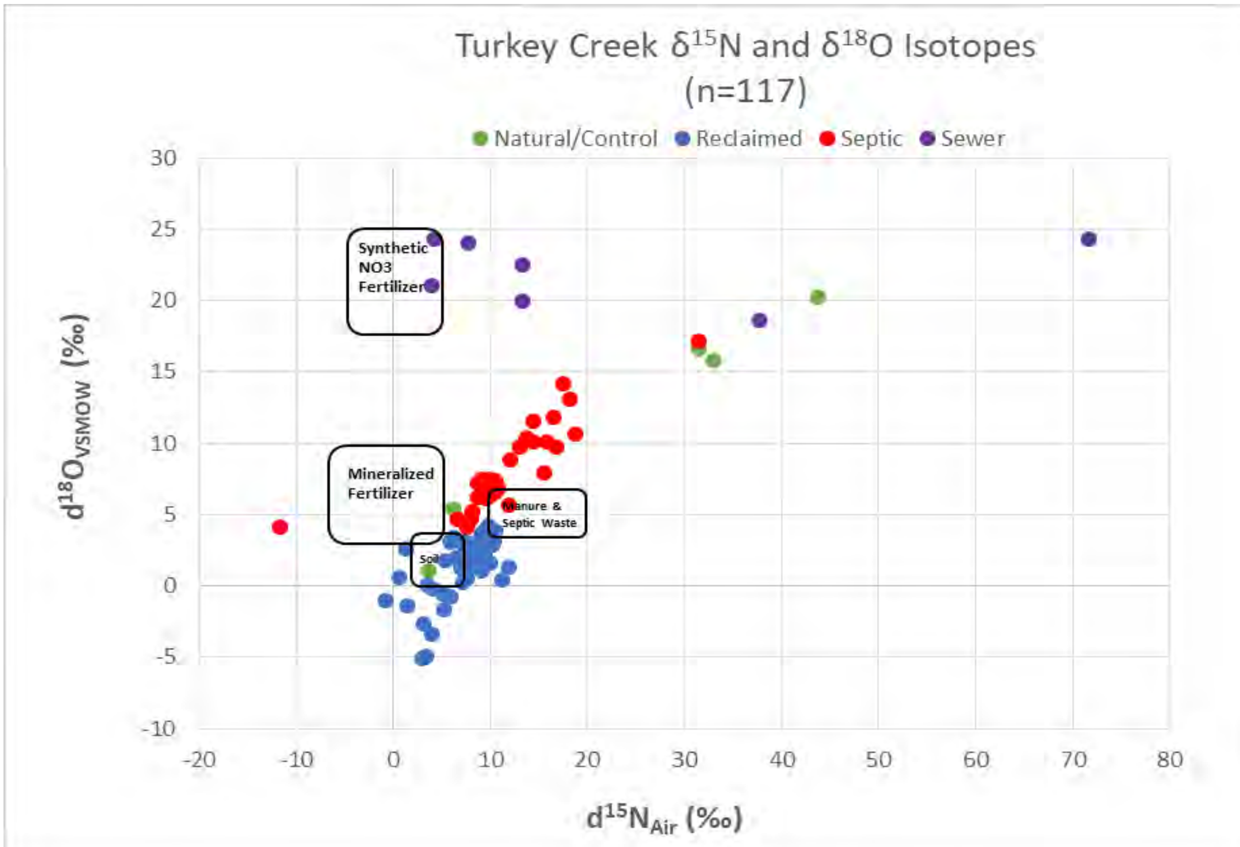

Figure 66: Source characteristics of Turkey Creek treatments using $\delta^{15}N$ and $\delta^{18}O$ isotopes.

Beaches

The Beaches region includes the wells located in a Satellite Beach sewer community and Melbourne Beach reclaimed and septic communities. In Figure 67, the first pattern that emerges is the powerful denitrification process that is enriching $\delta^{15}N$ and $\delta^{18}O$. Upon further examination, the denitrification reclaimed line is shorter than the other two, but not nearly as clustered as what we saw in Turkey Creek. The reclaimed community isotopic signatures are much more enriched than those observed in the Turkey Creek reclaimed community. The reclaimed community data are scattered above the manure/septic waste source box, suggesting the mixing of waste and fertilizer sources. This is confirmed by plotting the trend line, which shows the $\delta^{15}N$:$\delta^{18}O$ linear relationship has a linear slope of 0.47 (< 2:1) with a low r^2 (0.59) confirming the plotted points do not fit strongly to the trend line and that the denitrification trend line explains about 59% of the variance in $\delta^{15}N$ and $\delta^{18}O$. From the denitrification plot, it appears that fertilizer is the source of variance, since the mixing is resulting in less enriched $\delta^{15}N$ and $\delta^{18}O$.

The sewer data has a similar clustered configuration, but the trend lines clearly initiates in the less enriched soil and mineralized fertilizer portion of the graph and trends up the denitrification line at a slope of 0.55 ($r^2 = 0.75$), suggesting less mixing and more atmospherically derived source inputs (fertilizer). In the sewer community, the denitrification trend line explains 75% of the variance in $\delta^{15}N$ and $\delta^{18}O$ signatures. We would expect to see less mixing in the sewer community if the sewer lines are intact and not leaking. In the sewer dataset, there are some isotopic data that suggest a wastewater input, including one highly enriched with $\delta^{15}N$ at + 48.54 ‰ and $\delta^{18}O$ at +26.61 ‰.

The septic community isotopic signature most closely fits the denitrification line ($r^2 = 0.84$), but the slope is slightly lower than expected for denitrification (m = 0.46), suggesting that there are still mixed sources that are lowering the overall enrichment values. There are depleted septic community isotope points located around the mineralized fertilizer and the soil source box that are denitrifying along a higher trend line. Curiously, we don't see a lot of denitrification initiating in the manure and septic waste box. Most of the plotted data for the septic community appear to come from the soil, suggesting that denitrification in the soil is the greatest source of NO_X in this septic community. The organic content in the soil in the Melbourne Beach septic community is relatively high, ranging from 1.56 – 4.85% in soil cores.

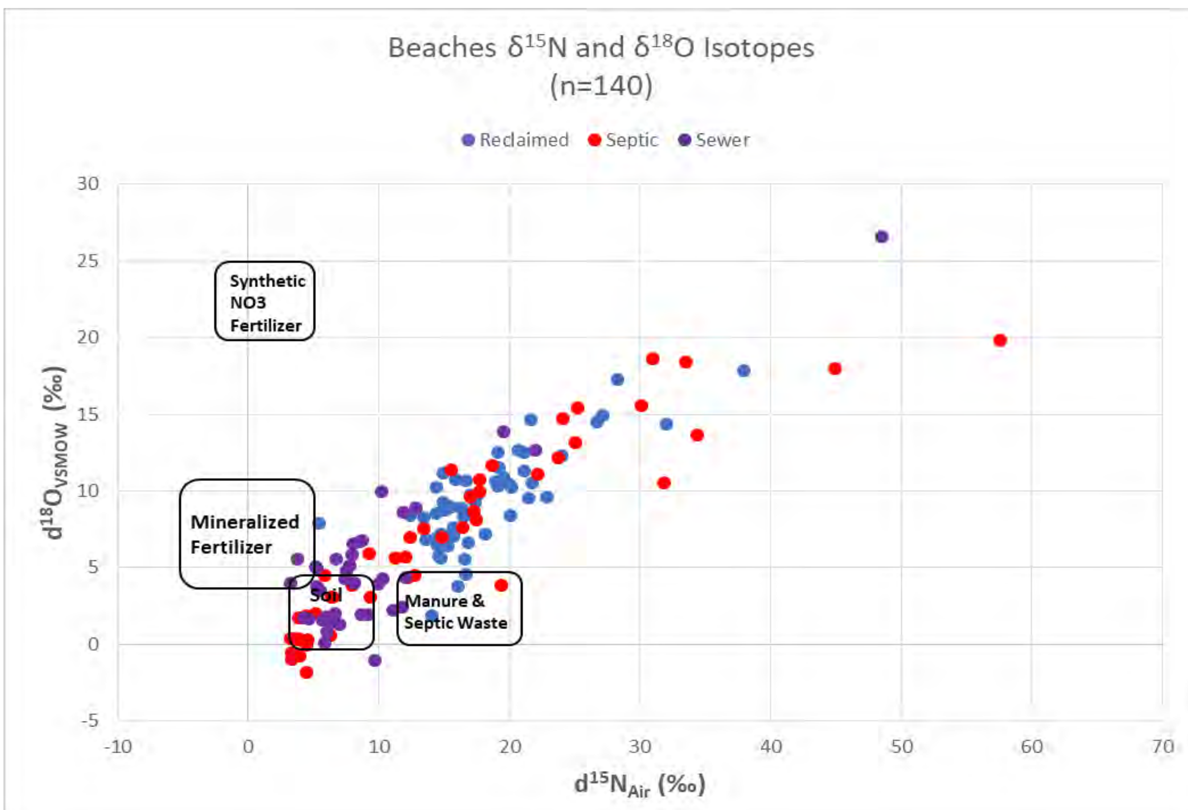

Figure 67: Source characteristics of Beaches treatments using $\delta^{15}N$ and $\delta^{18}O$ isotopes.

Merritt Island

In the Merritt Island region, there are wells located in adjacent septic and sewer communities. Plotting the isotopic signatures of these communities shows two distinct denitrification trend lines with very different source signatures (Figure 68). The sewer community tracks closer to the mineralized fertilizer source box, with some points dropping into the depleted soil range. This indicates that fertilizer is a clear source of NO_X in this community. Fertilizer is easier to see in the groundwater of communities that aren't receiving a constant input of nitrogen from septic tanks or reclaimed irrigation water.

The septic community denitrification line appears in a much more enriched area of the plot, slightly more enriched than the typical septic waste source box. There is one septic data point in the range of the mineralized fertilizer box, indicating that fertilizer is also an input in Merritt Island septic communities.

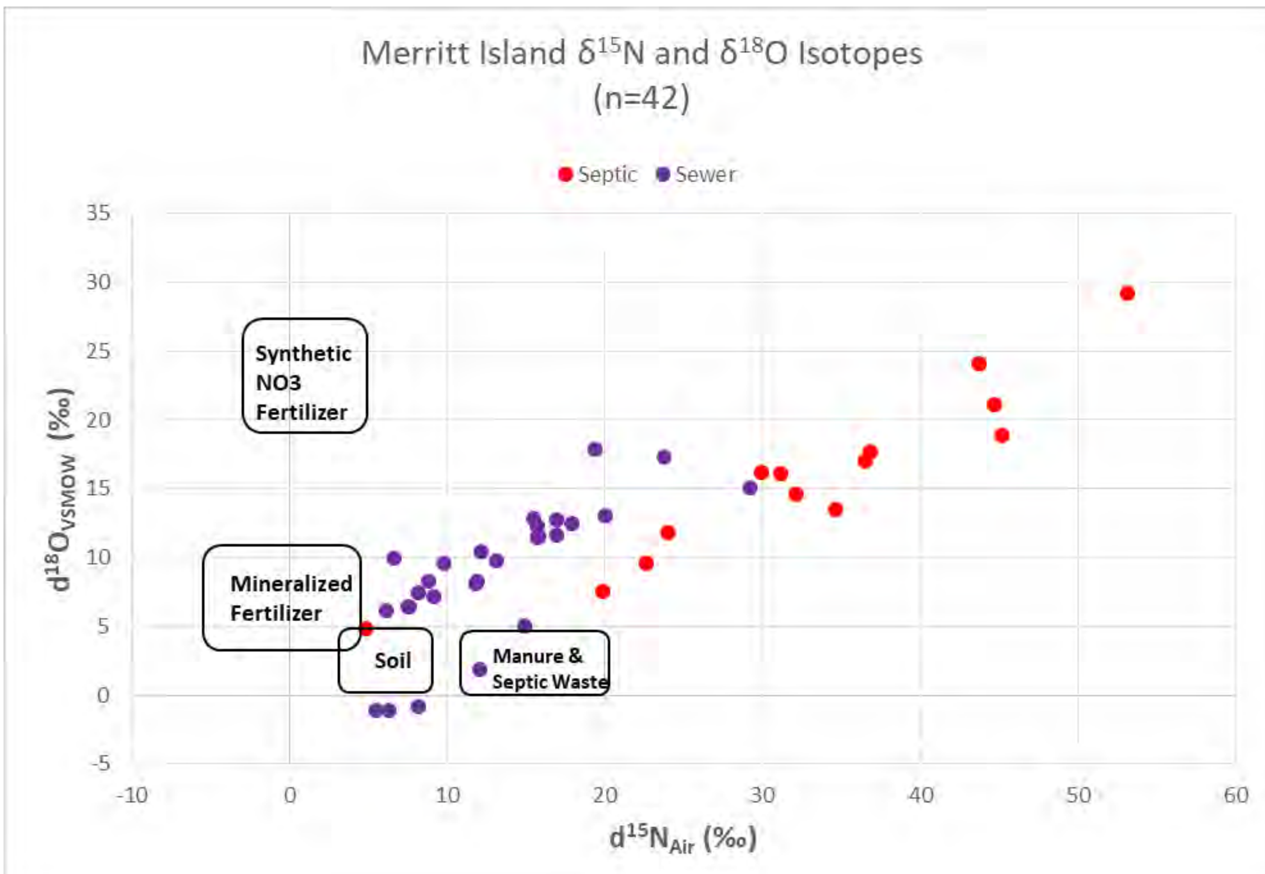

Figure 68: Source characteristics of Merritt Island treatments using $\delta^{15}N$ and $\delta^{18}O$ isotopes.

Suntree

Septic, reclaimed (reuse), and sewer communities were monitored in this region and isotopic data were plotted to understand denitrification and differentiate sources (Figure 69). Denitrification is clear, but there is a great deal of scattering, indicative of source mixing between waste derived and fertilizer derived sources of nitrate. As expected, the septic community sources primarily in the manure and septic waste portion of the graph, but there are some outlier data that appear in the areas of mineralized and synthetic fertilizer.

The sewer data appears to initiate firmly in the mineralized fertilizer area of the graph and denitrification continues to show enrichment with values approaching a $\delta^{15}N$ signature of $+70^{o}/_{oo}$. Again, this is an incredibly high enrichment number that we keep seeing in sewer communities, further supporting the speculation that sewer lines may be leaking highly enriched NO_X or highly enriched ammonia that is further enriched through denitrification processes in the pipe and the surrounding soil.

The reclaimed (reuse) community denitrification line initiates almost exactly between the organic waste and mineralized fertilizer boxes. The isotopic signatures are lighter than expected for a waste by-product. This might be an indication of nitrification processes in the wastewater treatment plant, which would result in a lighter NO_X and more enriched NH_3. Alternatively, this could be an indication of source mixing between enriched waste product NO_X and mineralized fertilizer.

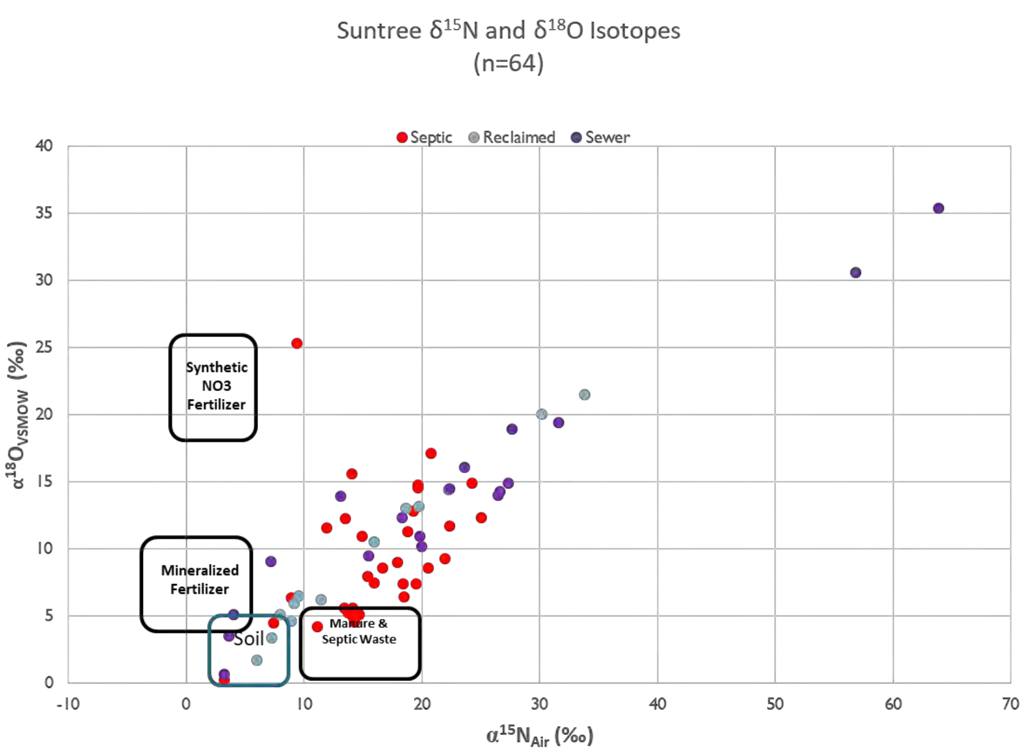

Figure 69: Source characteristics of Suntree treatments using $\delta^{15}N$ and $\delta^{18}O$ isotopes.

Titusville

In the Titusville region, there are reclaimed, sewer, and natural areas. Plotting $\delta^{15}N$ and $\delta^{18}O$ in this region shows two distinct denitrification lines that are similar to Merritt Island sewer and septic plots, but with a little more mixing (Figure 70). As with Merritt Island, we see the sewer denitrification line trends higher than expected, initiating near the mineralized fertilizer box. The denitrification trend extends to reach a total $\delta^{15}N$ enrichment of nearly +40 ‰.

The reuse community is lighter in $\delta^{18}O$ overall, with a series of data that appears to source in the manure and septic box and another series of data that appear to source in the soil NO_X box. Further investigation of how the wastewater process impacts isotopic signatures of NH_3 and NO_X are warranted.

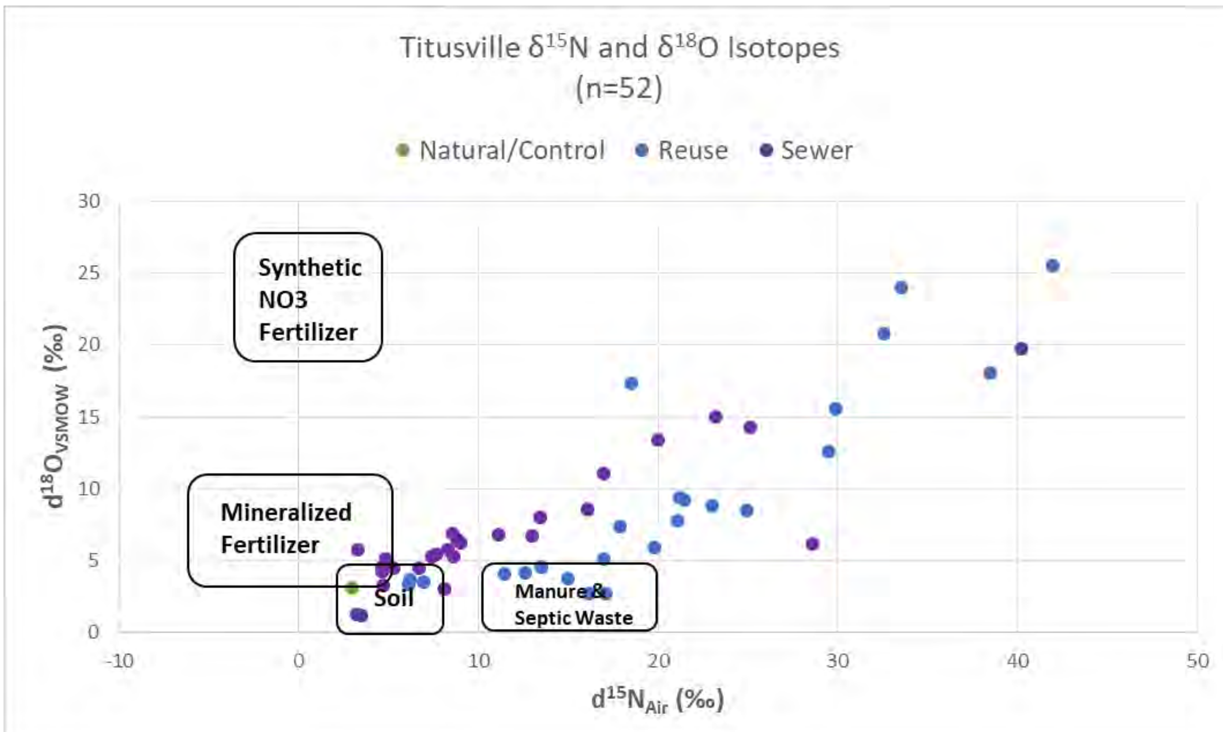

Figure 70: Source characteristics of Titusville treatments using $\delta^{15}N$ and $\delta^{18}O$ isotopes.

Individual Wells

The denitrification effect overwhelms the dataset making it is necessary to examine isotopic signatures in relation to NO_X concentrations individually instead of as an aggregate dataset. To accomplish this, we selected all of the wells in the study that had at least 12 isotope data to plot and analyze. Table 28 summarizes the 17 monitoring wells that had adequate data. The range and variability of NO_X and $\delta^{15}N$ are consistent, although there appears to be greater variability and enrichment in septic and sewer communities than in the reclaimed communities.

Table 28: Monitoring well NO_X concentration and $\delta^{15}N$ signatures by treatment type for 17 monitoring wells that have at least 12 results.

Monitoring Well	Treatment	n	NO_X (mg/L) Min	NO_X (mg/L) Max	α 15N Min	α ^{15}N Max	^{15}N Mean	$^{15}N/^{18}O$ Slope
MW SP 1739	Septic	13	0.23	36.70	19.86	53.15	34.94	0.56
MW SP 1127	Septic	27	0.66	37.60	6.58	31.41	12.53	0.56
MW SP 250	Septic	17	0.31	18.30	9.37	30.97	17.94	0.68
MW SP 270	Septic	14	0.34	4.50	5.87	57.61	24.32	0.35
MW SP 275	Septic	17	0.13	1.40	3.25	14.80	5.14	0.65
MW SP 6215	Septic	18	1.10	32.40	11.11	25.05	16.93	0.60
MW SP 6155	Septic	12	0.12	2.80	3.22	24.28	15.46	0.64
MW RE C	Reclaimed	30	24.40	30.00	7.02	9.02	8.04	0.61
MW RE 2456	Reclaimed	21	0.14	9.40	-0.75	8.54	4.66	0.39
MW RE C3	Reclaimed	18	9.00	21.10	7.49	11.95	9.86	0.20
MW RE 158	Reclaimed	18	5.60	9.20	12.36	20.71	15.66	0.56
MW RE 182	Reclaimed	18	2.70	10.60	7.87	14.64	7.68	0.39
MW RE 239	Reclaimed	18	1.70	10.30	14.41	37.94	22.21	0.42
MW RE 549	Reclaimed	18	1.30	20.30	6.93	38.49	19.82	0.54
MW SE 1710	Sewer	17	0.75	2.20	7.51	29.16	14.11	0.43
MW SE 540	Sewer	18	0.13	2.30	3.20	25.12	10.29	0.54
MW SE C1	Sewer	13	0.19	3.50	15.48	63.88	29.23	0.53

Monitoring Well MW SP 1739

Monitoring well MW SP 1739 is located in a septic tank community in Merritt Island adjacent to the septic tank drain field. In addition to the monitoring well, 14 push points were completed to better understand the extent of the contamination plume. Figure 71 shows the placement of the monitoring well and push points on the property. This monitoring well is consistently very enriched, with a mean $\delta^{15}N$ signature of 34.94 $^{0}/_{00}$

Figure 71: Merritt Island push point and monitoring well locations.

Plotting the change in NO_X and $\delta^{15}N$ over time helps clarify denitrification trends and may illuminate the timing of source inputs (Figure 72). Overall, the NO_X is enriched with $\delta^{15}N$ which is consistently above +20 $\delta^{15}N$. The only time during the sampling timeframe that the isotopic value falls below +20 $\delta^{15}N$ is when there is a sharp spike in NO_X concentration. The dramatic increase in NO_X from 0.23 mg/L on 01/07/19 to 36.7 mg/L on 02/11/19 is followed by a decrease to 1.1 mg/L on 03/11/19. During that same timeframe, we see a decrease in $\delta^{15}N$ from +32.11 to +19.86 followed by an increase to +53.15. This could be indicative of a less enriched source input of NO_X which mixed with the existing enriched NO_X groundwater resulting in a lower enrichment overall. This new addition was then reduced through denitrification, resulting in lower NO_X concentration with a higher $\delta^{15}N$ signature. If this is the case, then the denitrification of 35 mg/L NO_X occurred in less than a month, demonstrating an efficient denitrification process. We see a similar, albeit smaller, increase in NO_X concentration from 1.7 to 12.1 mg/L that occurred between 04/08/19 to 05/06/19 with an associated decrease in $\delta^{15}N$ signature followed by an increase.

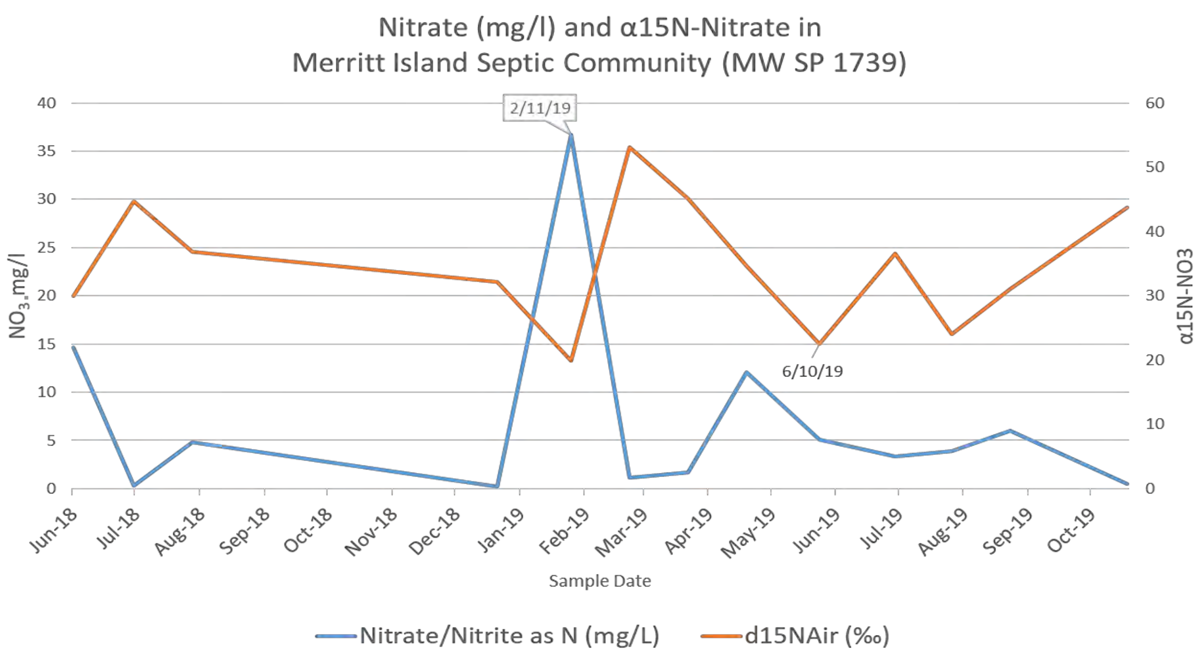

Figure 72: Merritt Island MW SP 11739 NO_X and $\delta^{15}N$ over time.

Looking at the $\delta^{15}N$ and δ18O signatures for this well can clarify the source of nitrogen contributing to the NO_X concentration increases. Plotting the two isotopes in Figure 73 shows the denitrification slope for MW SP 1739, along with the potential source contributions as described by their ^{15}N:^{18}O ratios. The denitrification line is within the expected 2:1 slope range and has strong internal consistency, demonstrating that denitrification is a strong driver at this location. Notice that the two sampling dates where NO_X concentration increased are at the beginning of the denitrification line. This demonstrates that those are sources that will follow the same denitrification trajectory. In this case, the source appears to be the septic tank.

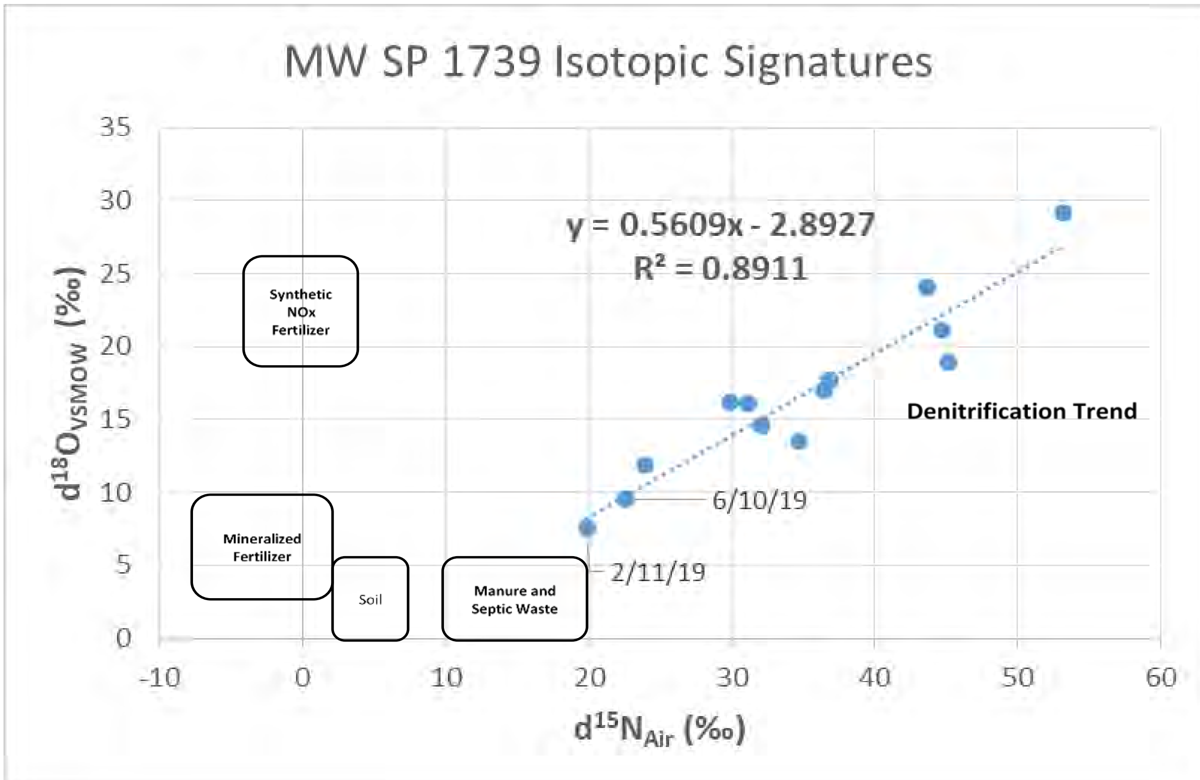

Figure 73: Merritt Island septic monitoring well MW SP 1739 source characteristics.

To understand the driver of the two NO_X peaks, we looked further into the data and found that within 3-days prior to these two sampling events, the area experienced rainfall events. On February 10-11, over one inch (1") of rainfall occurred and from June 9-11, there was five inches (5") of rainfall. These two events each resulted in about a one foot (1') increase in water level in the monitoring well. It could be that this rise in water level saturated previously unsaturated soil where denitrification was occurring, releasing the enriched nitrate into the water column. Curiously, dissolved oxygen (DO) in the well also increased substantially, from near 0 mg/L to over 2 mg/L after each rain event. The DO concentration quickly returned to near 0 following the sampling. Denitrification is more efficient in an anaerobic environment.

Comparing Different Treatments in Turkey Creek

Eleven monitoring wells were installed in four different treatment areas in the Turkey Creek region, providing an opportunity to compare nutrient dynamics in septic, reuse, and sewered communities relative to a natural area. The natural area (Turkey Creek Sanctuary) includes two monitoring wells, TC-1 and TC 2. Monitoring well TC-1 samples never exceeded the minimum NO_X concentration of 0.12 mg/L to analyze for $\delta^{15}N$ and $\delta^{18}O$. The second natural area (TC2) had seven (7) samples with NO_X concentrations high enough to analyze for $\delta^{15}N$ and $\delta^{18}O$. We suspect that this well might be receiving inputs of nitrogen from the neighborhood directly west of it, but we can't be sure. Although the majority of the adjacent neighborhood is on city sewer, the house within about 1,000 ft (304 m) of the monitoring well had a septic tank.

Only seven samples in the Turkey Creek sewer community had NO_X concentrations high enough to run isotopic analyses. These included four samples in monitoring well MW SE 841 and 3 samples in monitoring well MW SE 849. Of the septic community wells, a single well (MW SP 1127) makes up 90% of the 31 samples that had adequate NO_X concentrations to run isotope analysis. In the reuse community, there were 165 samples collected from all three monitoring wells with high enough NO_X to run isotope analysis.

In Figure 74, mean NO_X and $\delta^{15}N$ values are compared across the four treatments. The sewer area had the lowest average NO_X concentration (0.64 mg/L) and the highest average $\delta^{15}N$ value (+21.72). The natural area well had the second-lowest NO_X concentration (1.10 mg/L) and the second highest $\delta^{15}N$ value (+20.96). The Reuse community had the highest average NO_X concentration (10.88 mg/L) and the lowest average $\delta^{15}N$ (7.32) followed by the Septic community (NO_X 10.15 mg/L, $\delta^{15}N$ 11.81).

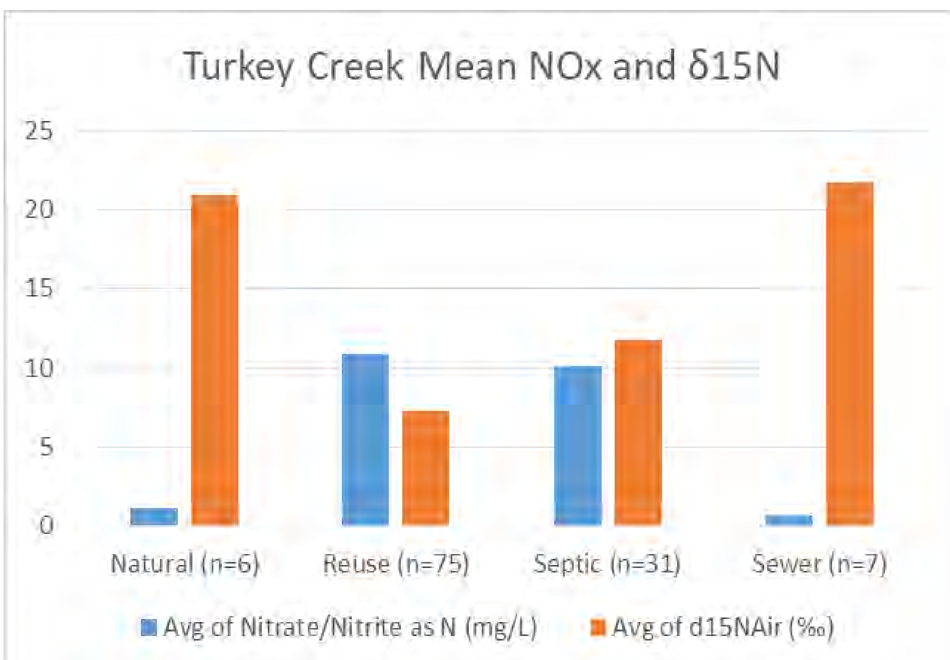

Figure 74: Turkey Creek NO_X and $\delta^{15}N$.

The fact that groundwater wells in the natural and sewer areas have more enriched $\delta^{15}N$ values than the septic and reuse community wells demonstrate the challenge of looking at the enrichment of a single isotope ($\delta^{15}N$) to understand sources of nitrogen. One has to look at it the context of denitrification processes and the signature of new inputs. In the following section, we will examine one well in each community.

Turkey Creek Septic Monitoring Well MW SP 1127

Monitoring Well MW SP 1127 is located adjacent and downstream of the mounded septic tank and drain field located to the west of the house in Figure 75, between push point locations PP15, PP16, PP17, and PP18.

Figure 75: Turkey Creek septic monitoring well MW SP 1127 and push point locations.

Graphing of NO_X and $\delta^{15}N$ over time shows the tremendous variability in this monitoring well (Figure 76). We see three big peaks in the time series during which NO_X concentrations went from < 2 mg/L to > 30 mg/L. Similarly, to what we saw in the Merritt Island septic tank well MW SP 1739, each NO_X peak coincides with a reduced $\delta^{15}N$ value that is followed by a reduce NO_X concentration and an increased $\delta^{15}N$ value. This looks like denitrification is occurring, but what is the source of those peaks in NO_X?

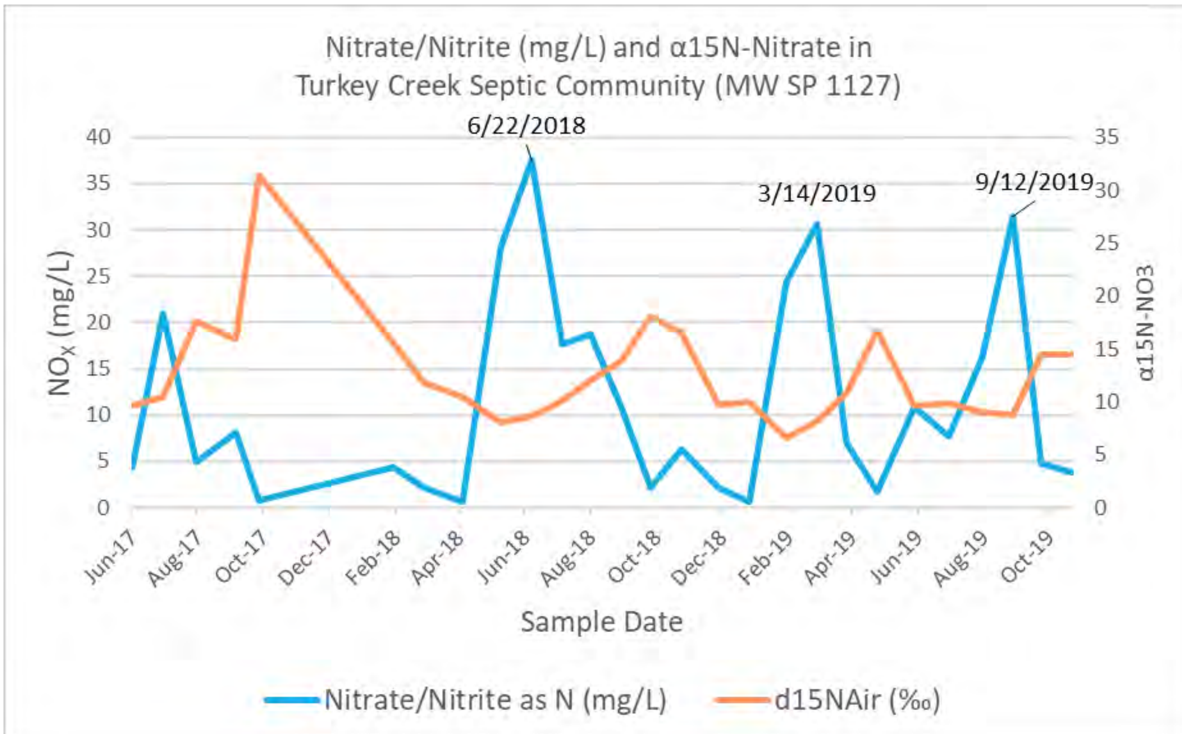

Figure 76: Turkey Creek septic well MW 1127 NO_X and $\delta^{15}N$ over time.

Plotting $\delta^{15}N$ and $\delta^{18}O$ demonstrates the denitrification potential in this well is a powerful driver of NO_X concentrations (Figure 77). The relationship between $\delta^{15}N$ and $\delta^{18}O$ is strongly linear ($r^2 = 0.83$) with a slope of 0.56, which is indicative of denitrification occurring. Labeling the sample dates shows that the inputs of nitrogen that contributed to the three major NO_X concentration spikes appear at the beginning of the denitrification line, in the range of $\delta^{15}N$ +6.5 to +8.66 $^{o}/_{oo}$. This is much less enriched than the new input nitrogen that we saw in the Merritt Island septic well MW SP 1739. After adding the source boxes described by, the new sources of nitrogen appear to be coming from soil nitrogen sources or possibly mineralized fertilizer (Figure 78).

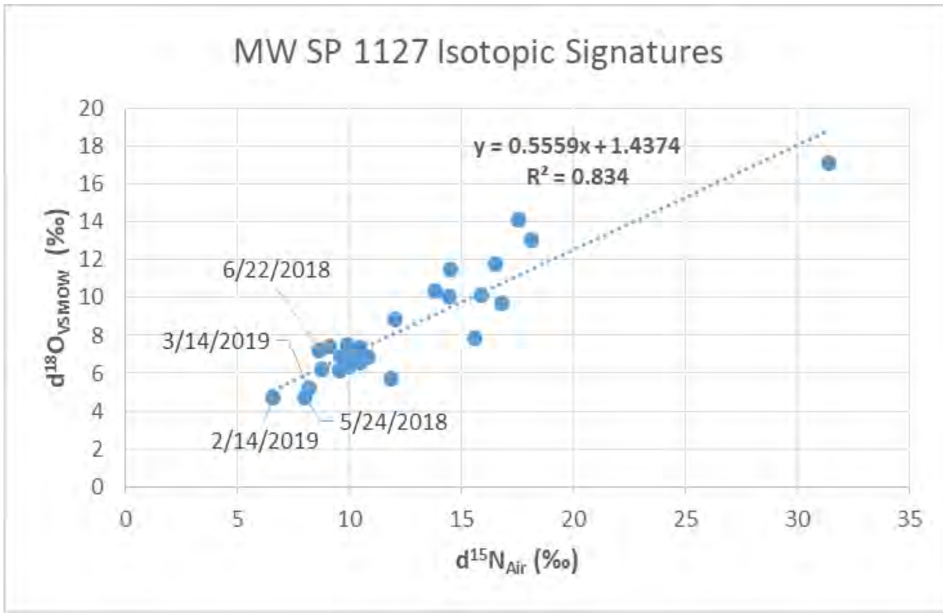

Figure 77: Turkey Creek septic well MW SP 1127 denitrification trend line.

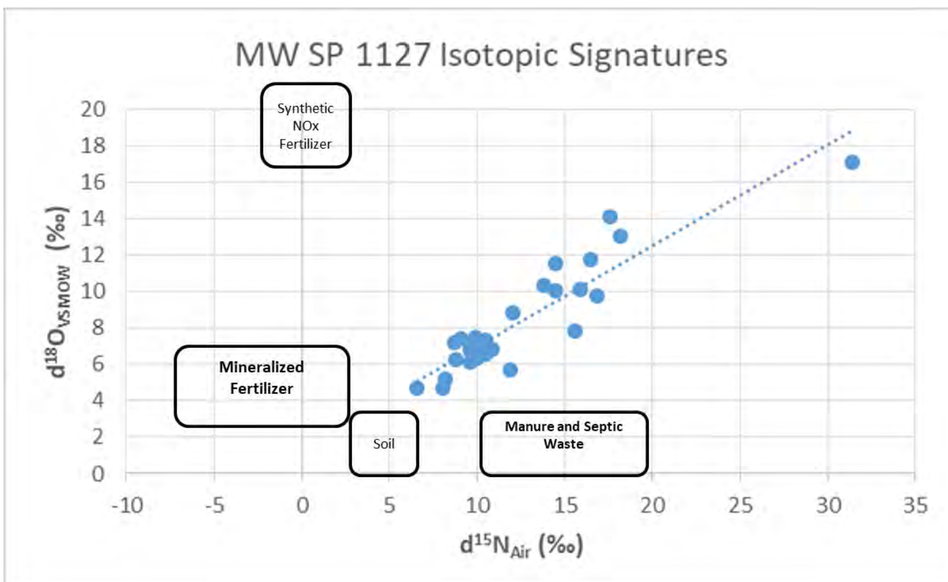

Figure 78: Turkey Creek septic well MW SP 1127 denitrification trend line and source characterization

Considering the power of denitrification at this site, and the Merritt Island site, the question can be asked, "Is the nitrogen being completely reduced during denitrification before it interacts with surface waters?" Mapping the push point data helps illuminate the fate and transport of the nitrogen coming out of the tank (Figure 79). This map shows the results of a single sampling event that took place on 10/8/2019, when NO_X concentrations were relatively low (4.8 mg/L) in the monitoring well, and many of the push point samples were too low in NO_X to analyze for $\delta^{15}N$ and $\delta^{18}O$.

Figure 79: Mapped Turkey Creek MW SP 1127 NO_X and $\delta^{15}N$.

Push point #PP15 located directly up-gradient of the septic drain field has the highest NO_X concentration of the three push points and a high $\delta^{15}N$ of +15.79 ‰. Further up-gradient is PP 16, which has a much lower NO_X concentration (0.5 mg/L) and lower $\delta^{15}N$ of +6.52 ‰, which does not appear to be a product of denitrification. This up-gradient source of nitrogen appears to be a lighter, less enriched source that is possibly a mixed soil source or atmospheric source like mineralized fertilizer. Downgradient of the septic tank, between the tank the adjacent surface water body, is PP 21, which has a NO_X concentration of 1.8 mg/L and a $\delta^{15}N$ of +18.66 ‰, which is what we would expect to see if denitrification was occurring. Although not conclusive, it appears NO_X of 1.8 mg/L is approaching the canal located behind the house, suggesting that even with highly active denitrification at work, nitrate is still reaching surface waters. More push point samples conducted seasonally can help confirm this suggestion.

Turkey Creek Sewer Community Aggregate Data from MW SE 841 & MW SE 849

Because NO_X concentrations were so low in the sewer community, the data from two monitoring wells were aggregated for the plots in this section. Nitrate concentrations in these wells are relatively low compared to the septic and reclaimed communities, but the enrichment values are incredibly high (Table 29). In fact, the lab analyzing our data called to determine if these samples were artificially enriched and they reran the samples to confirm these high values.

Table 29: Turkey Creek Sewer Isotope Data

Well ID	Sample Date	$\delta^{15}N_{Air}$ (‰)	$\delta^{18}O_{VSMOW}$ (‰)	NO_X (mg/L)
MW SE 841	7/14/2017	13.27	22.49	0.39
MW SE 841	8/14/2017	3.95	21.02	0.18
MW SE 841	9/21/2017	37.78	18.61	2.7
MW SE 841	10/12/2017	71.69	24.33	0.74
MW SE 849	6/16/2017	7.72	24.08	0.12
MW SE 849	7/14/2017	4.29	24.33	0.24
MW SE 849	10/12/2017	13.34	19.93	0.12

Plotting the $\delta^{15}N$ and $\delta^{18}O$ values show that there is not a denitrification trend line (Table 29) and that the $\delta^{15}N$ values range dramatically (from +3.95 ‰ to 71.69 ‰), while the $\delta^{18}O$ valued remained relatively constant (from +18.61 ‰ to 24.33 ‰). Each of these points could be representative of a new source signature. The cluster located near the synthetic fertilizer box may be indicative of a fertilizer source, but there is really not enough data here to understand the nature of the nitrogen source (Figure 80). It could be that the NO_X in these wells is the result of nitrified ammonia that has been nearly completely denitrified. This is an unusual and interesting pattern not seen elsewhere.

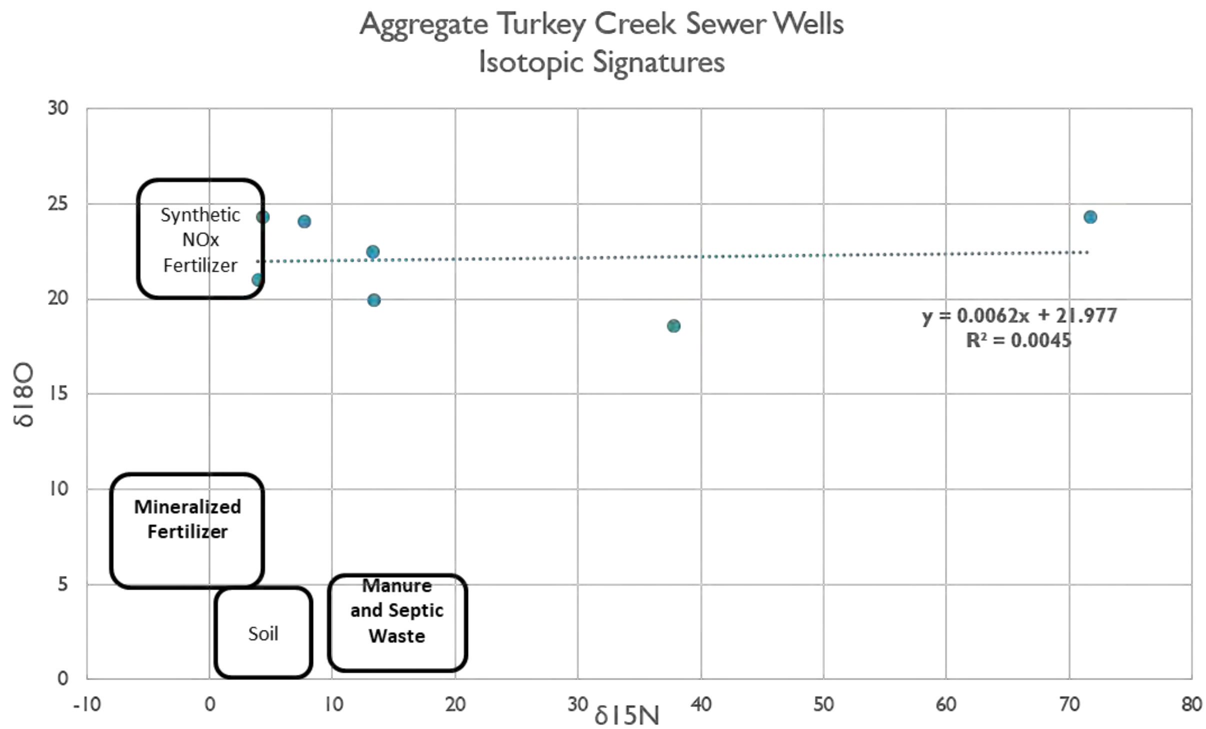

Figure 80: Turkey Creek sewer wells isotopic signatures and source characteristics.

Turkey Creek Reuse Monitoring Well MW RE C

The monitoring well centrally located in the reuse community was selected for the analysis, because it had the largest number of data and because its location makes it more representative. Graphing NO_X concentrations and $\delta^{15}N$ over time shows that there is not tremendous variation of either in this well (Figure 81). NO_X concentrations are relatively high, exceeding the drinking water standard of 10 mg/L in 29 of the 30 samples. The range of $\delta^{15}N$ varies little and stays within a range of 7.02-9.02 $^{o}/_{oo}$.

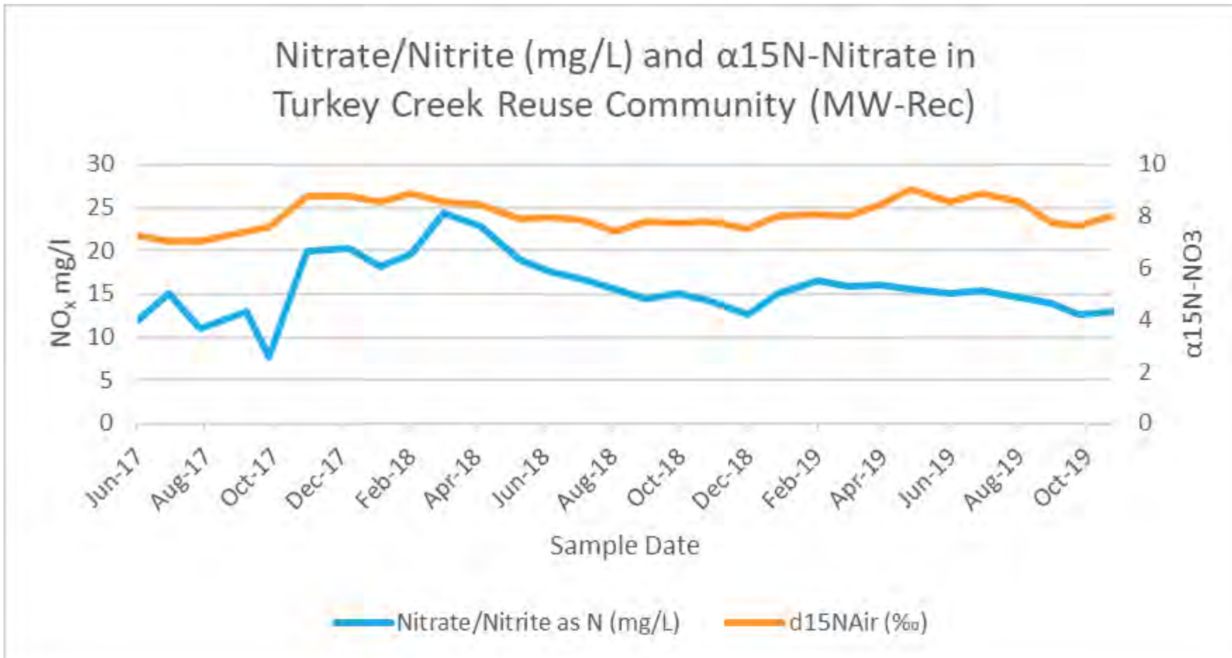

Figure 81: Turkey Creek reclaimed monitoring well MW RE C NO_X and $\delta^{15}N$ over time

An interesting denitrification line appears in the plot of $\delta^{15}N$ and $\delta^{18}O$ values (Figure 82), although the $\delta^{18}O$ values never exceed $3^o/_{oo}$ and the $\delta^{15}N$ range is so tight (within $2^o/_{oo}$). There is a trend line with a slope that indicates denitrification is at work here, but there also appears to be new inputs.

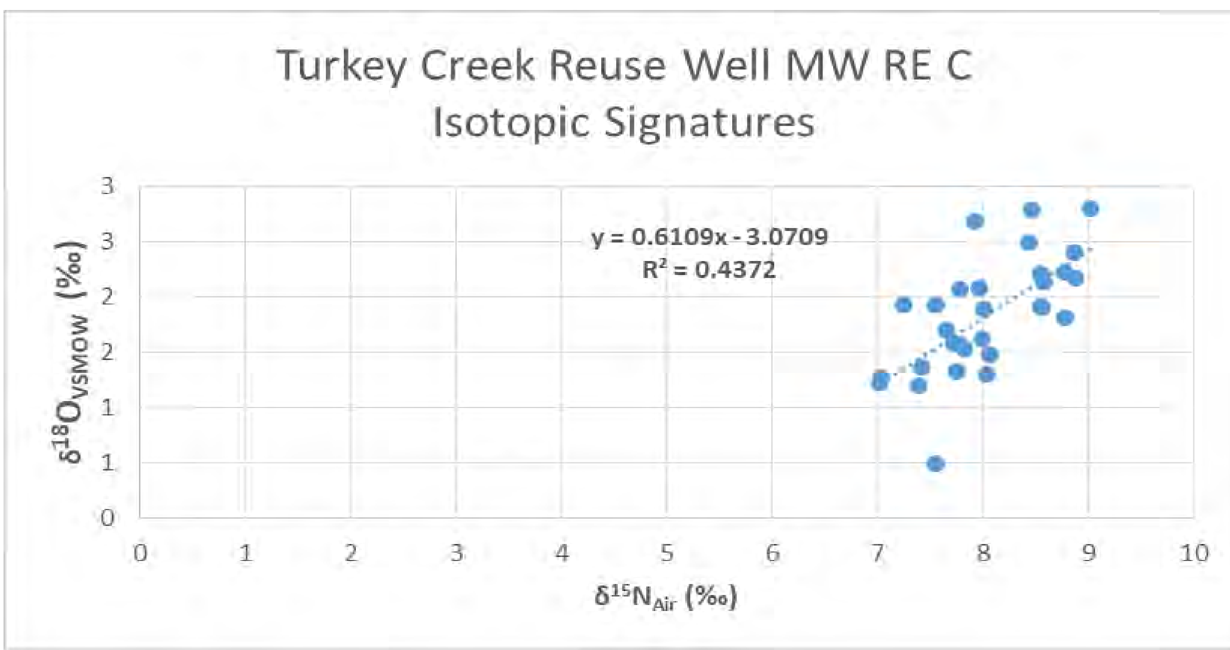

Figure 82: Turkey Creek reclaimed monitoring well RE C denitrification line.

Adding the source boxes reveals little in terms of a clear source (Figure 83). This community is irrigated with water that is high in nitrogen content after treatment at the wastewater facility. The single irrigation sample collected in the community had low NO_X (1.8 mg/L) that was depleted in δ15N (-.027 $^{o}/_{oo}$) and δ18O (-11.71 $^{o}/_{oo}$). The majority of nitrogen in the irrigation was in the form of ammonia (28.6 mg/L). So is the nitrate denitrified ammonia, or some other source? Additional analysis is needed to see the extent that this is denitrification or mixing.

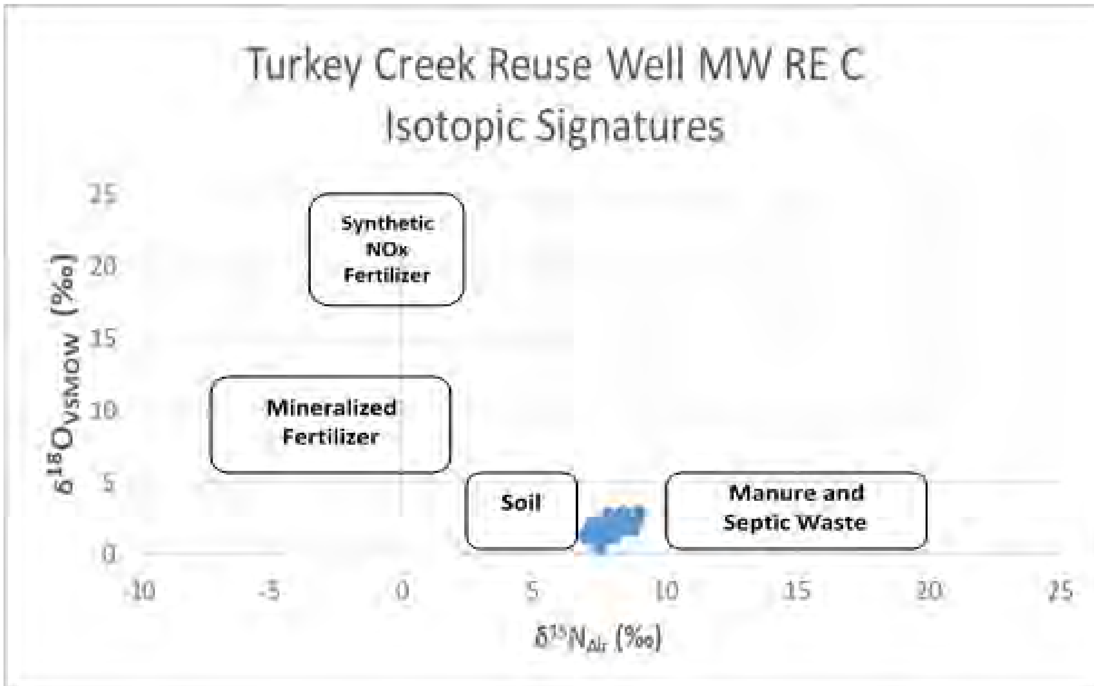

Figure 83: Turkey Creek reclaimed monitoring well MW RE C source characteristics.

Turkey Creek Natural Area Aggregate Data from Monitoring Wells MW TC 1 & MW TC 2

Of the seven (n=7) data collected in the natural wells that had high enough NO_X concentrations to run isotopes, six were collected from MW TC 2, the monitoring well closest to the neighborhood. As such, there may be upgradient sources that are influencing the nitrogen concentrations. In Table 30, it is clear that the one sample from monitoring well MW TC 1 differs from the others when you look at NO_X concentration relative to the isotopic signatures. The MW TC 1 well sample had a low NO_X concentration (0.034 mg/L) and low $\delta^{15}N$ enrichment (3.27 $^{o}/_{oo}$). In well MW TC 2, we see much more enrichment, with several values of $\delta^{15}N$ over 30 $^{o}/_{oo}$. This could mean that well is somehow influenced by another, more enriched, source of nitrogen or that denitrification is at work.

Table 30: Turkey Creek Natural Area Combined Well NO_X and $\delta^{15}N$ Data

Well ID	Sample Date	$\delta\ ^{15}N_{Air}$ (‰)	$\delta\ ^{18}O_{VSMOW}$ (‰)	NO_X (mg/L)
MW TC 1	6/15/2017	3.27	8.41	0.034
MW TC 2	8/9/2017	6.23	5.35	0.39
MW TC 2	9/20/2017	3.63	1.09	3.5
MW TC 2	10/11/2017	7.53	2.90	1.6
MW TC 2	11/14/2017	31.51	16.60	0.33
MW TC 2	11/8/2018	33.06	15.76	0.28
MW TC 2	9/12/2019	43.81	20.20	0.47

Plotting the $\delta^{15}N$ and $\delta^{18}O$ values for monitoring well MW TC 2 shows that in fact, denitrification is a strong force in this well (Figure 84). Adding the source boxes suggests that this could be a representative natural area well that is simply experiencing denitrification of the naturally occurring soil nitrate. This demonstrates the fact that a single sample enriched in δ15N does not necessarily mean there is an enriched source. It could just be natural denitrification processes working to reduce soil nitrogen.

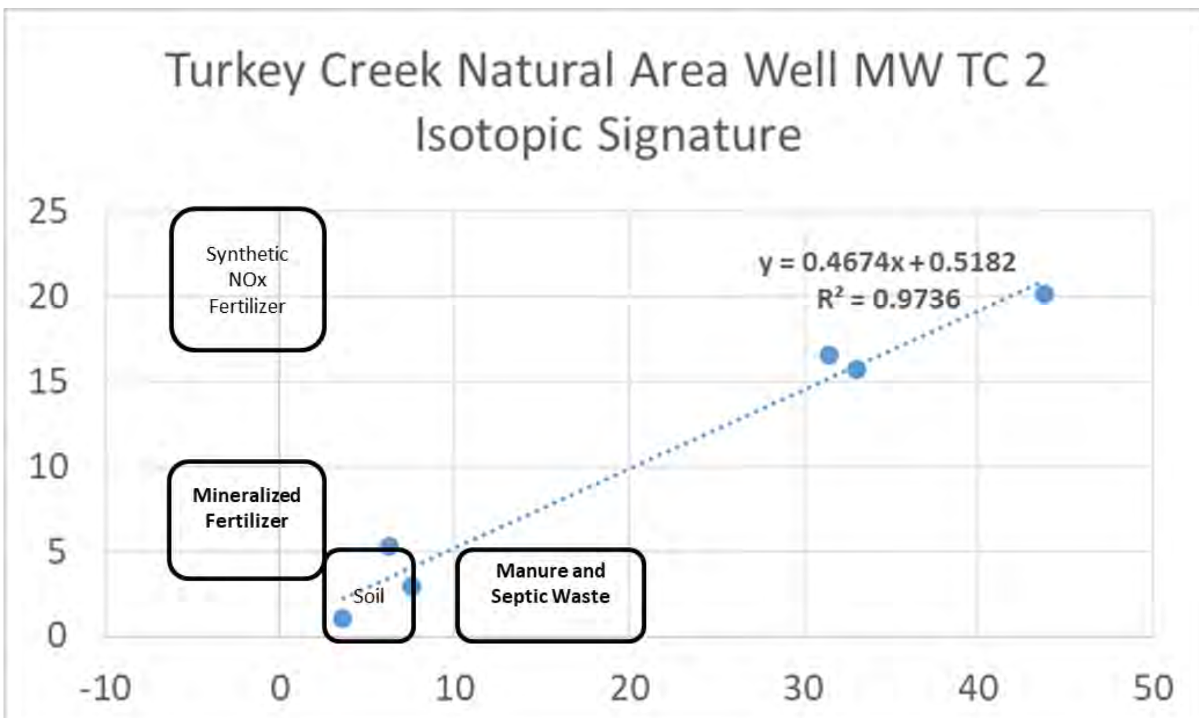

Figure 84. Turkey Creek reclaimed monitoring well MW TC 2 source characteristics.

Further Research and Study

Continued sampling for temporal influences

A continuation of sampling would allow for identification and assessment of seasonal influences on these regions, such as Florida's wet and dry seasons, seasonal residents, and extreme events. Continued isotope sampling also provides clarification on denitrification changes over time.

Increased sampling density

The three permanent monitoring wells installed in each community did not capture the community-wide conditions, based on the extreme variability between wells. This was especially true in the septic communities which had the largest variation of nutrient concentrations between wells. The installation of additional wells in each community would allow greater confidence in treatment level reporting, especially in communities where a seepage project could be implemented as well. It would also help clarify differences between treatments in different regions. The use of push points could supplement this in communities where groundwater is accessible. Push points could also help better delineate septic tank plumes.

Estimating denitrification rates

Denitrification effectively converts groundwater dissolved NOx to atmospheric gasses. We suspect that it greatly reduced nitrate concentrations in our study. From the isotopic research, we see major differences in denitrification within wells in the septic communities. Current septic loading models like ArcNLET, use a single decay coefficient to estimate the denitrification of nitrate. This study demonstrates that denitrification is a powerful driver of nutrient concentrations that varied tremendously between septic communities and individual wells. Additional laboratory research that calculates denitrification rates in different groundwater samples can demonstrate variations in denitrification scenarios and refine model decay coefficients with actual data. The samples could be selected form a subset of communities or wells and varying time periods, targeting the regular monthly events or specific rain events. This effort might allow more prioritization of septic communities for retrofit to take the regional denitrification rates into account.

Survey homeowners for landscaping and irrigation practices

Across every region and treatment there were potential sources of variation that may originate with actions that the homeowner takes. The most evident of these are the use of lawn fertilizers on the property and the use of reclaimed water for irrigation. A representative survey of homeowners in Brevard County could be conducted to better understand the timing, types, and amount of fertilizer being applied to residential lawns as well as irrigation practices. In addition, understanding the practices of the monitored homeowners, such as the number of residents per house, seasonal long-term visitors, and landscape practices might help explain some of the variability encountered in the dataset.

Survey existing monitoring wells so regional groundwater elevations can be compared

The research suggests that rainfall has an impact on groundwater nutrient concentrations. Because the existing groundwater monitoring wells were not initially surveyed, we only had a relative depth to water measurement for each well that could not be compared with other wells. A better understanding of groundwater elevation will allow better estimates of groundwater flow and velocity to refine loading estimates and better delineate contaminant plumes. Groundwater elevation is also comparable within and between regions.

Connecting groundwater to surface water

This study should be linked to other surface water studies to better understand the link between groundwater and receiving surface waters. Turkey Creek provides an opportunity to work with FIT scientists who are already examining surface water nutrients. Seepage meters can be installed in Turkey Creek and smaller Turkey Run near the groundwater study communities to measure the volume, concentration, and form of nitrogen entering the lagoon through seepage. Piezometers could be collocated with seepage meters to provide hydraulic head information. Additional wells installed in a transect connecting drain field to receiving waterbody would also help trace the nitrate and ammonia transport in a field setting. Both of these types of studies would provide a critical missing link between load estimates and actual conditions.

Baseflow component refinement of SWIL and Septic Moratorium Refinement

Expanding the refinement of the baseflow model component of the SWIL with field collected data by treatment type (and possibly also region) might provide to also better refine basin prioritization efforts for retrofits. This would be better accomplished with an additional year of field data, so monthly median values used provide representation of a minimum of 2 wet and dry seasons each. Seepage and well transect-based data (study described above) as well as a better understanding of denitrification rates would be useful to verify if the measured nitrogen and phosphorus concentrations should be buffered or reduced in specific regions. Volumes of baseflow reaching the Lagoon could also be calibrated based on a more extensive seepage study.

In addition, the newly acquired groundwater dataset might inform the spatial analysis effort to define the septic moratorium ordinance overlay. The full groundwater dataset can provide *in situ* data to calibrate the estimated nutrient input loading based on distance to the water, soil type, and region. Seepage data to determine the potential attenuation of the measured septic concentration data and actual denitrification rates might further help in this refinement effort.

Appendix A: STRATEGIC INITIATIVE RESULTS SUMMARY REPORT

STRATEGIC INITIATIVE RESULTS SUMMARY REPORT

Groundwater Pollution: Engaging the Community in Solutions Task 6 Deliverable

Results of community study to examine the customer satisfaction, usability and effectiveness of a proprietary septic tank maintenance product in reducing groundwater nutrient concentrations.

Leesa Souto, Ph.D
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Marine Resources Council

GROUNDWATER POLLUTION: ENGAGING THE COMMUNITY IN SOLUTIONS

Strategic Initiative Results Summary Report Deliverable for Contract Task 6

Executive Summary

The goal of Task 6 of the legislatively funded project titled, "Groundwater Pollution: Engaging the Community in Solutions" was to engage homeowners in an inexpensive intervention strategy that could potentially reduce the pollutants leaching from their septic tanks. To address this goal, an in-situ septic treatment product called BiOWiSH was distributed to nearly every resident located in the Turkey Creek septic tank community. BiOWiSH is described as an advanced enzyme technology that rapidly breaks down waste materials and reduces odor-causing compounds. It is a readily available, inexpensive, and easy-to-use product that is flushed down the toilet by homeowners quarterly. BiOWiSH advertises that it can reduce total nitrogen by 52.9%, chemical oxygen demand (COD) by 76.6%, and suspended solids by 89.2%. The Turkey Creek septic community was selected to receive the BiOWiSH because there was sufficient monthly data to constitute a pre- BiOWiSH condition. Turkey Creek sampling initiated in June 2017 as a pilot project to test the research methodology before implementing the County-wide project. The BiOWiSH intervention was initiated in the second quarter of this study and continued until the end of the sampling program (November 2019). A total of 76 homeowners (96%) within the community of interest agreed to actively participate in this intervention study and apply the product to their toilet every three months. The BiOWiSH product was delivered to homeowners quarterly for five quarters on October 22- 23, 2018, January 23, 2019, April 25, 2019, July 24, 2019, and October 25, 2019. Post-interventional changes in the sampled parameters of the three Turkey Creek septic community wells, particularly nitrogen constituents, were examined for any potential changes in concentrations. Monthly groundwater sampling continued as normal to determine if the BiOWiSH had any effect on total nitrogen concentrations. We found too much seasonal variation to clearly establish if the BiOWiSH is having a positive effect on groundwater nutrient concentrations. According to survey responses, participating residents were overall pleased, they found the project easy to use, they believed it was having a positive impact and they were willing to pay to continue using the product. Future research may focus on sampling immediately after product application, or continuing monthly sampling to assess seasonal variations.

Participant Recruitment

Brevard County Property Appraiser data were acquired for the geographic boundaries of the Turkey Creek septic tank community to create a list of 103 addresses. Eight (8) of these were determined to be vacant parcels, 14 were hooked up to sewer and 1 was not accessible for a total of 80 homes to recruit for the BiOWiSH study. Recruitment was initiated with a letter that was mailed to homeowners explaining the project goals and announcing when team members would be distributing the BiOWiSH product. Thereafter, all of the Turkey Creek homes were visited by teams of MRC staff members and volunteers. A pledge card was provided to participants with details on how to use the product and contact information was collected to notify them of future deliveries. Those who were not home were left a "Sorry we missed You" flyer that explained the project and requested participation. Of the 80 homes visited, 96% (n = 76) pledged to participate. Six of the homes had two septic tanks and they were provided two bags of BiOWiSH each quarter (one for each tank).

BioWish Implementation

Participating homeowners were instructed to flush the BiOWiSH product down one toilet that leads to each septic tank at the end of the day. MRC personnel stayed in regular contact with them to remind them how and when to use the product and to announce when quarterly deliveries would occur. Volunteers were engaged to assist MRC staff with the distribution of product and instructions over five quarters October 22- 23, 2018, January 23, 2019, April 25, 2019, July 24, 2019, and October 25, 2019. Pledge cards were reviewed to ensure homeowners had flushed their BiOWiSH as instructed. A behavior and willingness to pay survey was included with the final delivery package with postage paid for participants to return.

Results

Total Nitrogen

Previous independent research of the BiOWiSH product found a 52.9% reduction of Total Nitrogen (TN) concentrations in leachate immediately after the product was used. The average TN concentration of the Turkey Creek septic wells shows seasonal variations with the average TN peaking during the wet season and lower during the dry season. In 2019, we also see an abnormal peak during the months of February and March 2019, possibly due to an out of season rainfall event of greater than 6 inches before the sampling event (Figure 1).

After the first delivery of BiOWiSH, there was a slight increase in average TN concentration in November 2018, but the concentration decreased in both December 2018 and January 2019. After the second delivery, there was a sharp increase in average TN in February 2019, likely due to a high rainfall event (>4") just prior to the sampling effort. After the third delivery of the product in May, average TN concentrations appear to decline, however, a declining trend was already in place (April 2019). The expected increase in TN concentration in June and July 2019 was not as drastic as the increases measured in the two previous years (2017 and 2018). This may be associated with a BiOWiSH mitigating effect. There was a slight increase in TN concentrations measured in the community wells after the August BiOWiSH delivery, but this is likely a result of the 2.24 inches of rainfall which occurred over a three-day period directly preceding the sampling event. TN concentrations decreased in November, after the delivery of the October BiOWiSH package.

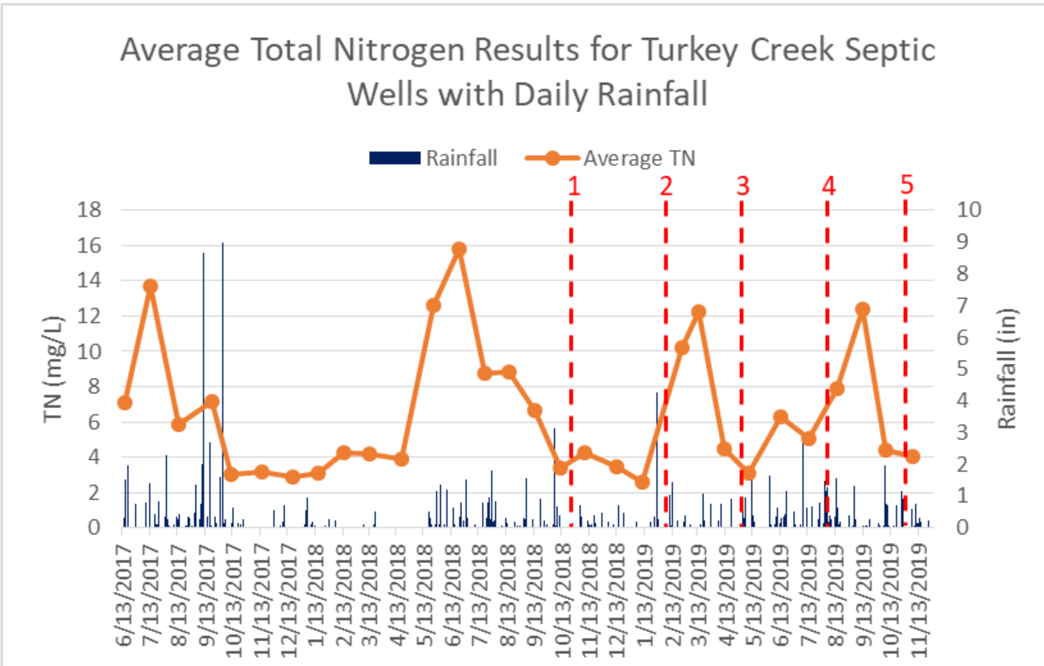

Figure 1. Average TN concentration for the Turkey Creek septic community wells, from June 2017 through November 2019. The red dotted lines represent the delivery dates of the BiOWiSH product and blue lines represent daily rainfall.

Each of the three wells in Turkey Creek community is unique and it is helpful to examine them independently (Figure 2). Two of the wells (MW SP 981 an MW SP 1099) had fairly stable TN concentrations throughout the sampling period. MW SP 981 had lower TN concentrations (0.79-1.90 mg/L) than MW SP 1099 (4.10-9.30 mg/L). After the first four deliveries of BiOWiSH, there was little to no change in the TN concentrations at either MW SP 1099 or MW SP 981. In contrast, MW SP 1127 displayed an extreme seasonal variation of TN concentrations ranging from 0.71 to 37.60 mg/L. After the first delivery of BiOWiSH, MW SP 1127 there was a slight increase in concentration in November 2018, but then the TN concentrations clearly decreased in the two subsequent months. This reduction in concentrations could be due to a relatively dry period. After the second delivery in late January 2019, concentrations of measured TN sharply increased to 24.5 mg/L in February and to 31.2 mg/L in March 2019. The increase from January to February 2019 is likely due to an unusually wet period, which included a heavy rainfall day. However, the continuation of this increasing TN concentration trend into March 2019 does not appear to be the result of increased precipitation. In April 2019, there was a sharp decrease in TN concentration at MW SP 1127 and after the third delivery of the product the decreasing trend continued into May 2019. In June and July 2019, there was an increasing trend in the TN concentration at MW SP 1127 and the trend continued after the fourth delivery and into August 2019. There was a continued decrease in TN concentrations in MW SP 1127 after the final October delivery.

The tremendous variability in monitoring well MW SP 1127 is worth further investigation. Increasing TN concentrations peak well above the average concentrations on 7/13/17, 6/13/18, 3/13/19, and 9/13/19. The Total Nitrogen concentration in this well is made up almost entirely of organic nitrogen that include TKN and ammonia, suggesting incomplete nitrification.

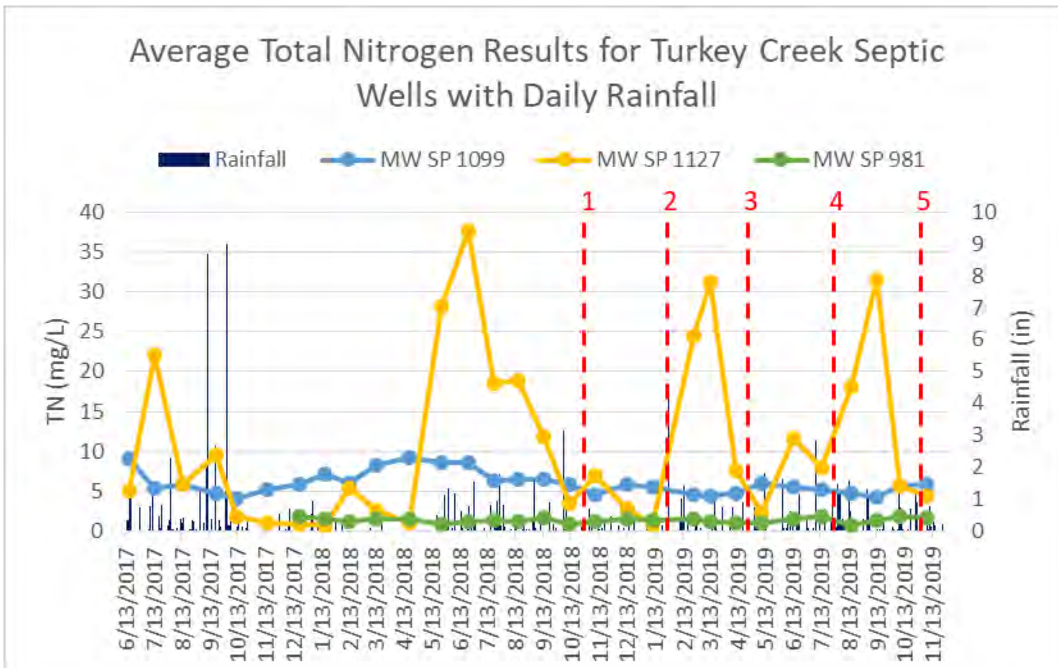

Figure 2. TN concentrations for the three Turkey Creek septic community wells, from June 2017 through November 2019. The red dotted lines represent the delivery dates of the BiOWiSH product and blue lines represent daily rainfall.

Total Phosphorus

Total Phosphorus (TP) concentrations were measured bi-monthly from May 2018 to November 2019 providing limited sampling events after each of the BiOWiSH interventions. The average TP concentrations ranged from 0.52 mg/L to 1.25 mg/L for the first nine months that TP was sampled (Figure 3). After delivering BiOWiSH in October 2018, average TP concentration decreased slightly over the next two months and decreased substantially after the second product delivery in January. However, from February to April 2019, this trend reversed and TP concentrations rebounded, but never returned to the pre-intervention concentrations. After the third delivery of the product, there was a slight increase in TP concentration (0.02 mg/L) from April to June 2019. After the fourth delivery of the product, there was a sharp decrease in TP concentration from June to August 2019 (0.30 mg/L). The final sampling event in November demonstrated a decrease in TP after the fifth and final delivery date.

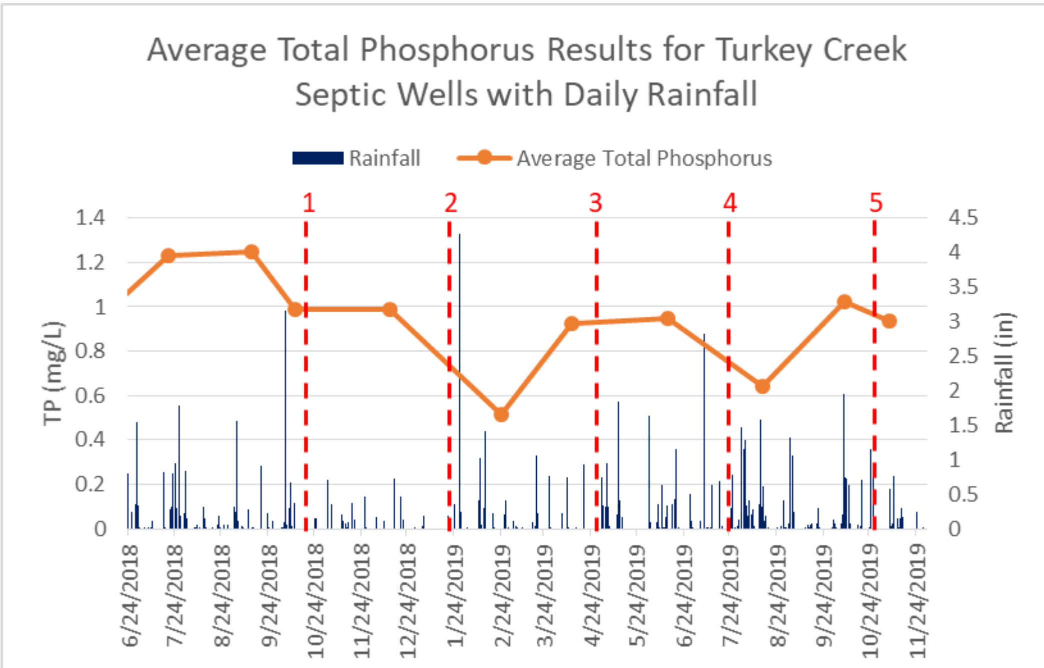

Figure 3. Average TP concentration for the Turkey Creek septic community wells, from May 2018 through November 2019. The red dotted lines represent the four delivery dates of the BiOWiSH product and the blue lines represent daily rainfall.

As with TN concentrations, it is useful to examine each well individually (Figure 4). TP concentrations in monitoring wells MW SP 1099 and MW SP 1127 decreased after the first and second deliveries of BiOWiSH but MW SP 981 presented an increase in TP concentration after the first delivery and a decrease after the second delivery. After the third delivery of the product, all three wells showed little change in concentration ranging from a 0.1 mg/L increase to a 0.1 mg/L decrease. After the fourth delivery of the product, MW SP 1127 had no change in TP concentration and MW SP 1099 had a small increase in concentration. MW SP 981 demonstrated a substantial decrease throughout the sampling period, with a TP reduction of 0.93 mg/L.

With only five deliveries in one community and uncertainties related to seasonal variabilities, it is impossible to draw any definite conclusions about the effectiveness of the proprietary product in improving groundwater quality. Based on preliminary data, no consistent, across the board reduction in TN and/or TP concentrations are apparent after the BiOWiSH product is delivered.

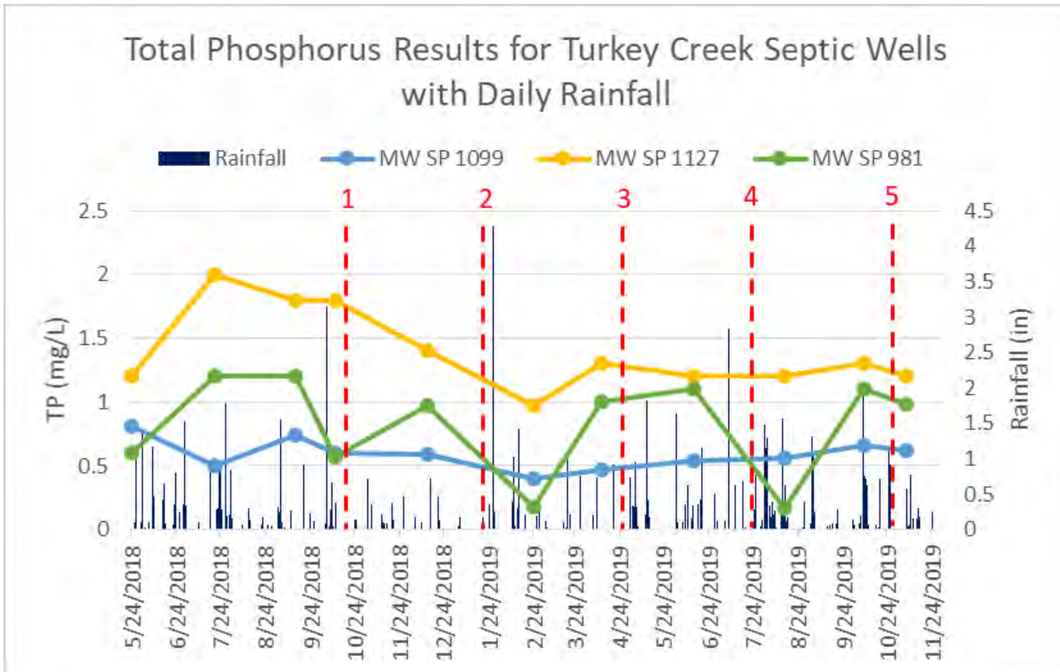

Figure 4. TP concentrations for the three Turkey Creek septic community wells, from May 2018 2017 through November 2019. The red dotted lines represent the delivery dates of the BiOWiSH product and blue lines represent daily rainfall.

Social Survey

A brief 6-question survey card was distributed to product recipients with their final delivery of BiOWiSH and about 33% of the surveys were completed and returned. The goal of the survey was to evaluate the extent that participants liked the product, that they received the product, they found it easy to use, the believed it was making a difference, and they were willing to pay for it in the future. All respondents (100%) indicated that they received and found it very easy to use. All respondents except one had flushed every bag of product provided - some received two bags/household and one missed a delivery. When asked if they felt the product was having a positive impact on septic tank function, nearly 1/3 of respondents answered that they didn't know. Those who responded, were favorable, believing the product was having a somewhat to very positive impact on septic tank function. There was a similar response to the questions that asked if they felt the product had a positive impact on water quality. Over 1/3 responded that there were unsure, the remainder thought it was having somewhat of a positive impact on water quality. Lastly, the survey asked how much the homeowner would be willing to pay every three months for the product. The average amount respondents would be willing to pay was $6.67 every three months. Four respondents indicated they would be unwilling to pay anything ($0).

Appendix B: Groundwater Modeling Report

Groundwater Modeling Memorandum Report

Incorporating In Situ Groundwater Quality Data into Groundwater Nutrient Transport and Watershed Loading Models

FDEP CONTRACT #LP05112, FINAL DELIVERABLE TASK 2

PREPARED FOR:

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PREPARED BY:

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1 OBJECTIVES

In June 2018, Marine Resources Council (MRC) and Applied Ecology, Inc. (AEI) initiated an extensive groundwater sampling effort that included installing and monitoring 45 shallow groundwater wells throughout Brevard County. A large portion of this effort is a legislative-funded groundwater pollution investigation called "Groundwater Pollution, Engaging the Community in Solutions Study" (GW Pollution) which addressed the need to identify critical areas where groundwater contamination is most negatively impacting the Indian River Lagoon (IRL). The legislative study funded the installation of 20 shallow groundwater wells and the continuous monitoring of a total of 30 wells in 11 communities, as well as spatial data acquisition and analyses, groundwater modeling, a representative behavioral survey, and reporting. An additional 15 shallow groundwater wells in five communities were funded by the Brevard County Save Our Indian River Lagoon (SOIRL) Trust Fund. The data from these five additional communities were included with the data from the larger groundwater study to compare nutrient and bacteria concentrations in communities with septic systems, those with sewage, and those with sewage and reclaimed water. This Groundwater Modeling Memorandum Report provides the final deliverable for FDEP Contract LP05112, Task 2.

To identify the best predictive resources available for this project, a literature review was performed to compare various available models capable of estimating nitrogen transport and transformation through soil and groundwater. The three models examined in this review include: 1) STUMOD-FL-HPS, 2) ArcGIS-based Nitrate Load Estimation Toolkit (ArcNLET), and the 3) Loading Simulation Program in C++ (LSPC).

The first model, STUMOD-FL-HPS, was developed as part of the Florida Onsite Sewage Nitrogen Reduction Strategies (FOSNRS) project. The project was commissioned in 2009 by the Florida Department of Health (FDOH) and led by Hazen and Sawyer. This model is a modification of the original STUMOD model and has been specifically designed to estimate nitrogen contributions from onsite sewage treatment and disposal systems (OSTDS) into Florida soils and aquifers. The model has a user-friendly Excel-based interface; can simulate multi-dimensional nitrogen movement, through both soil and groundwater, from either a single OSTDS or multiple OSTDS sources; and includes Florida specific soils and climate conditions. While the model is designed for more general analyses, it allows for a wide range of parameter configurations to better represent site-specific conditions.

The STUMOD_FL_HPS model was tested at the Gulf Coast Research & Education Center. Model calibration achieved an R^2 value of 0.66 after increasing the input nitrate concentration at the water table above both predicted and observed results; discrepancies in the model were attributed to the heavy influence from the agricultural areas' nitrate plumes. The model estimates were found to be conservative, with measured levels of denitrification being higher than model predictions. It should be noted that the simplified modeling approach used by STUMOD-FL-HPS makes it incapable of adequately predicting all environmental and OSTDS configurations. In high-risk scenarios, the model uncertainty may be unacceptable. In addition, the model does not account for other sources of nitrogen (such as agricultural inputs) or interactions between multiple nitrogen plumes.

The second model, ArcNLET, is an ArcGIS based toolkit developed alongside the STUMOD-FL-HPS model by Rios, Ye, Wand, and Lee (2011). This toolkit uses the same processes as STUMOD to estimate nitrogen transport through soil in a GIS environment. The GIS-environment allows the import of various layers of information, such as terrain elevation, soil survey data, location of water bodies, and location of parcels served by OSTDS. The concentrations of each nitrogen plume are mapped onto a raster layer, allowing representation of multiple plumes on a map. Users should be aware that this model assumes the concentration reaching the water table is the same as the initial concentration. This assumption can result in over or underestimations of the mass loading from the system. Additional limitations of this model include: 1) treating the water table as a subdued replica of topography and representing groundwater flow in 2-D and a steady-state and 2) the need for an empirical or calibrated value for the decay coefficient.

The third model, Loading Simulation Program in C++ (LSPC), is a comprehensive data management and modeling system capable of simulating in-stream processes and representing loading, water flow, and water quality from point and non-point sources. LSPC has been successfully used in many case studies across the country and can be tailored to many different environmental conditions. However, due to the program's complexity and extensive calibration, a higher level of user expertise is necessary. In addition, at least ten years of historical data is recommended to calibrate the model. An additional year of data prior to the simulation period is also recommended for use as a "spin-up" run, making it less suited to areas lacking such data records.

Based on the literature review, it was decided that the ArcNLET model would be used to assess the potential contribution of OSTDS to the overall nitrate and ammonium loading of the study area. ArcNLET was selected based on the following rationale: 1) it is a relatively simple model that required limited input data but still incorporates key hydrogeological processes of groundwater flow and nutrient transport as well as spatial variability, 2) it is the model currently accepted by the FDEP to receive BMAP credit for removing or retrofitting septic tanks within a watershed with a Total Maximum Daily Load (TMDL), and 3) it can be calibrated with *in-situ* measured data for hydraulic head and nitrate and ammonium concentrations which are key to providing realistic results..

In addition, this Memo Report also includes a brief overview of the potential impact of the field collected groundwater quality data as input information for the baseflow component of a regional watershed loading model, the Spatial Watershed Iterative Loading Model (SWIL). For this baseflow model refinement, median measured water quality concentrations were assigned by land use and type of wastewater system (i.e. OSTDS, centralized sewer, and centralizer sewer with on-site reclaimed) to a selected basin in mainland Brevard County. A comparison of total estimated nitrogen and phosphorus loads for the basin between the original SWIL model run and the refined model run are further described below.

The overall purpose of this Groundwater Modeling Memorandum Report is to present the results of the predictive groundwater model runs, which include the calibrated ArcNLET runs and SWIL baseflow components. This memo report synthetizes results from several tasks, as described below.

1.1 TASK 1

The purpose of this task was to establish input data and run a series of preliminary groundwater model runs based on a modification of ArcNLET. The preliminary model runs were basin specific and only used the historically available data (*i.e.*, water levels and surficial water quality data) for calibration. These initial outputs were also used to guide well installation efforts to enhance the value of the collected groundwater quality data to represent the community of interest and calibrate these initial ArcNLET model runs.

1.2 TASK 2

The purpose of this task was to refine and calibrate the preliminary ArcNLET model using the data collected during the 18-19 months of groundwater monitoring. The focus of the calibrating effort was on communities on septic, since ArcNLET was built to specifically allow user to estimate the nutrient contribution of individual septic tanks to local surface water. ArcNLET requires the calibration of various input variables (*i.e.*, soil hydraulic conductivity, soil porosity, smoothing factors, source concentrations, *etc.*). Predicted nitrogen loading between the uncalibrated and calibrated model ArcNLET model runs is compared by community of interest.

1.3 TASK 3

This task included exploring a unique ArcNLET function to estimate model uncertainty, called the Monte Carlo Simulation for Uncertainty Quantification, hereafter called the MC Simulation. Critical driving factors of the nitrate transport were evaluated as part of this effort, to better understand the magnitude of uncertainty inherent to nitrate load estimates developed for management and planning purposes. The following single parameters were explored using the MC Simulation: Smoothing Factor, Hydraulic conductivity, Porosity, and septic tank source nitrate concentration. The simulation was applied to two study areas of interest, one representative of Barrier Island conditions (Melbourne Beach) and another mainland conditions (Suntree). Results were synthetized to describe the variability of the estimated loadings based on randomized runs of parameters of interest, highlighting the impact of each the environmental variables on the predicting nitrate from ArcNLET based on Brevard County conditions.

The task also involved a comparative analysis between original SWIL model and a refined version of SWIL using site specific data within a specific community in the study area (Suntree). The original SWIL model used well studied and approved event mean concentrations (EMCs) that associated with land use for predicting direct runoff nutrient loading to the IRL. However, the original model relied on a limited number of data to develop land use-related groundwater concentrations, and a single set of EMC values (one for nitrogen and another for phosphorus) were developed to estimate the groundwater loading to the Lagoon. Under this task, the set of static groundwater EMC values were replaced by the *in situ* TN and TP concentration data for four critically different types of areas: natural (undeveloped), those serviced by OSTDS, those serviced by central sewer without reclaimed water, and finally those with central sewer and reclaimed.

2 GROUNDWATER NUTRIENT TRANSPORT MODEL PREDICTIONS

2.1 ARCNLET

2.1.1 BACKGROUND

The ArcGIS-based Nitrogen Load Estimation Toolkit (ArcNLET) model was developed by the Florida Department of Environmental Protection (FDEP) and Florida State University (FSU) to model the fate and transport of nitrate and ammonia in surficial groundwater, originating from onsite sewage treatment and disposal systems (OSTDS), also known as septic tanks (Rios, Ye, Wand, and Lee, 2011; Rios, Ye, Wang, Lee, Davis, and Hicks, 2013) . ArcNLET was originally designed to estimate nitrate loads to surface water bodies from OSTDS, and it was updated to simulate ammonia, critical to better understanding total nitrogen loading to surface water bodies (Zhu, Ye, Roeder, Hicks, Shi, and Yang, 2016). ArcNLET requires the calibration of various input variables (*e.g.*, soil hydraulic conductivity, soil porosity, smoothing factors, source concentrations, *etc.*). This section will describe the efforts used in calibrating the ArcNLET model runs using site specific acquired data (i.e. soil information) and 18 months of groundwater quality monitoring data across Brevard County. The importance of using relevant and site-specific data for forced model calibration is visible when comparing the uncalibrated with the calibrated model runs.

The following sections describe the inputs, methodology, and resulting initial and calibrated nitrogen loading estimates using a custom ArcNLET model for various communities throughout the groundwater monitoring study area.

2.1.2 MODEL INPUTS AND METHODOLOGY

The ArcNLET model requires several input parameters, some with widely available data and others that require site-specific information that is mostly unknown. The typical input datasets generated from available data sources for use in ArcNLET include the following parameters, with unknown parameters denoted with an asterisk (*):

- Topography (digital elevation model (DEM) data acquired from the United States Geological Survey (USGS))
- Soil hydraulic conductivity data (United States Department of Agriculture (USDA) Soil Survey Geographic database (SSURGO) database)
- Soil porosity data (USDA SSURGO database)
- Septic tank and drain field location data placed using recent high-resolution aerials (developed by Applied Ecology based on an assessment of data from the Florida Department of Health (FDOH), FDEP, USDA, USGS, Brevard County, and a majority of the cities located within the County)

- Waterbody location data (developed by Applied Ecology based on an assessment the USGS National Hydrography Dataset (NHD), culverts/open channels obtained from Brevard County and various cities located within the County, and aerial photointerpretation)
- Soil dispersivity*
- Decay coefficient of denitrification*
- Source load and concentration*

The unknown parameters can only be determined, typically, through an extensive calibration effort based on locally collected groundwater quality data. To determine the importance of incorporating *in situ* data to better fine tune the above referenced unknown parameters into the ArcNLET model, uncalibrated and calibrated model runs were performed.

An uncalibrated version of ArcNLET was performed prior to the collection of groundwater monitoring data to assist in site selection for well placement within the selected monitoring communities throughout the County (selection of communities is included in the Groundwater Quality Final Report). Water level data for nearby groundwater monitoring wells were retrieved, when available, from the FDEP Petroleum Program and incorporated into ArcNLET for a better representation of the water table within specific areas when available. However, sources for long-term recent relevant data within out study areas were extremely limited. It should be noted that the uncalibrated model run for the Turkey Creek region used for this analysis was originally performed during a Florida Tech Legislatively funded pilot study (DEP Grant Agreement No. S0714 – Brevard County Muck Dredging); results from this run were used for equal comparisons to the other study areas.

Once site-specific data for the 18 sampling events were collected, median concentration data of nitrate and ammonia were incorporated into ArcNLET to refine and individually calibrate nitrate and ammonium predicted loads for the Merritt Island, Suntree, Melbourne Beach, and Turkey Creek study areas. Drain field locations were slightly modified from the uncalibrated model run after more accurate locations, using field collected knowledge, were determined. Model boundaries for each region were reduced for this post-sampling model run, allowing calibration to take place only in the area of interest, which focused on the monitored septic communities.

2.1.3 PREDICTED GROUNDWATER NUTRIENT LOADING BY COMMUNITY

Predicted loads of nitrate and ammonia were produced from the uncalibrated and calibrated ArcNLET model runs for each model area. For ease of comparison, ammonia and nitrate loads were also summed to provide closer estimate of predicted “nitrogen” loads per model run. Other forms of nitrogen, such as urea, are not included in the ArcNLET model estimates and hence the “nitrogen” referred to below is likely a less than total nitrogen.

For comparison purposes, only predicted nitrogen (ammonia + nitrate) loads being transported into waterbodies directly connected to either the Banana River, North IRL, or Central IRL are reported in the summary results included below. Additionally, a select number of these waterbodies were classified as wetlands with high connectivity to the Lagoon. In these cases, reductions in overall loading potential could take place due the inherent nutrient attenuation occurring in the wetlands. These attenuations were not included in the estimated provided below, since they are difficult to estimate without appropriate monitoring data.

Although the ArcNLET calibration process allows for a more accurate prediction of output loads within the targeted areas of interest (*i.e.*, the monitored septic communities of each region), the reduction in model boundaries also reduces the input number of septic tanks and prevents the opportunity of having truly comparable results between the uncalibrated and calibrated model runs. For more realistic comparisons to take place, the total loads from each area must be converted to loads per septic tank.

The average load per septic tank was calculated by dividing the predicted loading values of nitrate, ammonia, and calculated nitrogen by the number of modeled septic tanks in that particular area. Equation 1 provides an example of the per septic tank normalization process undertaken to determine the average annual nitrate load (lbs./year) per septic tank using the results from the Melbourne Beach uncalibrated model run.

Equation 1. Calculation of the average annual nitrate load (lbs./yr) per septic tank for the Melbourne Beach uncalibrated ArcNLET model run.

$$\frac{452.06 \ \ lbs/yr}{213 \ \ Modeled\ Septic\ Tanks} = 2.12\ lbs\ /\ year\ /\ septic\ tank$$

Next, the average estimated loading values per septic tank were multiplied by the number of septic tanks within the monitored septic communities. Equation 2 provides an example of this process to determine the average annual nitrate load (lbs./year) per septic tank using the results from the Melbourne Beach uncalibrated model run.

Equation 2. Calculation of the predicted annual nitrate loads (lbs./yr) from the uncalibrated ArcNLET model run within the monitored Melbourne Beach septic community.

$$2.12\ lbs/year/septic\ tank \times 79\ septic\ tanks\ = 167.67\ lbs/yr$$

While this process underestimates the predicted nitrogen loads of particular septic tanks, especially those within close proximity to a waterbody, the results are sufficient to provide the necessary comparisons between the ArcNLET model runs and highlight the importance of calibration efforts. Data and specific details from the uncalibrated and calibrated model runs for each study area are provided in the subsections below.

2.1.3.1 MERRITT ISLAND

Median concentration values of ammonia and nitrate for the three monitoring wells of the Merritt Island septic community during the 18-month study are presented in Table 1. Measured concentration values were generally highest at MW SP 1739 and lowest at MW SP 1688, with ammonia being the dominant nitrogen constituent, anticipated due to shallow water table and likely anaerobic conditions of the drain fields. These data were used to calibrate the input source concentrations of the ArcNLET model for the Merritt Island study region.

Table 1. Median concentration values (mg/L) of ammonia and nitrate-nitrite measured during the first 18 months of sampling at each monitoring well within the Merritt Island septic community. Median measured values of ammonia and nitrate-nitrite were summed to create a combined "nitrogen" value (mg/L), similar to the values provided by ArcNLET.

Parameter	**MW SP 1655**	**MW SP 1688**	**MW SP 1739**
Ammonia (mg/L)	1.35	0.63	2.45
Nitrate-Nitrite (mg/L)	0.025	0.025	1.4
Nitrogen (Ammonia + Nitrate-Nitrite) (mg/L)	1.375	0.655	3.85

Final calibration run estimated outputs at the monitoring well location against the measured water quality at each of the locations are provided in Figure 1 for nitrate and Figure 2 for ammonia. Calibration was only successfully achieved for two wells for either nitrate or ammonia. The model overestimated nitrate for SP 1739 and ammonia for SP 1655, likely due to an anomaly in the plume generation component of ArcNLET. Development of extraneous plumes in areas where no septic tanks have been designated by the user has been one of the issues noted in a few of the modeled areas of interest using ArcNLET. This was observed, in particular, for the ammonia plume generation during both the uncalibrated and calibrated model runs (Figure 3).

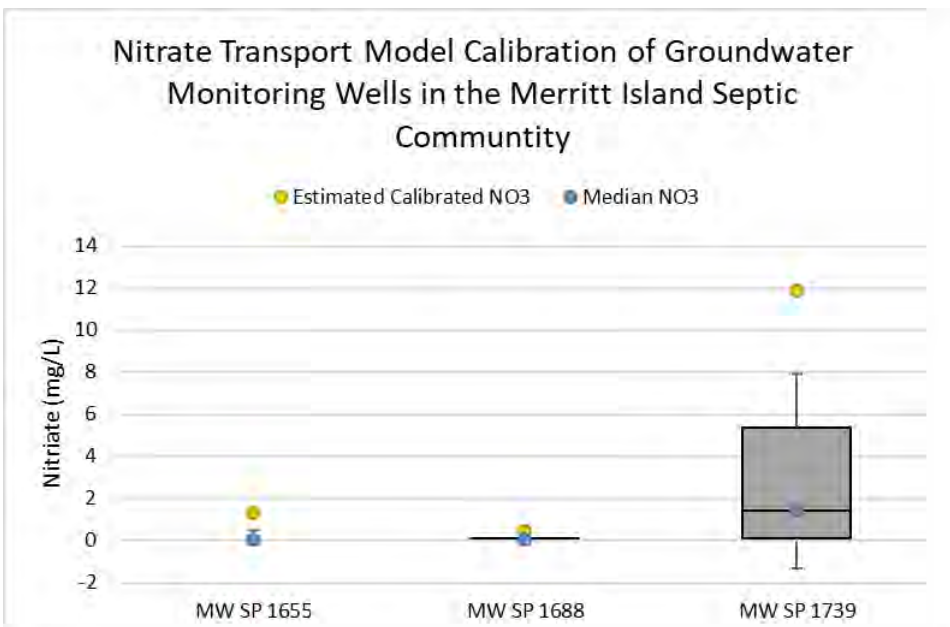


Figure 1. ArcNLET nitrate transport model calibration of groundwater monitoring well in the Merritt Island Septic Community.

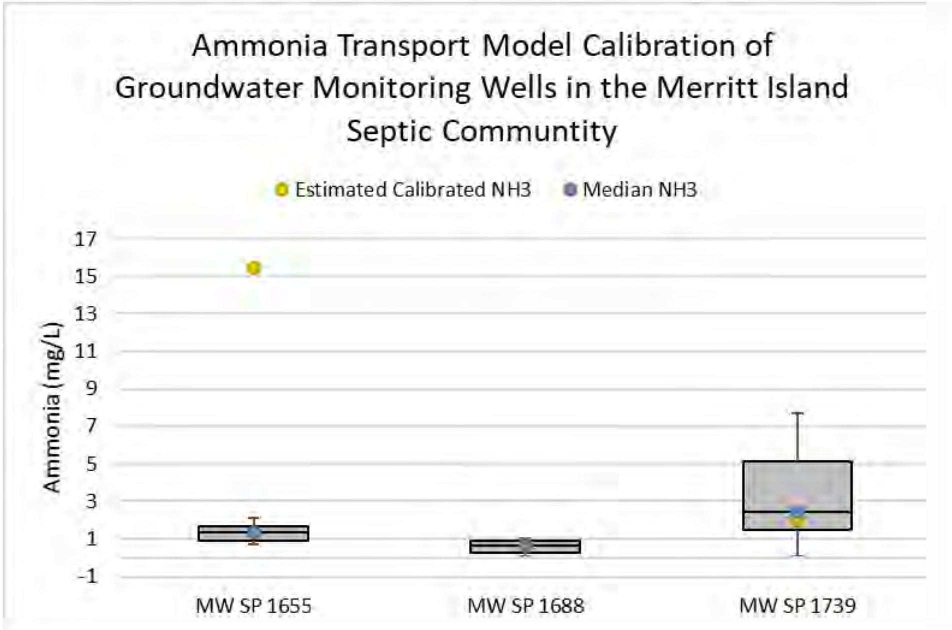


Figure 2. ArcNLET ammonia transport model calibration of groundwater monitoring well in the Merritt Island Septic Community.

The Merritt Island study area was reduced from 241 septic tanks in the uncalibrated model run to 132 septic tanks during the calibration process. The remainder of this discussion will focus on the 80 septic tanks located directly in the monitored septic community. Once calibrated, the ammonia plumes had slightly higher magnitudes with narrower diameters than the original uncalibrated model runs (Figure 3). The nitrate plumes had lower magnitudes with similar diameters to the uncalibrated run; this is anticipated as nitrate-nitrite concentrations are typically much lower than ammonia in this region, and the calibration process allowed for a correction of the over-estimation of this parameter (Figure 4). Differences in plume directions for ammonia and nitrate between the model runs demonstrate the true variation in groundwater flow within the area, which would not have been known prior to the calibration process. Plumes from both nitrogen constituents were shorter than those predicted in the original uncalibrated model runs, with longer distanced predicted for ammonia plumes than for nitrate plumes.

Values encountered in literature report typical plume lengths ranging between 20 and 60 m (Ye, Sun, and Hallas, 2017), many of the plume lengths modeled for the Merritt Island community are well above this range. Simulated plume lengths are often a result of soil type, particularly soil hydraulic conductivity and porosity characteristics, which can vary between locations. Prior to calibrating and monitoring the concentrations in these types of communities, only septic tanks adjacent to waterbodies of concern were considered to have any pollution potential to the Lagoon. Static distances of 50-55-m were historically used in prioritizing septic tanks for upgrade or connection to sewer lines. High hydraulic conductance, particularly when coupled with high hydraulic head, might mean that septic systems further away from the Lagoon have a significant pollution potential and should not be dismissed.

Overall, plumes eventually decrease in concentration intensity with distance from the septic tank, with much higher plume concentrations observed for nitrate than for ammonia. Differences in ammonia and nitrate plume concentrations are a direct result of typical source concentrations (N_0) assigned as inputs from the drain field of each septic tank to the nutrient transport within the vadose zone (40 mg/L for nitrate and 10 mg/L for ammonia). The calibrated model predicted that a sizeable portion of the ammonia plumes are nitrified to nitrate over a short distance, particularly for the septic tanks located upgradient from the monitored locations. However, for the plumes located closest to the canals leading into the Banana River, ammonia has higher concentration intensities, likely due to the shallow water tables and high velocity, reducing the ability for nitrification processes to take place.

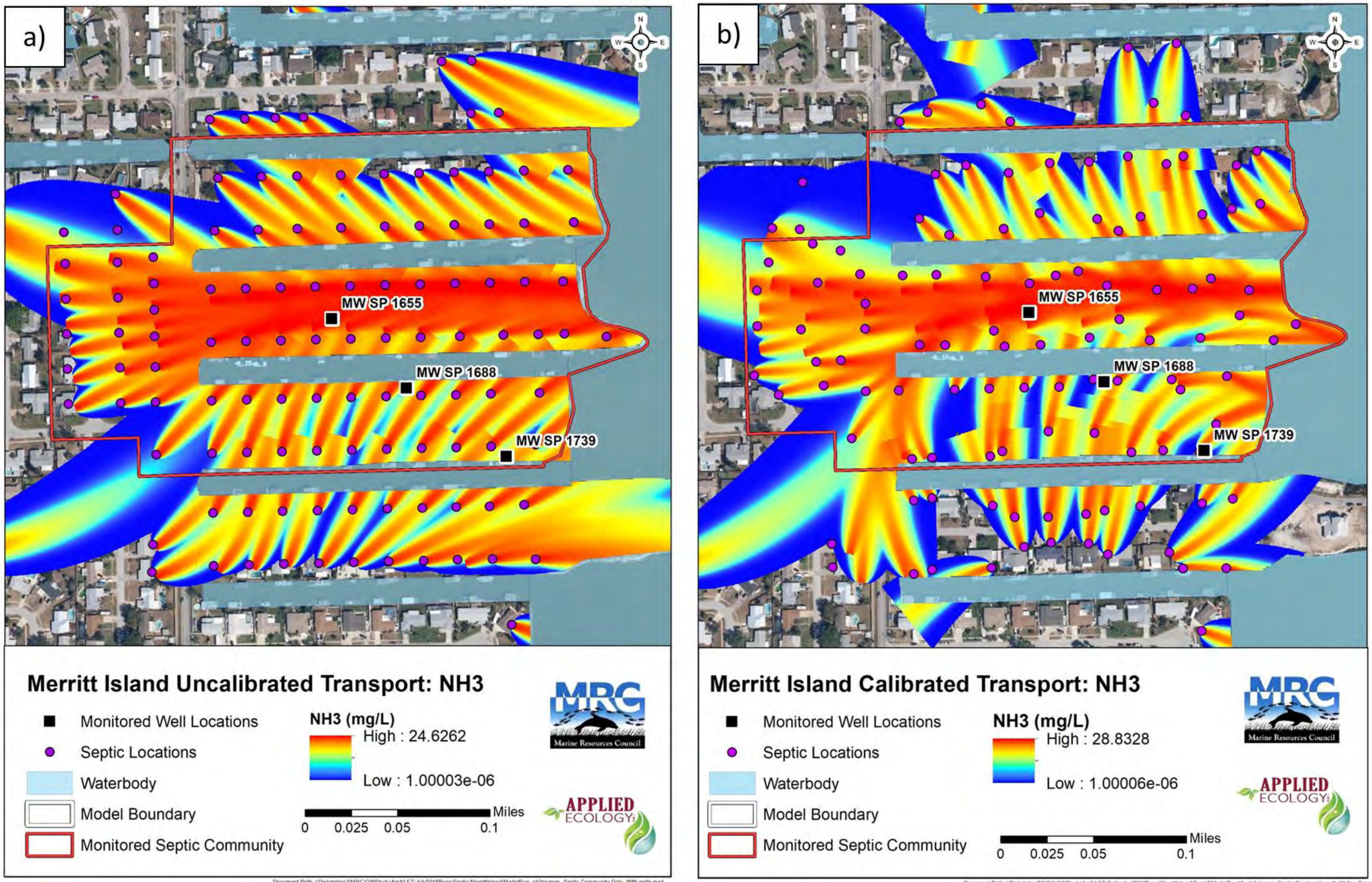


Figure 3. ArcNLET Model ammonia plume outputs in the monitored septic community of Merritt Island before and after calibration with measured concentration data. (a) Ammonia plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1.00003 x 10^{-6} mg/L in blue to 24.6 mg/L in red and (b) ammonia plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.00006 x 10^{-6} mg/L in blue to 28.8 mg/L in red.

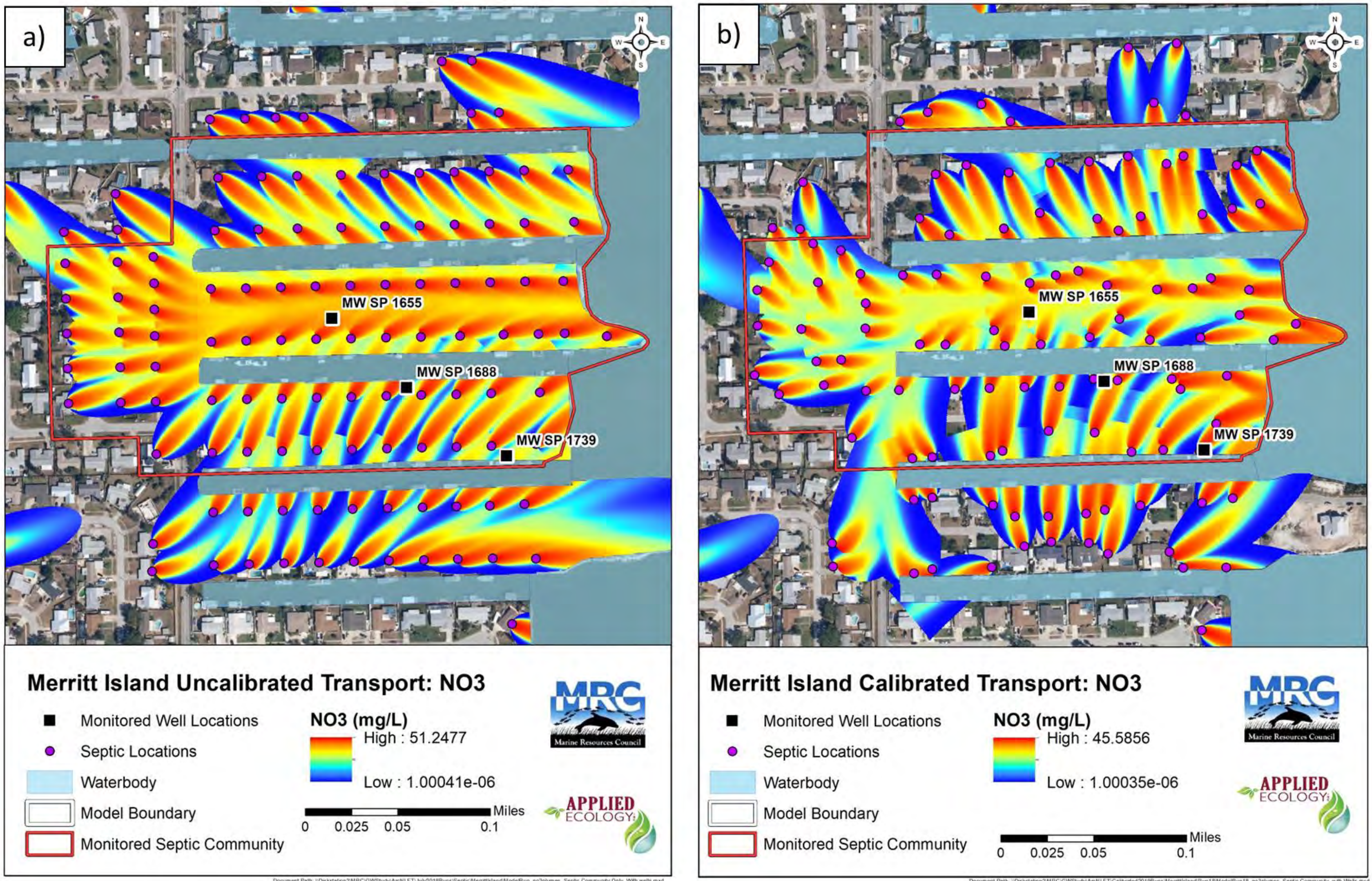


Figure 4. ArcNLET Model nitrate plume outputs in the monitored septic community of Merritt Island before and after calibration with measured concentration data. (a) nitrate plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1.00041 x 10^{-6} mg/L in blue to 51.2 mg/L in red and (b) nitrate plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.00035 x 10^{-6} mg/L in blue to 45.6 mg/L in red.

Comparison between average loading potentials from individual septic tanks and the entire monitored septic community of Merritt Island are presented in Table 2 and Table 3, respectively. Average predicted potential loads (lbs./year) per septic tank increased by at least 92% for all nitrogen constituents (92% for ammonia, 103% for nitrate, and 97% for the combined nitrogen loads) (Table 2). When these annual averages were applied to the 80 septic tanks with the Merritt Island study area, the overall nitrogen load increased by 131 lbs./year (Table 3). Shifts in the percent composition of nitrogen constituents were negligible (<2%), with nitrate making up a slightly larger proportion of the overall nitrogen load (54% for the uncalibrated, and 52% for the calibrated). The calibration process demonstrates that ArcNLET was previously underestimating the loading potentials of both nitrogen constituents from individual septic tanks as well as complete communities. In general, even after calibration, could be underestimating the average septic tank loading, often typical averaged to be closer to 19g nitrogen per septic system per day entering the groundwater (Zhu et al., 2016). Small changes in the denitrification coefficient used in the ArcNLET model have the greatest impact on the predict among of septic loading reaching the waterbodies. In previous modeling applications of ArcNLet, denitrification coefficient decreases from 0.011 d^{-1} to 0.001 d^{-1} result in reductyion ration changes from 78% to 36% (Sayemuzzaman and Ye, 2014). According to the Arcnlet model developers, more effort should spent to determeine appropriate value of the neiytification paraemetr for more accurate estimation of load reduction (Ye and Sun 2013). Further discussion of this systematic underestimation is included in the Conclusion section of this Memorandum.

Table 2. Annual average ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET run for each septic tank within the Merritt Island model boundaries. Differences in loads (lbs/year/tank) between model runs are also provided.

Parameter	**Uncalibrated Average Septic Tank Load**	**Calibrated Average Septic Tank Load**	**Septic Tank Load Difference**
Ammonia (lbs./year/tank)	0.91	1.75	0.84
Nitrate (lbs./year/tank)	0.77	1.57	0.80
Nitrogen (Ammonia + Nitrate) (lbs./year/tank)	1.68	3.32	1.64

Table 3. Annual ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET model run for the Merritt Island monitored septic community. Differences in loads (lbs/year/community) between model runs are also provided.

Parameter	**Uncalibrated Monitored Community Load**	**Calibrated Monitored Community Load**	**Monitored Community Load Difference**
Ammonia (lbs./year/community)	72.81	139.95	67.14
Nitrate (lbs./year/community)	61.75	125.46	63.71
Nitrogen (Ammonia + Nitrate) (lbs./year/community)	134.56	265.40	130.84

2.1.3.2 SUNTREE

Median concentration values of ammonia and nitrate for the three monitoring wells of the Suntree septic community during the 18-month study are presented in Table 4. Measured concentration values were generally highest at MW SP 6215 and lowest at MW SP 6155, with a large variety in median values for measured ammonia and nitrate-nitrite. The composition of nitrogen constituents varied between the wells, with MW SP 6215 nitrogen values dominated by nitrate-nitrite, MW SP 6398 predominately comprised of ammonia, and almost even contributions from both at MW SP 6155.

Table 4. Median concentration values (mg/L) of ammonia and nitrate-nitrite measured during the first 18 months of sampling at each monitoring well within the Suntree septic community. Median measured values of ammonia and nitrate-nitrite were adding together to create a combined "nitrogen" value (mg/L).

Parameter	**MW SP 6155**	**MW SP 6215**	**MW SP 6398**
Ammonia (mg/L)	0.455	0.035	7.150
Nitrate-Nitrite (mg/L)	0.490	9.950	0.060
Nitrogen (Ammonia + Nitrate-Nitrite) (mg/L)	0.945	9.985	7.210

Final calibration run estimated outputs at the monitoring well locations against the measured water quality at each of the locations are provided in Figure 5 for nitrate and Figure 6 for ammonia. Estimated values at the individual well locations were present within the 25th-75th percentile of the monitoring well data for nitrate for two wells, but overestimated for SP 6398. Ammonia calibration was more difficult, with underestimated values for SP 6398 and slightly overestimated values for the other two monitoring wells.

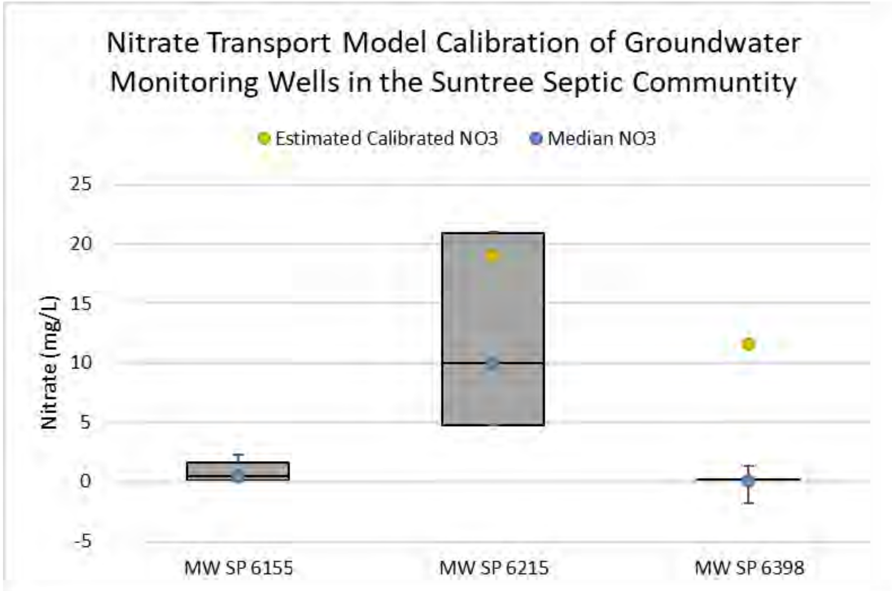


Figure 5. ArcNLET nitrate transport model calibration of groundwater monitoring well in the Suntree Septic Community.

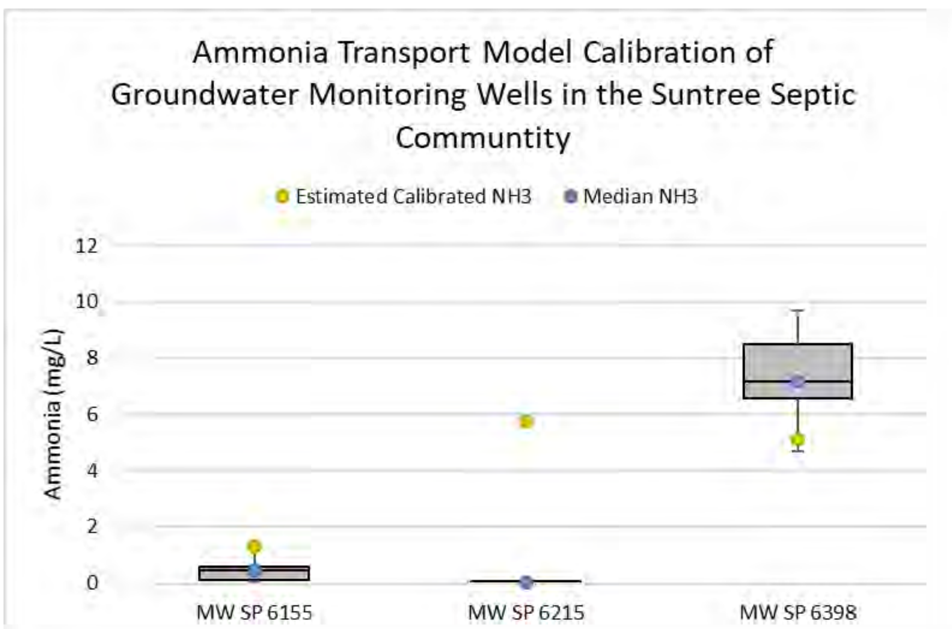


Figure 6. ArcNLET ammonia transport model calibration of groundwater monitoring well in the Merritt Island Septic Community.

The Suntree study area was reduced from 233 septic tanks in the uncalibrated model run to 186 septic tanks during the calibration process. The remainder of this discussion will focus on the 128 septic tanks located directly within the monitored septic community. Once calibrated, the ammonia and nitrate plumes had slightly higher magnitudes at shorter distances than the original uncalibrated model runs (Figure 7 and Figure 8). Increases in plume intensity of nitrate appear to be slightly higher than those of ammonia with calibration. Plume directions did not change significantly for this area.

Patterns of the fate of nitrogen in Suntree are similar to those observed in Merritt Island including: the anticipated decreased in plume intensity with increased distance from the septic tank, higher plume concentrations for nitrate than ammonia, relatively rapid nitrification of ammonia into, and higher ammonia concentration intensities for plumes of septic tanks located closest to the canals leading into the North Indian River Lagoon (likely due to shallower water tables and high velocity). Overall, it appears as though the model was previously underestimating the transport of these variables.

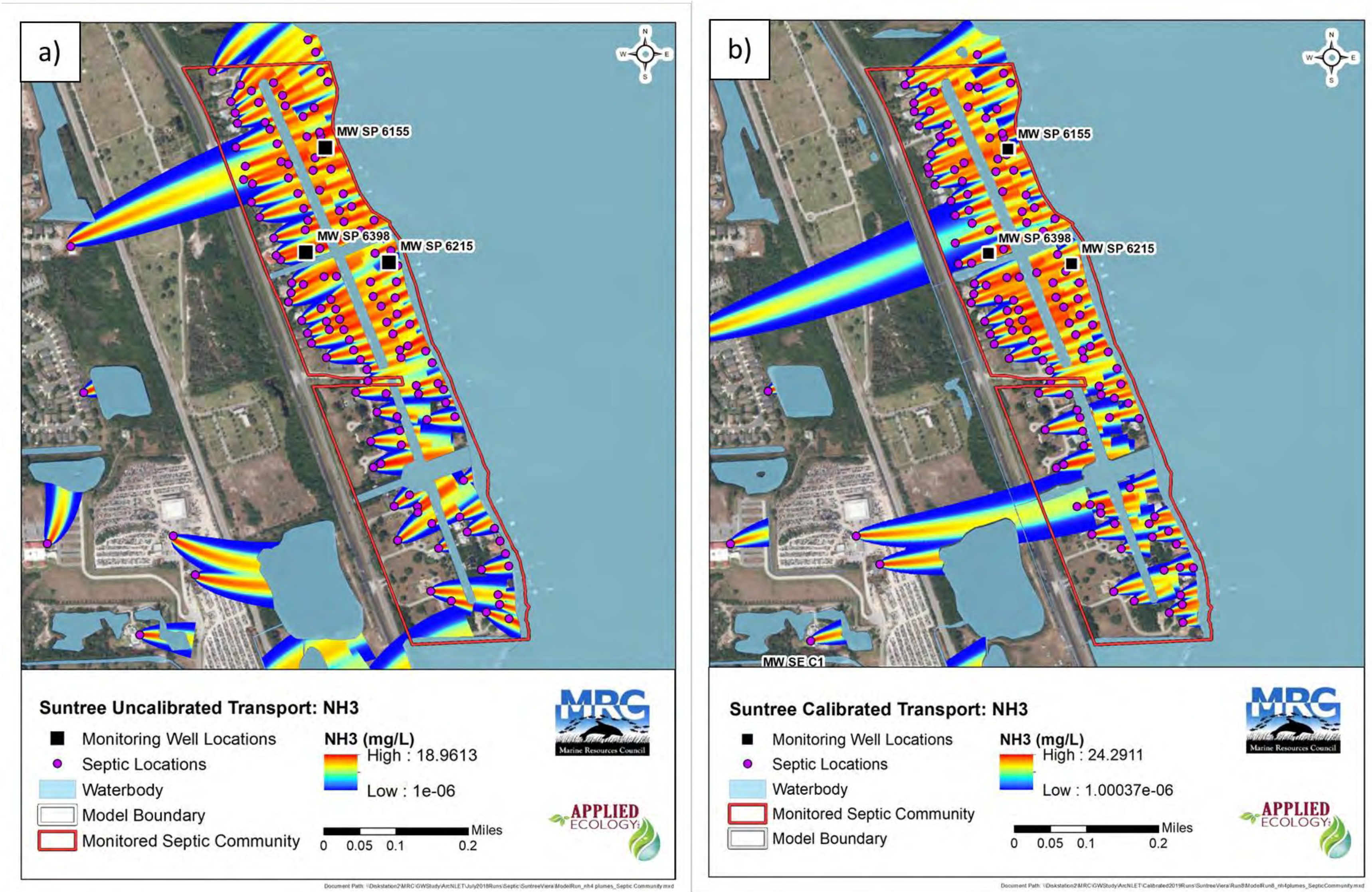


Figure 7. ArcNLET Model ammonia plume outputs in the monitored septic community of Suntree before and after calibration with measured concentration data. (a) Ammonia plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1 x 10^{-6} mg/L in blue to 18.96 mg/L in red and (b) ammonia plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.00037 x 10^{-6} mg/L in blue to 24.29 mg/L in red.

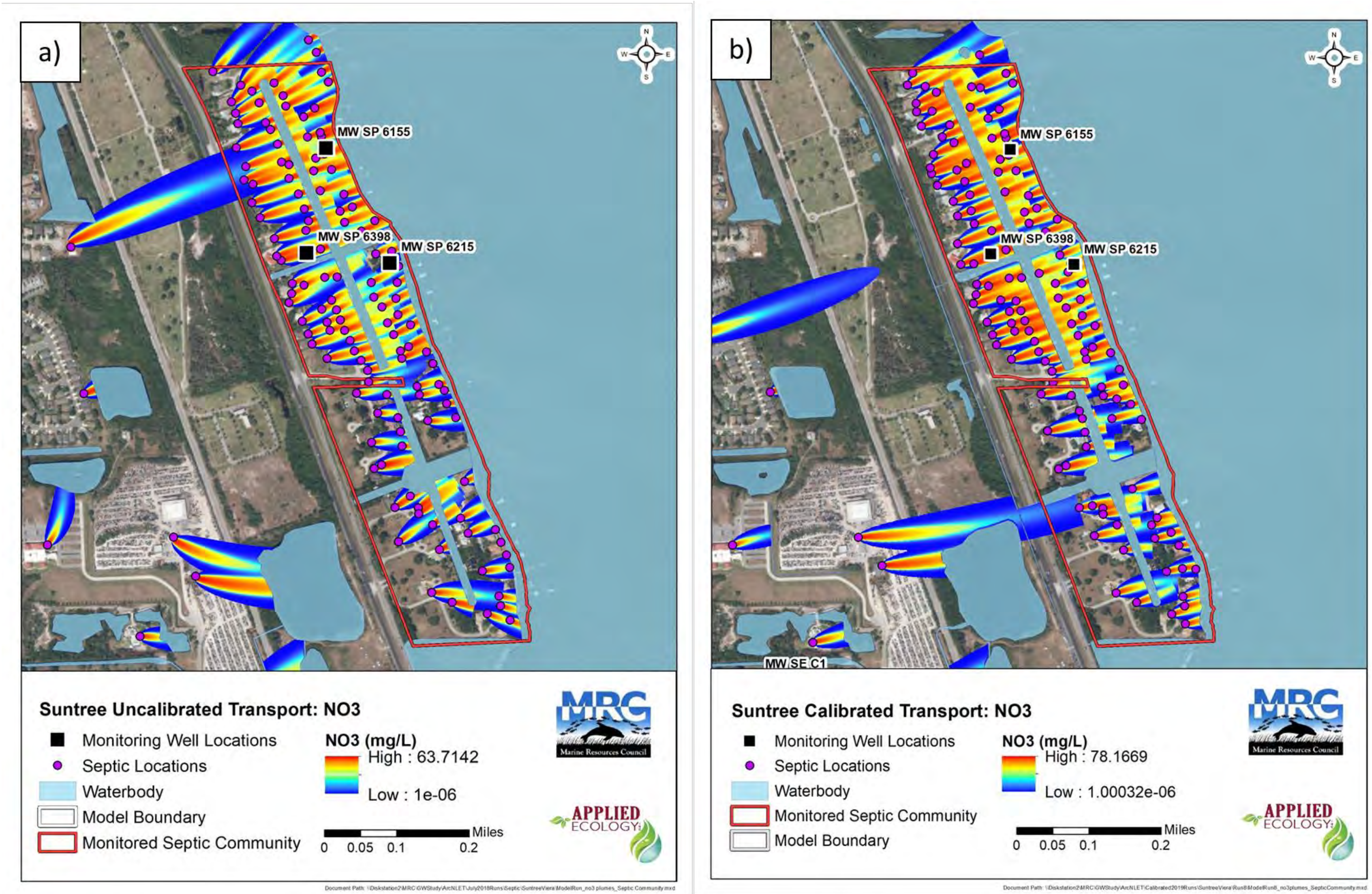


Figure 8. ArcNLET Model nitrate plume outputs in the monitored septic community of Suntree before and after calibration with measured concentration data. (a) nitrate plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1 x 10^{-6} mg/L in blue to 63.71 mg/L in red and (b) nitrate plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.00032 x 10^{-6} mg/L in blue to 78.17 mg/L in red.

Comparison between average loading potentials from individual septic tanks and the entire monitored septic community of Suntree are presented in Table 5 and Table 6, respectively. Average predicted potential loads (lbs./year) per septic tank increased for all nitrogen constituents (26% for ammonia, 22% for nitrate, and 24% for nitrogen), although not as dramatically as Merritt Island (Table 5). When these annual averages were applied to the 128 septic tanks with the Suntree study area, the overall nitrogen load increased by 108 lbs./year (

Table 6). Shifts in the percent composition of nitrogen constituents were negligible (~1%), with ammonia slightly providing more contribution to the nitrogen load (51% for the uncalibrated, and 52% for the calibrated). As with Merritt Island, the calibration process demonstrates that ArcNLET was previously underestimating the loading potentials of both nitrogen constituents from individual septic tanks as well as entire communities.

Table 5. Annual average ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET run for each septic tank within the Suntree model boundaries. Differences in loads (lbs/year/tank) between model runs are also provided.

Parameter	Uncalibrated Average Septic Tank Load	Calibrated Average Septic Tank Load	Septic Tank Load Difference
Ammonia (lbs./year/tank)	1.78	2.25	0.47
Nitrate (lbs./year/tank)	1.69	2.07	0.37
Nitrogen (Ammonia + Nitrate) (lbs./year/tank)	3.47	4.32	0.84

Table 6. Annual ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET model run for the Suntree monitored septic community. Differences in loads (lbs/year/community) between model runs are also provided.

Parameter	Uncalibrated Monitored Community Load	Calibrated Monitored Community Load	Monitored Community Load Difference
Ammonia (lbs./year/community)	227.67	287.94	60.27
Nitrate (lbs./year/community)	216.90	264.48	47.58
Nitrogen (Ammonia + Nitrate) (lbs./year/community)	444.57	552.42	107.85

2.1.3.3 MELBOURNE BEACH

Median concentration values of ammonia and nitrate for the three monitoring wells of the Melbourne Beach septic community during the 18-month study are presented in Table 7. Measured concentration values were generally highest at MW SP 250 and lowest at MW SP 275, with nitrate being the predominant nitrogen constituent for all wells.

Table 7. Median concentration values (mg/L) of ammonia and nitrate-nitrite measured during the first 18 months of sampling at each monitoring well within the Melbourne Beach septic community. Median measured values of ammonia and nitrate-nitrite were adding together to create a combined "nitrogen" value (mg/L).

Parameter	MW SP 250	MW SP 270	MW SP 275
Ammonia (mg/L)	0.2100	0.0385	0.0820
Nitrate-Nitrite (mg/L)	4.9500	0.9000	0.4750
Nitrogen (Ammonia + Nitrate-Nitrite) (mg/L)	5.1600	0.9385	0.5570

Final calibration run estimated outputs at the monitoring well locations against the measured water quality at each of the locations are provided in Figure 9 for nitrate and Figure 10 for ammonia. In the Melbourne Beach study area, calibration was easier, with only one well (SP 275) being overestimated for nitrate. Ammonia calibration efforts successfully estimated medians of the measured data for each of the three monitoring wells.

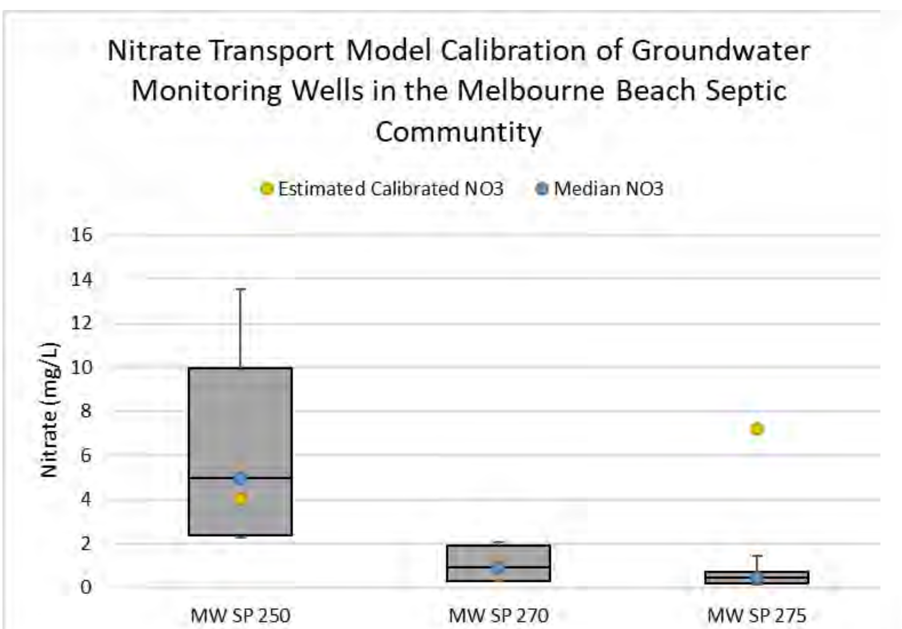


Figure 9. ArcNLET nitrate transport model calibration of groundwater monitoring well in the Melbourne Beach Septic Community.

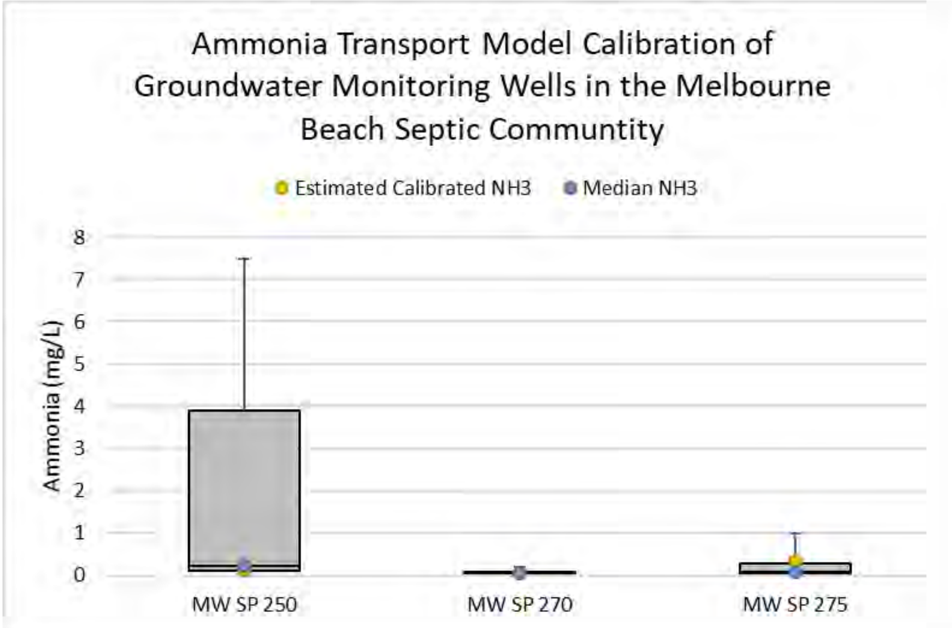


Figure 10. ArcNLET ammonia transport model calibration of groundwater monitoring well in the Melbourne Beach Septic Community.

The Melbourne Beach study area was reduced from 213 septic tanks in the uncalibrated model run to 121 septic tanks during the calibration process. The remainder of this discussion will focus on the 79 septic tanks located directly in the monitored septic community. Once calibrated, the ammonia and nitrate plumes had a lower magnitude with wider diameters than the original uncalibrated model runs (Figure 11). This was anticipated since measured ammonia concentrations were typically much lower than measured nitrate concentrations in Melbourne Beach, and the calibration process allowed for a correction of the overestimation of this parameter. While the changes in distance and width of plumes experienced by ammonia during calibration were similar to those for the nitrate plumes, changes in magnitudes between calibrated and uncalibrated runs were significantly different for ammonia versus nitrate plumes (Figure 12). Plume direction for ammonia and nitrate were unidirectional prior to model calibration. The calibration process, which included relocating known septic/drain field placements, demonstrated the impact of input data in both direction, velocity, and magnitude of the predicted nutrient plumes.

The calibrated model clearly indicated the rapid decrease in plume intensity with increases in distance from the septic tank. As previously described for most model areas, nitrate demonstrated higher concentrations, with predictions that most ammonia is rapidly nitrified to nitrate in this environment. Concentration intensities for ammonia plumes were higher at septic tanks located closest to the Central IRL, likely due to shallower water tables and high velocity. Overall, it appears as though the model previously overestimated ammonia transport and underestimated nitrate transport in the Melbourne Beach area.

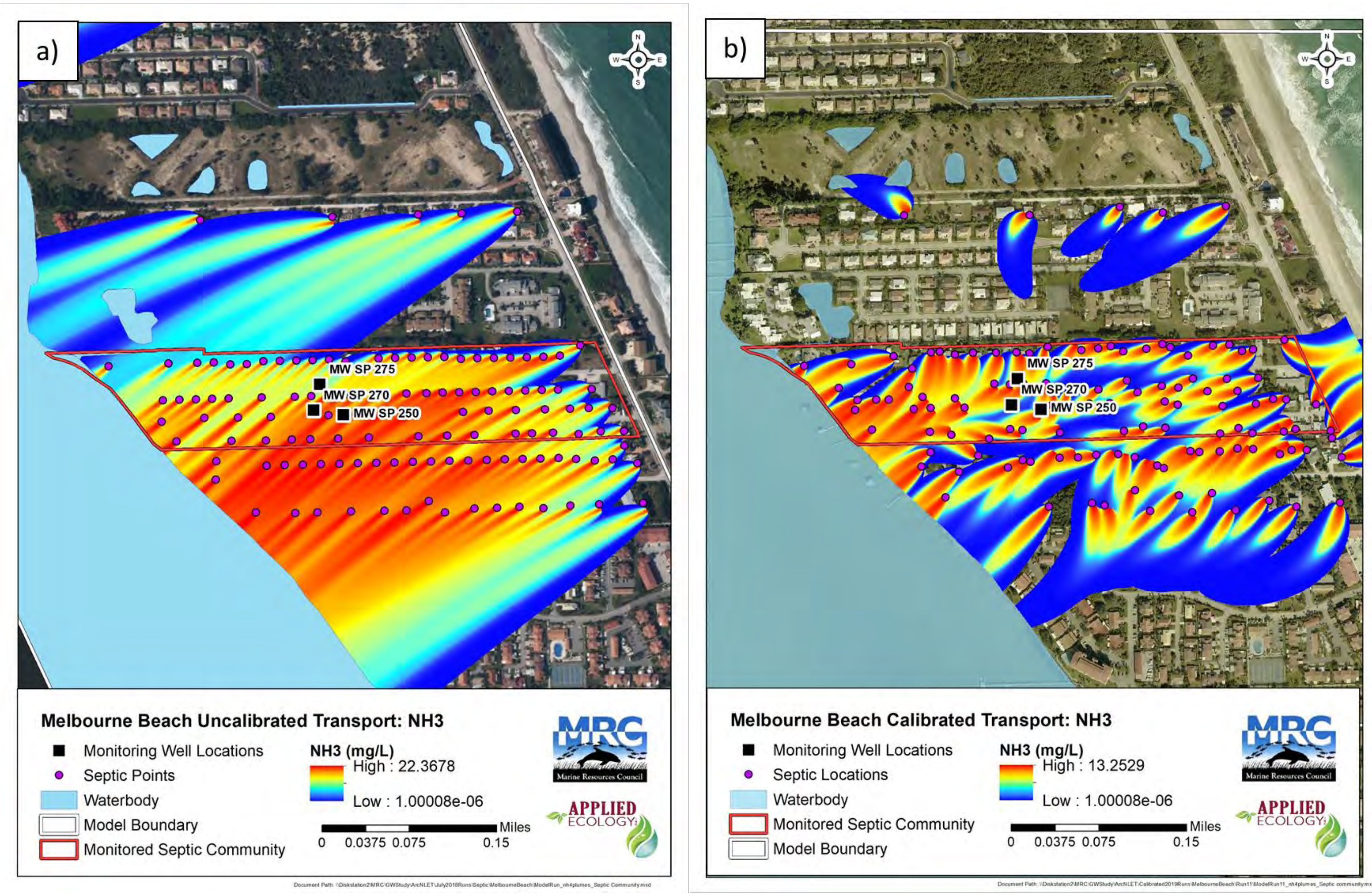


Figure 11. ArcNLET Model ammonia plume outputs in the monitored septic community of Melbourne Beach before and after calibration with measured concentration data. (a) Ammonia plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1.00008 x 10^{-6} mg/L in blue to 22.37 mg/L in red and (b) ammonia plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.00008 x 10^{-6} mg/L in blue to 13.25 mg/L in red.

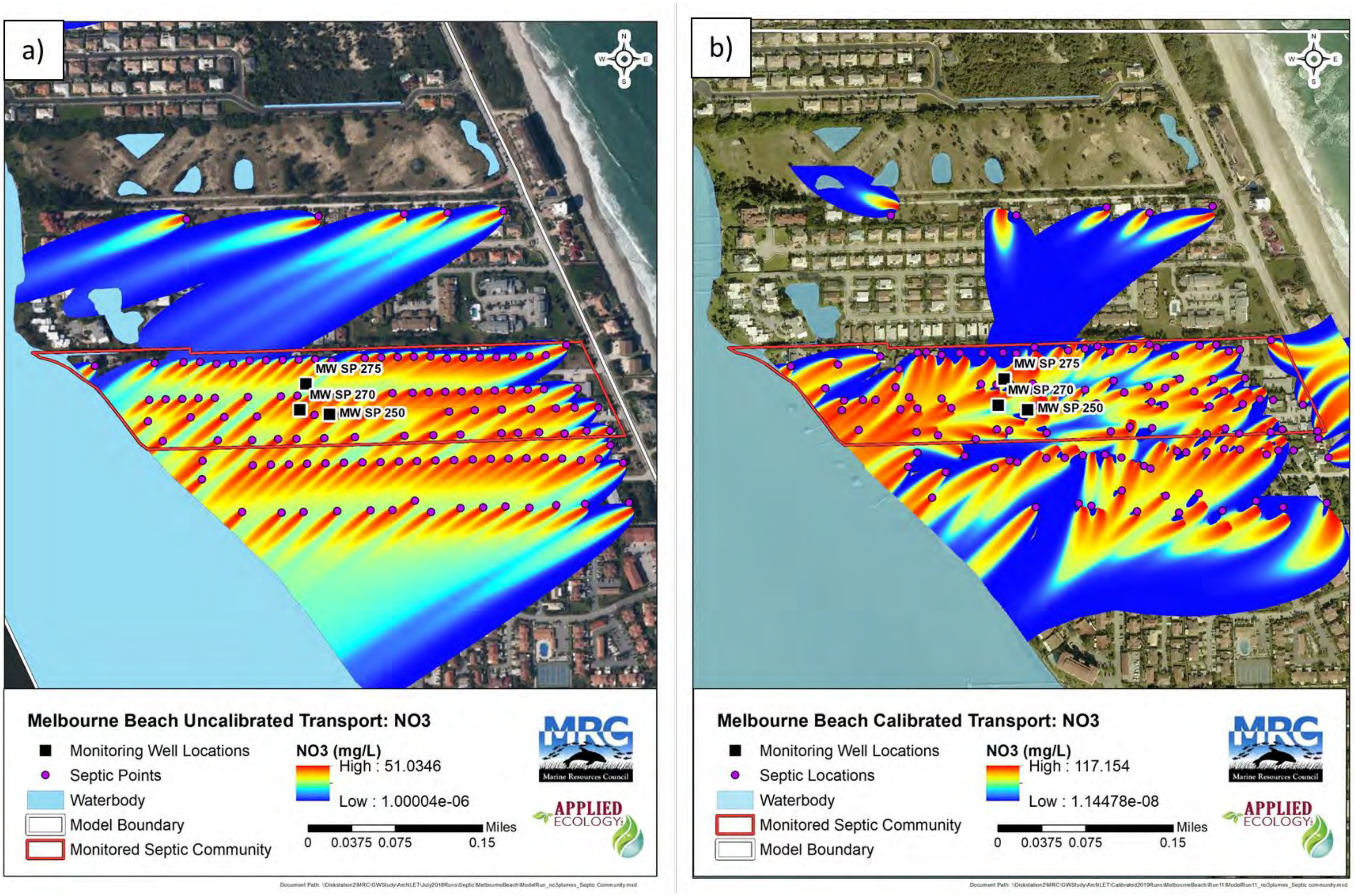


Figure 12. ArcNLET Model nitrate plume outputs in the monitored septic community of Melbourne Beach before and after calibration with measured concentration data. (a) nitrate plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1.0004 x 10^{-6} mg/L in blue to 51.03 mg/L in red and (b) nitrate plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.14478 x 10^{-8} mg/L in blue to 117.15 mg/L in red.

Comparison between average loading potentials from individual septic tanks and the entire monitored septic community of Melbourne Beach are presented in Table 8 and Table 9, respectively. Average predicted potential loads (lbs./year) per septic tank increased for nitrate and nitrogen (81% and 15%, respectively), and decreased for ammonia (71%) (Table 8). A dramatic shift between ammonia and nitrate (33%) is a result incorporating measured concentration values during model calibration, with the percent composition of ammonia decreasing from 44% to 11% during the calibration process and nitrate increasing from 56% to 89%. When these annual averages were applied to the 79 septic tanks with the Melbourne Beach study area, the overall nitrogen load increased by 43 lbs./year (Table 9). As suspected from the output plumes, ArcNLET was previously underestimating loading potentials of nitrate and overestimating those of ammonia from individual septic tanks as well as entire communities.

Table 8. Annual average ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET run for each septic tank within the Melbourne Beach model boundaries. Differences in loads (lbs/year/tank) between model runs are also provided.

Parameter	**Uncalibrated Average Septic Tank Load**	**Calibrated Average Septic Tank Load**	**Septic Tank Load Difference**
Ammonia (lbs./year/tank)	1.64	0.47	-1.17
Nitrate (lbs./year/tank)	2.12	3.84	1.72
Nitrogen (Ammonia + Nitrate) (lbs./year/tank)	3.76	4.32	0.55

Table 9. Annual ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET model run for the Melbourne Beach monitored septic community. Differences in loads (lbs/year/community) between model runs are also provided.

Parameter	**Uncalibrated Monitored Community Load**	**Calibrated Monitored Community Load**	**Monitored Community Load Difference**
Ammonia (lbs./year/community)	129.76	37.25	-92.51
Nitrate (lbs./year/community)	167.67	303.65	135.98
Nitrogen (Ammonia + Nitrate) (lbs./year/community)	297.42	340.90	43.47

2.1.3.4 TURKEY CREEK

Median concentration values of ammonia and nitrate for the three monitoring wells of the Turkey Creek septic community during the 18-month study are presented in Table 10. Measured concentration values were generally highest at MW SP 1127 and lowest at MW SP 981. Composition of nitrogen constituents varied between the wells, with the nitrogen of MW SP 1127 dominated by nitrate-nitrite, and MW SP 981 and MW SP 1099 predominately comprised of ammonia.

Table 10. Median concentration values (mg/L) of ammonia and nitrate-nitrite measured during the first 18 months of sampling at each monitoring well within the Turkey Creek septic community. Median measured values of ammonia and nitrate-nitrite were adding together to create a combined "nitrogen" value (mg/L).

Parameter	**MW SP 981**	**MW SP 1099**	**MW SP 1127**
Ammonia (mg/L)	0.930	5.200	0.035
Nitrate-Nitrite (mg/L)	0.025	0.030	10.800
Nitrogen (Ammonia + Nitrate-Nitrite) (mg/L)	0.955	5.200	10.835

Final calibration run estimated outputs at the monitoring well locations against the measured water quality at each of the locations are provided in Figure 13 for nitrate-nitrite and Figure 14 for ammonia. Data from all three monitoring wells were used to successfully calibrate the input source concentrations of the ArcNLET model for the Turkey Creek study area. Estimated nitrate values for the monitoring wells were closely within or within the measured data distributions for all three locations, while calibrated model estimates slightly overestimated ammonia at one of the monitoring wells (SP 1127).

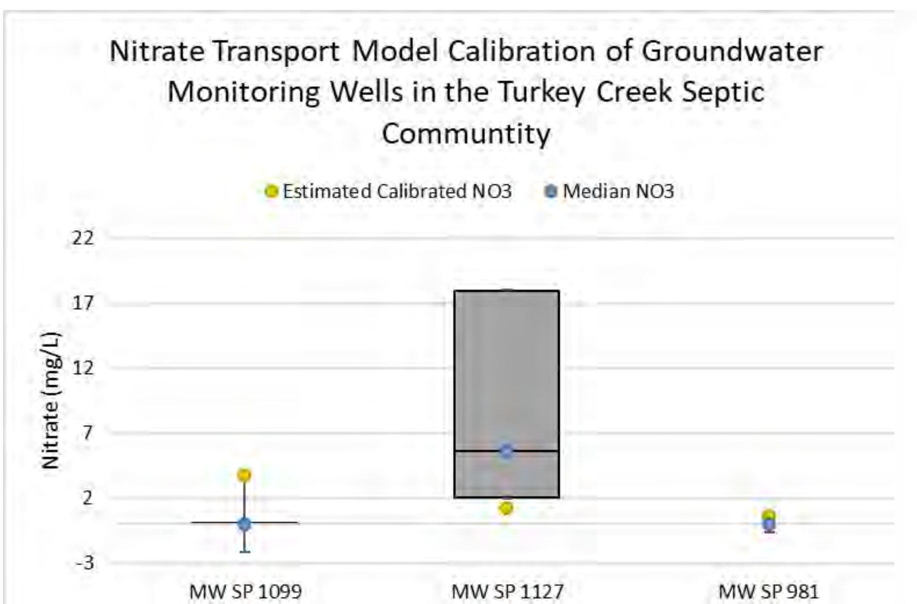


Figure 13. ArcNLET nitrate transport model calibration of groundwater monitoring well in the Turkey Creek Septic Community.

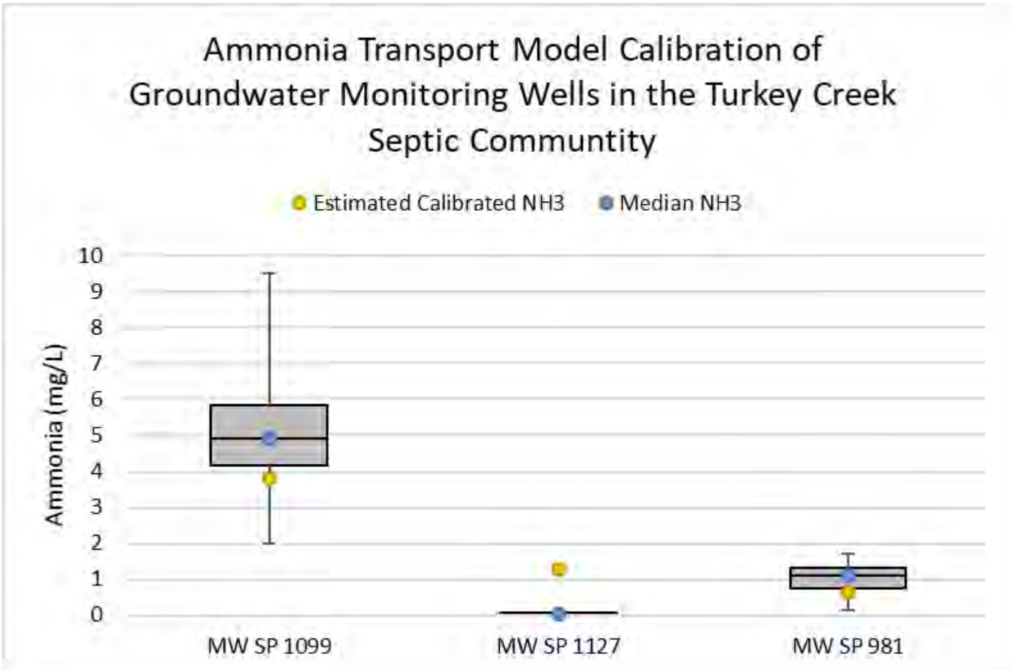


Figure 14. ArcNLET ammonia transport model calibration of groundwater monitoring well in the Turkey Creek Septic Community.

The Turkey Creek study area was reduced from 172 septic tanks in the uncalibrated model run to 85 septic tanks during the calibration process. The remainder of this discussion focuses on the 63 septic tanks located directly in the monitored septic community. Once calibrated, the ammonia and nitrate plumes had a higher magnitude with wider diameters at greater distances than the original uncalibrated model runs (Figure 15 and Figure 16). Plume directions for both ammonia and nitrate remain consistent between the model runs. Eventually, plumes decrease in pollutant concentration intensity with distance from the septic tank, with much higher plume concentrations observed for nitrate than ammonia. The calibrated model, unlike the uncalibrated version, predicted that a large portion of the ammonia plumes to be nitrified to nitrate, particularly for the septic tanks located upgradient from the monitored location. However, for the plumes located closest to the Turkey Creek, the ammonia plumes have higher concentration intensities, likely due to the shallow water tables and high velocity, reducing the ability for nitrification processes to take place. Due to the larger, more concentrated plumes, most of the septic tanks in these communities adjacent to Turkey Creek are predicted to deliver nitrogen loads to the Creek. Overall, it appears as though the model previously overestimated ammonia transport and underestimated nitrate transport in the Turkey Creek area.

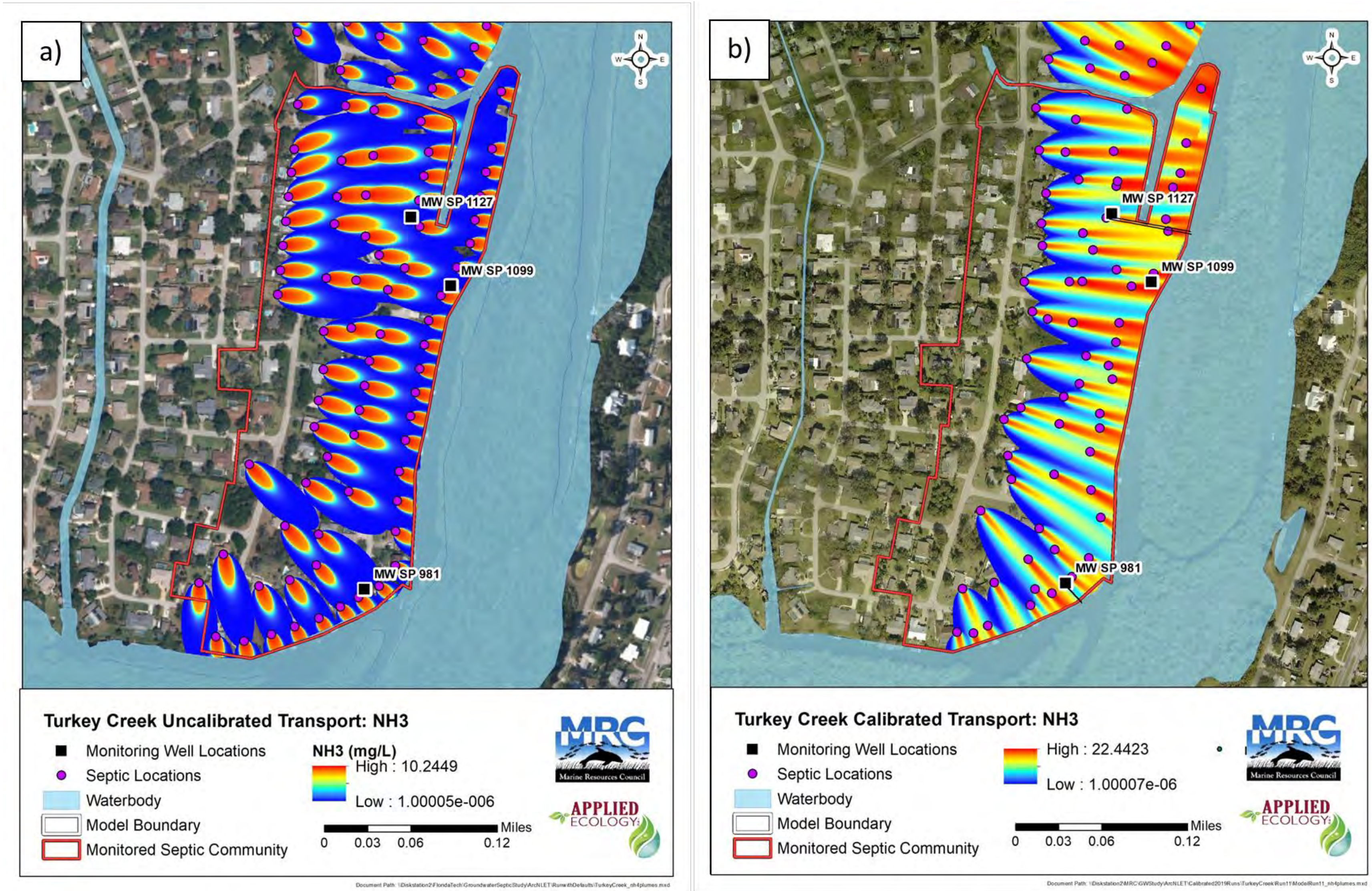


Figure 15. ArcNLET Model ammonia plume outputs in the monitored septic community of Turkey Creek before and after calibration with measured concentration data. (a) Ammonia plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1.00005 x 10^{-6} mg/L in blue to 10.24 mg/L in red and (b) ammonia plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.00007 x 10^{-6} mg/L in blue to 22.44 mg/L in red.

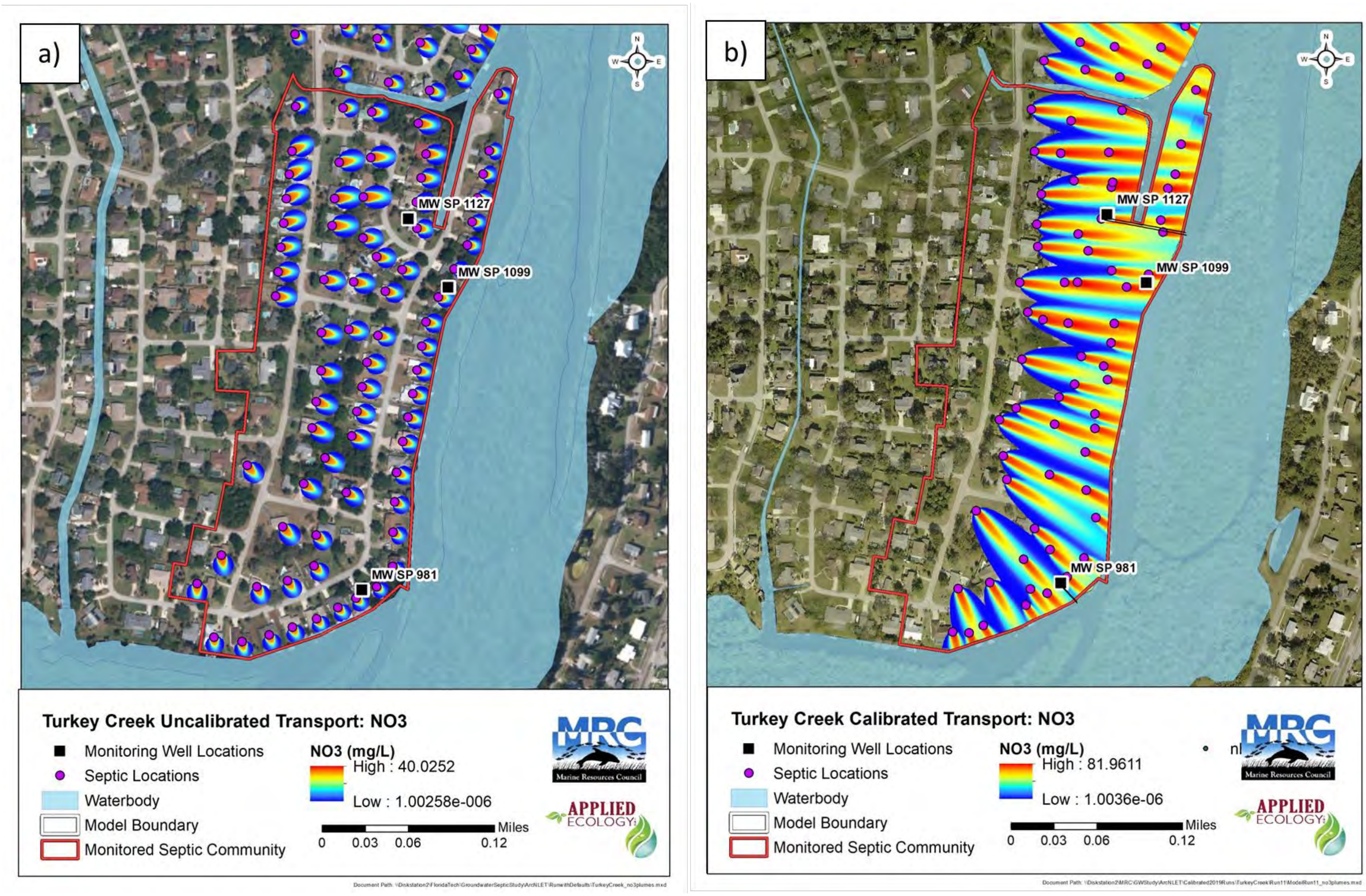


Figure 16. ArcNLET Model nitrate plume outputs in the monitored septic community of Turkey Creek before and after calibration with measured concentration data. (a) nitrate plume direction and intensity from the uncalibrated model run is provided with concentrations ranging from 1.0028 x 10^{-6} mg/L in blue to 40.03 mg/L in red and (b) nitrate plume direction and intensity from the calibrated model run is provided with concentrations ranging from 1.0036 x 10^{-8} mg/L in blue to 81.96 mg/L in red.

Comparison between average loading potentials from individual septic tanks and the entire monitored septic community of Turkey Creek are presented in Table 11 and Table 12, respectively. Average predicted potential loads (lbs./year) per septic tank dramatically increased for all nitrogen constituents (412% for ammonia, 5,040% for nitrate, and 837% for nitrogen) (Table 11). A dramatic shift between ammonia and nitrate (41%) is a result of the calibrated model incorporating measured concentration value, with the percent composition of ammonia decreasing from 91% to 50% during the calibration process and nitrate increasing from 9% to 50%. When these annual averages were applied to the 63 septic tanks within the Melbourne Beach study area, the overall nitrogen load increased by 299 lbs./year (Table 12). As suspected from the output plumes, ArcNLET was previously underestimating loading potentials of nitrate and overestimating those of ammonia from individual septic tanks as well as entire communities. Calibrated per septic tank contributions are a little higher in Turkey Creek than previous study areas (6.6 g/tank/day).

Table 11. Annual average ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET run for each septic tank within the Turkey Creek model boundaries. Differences in loads (lbs/year/tank) between model runs are also provided.

Parameter	Uncalibrated Average Septic Tank Load	Calibrated Average Septic Tank Load	Septic Tank Load Difference
Ammonia (lbs./year/tank)	0.51	2.63	2.12
Nitrate (lbs./year/tank)	0.05	2.68	2.62
Nitrogen (Ammonia + Nitrate) (lbs./year/tank)	0.57	5.31	4.74

Table 12. Annual ammonia, nitrate, and nitrogen (ammonia + nitrate) loads (lbs./year) predicted by the uncalibrated and calibrated ArcNLET model run for the Turkey Creek monitored septic community. Differences in loads (lbs/year/community) between model runs are also provided.

Parameter	Uncalibrated Monitored Community Load	Calibrated Monitored Community Load	Monitored Community Load Difference
Ammonia (lbs./year/community)	32.41	165.98	133.56
Nitrate (lbs./year/community)	3.28	168.55	165.27
Nitrogen (Ammonia + Nitrate) (lbs./year/community)	35.69	334.53	298.83

2.2 UNCERTAINTY MODELS

2.2.1 BACKGROUND

The ArcNLET model version 2.0 is equipped with a function to estimate uncertainty of a subset of key input parameters, using Monte Carlo (MC) simulations which allows for the quantification of uncertainty for the input variables. However, it should be noted that the MC simulation does not support the modeling of ammonia, since the module was built for an older version of ArcNLET, prior to the development of the ammonia transport version. Thus, results within this section focus on predicting nitrate exclusively.

The ability to measure uncertainty is critical, since it provides a quantitative indication of the quality of measurement results, without which these could not be compared between themselves, with specified reference values or to a standard. The Monte Carlo method is a well-accepted method of estimating uncertainty (Papadopoulos and Yeung, 2001), since it is relatively simple and can be used for complex systems, such as the one system used to predict nutrient transport in the vadose zone. MC is an especially useful technique that handles non-normal distributions, complex algorithms, and correlations between input factors.

The MC simulation function built into the ArcNLET is used to quantify uncertainty as the nitrate loads estimated by ArcNLET are inherently uncertain before they (and/or their statistics) are used for environmental management and planning. For a process like ArcNLET there are four key sources of uncertainty: model parameters (e.g. hydraulic conductivity), model structure (e.g. flow and transport models), model input data or scenario uncertainty (e.g. water use per household), and measurements of model parameters (e.g. hydraulic head). The MC simulation addresses the first source of uncertainty, the parametric uncertainty, which includes the following model parameters: smoothing factor, longitudinal dispersivity (dispL), horizontal transverse dispersivity (dispTH), first-order denitrification coefficient (k), soil hydraulic conductivity, soil porosity, and source nitrate concentration. In the MC simulation, the first four parameters are randomly homogeneous, and the latter three are randomly heterogeneous. Recommended data distributions for each of the seven variables vary and include uniform (smoothing factor), normal (longitudinal dispersivity and source concentration) , lognormal (decay coefficient), and triangular (hydraulic conductivity and porosity) distributions.

Running the MC simulation of ArcNLET allows users to understand and determine the random and deterministic parameters before running ArcNLET, and it is designed to be flexible as the user can consider single or multiple random parameters. For simplicity purposes and to start exploring the uncertainty of some key parameters, single parameters were evaluated, and results reported in the following sections.

2.2.2 MODEL INPUTS AND METHODOLOGY

The MC simulation was executed separately for two selected septic communities within the study region, one representative of mainland (Suntree) and another barrier island (e.g.: Melbourne Beach) conditions. Typically, mainland areas are dominated by soils with slightly higher organic levels and lower hydraulic conductivity, whereas barrier soils have relatively low organic contents and higher hydraulic conductance values.

The following MC simulation parameters were modeled using uncalibrated runs for the two selected areas: smoothing factor, hydraulic conductivity, porosity, and source nitrate concentration. While the MC simulation allows the user to randomly vary more than one parameter per model execution, only a single parameter was varied in order to better understand the effect of altering that parameter (Table 13). To simplify the interpretation of the results, for every test case, the following parameters were kept static with the following input values: longitudinal dispersity ("dispL") with a value of 2.113, horizontal dispersity ("dispTH") was computed by the MC simulation to be 10% of the dispL value, and denitrification coefficient ("k") with a value of 0.008.

Table 13. Minimum, maximum, and mode values for each input parameter of the ArcNLET MC simulation.

Parameter	Minimum Value	Maximum Value	Mode Value
Smoothing Factor	10	200	20
Hydraulic conductivity (µm/s)	2	20	10
Porosity	0.2	0.45	0.35
Source nitrate concentration (mg/d)	10	40	80

For the MC simulation of each model area, the layer containing the monitoring well locations across the entire study area was reduced to only include monitoring wells within the Suntree and Melbourne Beach boundaries (Figure 17). Additional simulated monitoring well locations were also added to allow a greater number (N) of estimated points to be used when evaluating the impact of changing the selected input variables on the predicted nitrate plume concentrations within the modeled areas. For the Melbourne Beach study area, 52 "simulated" wells were added to the three physical well locations within the septic community. In the Suntree study area, 38 simulated wells were added to the three physical well locations within the septic community. The simulated well locations were strategically place to represent each existing soil type within the modeled areas and distributed evenly throughout the area most likely to be impacted by septic tank plumes (Figure 17). A total of 100 realizations using randomly generated numbers within the data distribution of each parameters, as described in Table 13, were included as part of each MC Simulation. Results synthesize the results of all 100 simulation per parameter and study area below.

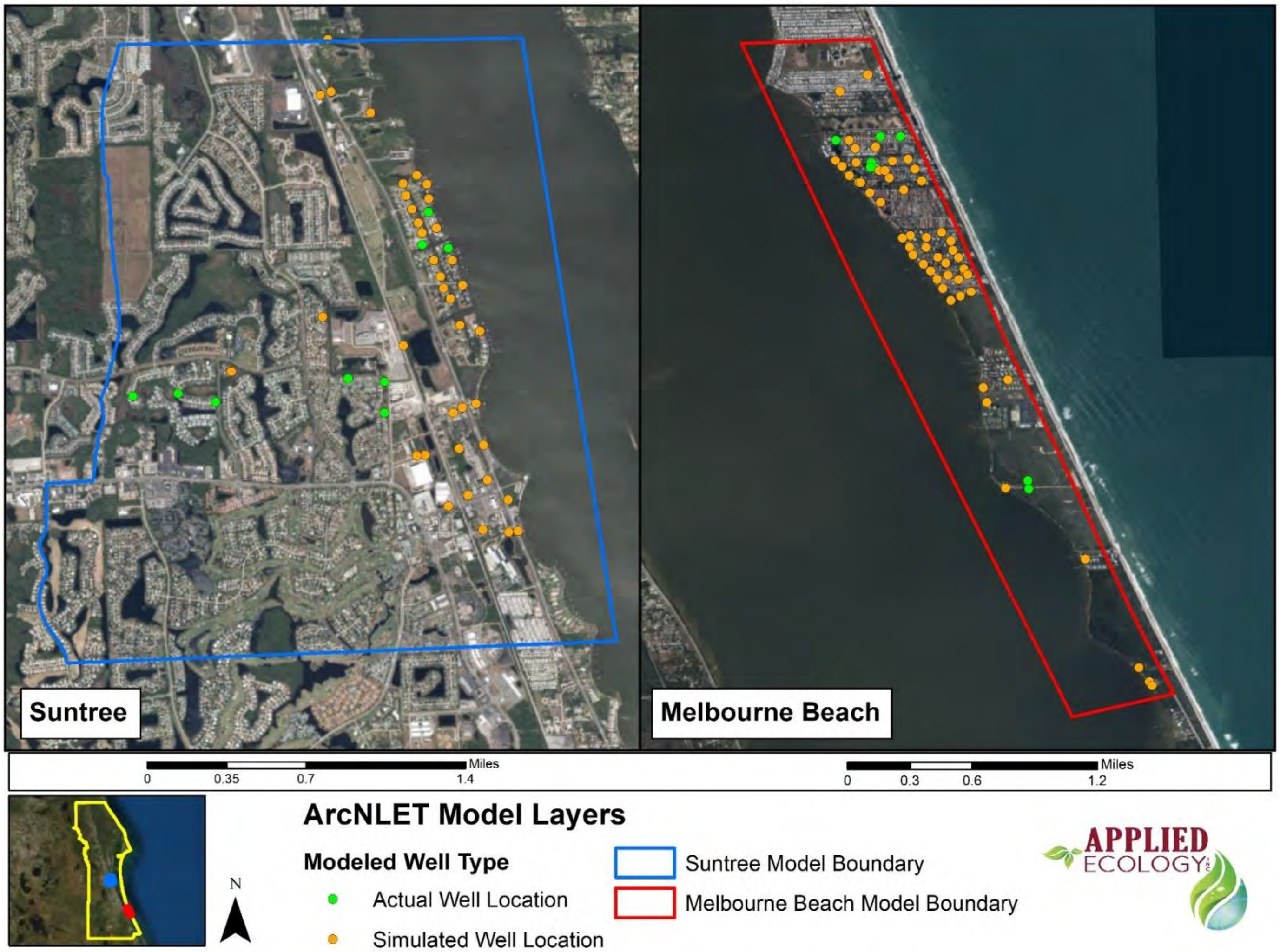


Figure 17. ArcNLET model boundaries, actual well locations, and simulated well locations for Suntree and Melbourne Beach.

The model supports at most 100 random features at a time per input layer, which include soils, waterbodies, and septic tanks. If MC simulation is attempted with over 100 features, the model will fail with a runtime exception. While modeling soil hydraulic conductivity or porosity for Melbourne Beach posed no issues, as there are only seven soil polygons in that study area, there are 217 soils polygons in the Suntree study area. The soils feature class from the Suntree model area was edited to reduce the polygon count to 100 through the removal of less important polygons (such as those near the western boundary or outside of any potential flow paths) to reduce the possibility of model failure. For both areas, selected septic tanks were modeled within the region to ensure the MC Simulation was successfully executed. As a result, total model outputs described below are not equivalent and should not be compared to those described in Section 2.1.3 of the report. Outputs should be evaluated for comparative analysis of the magnitude of change as a direct result of input changes, rather than absolute nitrate loading predictions.

During the MC Simulation of the ArcNLET model, just as during the traditional model runs, waterbody and soils input layers become rasterized during the execution process. Occasionally, rasterization leaves gaps between raster cells of the waterbody and soils. Gaps in raster outputs are a direct result of differenced in raster cell size and the angle of polygon segments and can result in the failure of the MC simulation to produce a nitrate plume. Locations that failed to produce plumes in various model execution attempts were fixed by slightly altering the soils layer in the GIS environment to ensure it overlaps the waterbody layer in problem locations. The waterbody layer was only modified when needed to ensure it was representative of the actual surface water.

The modeling of the smoothing factor, hydraulic conductivity, and porosity was successfully executed. However, the attempt to model source concentration failed due to unforeseen issues within the MC simulation software itself. Several attempts to simplify the model runs were attempted to evaluate the critical source concentration, including reducing the total number of input septic tanks in both areas. Even though the proper input files were generated containing random source concentrations, during the actual execution of the model, the model failed to use the randomly generated source concentrations that it produced. Instead, the simulation used the typical static source concentration value of 40 mg/day erroneously retrieved from the "N0_conc" (nitrate input concentration) field of the septic point layer. Since no model support, especially for the MC Simulation function is available, no outputs for this source concentration parameter are described in the results section below.

2.2.3 RESULTS

2.2.3.1 HYDRAULIC CONDUCTIVITY

Hydraulic conductivity is a soil-derived variable that is critical for nutrient transport estimates in groundwater (Rios *et al.* 2013; Wang, Ye, Rios, and Lee, 2012; Zhu *et al.*, 2016). Saturated hydraulic conductivity (Ksat, µm/s) refers to the ease with which pores in a saturated soil transmit water, or a measure of soil permeability (Amoozegar and Warrick, 1986). In order to treat wastewater effluent properly, soil in the absorption field must be able to move water away from the trenches fast enough to prevent the water from rising to the surface, yet slow enough to provide ample treatment of the effluent by the soil. Extremely low hydraulic conductivity increases the likelihood of septic water to rise to the surface (drain field failure), while high hydraulic conductivity increases the likelihood of groundwater pollution.

Hydraulic conductivity values can be converted to loading rates (loading rate = 0.22 x $(\text{Ksat})^{0.23}$) (Taylor, Yahner, and Jones*,* 1997) and compared to standards for onsite sewage treatment and disposal systems in Florida (State of Florida Department of Health, 2013). Loading rates are dependent on soil texture (coarse sand, loamy sand, clay, *etc.*) and percolation rates (in minutes per inch). The conceptual model development and application of ArcNLET predict larger nitrogen loads from septic systems to surface waterbodies with larger hydraulic conductivity (Ye, Zhu, and Sayemuzzaman, 2014). Variation of nitrate loads estimated by Wang *et al.* (2012) demonstrates an almost exponential relationship with hydraulic conductivity, with steeply increasing loading rates at conductivity values greater than 10 m/d.

2.2.3.1.1 MELBOURNE BEACH

The hydraulic conductance input values varied from 2.30 to 19.78 µm/s during the MC simulation of Melbourne Beach, a relatively wide range of input values. As a result of these input values, output nitrate loads for this region are normally distributed and range from 6.79 to 35.97 lbs./yr (Figure 18), with the majority of the runs yielding between 16-31 lbs./year of nitrate. These values are only a subset of the total uncalibrated nitrate loading estimates previously described (168 lbs./year), due to only representing the loading of a subset of the septic tanks in the Melbourne Beach community of interest.

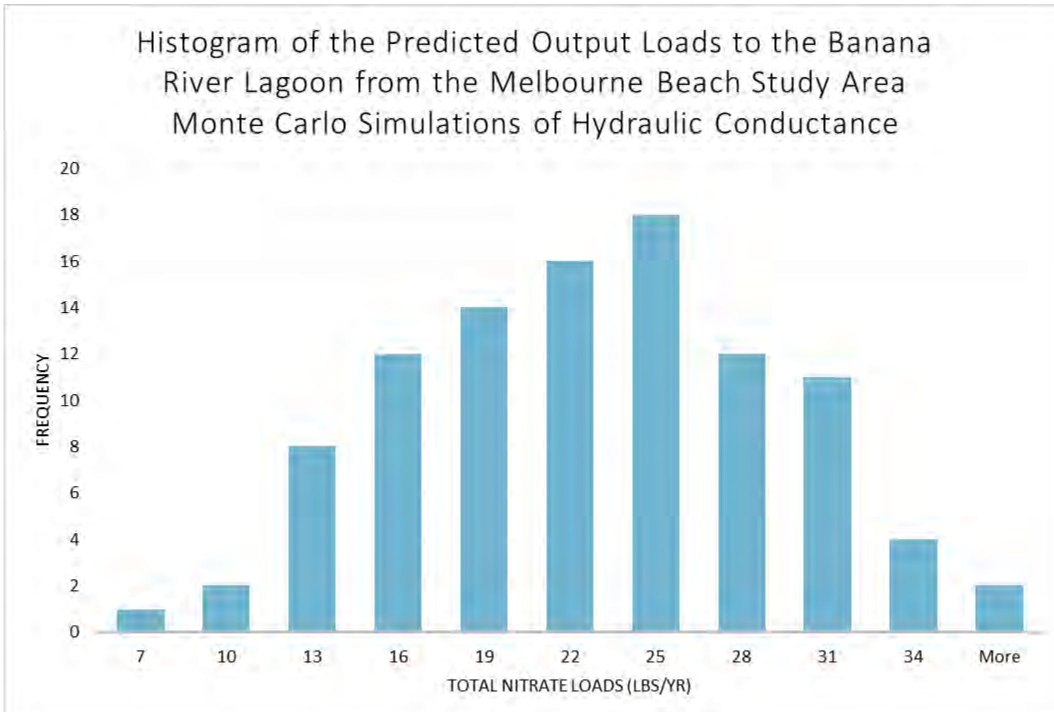


Figure 18. Histogram of the predicted input loads to the Banana River Lagoon from the Melbourne Beach study area based on Monte Carlo simulations of hydraulic conductance.

Most importantly, the resulting predicted nitrate concentrations for all 100 simulations present a logical relationship to hydraulic conductivity, with linear increases in nitrate concentrations with increases in hydraulic conductivity. In fact, very strong correlation coefficients ($R^2 \geq 0.943$) are reported between predicted nitrate load and hydraulic conductance (Figure 19) for all three monitoring well locations. This clearly indicates that, when removing any other contributing factors, at least 94% of the variance in nitrate loads can be explained by hydraulic conductivity. Furthermore, these changes clearly indicate a linear relationship between hydraulic conductivity and nitrate plume concentrations. Interestingly, output loads resulting from the same hydraulic conductance are different at the three well locations, with lower values for both SP 250 and SP 275 than at SP 270. Placement of wells in relationship to drain fields and proximity to the receiving surface water (e.g. Lagoon) likely explain these differences in magnitude of total nitrate concentration and slope of the linear relationship. Out of all three wells, SP 270 is located closest to the Lagoon.

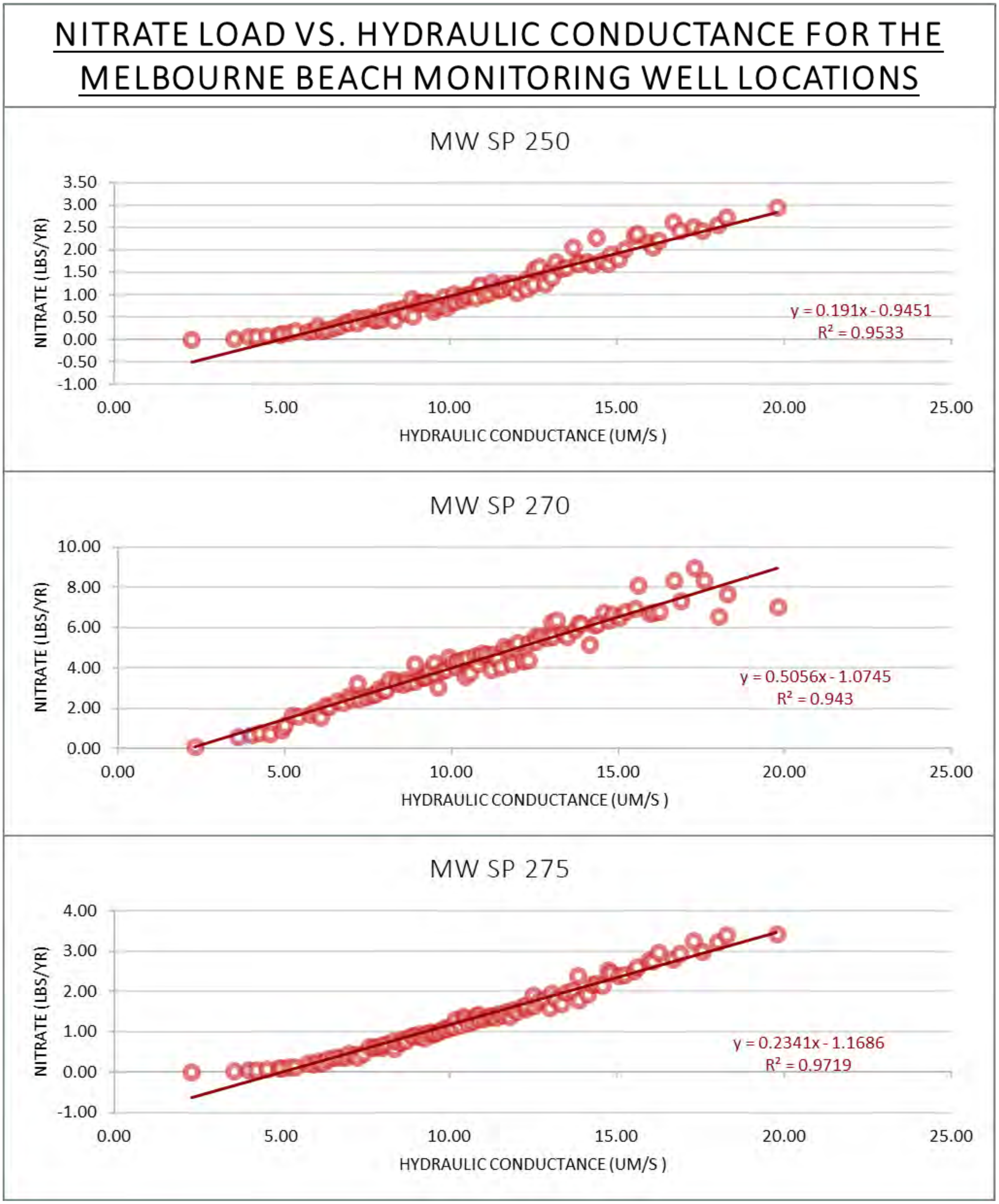
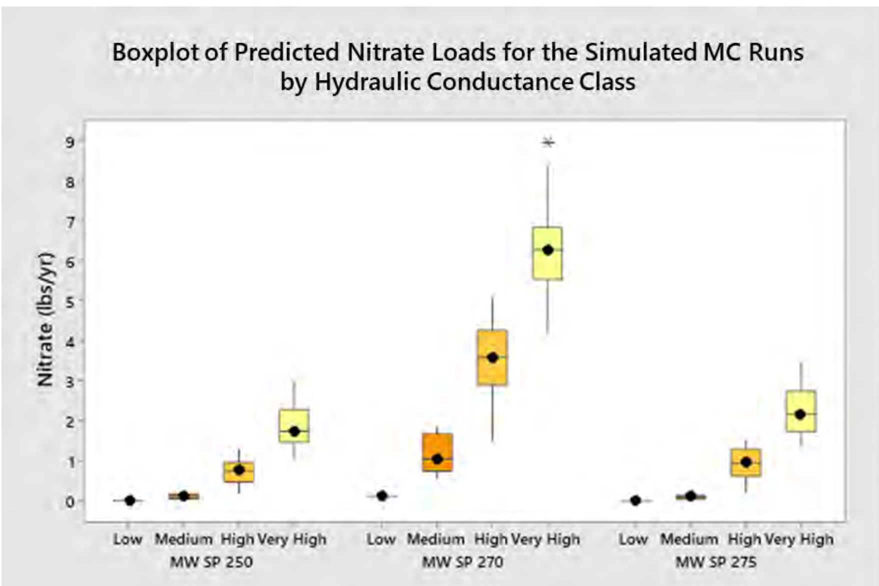


Figure 19. Scatterplots of nitrate loads (lbs./yr) vs. hydraulic conductance (μm/s) at the monitoring well locations within the Melbourne Beach study area.

A similar linear relationship between predicted nitrate concentration and hydraulic conductivity is also demonstrated in Figure 20, which synthesizes the nitrate results of the 100 MC simulations by hydraulic conductance class (i.e. very low through very high). As expected, as the hydraulic conductance increases, the median, interquartile range, and data distribution of predicted nitrate loads also increases. Interestingly, variability of outputs is highest for results in the high and very high hydraulic conductance classes; this means septic tank loading in soil areas of higher hydraulic conductance is not only higher, but also more difficult to pinpoint.



Soil Hydraulic Conductance Class	Soil Hydraulic Conductance Value
Very Low	≤ 1.274400
Low	1.274401 – 3.576960
Medium	3.576961 – 6.065280
High	6.065281 – 11.737440
Very High	≥ 11.737441

Figure 20. Boxplot of predicted nitrate loads by hydraulic conductance class (defined by the gradient table on the right) at monitoring well locations within the Melbourne Beach study area.

As there were only three monitoring well locations within the Melbourne Beach boundary, MC simulated nitrate plumes were extracted for an additional 52 locations ("simulated" well locations) to ensure similar relationships between hydraulic conductivity inputs and nitrate outputs are similar throughout the entire study area. Similar strong linear relationships ($R^2 > 0.94$) with varying slopes (0.4 to 0.5) were identified for these simulated locations (Figure 21). Output loads were very similar to those above observed for the monitoring wells with lower variability, even at the highest hydraulic conductivity class. The three simulated locations included in Figure 21 are located relatively close to the previously described monitoring well locations or slightly downstream and closer to the Lagoon waters.

MELBOURNE BEACH SIMULATED NITRATE LOADS EXTRACTED FOR SELECTED SIMULATED LOCATIONS (NEAR THE LAGOON)

a) Scatterplots: Nitrate Load VS. Hydraulic Conductance	b) Boxplots: Nitrate Loads by Hydraulic Conductance Class

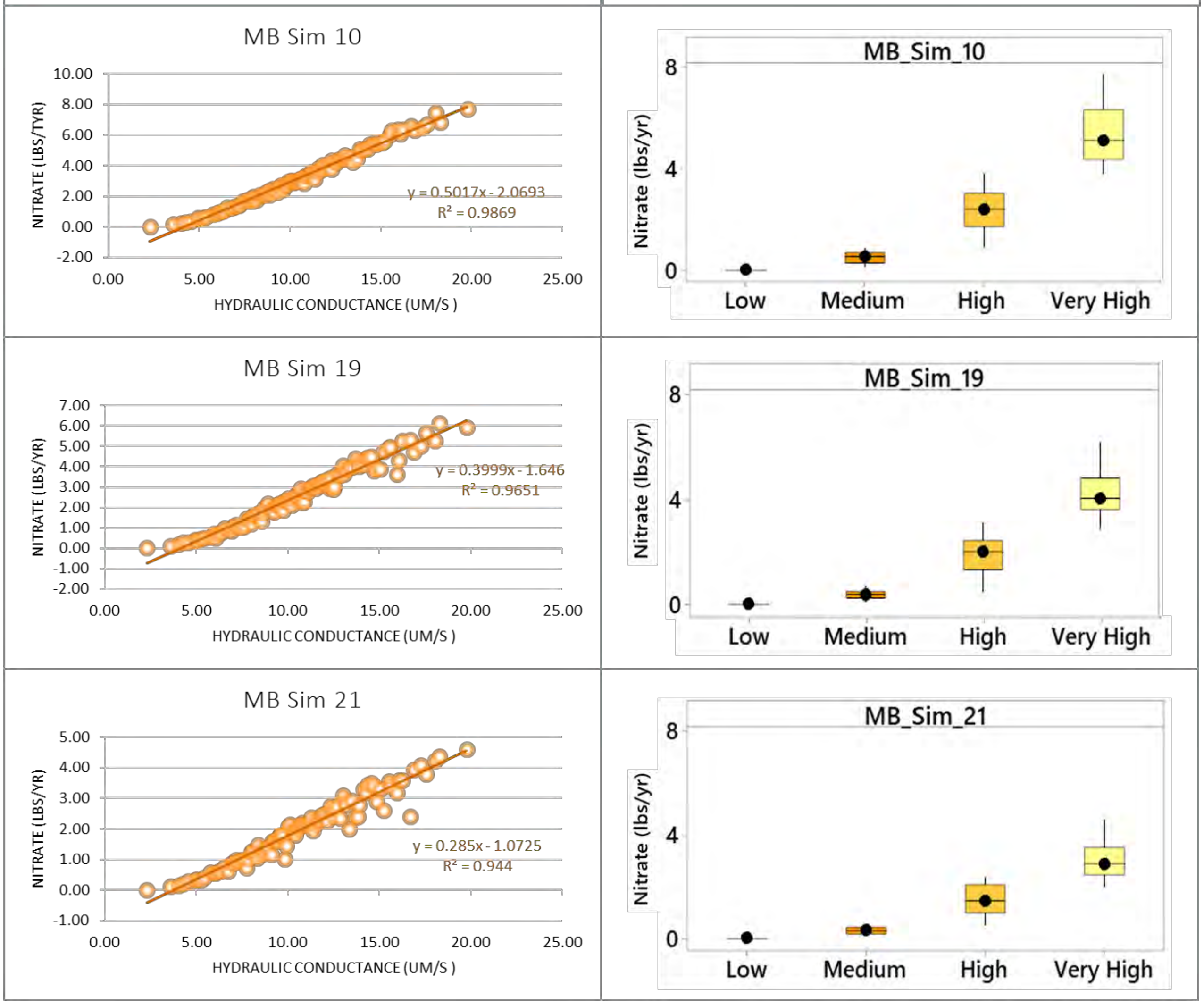


Figure 21. Resulting nitrate plume concentrations at selected locations near the Lagoon and monitoring wells within Melbourne Beach based on changes in input hydraulic conductance via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. hydraulic conductance (µm/s) and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input hydraulic conductance classes.

Predicted nitrate loads at locations further from both the monitoring wells and the Lagoon demonstrate a much weaker relationship between hydraulic gradient and nitrate load outputs (Figure 22). Even though there is a clear increase in predicted nitrate loads with increases in hydraulic conductivity, the correlation coefficients for these selected locations are relatively low, indicating hydraulic conductance is not able to explain the majority of the variance at these locations.

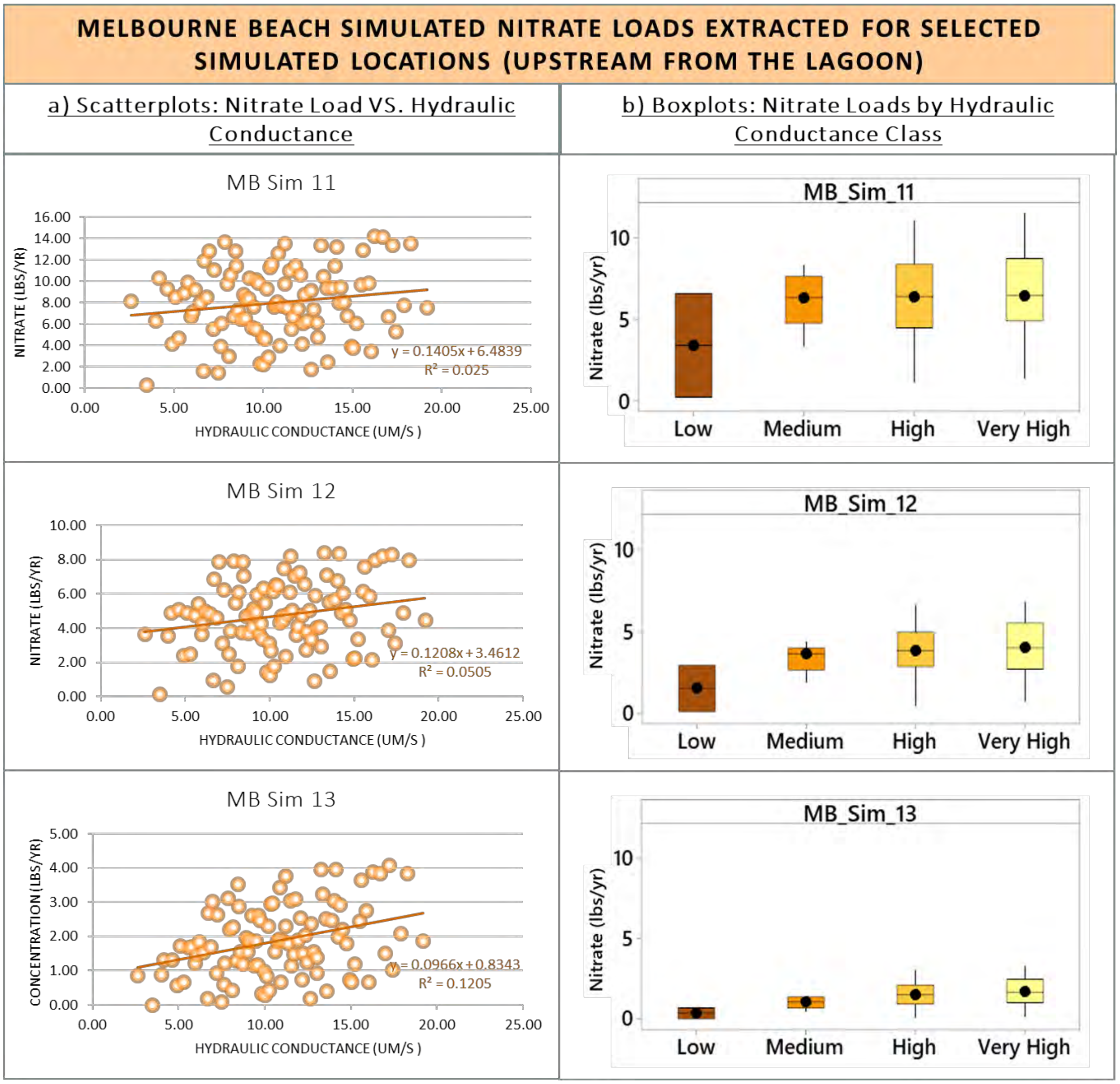


Figure 22. Resulting nitrate plume concentrations at selected locations further from the Lagoon within Melbourne Beach based on changes in input hydraulic conductance via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. hydraulic conductance (µm/s) and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input hydraulic conductance classes.

From the synthesis of the results presented above, it is apparent that hydraulic conductivity and nitrate output loads are related, with varying slopes and correlation coefficients based on spatial location (Figure 23). While there are some outlier septic points, such as those of the monitoring well locations, there is a general pattern of lower R^2 values for locations further from the Lagoon. Whereas 75% of the simulated locations within 150-m from the Lagoon had very strong relationships between hydraulic conductance and nitrate loads (R^2>0.80), most of the data extracted for locations located >150-m from the Lagoon had R^2 below 0.5.

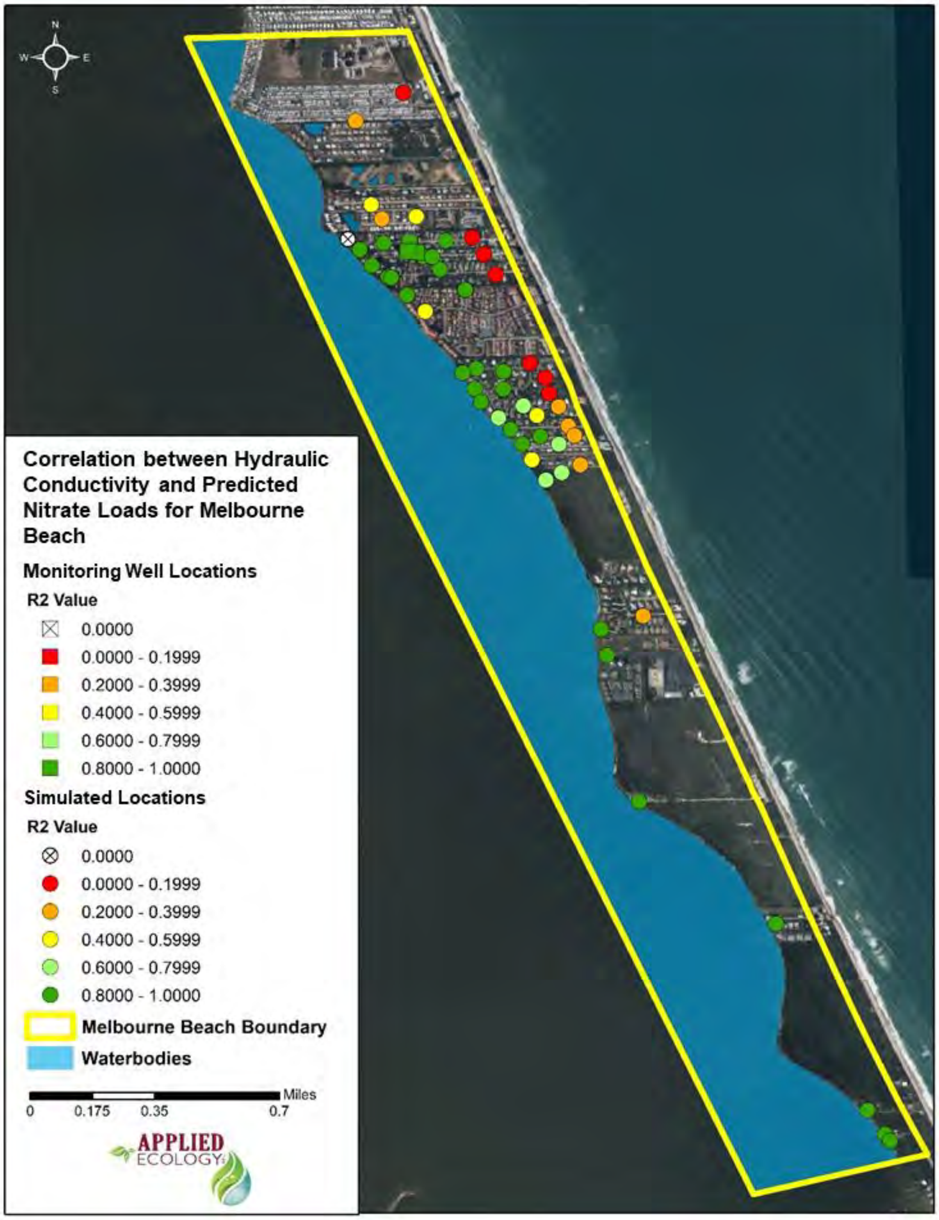


Figure 23. Spatial distribution of the correlation coefficients (R^2) between input hydraulic conductance and output nitrate loads for both the simulated and monitoring well locations within the Melbourne Beach study area.

2.2.3.1.2 SUNTREE

The hydraulic conductance input values varied from 2.19 to 19.76 µm/s during the MC simulation of the Suntree study area, a similar range to that one used in the Melbourne Beach simulation. Output nitrate loads for this region ranged from 34.17 to 49.38 lbs./yr, which are relatively higher in magnitude and lower in range than those described for the Melbourne Beach area (Figure 24). More than 50% of the 100 MC simulations performed for the Suntree area yielded total nitrate load outputs between 40 - 45 lbs./yr.

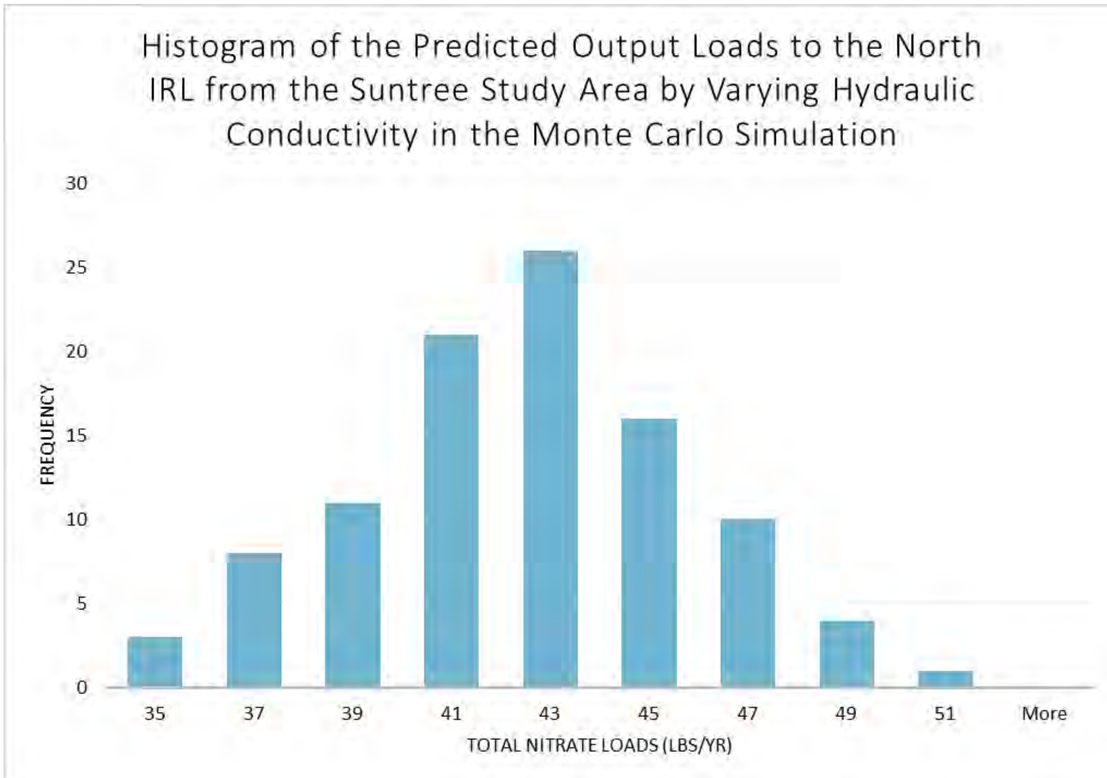


Figure 24. Histogram of the predicted output loads to the North IRL from the Suntree study area based on Monte Carlo simulations of hydraulic conductance.

Although the Suntree area had higher overall total output loads than the Melbourne Beach area when varying hydraulic conductance during the MC simulations, the mean individual plume magnitude is lower than then the mean for the plumes produced for Melbourne Beach. Extracted loads at the monitoring wells SP 6155 and SP 6215 are two orders of magnitude lower than those extracted for the SP 6398 location within Suntree and all three monitoring wells in Melbourne Beach. Additionally, the correlation coefficients between hydraulic loads and predicted nitrate loads were highly variable even just examining the data from the three monitoring well locations. Similar correlation coefficients for SP 6398 and SP 6215 indicate that at least 80% of the variance within nitrate loads can be explained by the variance in hydraulic conductance; however, only 43% of the variance can be explained at SP 6155 (Figure 25). Furthermore, it is also interesting to note that the scatterplot of predicted nitrate versus hydraulic conductivity clearly shows data lined up at two different slopes for SP 6155. If the data could be teased out, two regression lines with higher correlation coefficients would likely emerge. The fact that the data is distributed at two different slopes indicates that there is another factor driving, in selected simulation runs, driving a higher nitrate response from the same hydraulic conductance. This factor cannot be the location of input septic tanks, distance to waterbodies or input nitrate loading source, since these were held constant for all 100 simulations.

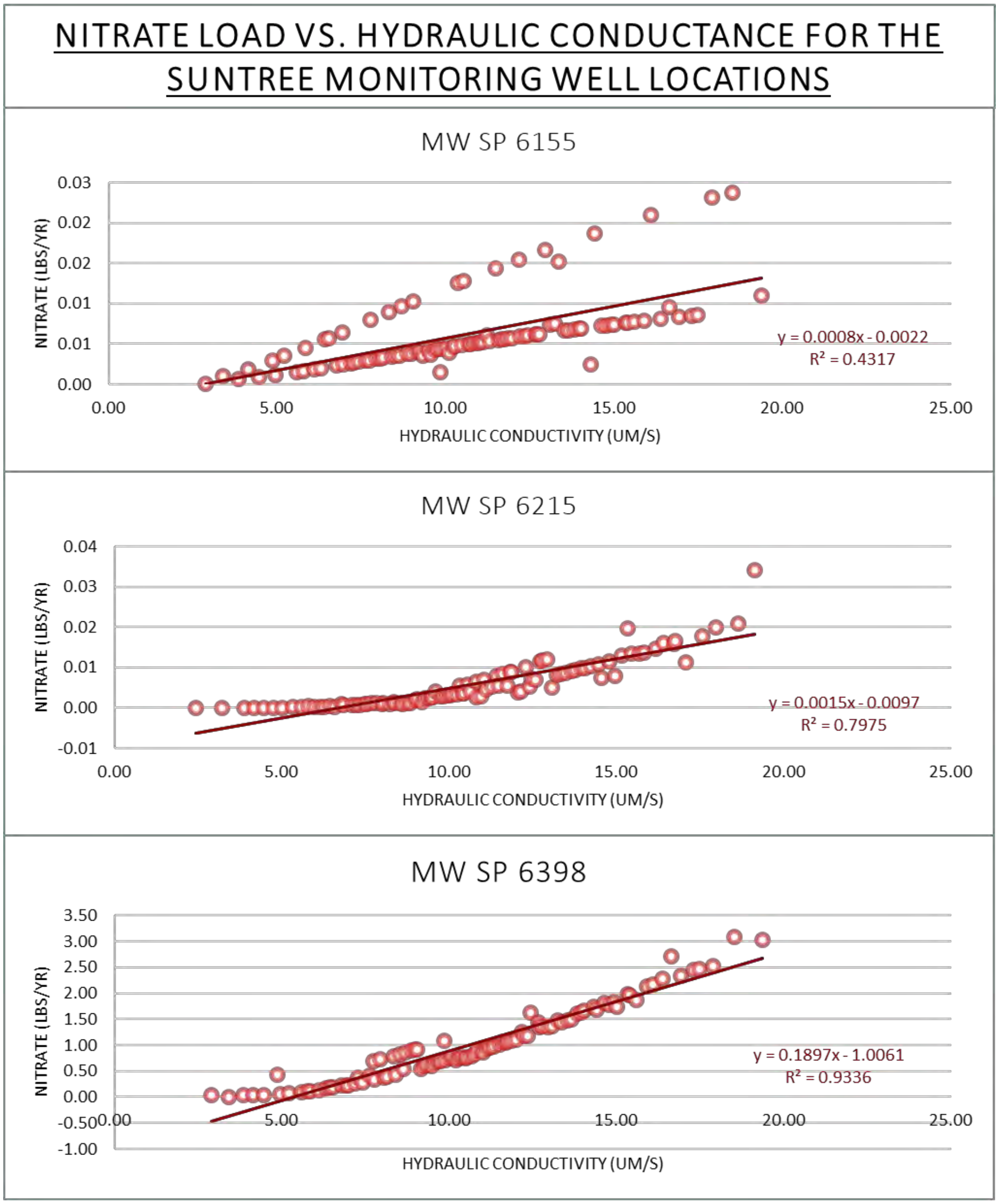
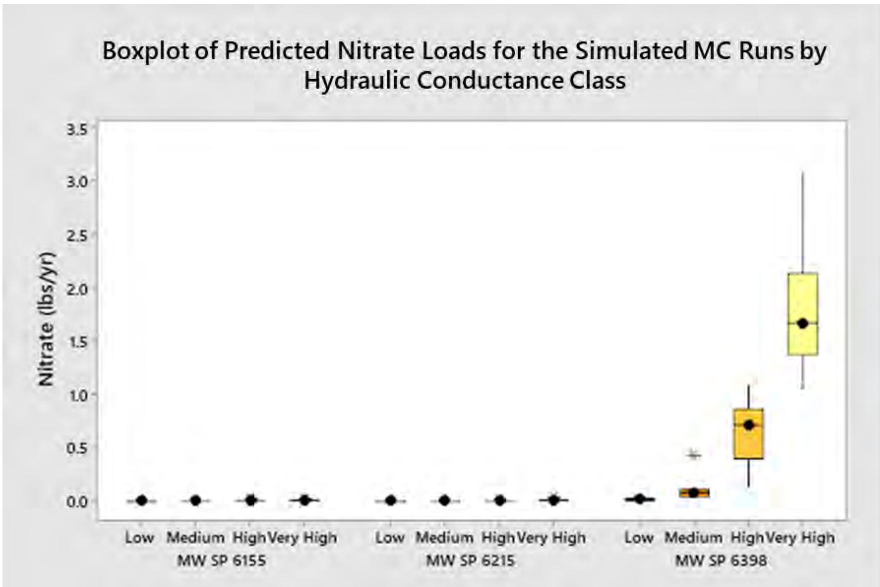


Figure 25. Scatterplots of nitrate loads (lbs./yr) vs. hydraulic conductance (µm/s) at the septic point at the monitoring well locations within the Suntree area.

The relationship between the input hydraulic conductance and predicted nitrate load outputs is also demonstrated in Figure 26, which shows the resulting nitrate loads from each MC simulation grouped by hydraulic conductance class. As the hydraulic conductance increases, the median, range, and variance of nitrate loads in each class also increase at the monitoring well locations. However, this pattern is difficult to visualize extracted at the SP 6215 and SP 6155 locations, since predicted nitrate outputs were much lower than those at the SP 6398 well location. The latter presents a similar pattern to those described for the Melbourne Beach well locations, with greater nitrate outputs and associated variability for the high and very high hydraulic conductivity classes.



Soil Hydraulic Conductance Class	Soil Hydraulic Conductance Value
Very Low	≤ 1.274400
Low	1.274401 – 3.576960
Medium	3.576961 – 6.065280
High	6.065281 – 11.737440
Very High	≥ 11.737441

Figure 26. Predicted nitrate loads by hydraulic conductance class (defined by the gradient table on the right) at monitoring well locations within the Suntree study area.

As was done with the Melbourne Beach study area, data from the simulated MC runs were also extracted and synthetized for an additional 38 locations throughout the study area to explore how location might impact the hydraulic conductivity relationship to predicted nitrate loads. Very similar pattern to that one described for Melbourne Beach emerged with locations closest to the Lagoon having stronger relationships between hydraulic conductivity and nitrate outputs (Figure 27) than for those further from the Lagoon (Figure 28). However, in the Suntree area, unlike in Melbourne Beach, there is more variability in the predicted nitrate plume simulations, with R^2 ranging between 0.43 and 0.94 for the locations closer to the Lagoon. For plumes generated well upgradient from the Lagoon (at further distances), correlations are even weaker and varied from 0.14 to 0.34 with a lot of unexplained variability (Figure 28).

While increasing hydraulic conductivity clearly appears to drive higher nitrate loads in the ArcNLET, slopes can appear to be shallower to most of those encountered for the Melbourne Beach data; this is true throughout the Suntree MC model simulation area, but even more noticeable for locations at further distances from the Lagoon.

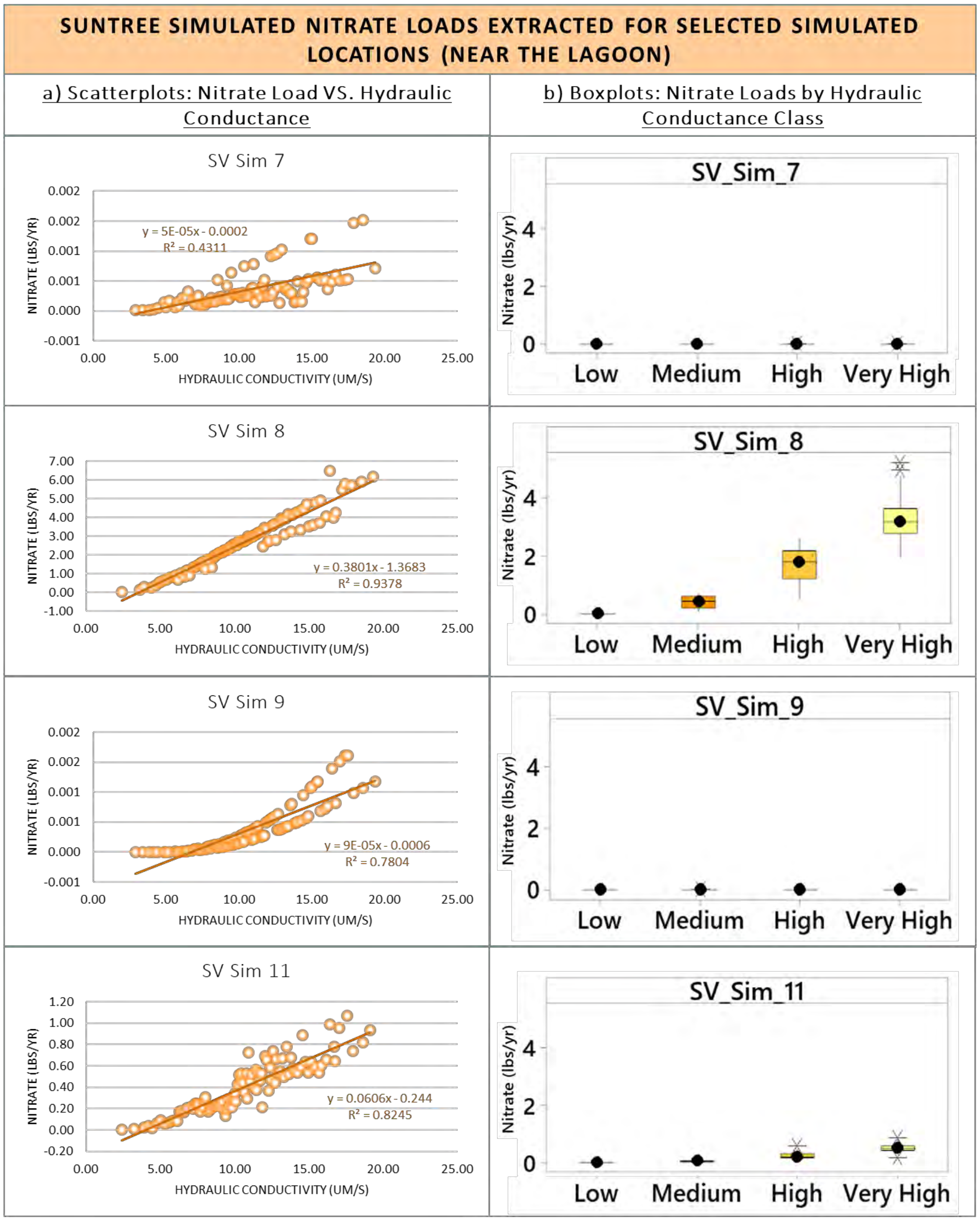


Figure 27. Resulting nitrate plume concentrations at selected locations near the Lagoon and monitoring wells within Suntree based on changes in input hydraulic conductance via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. hydraulic conductance (µm/s) and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input hydraulic conductance classes.

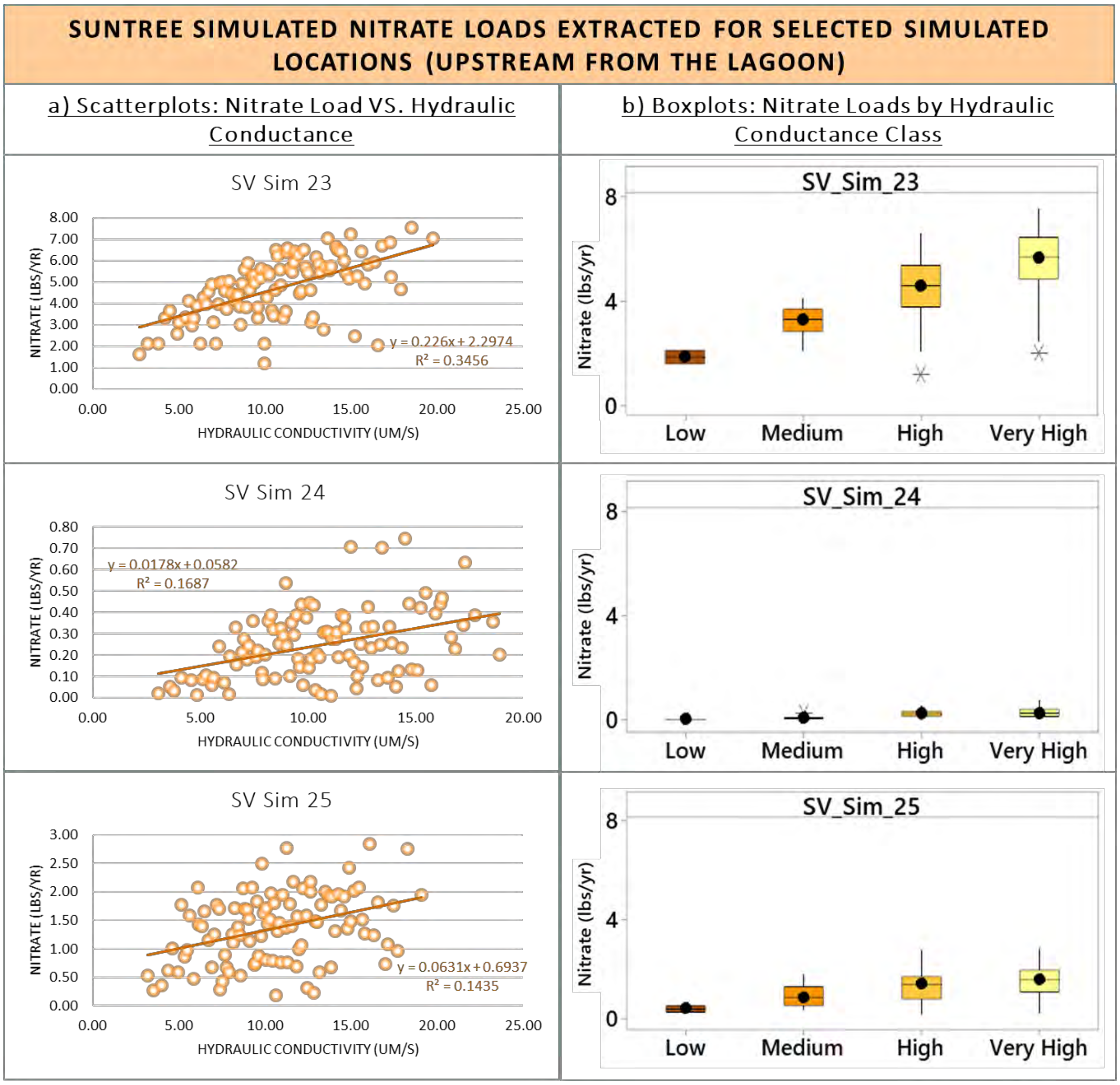


Figure 28. Resulting nitrate plume concentrations at selected locations farther from the Lagoons within Suntree based on changes in input hydraulic conductance via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. hydraulic conductance (µm/s) and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input hydraulic conductance classes.

2.2.3.2 POROSITY

Soil porosity has been used in numerous reports to better understand OSTDS pollution potential (Rios *et al.*, 2011; Wang *et al.*, 2012; Keene, 2015; Zhu *et al.*, 2016). Porosity is the measure of the void spaces between the soil as a percentage between 0% and 100%. Permeability is a function of porosity, particle size, and the arrangement of these particles. Typically, surface soil horizons have large void spaces and higher porosity than deeper soils due to compaction over time.

Soil porosity is used in the ArcNLET model to estimate seepage velocity using Darcy's velocity. Average velocity is increased by increasing hydraulic conductivity and/or a decrease in soil porosity. Soil porosity has an inverse exponential relationship with estimated loading from septic tanks, with highest loading values at porosity values around 25% and relatively low values at 45% or greater (Wang *et al*., 2012).

2.2.3.2.1 MELBOURNE BEACH

The porosity input values varied from 0.2048 to 0.4474 during the MC simulation of Melbourne Beach. Output nitrate loads for this region range from 16.24 to 31.97 lbs./yr (Figure 29). Approximately 51% of the total predicted output nitrate loads for this modeled region varied between 19.00 and 23.00 lbs./yr. The range of output nitrate loads, and the maximum total output load is lower in comparison to those predicted by varying hydraulic conductance in the MC simulation for this same area.

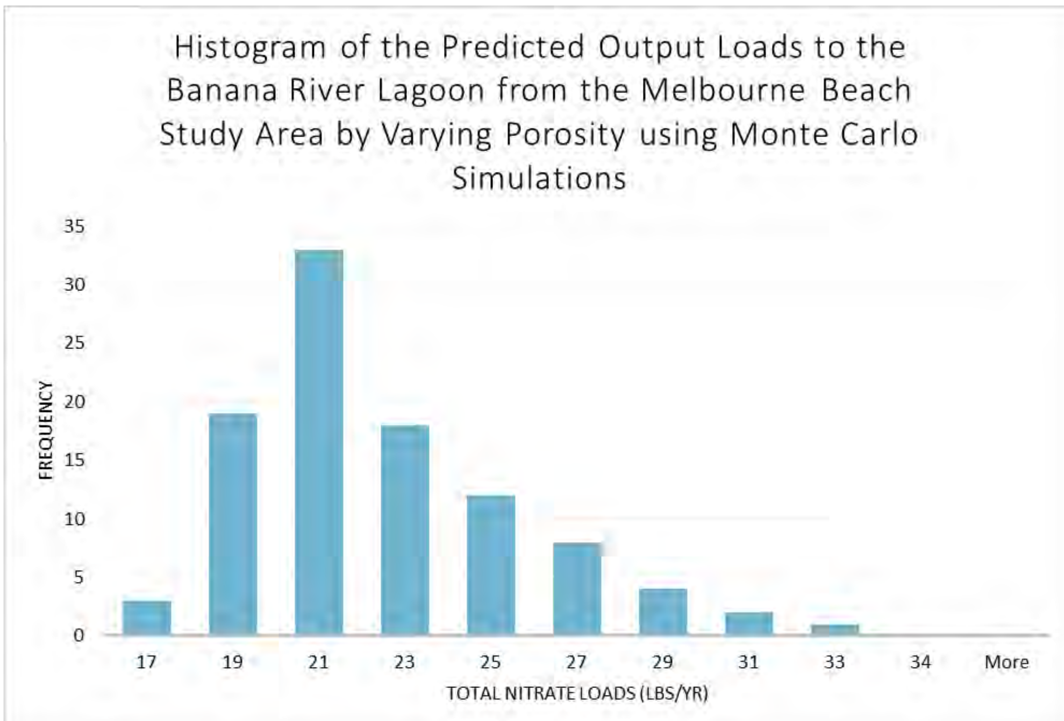


Figure 29. Histogram of the predicted input loads to the Banana River Lagoon from the Melbourne study area based on Monte Carlo simulations of porosity.

Relationships between input porosity and output nitrate loads extracted for the three monitoring well locations had fairly strong correlation coefficients ($R^2 \geq 0.784$), indicating at least 78% of the variance in nitrate concentrations can be explained by changes in porosity (Figure 30), when all other model parameters are kept static (hydraulic conductivity, septic input loads, denitrification coefficient, etc.). More specifically, decreases in nitrate will be accompanied by increases in porosity, anticipated by our understanding of hydraulics, where average velocity increases with a decrease in soil porosity. Greater velocity of nutrient transport leads to greater nitrate magnitudes at longer distances, since there is less time for denitrification to occur. Similar to what was described for hydraulic conductance, predicted nitrate plumes are greatest near SP 270, which is located closest to the Lagoon than at the other two well locations (SP 250 and SP 275).

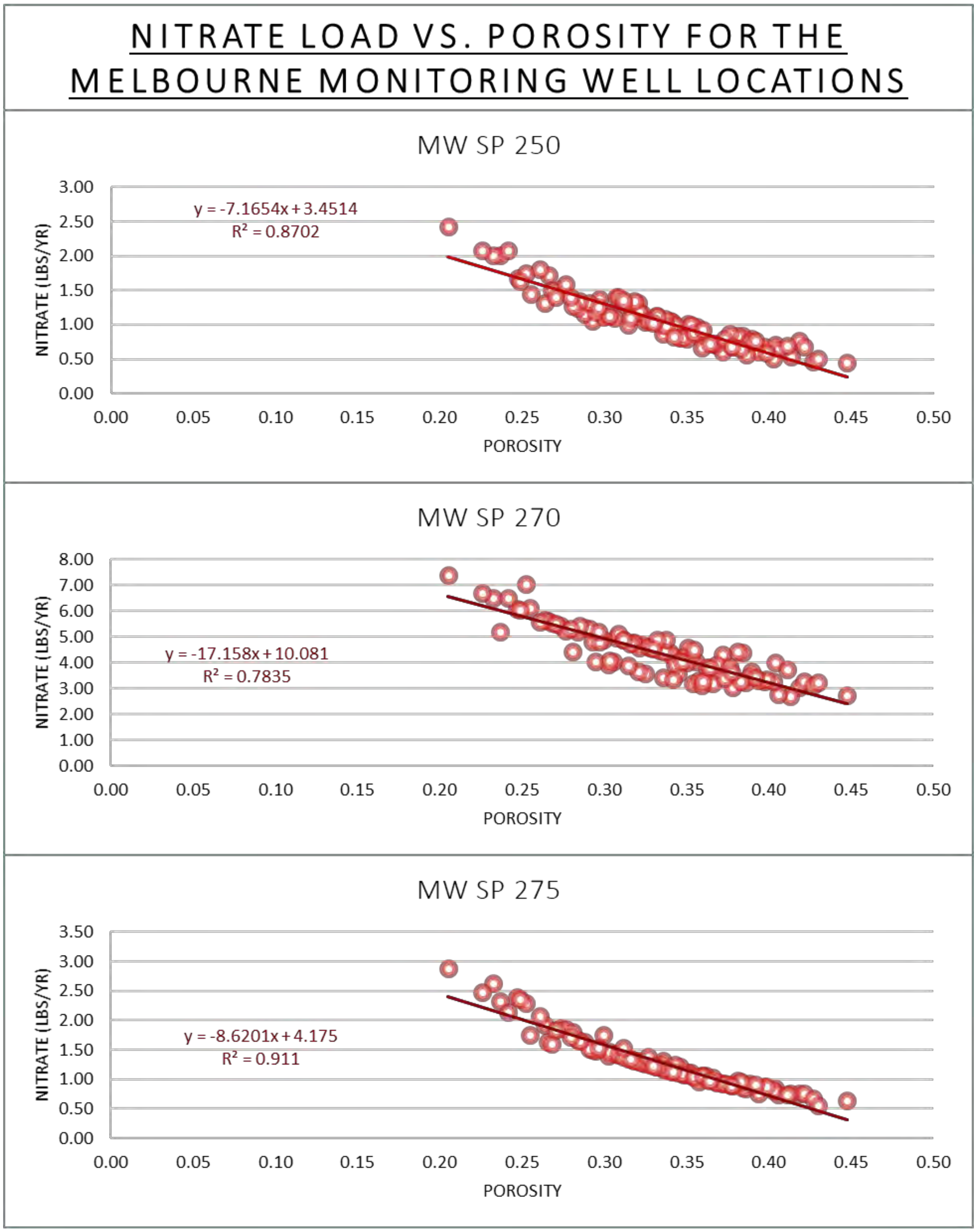
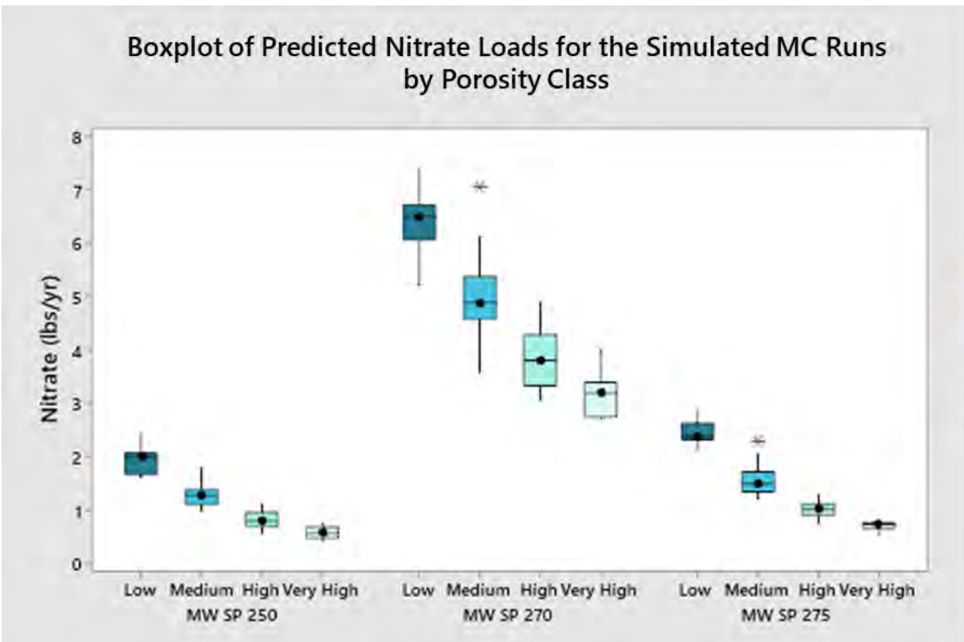


Figure 30. Scatterplots of nitrate loads (lbs./yr) vs. porosity at the monitoring well locations within the Melbourne Beach study area.

This relationship is also demonstrated in Figure 31, which shows the resulting nitrate loads at each of the monitoring well location from the MC simulations grouped by porosity class. As expected, median nitrate loads in each class decrease as the porosity increases. There are some decreases in range and variance by class, although these are not consistent across all of the monitoring well septic points. Higher output loads and larger variances were observed for plumes generated near the SP 270 monitoring well, while similar nitrate distributions were observed for the other two locations (SP 250 and SP 275).



Soil Porosity Class	Soil Porosity Value
Very Low	≤ 0.250
Low	> 0.250 - 0.330
Medium	> 0.330 - 0.400
High	>0.400 - 0.550
Very High	> 0.550

Figure 31. Boxplot of predicted nitrate loads by porosity class (defined by the gradient table on the right) at monitoring well locations within the Melbourne Beach study area.

Similar negative linear relationship between porosity and predicted nitrate outputs is also visible for several additional locations near the monitoring well locations within the Melbourne Beach study area (Figure 32). Output nitrate loads were also very similar to those reported above at the monitoring well locations, with steeper negative slopes and less scatter (corresponding to higher correlation coefficients, $R^2 > 0.87$).

Extracted nitrate load data for locations upstream from both the monitoring wells and the Lagoon demonstrate very weak relationships ($R^2 < 0.07$) between porosity and output nitrate loads (Figure 33). Even more obvious than the impact of location on hydraulic conductivity described in the previous section, location of the plumes changes the importance of the soil porosity as a driving factor in predicting nitrate loads. At distances > 150-m from the Lagoon, median nitrate plume concentrations are similar across the board for all porosity classes, clearly visible in the spatial map of correlation coefficients (Figure 34). Whereas approximately 71% of the predicted plume locations within the Melbourne Beach area have an R^2 of over 0.80 and are located within 150-

m of the IRL, most plumes generated at greater distances present R^2 values between porosity input values and nitrate outputs well below 0.5.

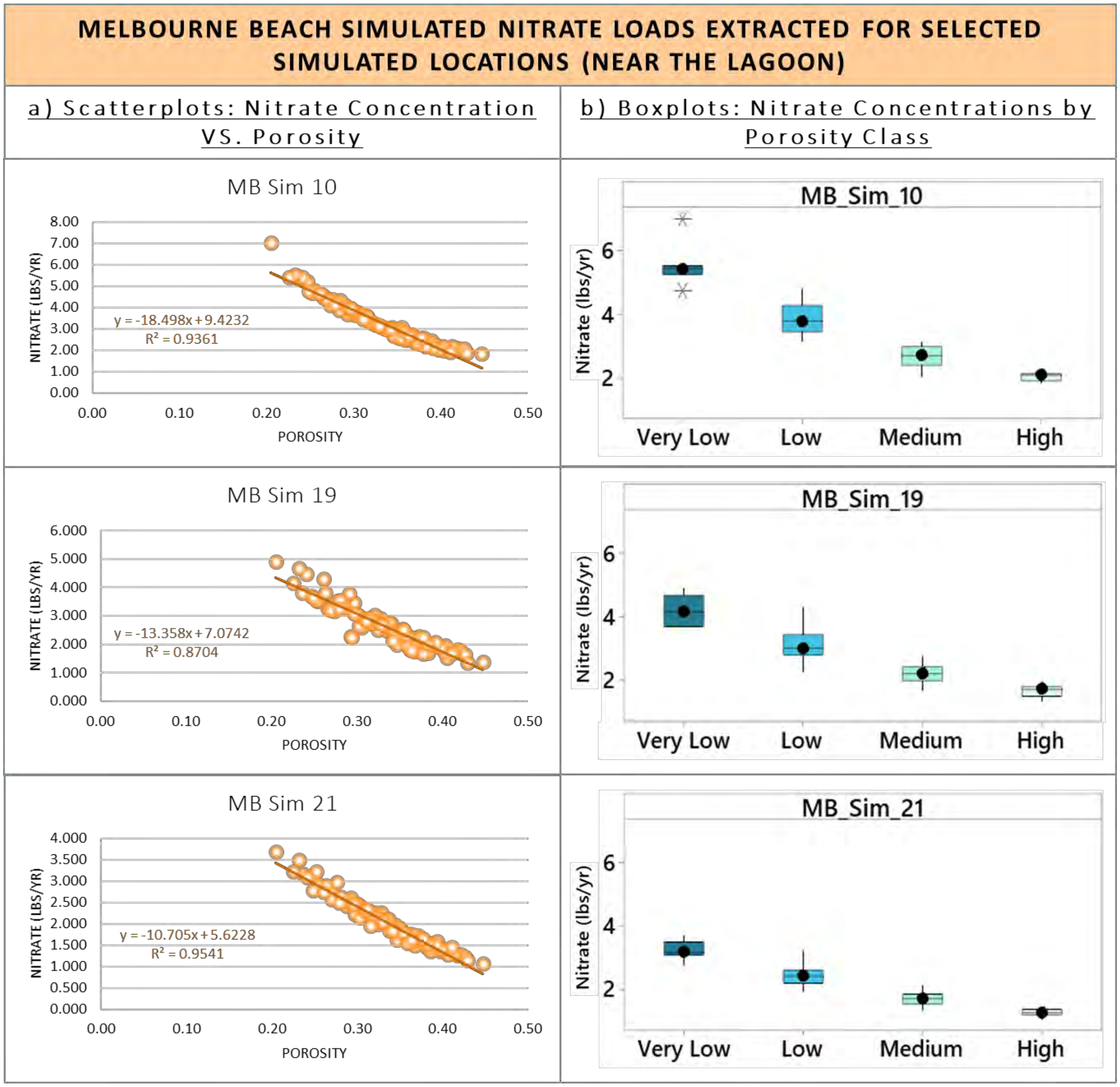


Figure 32. Resulting nitrate plume concentrations at selected locations near the Lagoon and monitoring wells within Melbourne Beach based on changes in input porosity via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. porosity and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input porosity classes.

MELBOURNE BEACH SIMULATED NITRATE LOADS EXTRACTED FOR SELECTED SIMULATED LOCATIONS (UPSTREAM FROM THE LAGOON)

a) Scatterplots: Nitrate Load VS. Porosity	b) Boxplots: Nitrate Loads by Porosity Class

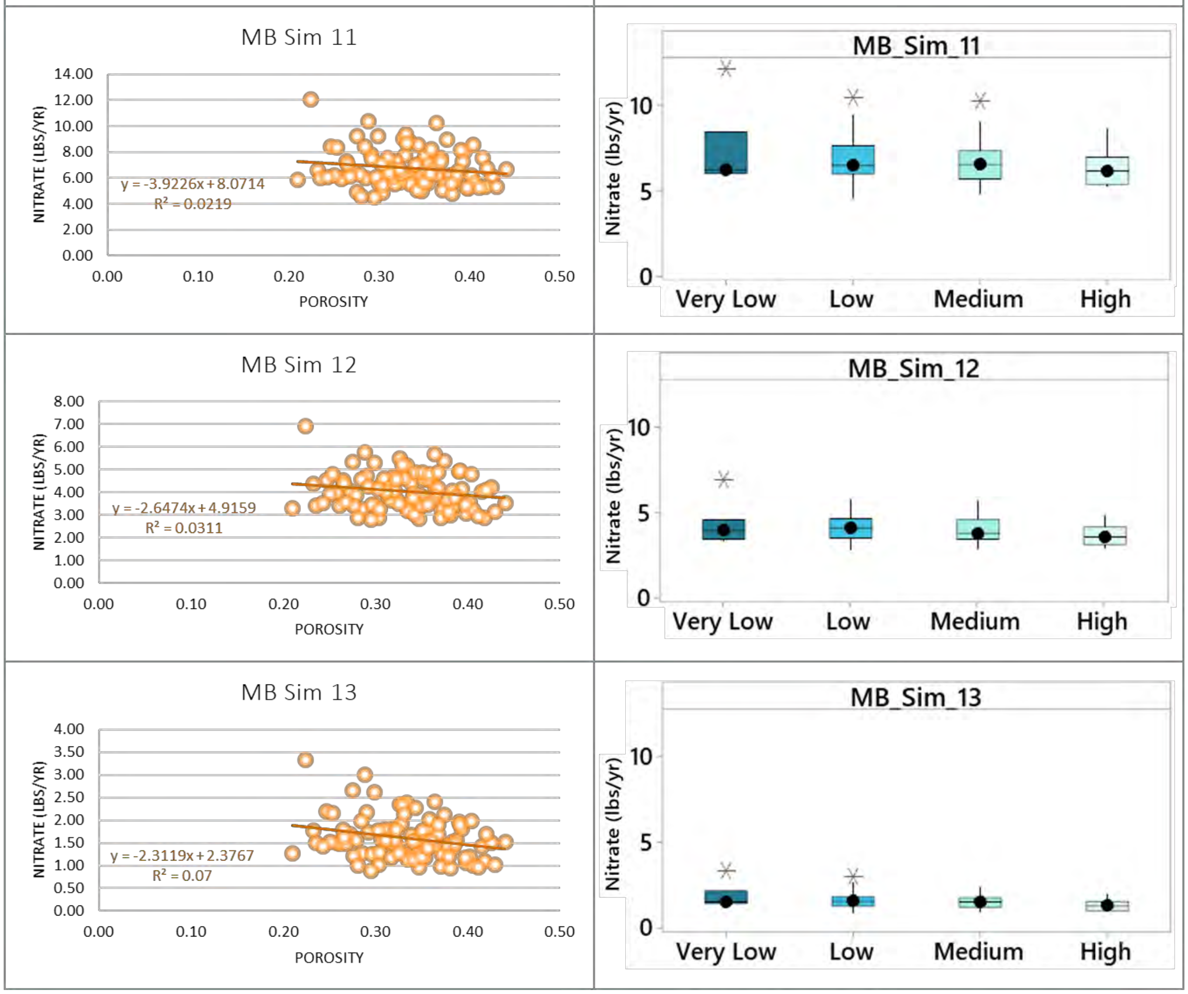


Figure 33. Resulting nitrate plume concentrations at selected locations further away from the Lagoon within Melbourne Beach based on changes in input porosity via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. porosity and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input porosity classes.

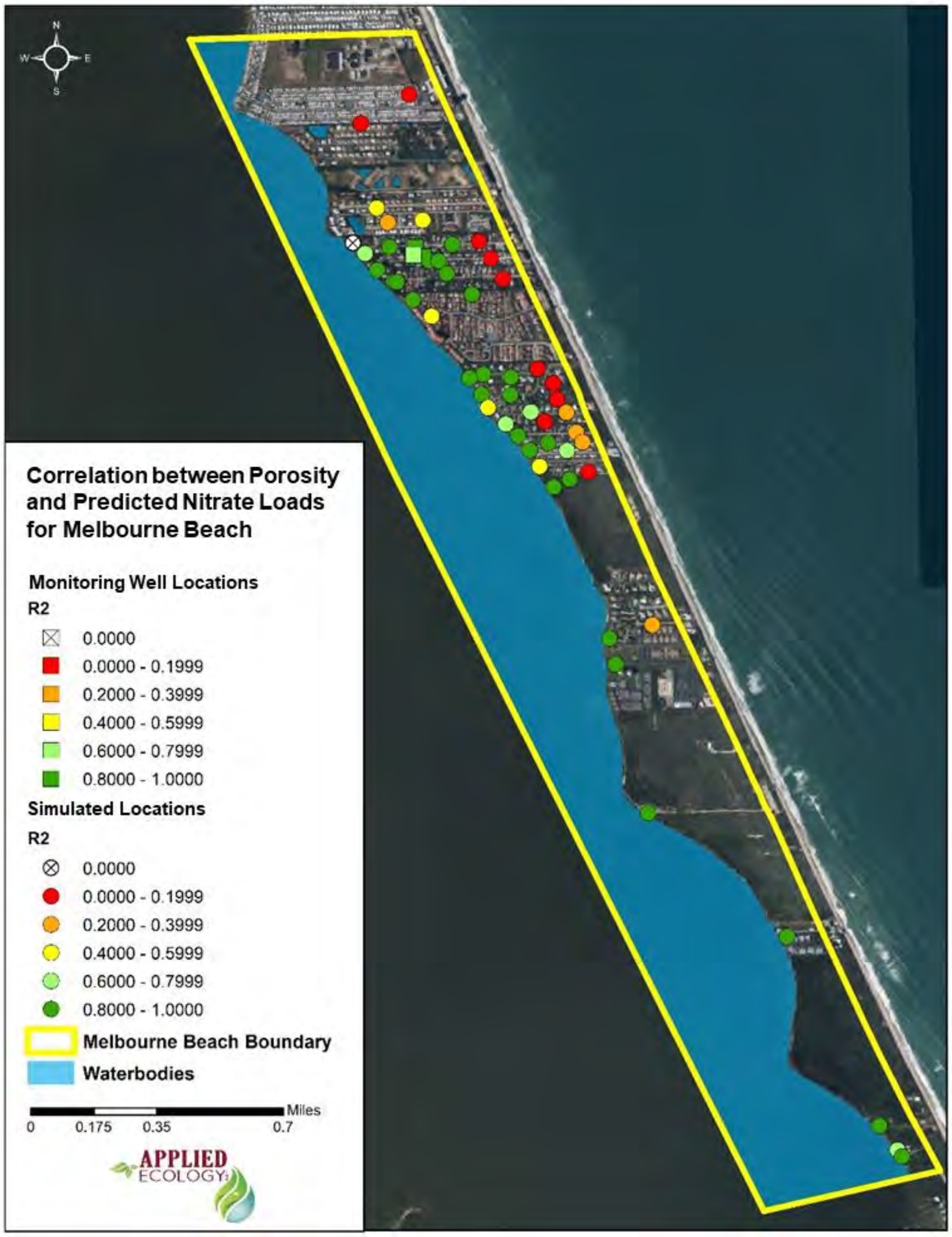


Figure 34. Spatial distribution of the correlation coefficients (R^2) between input soil porosity and output nitrate loads for both the simulated and monitoring well locations within the Melbourne Beach study area.

2.2.3.2.2 SUNTREE

The porosity input values for the MC simulation of the Suntree area were similar to those used for the Melbourne Beach simulation and varied from 0.203 to 0.447. However, output nitrate loads for this region were much higher than those described for Melbourne Beach, ranging from 39.21 to 46.73 lbs./yr (Figure 35). This is likely due to differences in the total number of septic tanks and area modeled for each of these two locations. Approximately 50% of the total predicted output nitrate loads for this modeled region varied between 42.00 and 44.00 lbs./yr.

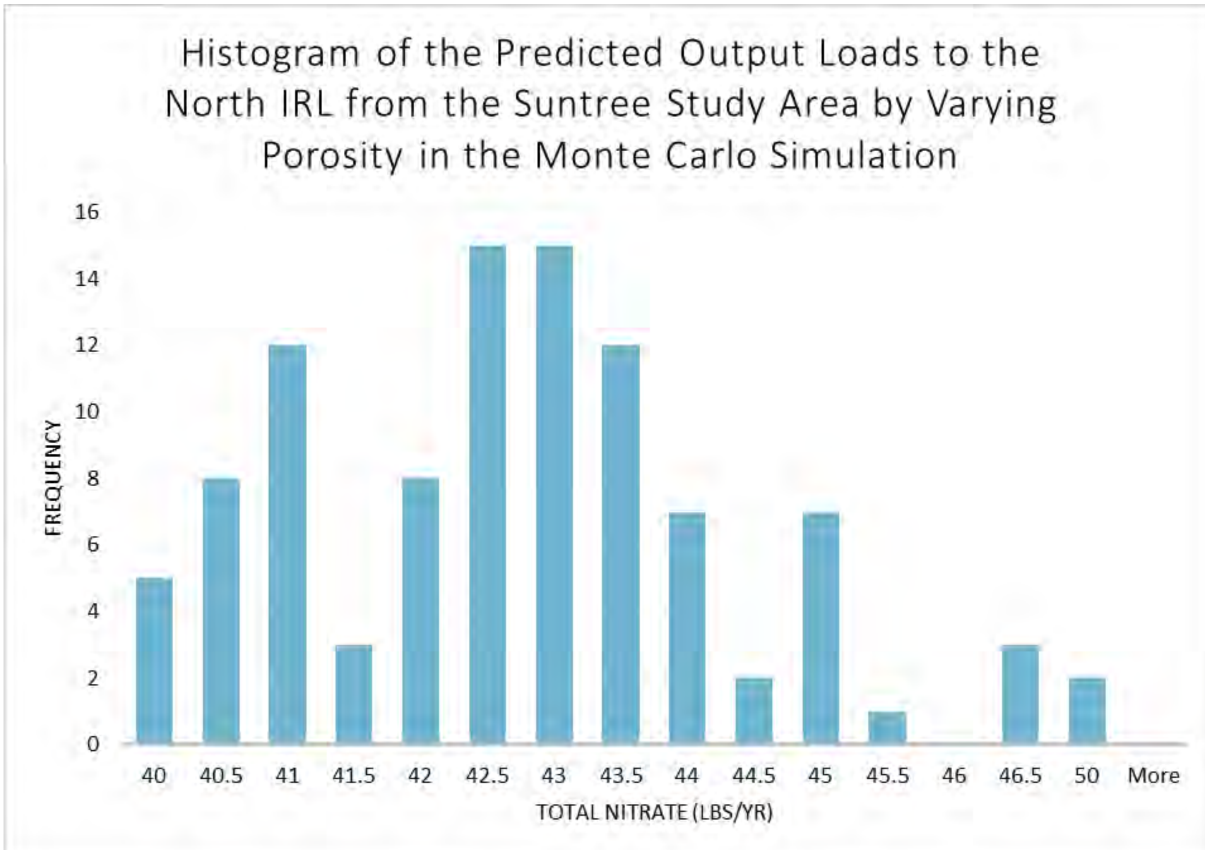


Figure 35. Histogram of the predicted output loads to the North IRL from the Suntree study area based on Monte Carlo simulations of porosity.

Even though total nitrate predictions are higher for Suntree than Melbourne Beach, extracted data that represents individual plume outputs present nitrate values well below those captured for Melbourne Beach in most cases. Extracted loads at the monitoring wells SP 6155 and SP 6215 are two orders of magnitude lower than those extracted for the SP 6398 location within Suntree and all three monitoring wells in Melbourne Beach. Additionally, the correlation coefficients between hydraulic loads and predicted nitrate loads were highly variable even just examining the data from the three monitoring well locations. Similar correlation coefficients for SP 6398 and SP 6215 indicate that at least 75% of the variance within nitrate loads can be explained by the variance in soil porosity; however, only 10% of the variance can be explained at SP 6155 (Figure 36). Just as described for hydraulic conductivity, two separate regression lines at different slopes appear to best describe the data for SP 6155. Similar patterns between soil hydraulic conductivity and porosity are anticipated since these two variables are typically autocorrelated with an inverse relationship.

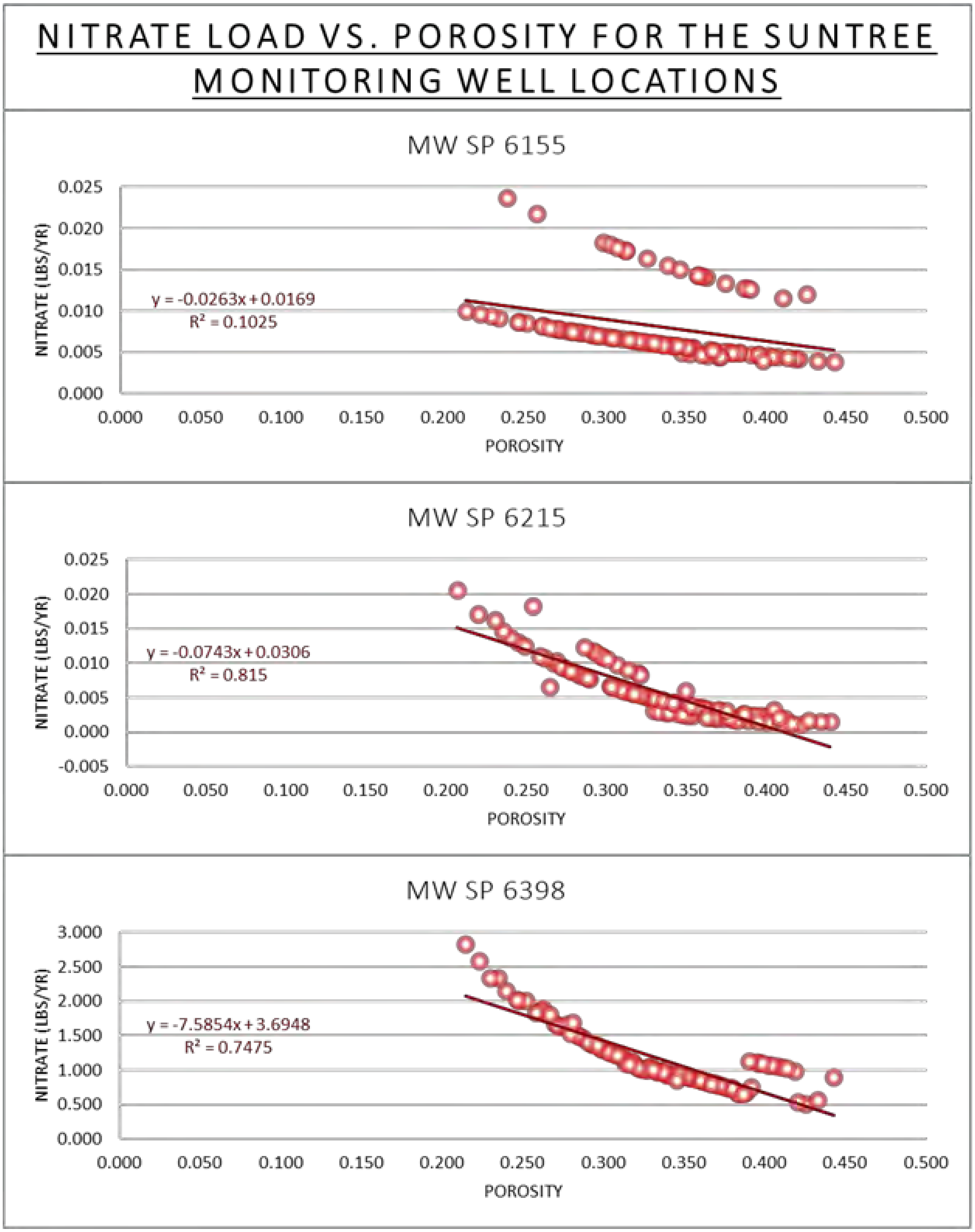
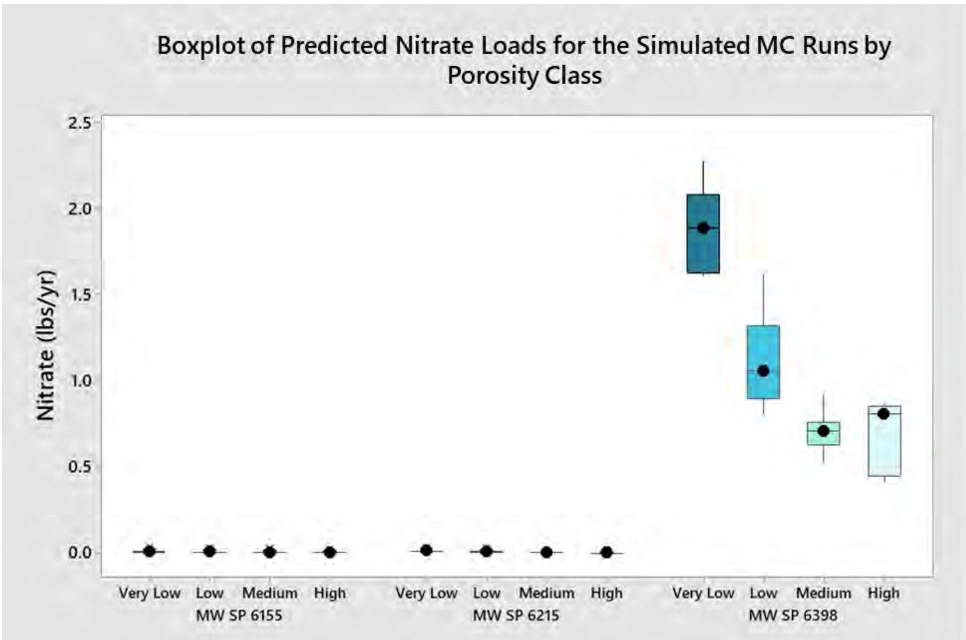


Figure 36. Scatterplots of nitrate loads (lbs./yr) vs. porosity at the monitoring well locations within the Melbourne Beach study area.

The inverse linear relationship between input porosity and nitrate predictions is clearly visible for SP 6398, but more difficult to discern due to negligible loads, even if present, for the other two well locations (Figure 37). As the porosity value of the soils decrease, the range of ArcNLET predicted nitrate loads generally increases.



Soil Porosity Class	Soil Porosity Value
Very Low	≤ 0.250
Low	> 0.250 - 0.330
Medium	> 0.330 - 0.400
High	>0.400 - 0.550
Very High	> 0.550

Figure 37. Boxplot of predicted nitrate loads by porosity class (defined by the gradient table on the right) at monitoring well locations within the Suntree study area.

Data from the simulated MC runs were also extracted and synthetized for an additional 38 locations throughout the Suntree study area to explore how location might impact the relationship between soil porosity and resulting nitrate loads. Similar to previously described spatial variability, locations closest to the Lagoon appear to have stronger relationships between porosity and nitrate outputs (Figure 38) in comparison to those located further from the Lagoon (Figure 39). However, in the Suntree area, unlike in Melbourne Beach, there is more variability in the predicted nitrate plume simulations, with R^2 ranging between R^2 ranging from 0.28 to 0.91 for the locations closer to the Lagoon. For plumes generated well upgradient from the Lagoon (at further distances), correlations are even weaker and varied from 0.14 to 0.39 with significant unexplained variability (Figure 39).

While decreasing soil porosity clearly drives higher nitrate loads in the ArcNLET simulations, slopes are overall shallower than those portrayed in the Melbourne Beach regressions; this is true throughout the Suntree MC model simulation area, but even more noticeable for locations at further distances from the Lagoon.

SUNTREE SIMULATED NITRATE LOADS EXTRACTED FOR SELECTED SIMULATED LOCATIONS (NEAR THE LAGOON)	
a) Scatterplots: Nitrate Load VS. Porosity	b) Boxplots: Nitrate Loads by Porosity Class

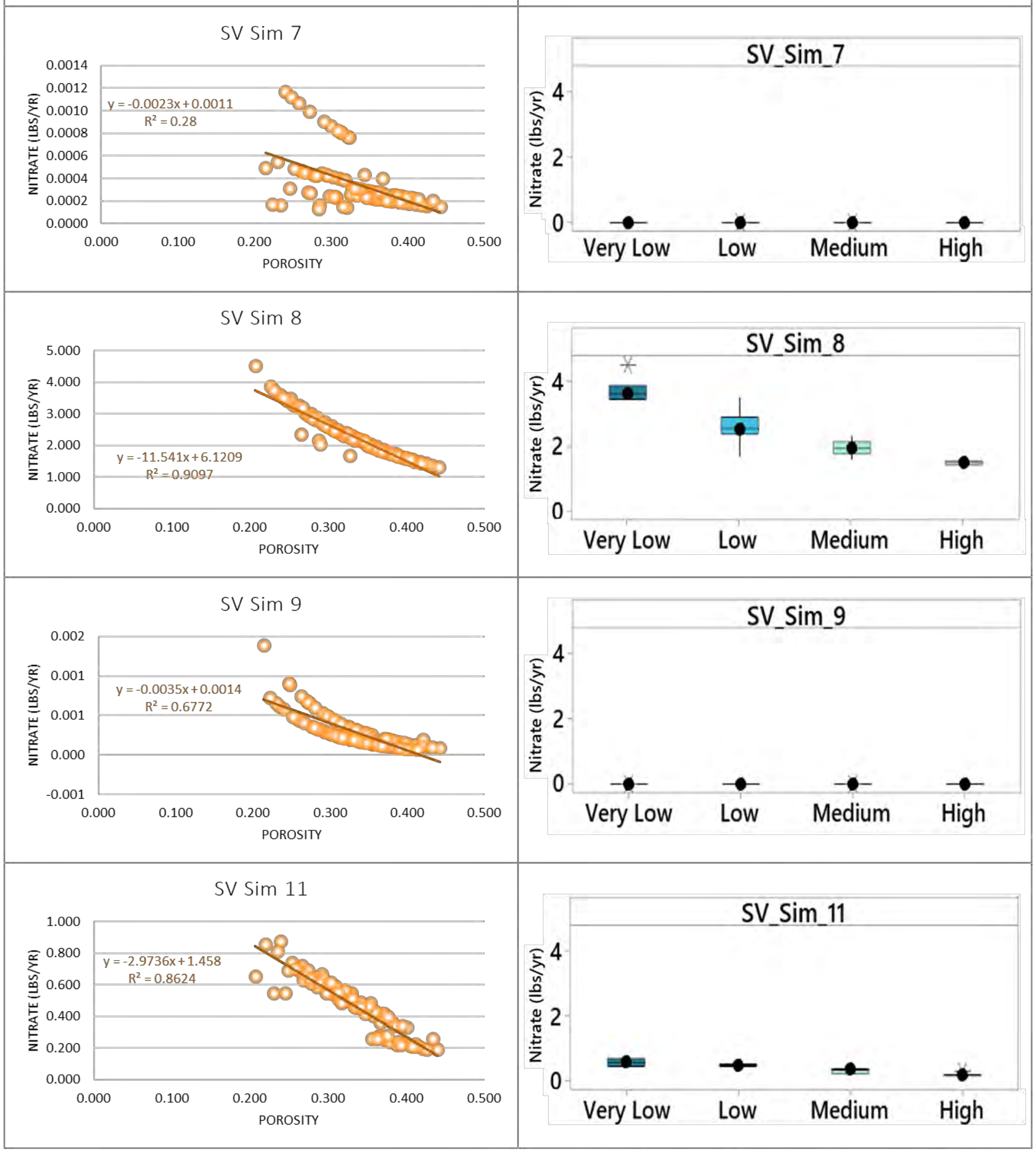


Figure 38. Resulting nitrate plume concentrations at selected locations near the Lagoon and monitoring wells within Suntree based on changes in input porosity via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. porosity and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input porosity classes.

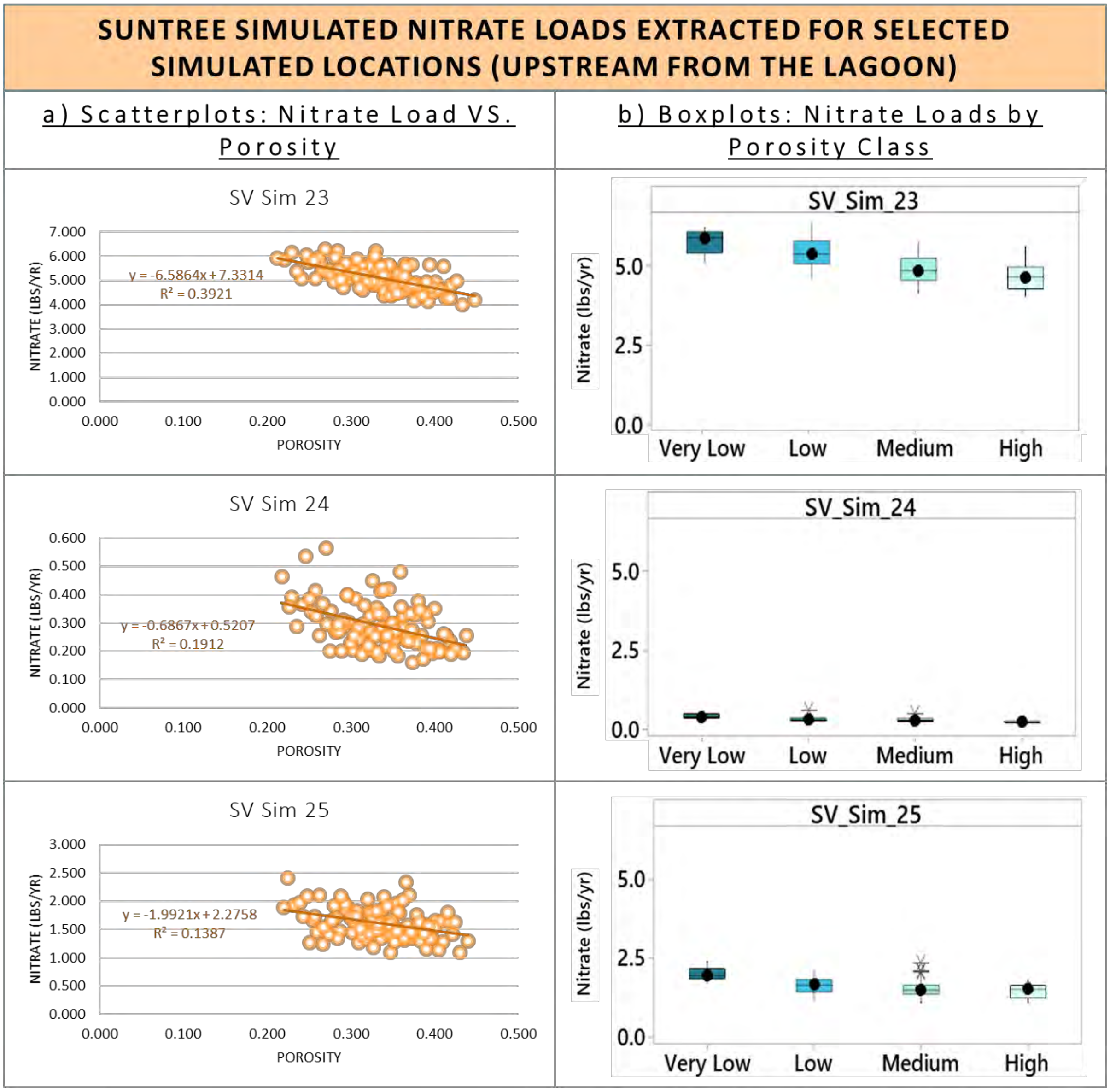


Figure 39. Resulting nitrate plume concentrations at selected locations more distance from the Lagoon within Melbourne Beach based on changes in input porosity via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. porosity and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input porosity classes.

2.2.3.3 SMOOTHING FACTOR

The smoothing factor controls the number of smoothing iterations that are performed on the digital elevation model (DEM) to generate the water table as a subdued replica of the topography (Rios, Ye, Wang, and Lee, 2011). Higher numbers indicate increased smoothing, resulting in decreased elevation gradients or a flatter replica; however, if the values are too high, it can shift peaks in elevations that are very different from the original values. In contrast, if the smoothing factors are too low, an unrealistic flow path of nutrients may result. The optimum value may be determined by comparing the smoothed DEM with hydraulic head observations. This is typically performed using locally collected groundwater water level data. Sometimes, the topography of a particular region has been greatly altered by anthropogenic alterations during the past decades, particularly with the use of dredge and fill traditional methods of residential development. This is particularly true in areas where canals were dredged for navigation and recreational access and the fill placed on-site to increase the elevation of the buildable lots. In these types of cases, the water table is likely not a close replica of the digital terrain model.

2.2.3.3.1 MELBOURNE BEACH

The input smoothing factor values used in the MC Simulations for this area varied from 10.47 to 199.29 , and predicted total loading outputs varied from 11.00 and 14.00 lbs./yr (Figure 40). Both the range and the maximum total output nitrate load from varying smoothing factors were relatively low to those predicted using the MC Simulations for hydraulic conductivity and soil porosity in the same area.

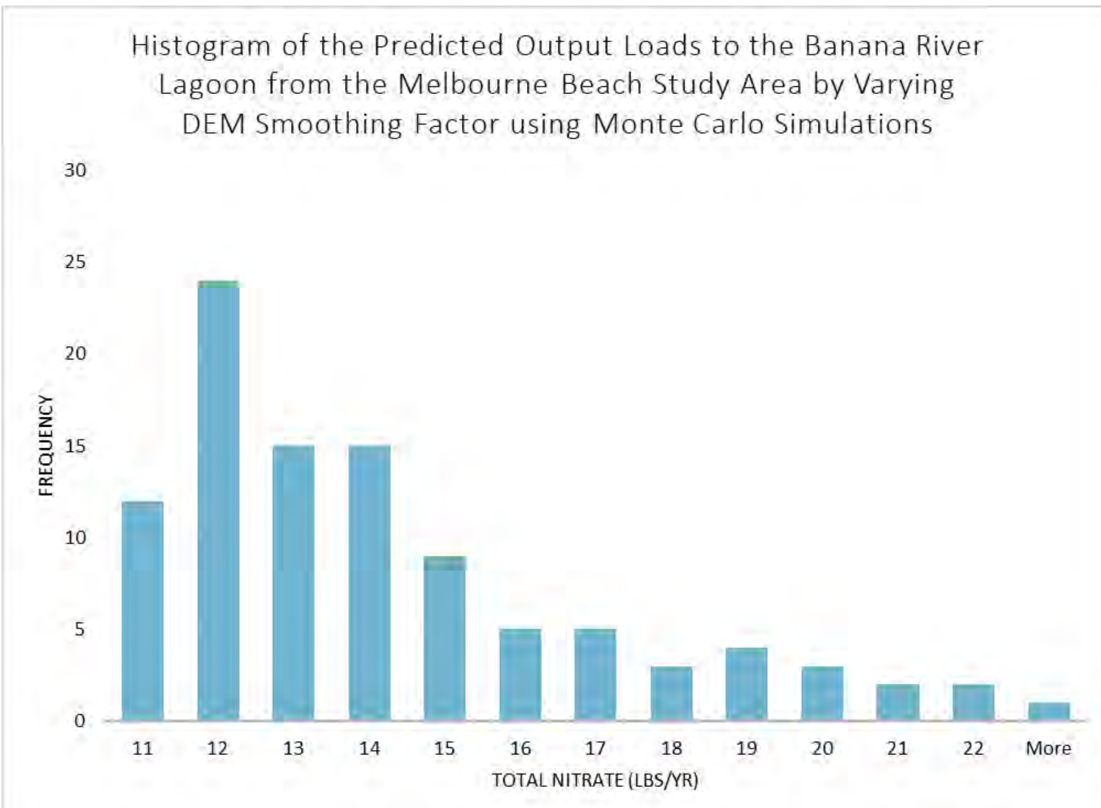


Figure 40. Histogram of the predicted output loads to the Banana River Lagoon from the Melbourne study area based on Monte Carlo simulations of smoothing factor.

While increases in smoothing factor typical appear to drive some decreases in predicted nitrate outputs, the relationship does not appear to be linear throughout the range of simulated smoothing factors and the slope is overall almost negligible. In fact, at both SP 250 and SP 275, for smoothing factors between 0 and 25, a reverse relationship appears to be visible, with increases in smoothing factor leading to increases in nitrate loads (Figure 41). For the remaining well and at smoothing factors >25, an inverse relationship between smoothing factor and nitrate loading is apparent. Overall, the correlation coefficients for this parameter are lower than those for both hydraulic conductance and porosity within the Melbourne Beach study area and slopes almost negligible. MW SP 270 and MW SP 275 had similar correlation coefficients (R^2 0.67 and 0.66, respectively), while MW SP 250 has the lowest correlation coefficient (R^2 = 0.34). Once again, the output loads are higher at SP 270 than at both SP 250 and SP 275, likely related to the individual placement of these wells to the water and septic drain fields.

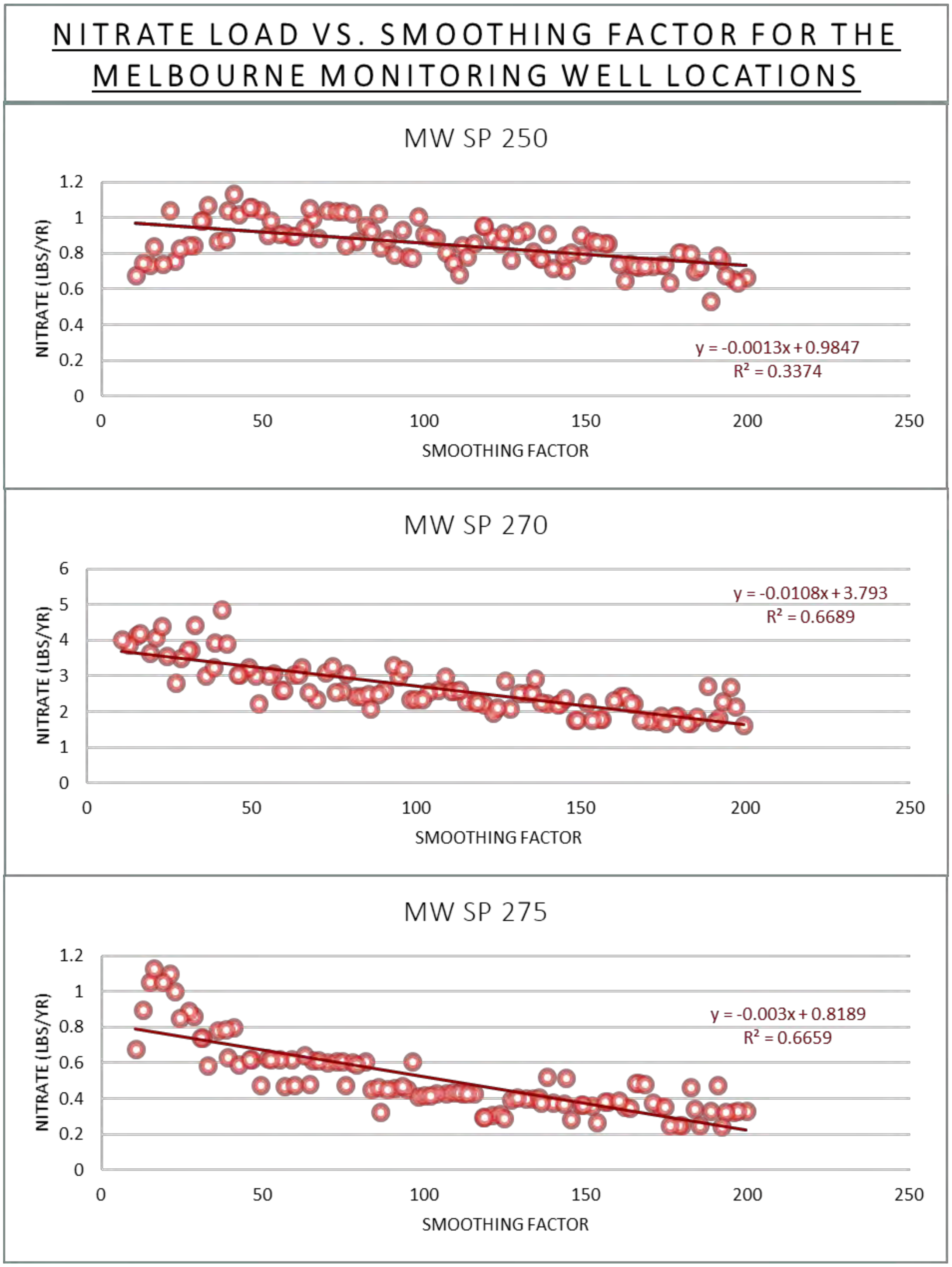
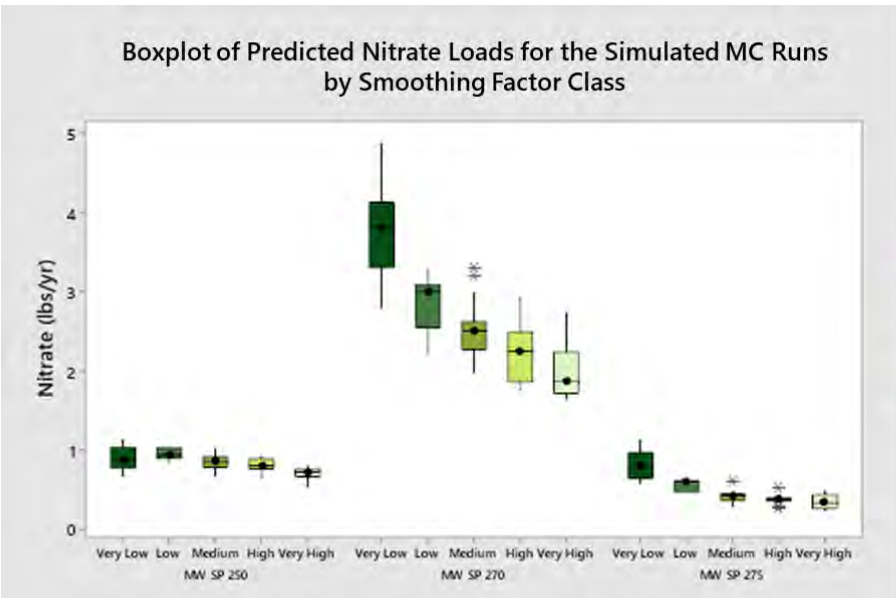


Figure 41. Scatterplots of nitrate loads (lbs./yr) vs. smoothing factor at the monitoring well locations within the Melbourne Beach study area.

This complex relationship between smoothing factor and predicted nitrate loads is more clearly demonstrated in Figure 42. While the data for SP 270 and SP 275 clearly show a unidirectional inverse relationship between smoothing factor and nitrate outputs, the same relationship is not apparent for SP 250, where medians and 25-75th percentile distribution large overlap for most smoothing factor classes. These differences indicate that in some cases, smoothing factor is likely a driver up to a certain range (likely near the optimum value to produce a water table) with little impact into resulting nitrate loads beyond that range.



Smoothing Factor Class	Smoothing Factor Value
Very Low	≤ 46.5915
Low	> 46.59154 - 84.77332
Medium	> 84.77332 - 124.23960
High	> 124.23960 - 162.35690
Very High	≥ 162.35690

Figure 42. Boxplot of nitrate loads by smoothing factor class (defined by the gradient table to the right) at monitoring well locations within the Melbourne Beach study area that resulted from varying input smoothing factor during the MC simulation.

Similar complex relationships are visible when extracting simulated nitrate loading values to a variety of simulated well locations, some close to the monitoring well locations (Figure 43) and further away from the monitoring wells and Lagoon (Figure 44). Distance from the Lagoon appear to not impact the relationship between smoothing factor and nitrate, with very variable correlation coefficients for all simulated locations regardless of location (R^2 ranging from 0.0374 to 0.8751). The trend of increasing nitrate load estimates at lower smoothing factor values is persistent and appears to be almost an exponential rather than linear curve at simulated septic points with lower R^2 values. Furthermore, the very low classes of smoothing factor present the most variable nitrate outputs from all smoothing classes,

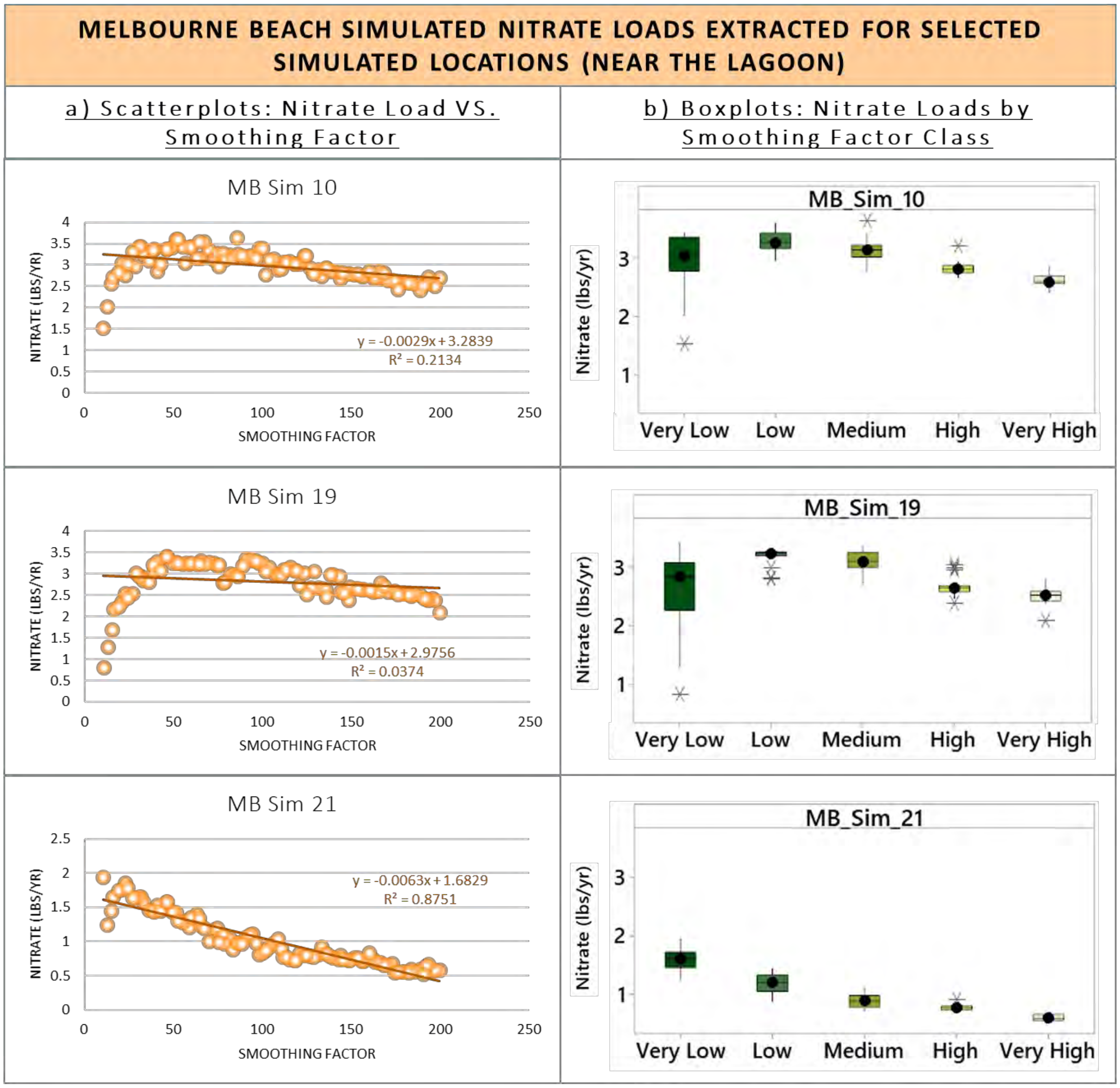


Figure 43. Resulting nitrate plume concentrations at selected locations near monitoring wells within Melbourne Beach based on changes in smoothing factor via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. smoothing factor and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input smoothing factor classes.

MELBOURNE BEACH SIMULATED NITRATE LOADS EXTRACTED FOR SELECTED SIMULATED LOCATIONS (UPSTREAM FROM THE LAGOON)

a) Scatterplots: Nitrate Load VS. Smoothing Factor	b) Boxplots: Nitrate Loads by Smoothing Factor Class

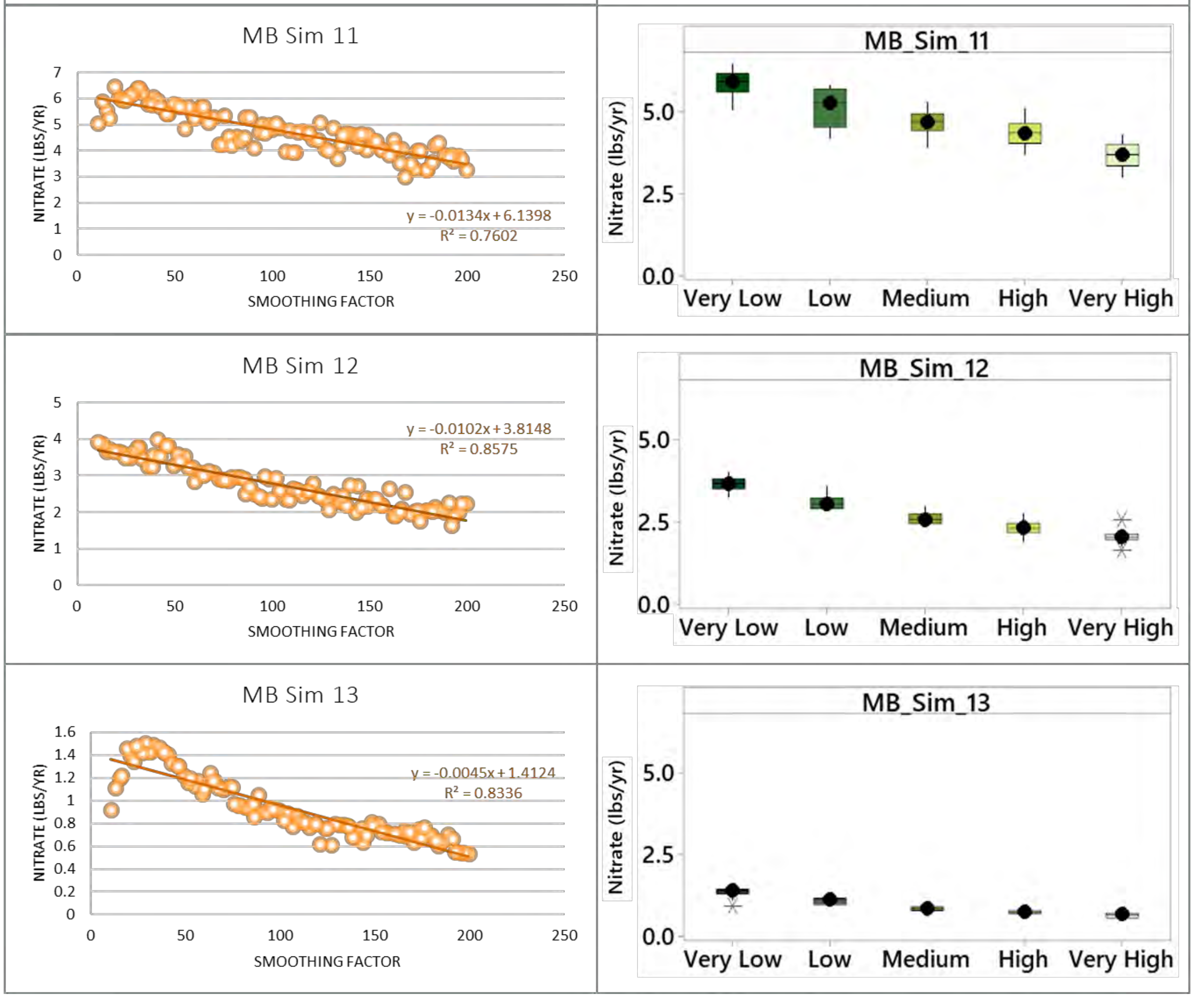


Figure 44. Resulting nitrate plume concentrations at selected locations upstream from the monitoring wells within Melbourne Beach based on changes in smoothing factor via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. smoothing factor and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input smoothing factor classes.

2.2.3.3.2 SUNTREE

Very similar input smoothing factors were used to simulate nitrate plumes in the Suntree area (10.45 to 199.46), with total estimated nitrate loads for this region ranging from 39.51 to 43.47 lbs./yr, three times higher than the Melbourne Beach total outputs. (Figure 45). Almost 50% of the simulated plumes total nitrate loads were between 43.00 and 43.50 lbs./yr.

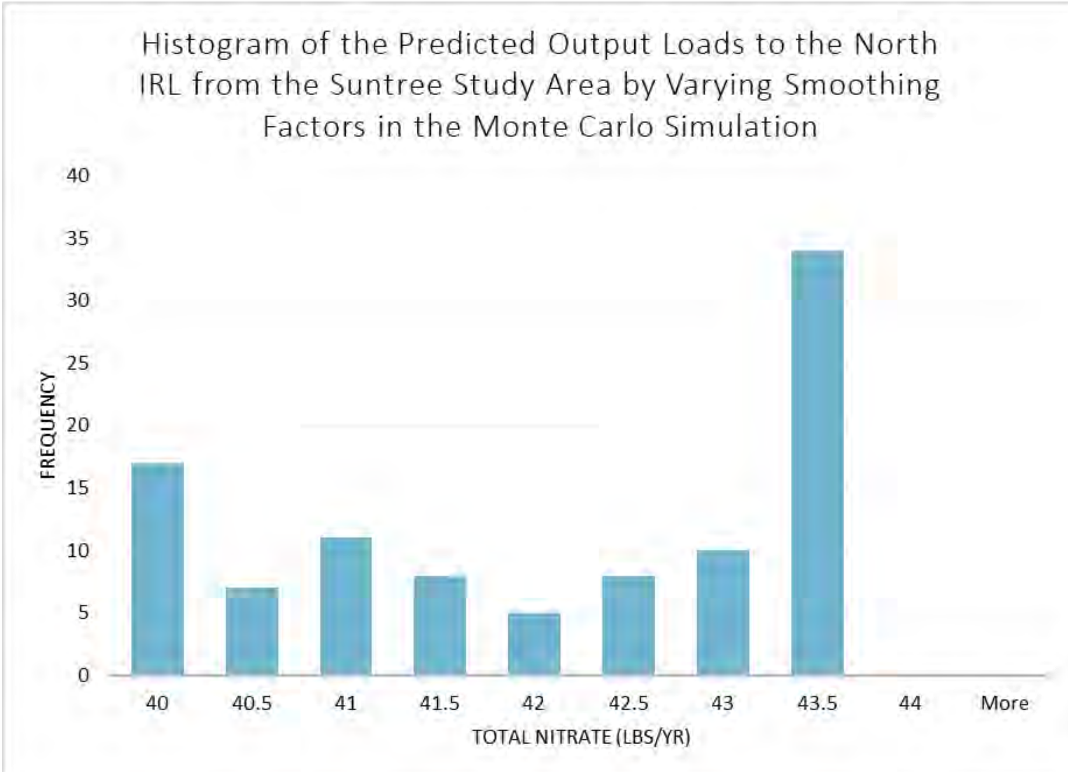


Figure 45. Histogram of the predicted output loads to the North IRL from the Suntree study area based on Monte Carlo simulations of smoothing factors.

Interestingly, unlike in Melbourne Beach, smoothing factor increases appear to have a unidirectional positive increase in predicted nitrate loads at the three monitoring well locations (Figure 46). The correlation coefficients were highly variable, with some well locations having relatively low R^2 values (0.24 and 0.37 for SP 6155 and SP 6398, respectively) and SP 6915 a very high R^2 (0.95). Magnitudes of predicted nitrate loads were similar for values extracted at both SP 6155 and SP 6215 locations; predicted nitrate loads were a couple orders of magnitude higher at or near the SP 6398 location.

The same relationship between the input smoothing variable and predicted nitrate load can be further confirmed in Figure 47. While the magnitude of loading differences between well locations masks some of the proportional increases in nitrate outputs with increases in smoothing factor, it is obvious that the relationship is slightly different for SP 6398. At this location, extracted plume nitrate loads appear to have slope changes with changes in ranges of smoothing factors: steep slope with smoothing factor from 0-30, linear shallow slope increases from 30-120 smoothing factor, and stable or even decrease (0 to negative slope) at the highest smoothing factors.

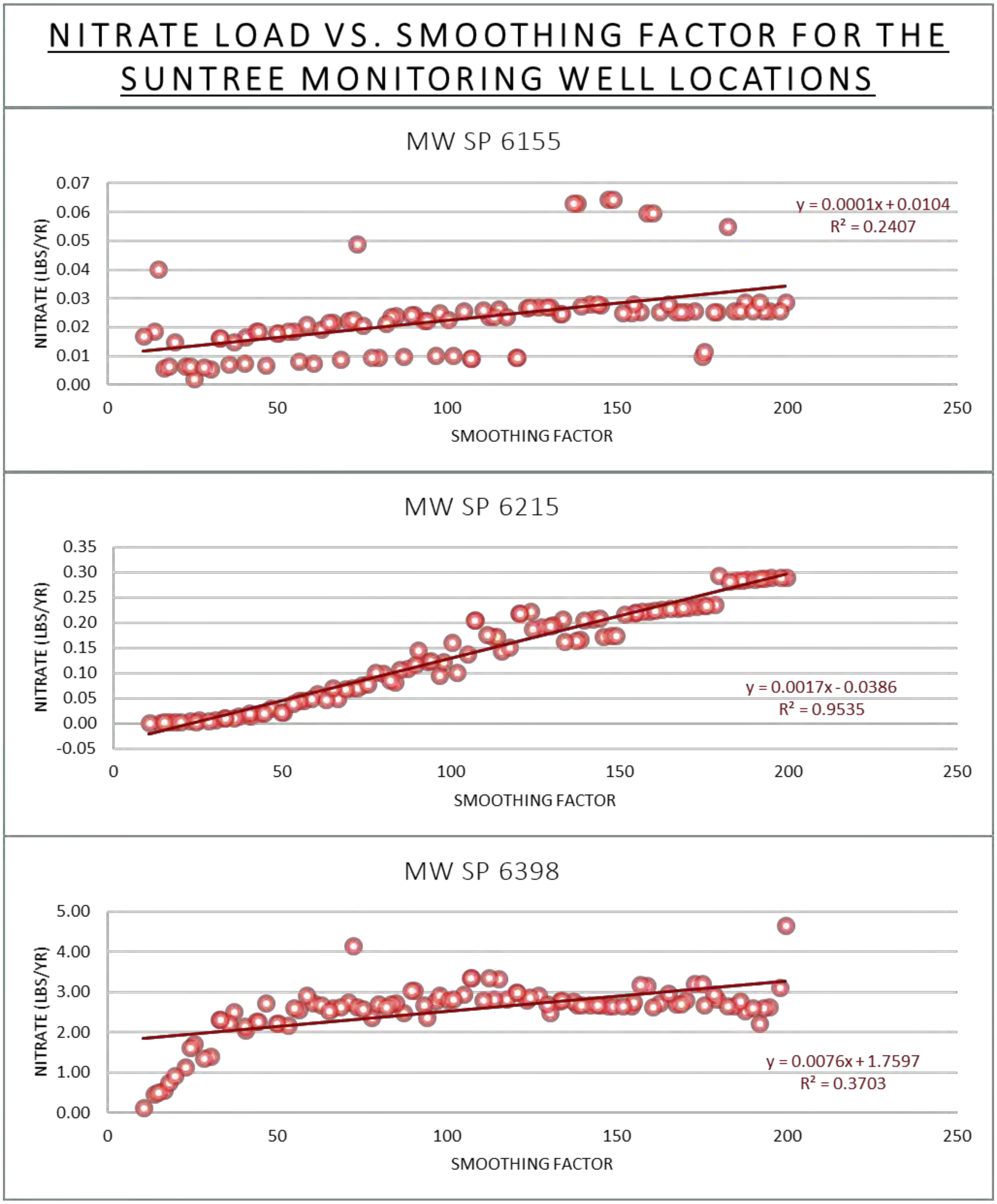
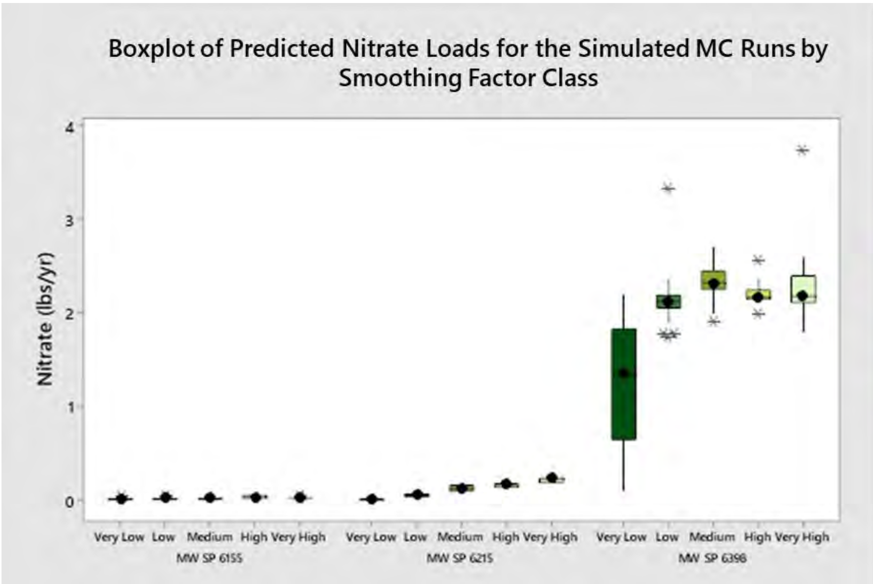


Figure 46. Scatterplots of nitrate loads (lbs./yr) vs. smoothing factor at the monitoring well locations within the Suntree area.



Smoothing Factor Class	Smoothing Factor Value
Very Low	≤ 46.5915
Low	> 46.59154 - 84.77332
Medium	> 84.77332 - 124.23960
High	> 124.23960 - 162.35690
Very High	≥ 162.35690

Figure 47. Boxplot of nitrate loads by smoothing factor class (defined by the gradient table to the right) at monitoring well locations within the Suntree study area that resulted from varying input smoothing factor during the MC simulation.

Examining additional plots of data extracted throughout the Suntree community at several locations, it clearly demonstrated that relationships between smoothing factors and predicted nitrate loads are very site-specific. While those extracted for location near the monitoring wells and close to Lagoon waters (Figure 48) present increasing relationships for most smoothing ranges, the relationships are contrastingly different for locations upstream from the Lagoon (Figure 49). Unlike the previous factors, hydraulic conductivity and porosity, smoothing factor can have an impact on the nitrate loads at different slopes and even directions for different ranges. Often, at the lower end ranges (0-30) the directionality of the relationship is the opposite of the one from the one described for the medium to high smoothing factor ranges. The slope is also steeper or even better described as an exponential relationship at the lower ranges of the smoothing factor than at the medium to high range. This impacts the overall regression coefficients which vary, from location to location, from very low (R^2= 0.20) to high (R^2= 0.885).

This indicates the importance of selecting an appropriate smoothing factor that better approximates the modeled water table to the measures hydraulic head. Small changes in smoothing factor, likely when close to the optimal factor, might lead to significant impacts in the prediction of nitrate loads via the fate transport modeling module of ArcNLET. Unfortunately, a one-size fits all approach is used in ArcNLET when the water table replica is produced from a Digital Elevation Model, and only one smoothing factor can be used per model run.

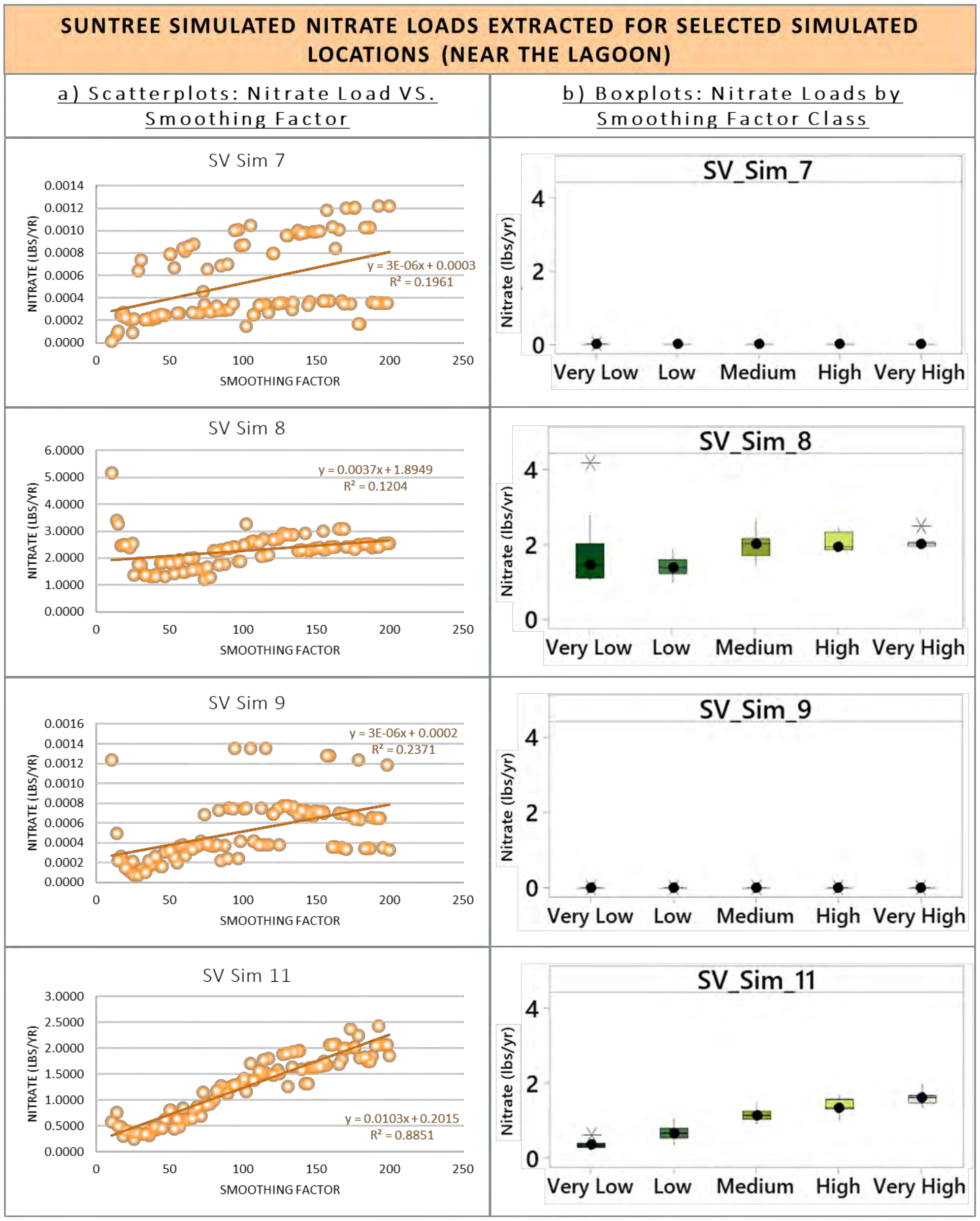


Figure 48. Resulting nitrate plume concentrations at selected locations near monitoring wells within Suntree based on changes in smoothing factor via MC Simulations. a) Scatterplots of nitrate loads (lbs./yr) vs. smoothing factor and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input smoothing factor classes.

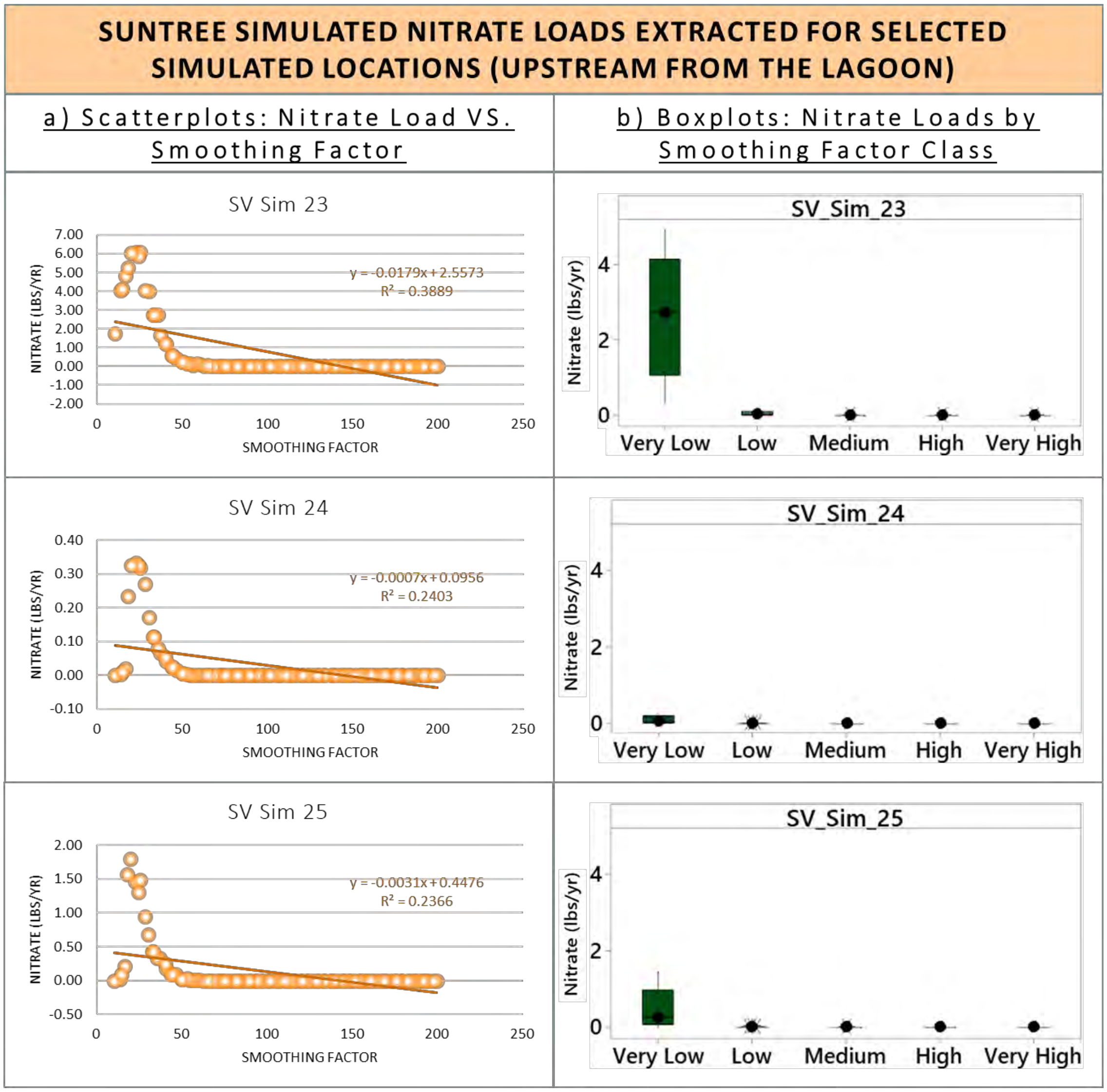


Figure 49. Results of the MC simulation when varying input smoothing factor of simulated septic points further away from the monitoring wells within the Suntree study area. a) Scatterplots of nitrate loads (lbs./yr) vs. smoothing factor and b) boxplot distributions of output nitrate loads (lbs./yr) grouped by input smoothing factor classes.

3 REFINING THE BASEFLOW COMPONENT OF THE SPATIAL WATERSHED ITERATIVE LOADING MODEL USING *IN SITU* GROUNDWATER QUALITY

3.1.1 BACKGROUND

The SWIL model was originally developed as part of a study aimed at refining and updating the TMDL set by FDEP and to address pertinent questions that arose regarding the pollutant loading and seagrass relationships in the IRL. Instead of refining the existing Pollutant Load Simulation Model (PLSM), the SWIL model was created to incorporate more available data, more recent conditions, and more temporally fine datasets.

SWIL is a custom ESRI ArcGIS toolset, originally designed to provide a continuous monthly simulation of runoff (surface and baseflows) over a 16-year period, yielding a more robust representation of pollutant loadings and freshwater volumes in the IRL. The SWIL model has been updated since the initial version was developed in 2012 (SWIL 1.0). By July 2014, SWIL 2.0 was released and focused on addressing initial FDEP comments, improving the ease of execution, and reducing the overall processing time. SWIL 3.0, released in April 2015, focused on improving model calibration to the measured available gage data, which included a change in the methodology to derive baseflow volumes and loads. SWIL 3.0 also incorporated the newly released evapotranspiration (ET) raster datasets, which were updated using the newly improved Mu, Zhao, and Running (2011) ET algorithm.

SWIL 4.0 was developed in support of the 3D Numerical Modeling effort for the IRL and Banana River led by Florida Institute of Technology and required three major changes: 1) expansion of the model extent to provide nutrient loadings from Ponce Inlet to Fort Pierce 2) temporal expansion from 2011 through August 2015, and 3) converting the model from two to three land use/treatment time steps. The most recent updates made to the SWIL model were to improve the efficiency of model run-time without compromising the validity of nutrient load estimates. In early 2018, SWIL 4.0 was further expanded spatially to cover most of the Indian River Lagoon Watershed, and temporally to span until December 2017.

The goal of the SWIL model development was to provide a GIS-based model that can be adaptive to changes in input and can batch complex processes through several months or years on demand. SWIL aims to provide both spatially and temporally fine-scale volumes and loads (TP and TN), allowing input data to be related to water quality parameters.

For incorporating the most recently collected *in situ* groundwater data, we used a recently developed grid layer comprised of 50x50-m cells as "basins" created for FDEP as an easy tool for load allocation for the Basin Management Action Plans (BMAP) of the North IRL, Central IRL, and Banana River. Background, methodology, and results from the data analysis using the original versus refined model run are discussed in the sections below.

3.1.2 MODEL INPUTS AND METHODOLOGY

Previous modeling efforts of the SWIL focused on providing loading estimated for the basins that made up the IRL watershed, as defined by St. Johns Water Management District (SJWMD) and the South Florida Water

Management District (SFWMD). During the Load Allocation Update to the SWIL for FDEP, the input basin layer was constructed of grid cells (50 x 50-m in cell size) to encompass the Banana River, North IRL, and Central IRL basins. The cell size was selected, in consultation with the FDEP, to provide high enough spatial resolution for the product to be used to quantify the loading reductions associated with retrofit projects (*i.e.*, Stormwater Best Management Practices or BMPs). Due to the limited timeframe and budget for the analysis described below, only a small subset (the mainland portion of the IR9-11-A basin) was selected to serve as the model boundary extent for comparison purposes with the following section (Figure 50). This area was chosen as it was inclusive and representative of the monitored Suntree communities.

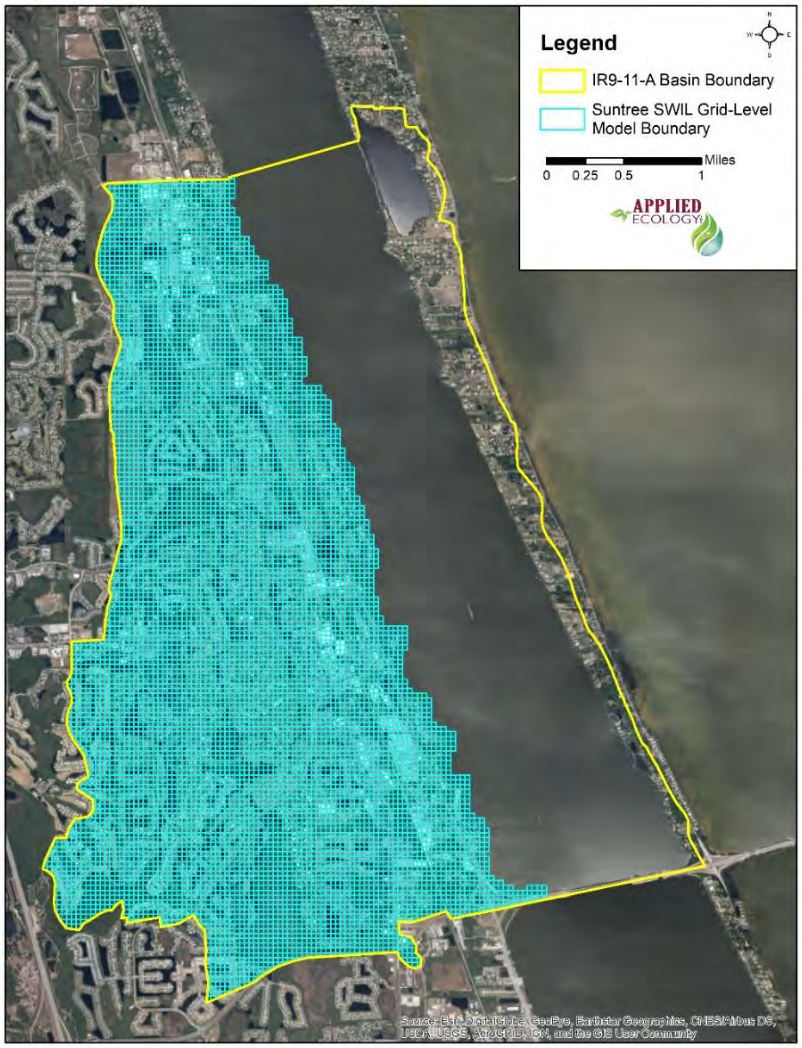


Figure 50. Grid layer composed of 50 x 50 m cell "basins" used within the Single-Year Grid-Level Allocation Run of the SWIL Model in the IR9-11-A basin.

Next, the grid-level basin layer was intersected with the Brevard County parcel layer to classify each cell as a particular treatment type (septic, sewer, reclaimed, or natural, Figure 51). This layer was checked against the latest Brevard County parcels to ensure the inclusion of recent development within the area. All roadways were classified as a treatment type (*i.e.*, if the neighborhood was mostly composed of sewer, then it was classified as sewer). Vacant lots without development were considered natural. Additionally, other land use types that are not necessarily "natural", but did not have wastewater services (*i.e.*, a recreation area without reclaimed water) were also considered natural. Reclaimed service areas were assigned to parcels using specific service lines provided by Brevard County; however, some additional classifications outside of these areas were made if the reclaimed water line and reclaimed water nodes extended into a parcel. Acres and percent of the total area are summarized by treatment type in Table 14.

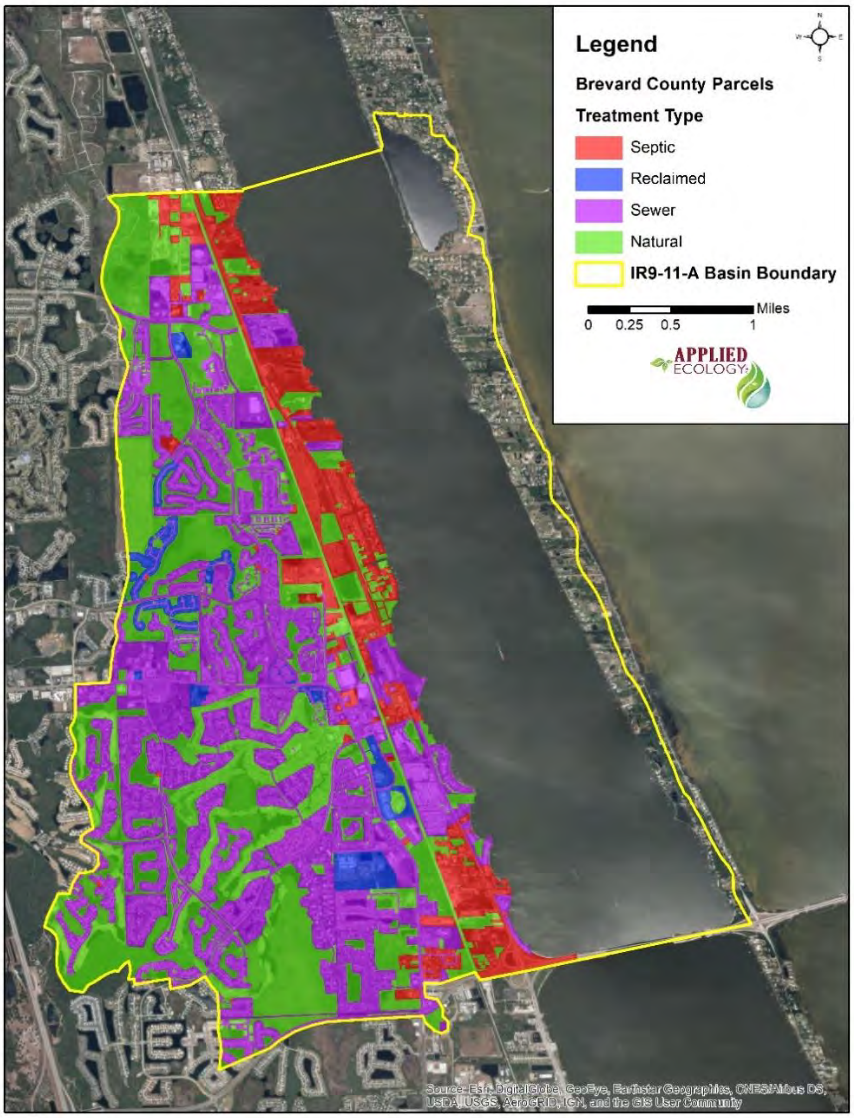


Figure 51. Brevard County parcels symbolized by treatment type within the IR9-11-A basin.

Table 14. Area in acres and percent of the total model area for each treatment type within the Mainland portion of the IR9-11-A basin.

Treatment	Area (Acres)	Percent of Total Area (%)
Natural	2,236	40.14
Septic	721	12.94
Sewer	2,428	43.59
Reclaimed	185	3.33

Other input data layers for this model version were consistent with those described in previous SWIL Methodology Manuals and included: monthly rainfall and ET data as raster inputs, land use and treatment layers for selected years, impoundment layer and associated management schedule, and soils layer. Output loads were provided as monthly nutrient loading values in addition to annual loading values. Outputs were representative of the direct runoff, baseflow, and total (direct runoff + baseflow) values of three parameters (volume, total nitrogen (TN), and total phosphorus (TP)) per basin. The monthly outputs were joined to the spatial grid cell layer, resulting in 12 layers. These were combined into a singular layer, hereafter referred to as the "Monthly Sum Grid SWIL Layer". The annual sum outputs per basin were also joined to the spatial grid layer and will herein be referred to as the "Annual Sum Grid SWIL Layer".

The SWIL model incorporates nutrient concentrations into both of its model components to account for loading from direct runoff as well as groundwater sources. Nutrient concentrations from direct runoff are based on the event mean concentrations (EMCs) for TN and TP from the BMPTRAINS 2020 recently released by UCF Stormwater Management Academy and are based on land use types. Unlike the dynamic direct runoff EMCs that are specific to land use type, only one set of TN and TP concentrations were used originally in the baseflow component of the SWIL model. Only including a homogenous groundwater concentration value for the entire Indian River Lagoon watershed was simply due to the extremely limited availability of site-specific groundwater quality data when the SWIL model was developed in 2015. The SOIRL Groundwater Study along with Brevard County's Legislative Study has provided, for the first time, water quality collected at a groundwater monitoring network of 45 wells located throughout Brevard County. Median water quality data from this network of wells was synthetized from the first 18 months of collection and used in the SWIL baseflow component of the model in lieu of the original one size fits all concentration values. Comparisons of results between the original model and refined model using recent baseflow concentration data are described in the sections below.

The Results section includes three types of results: 1) a comparison of countywide median measured TN and TP groundwater concentrations versus the original SWIL baseflow concentrations, 2) a comparison of region-specific median measured TN and TP versus the original SWIL concentrations, and 3) the simulated Suntree SWIL model results based on site-specific TN and TP measured data.

3.1.3 RESULTS

3.1.3.1 NITROGEN CONCENTRATIONS BASED ON COUNTYWIDE MONITORING

3.1.3.1.1 DIFFERENCES BETWEEN TREATMENT

Overall, the original TN concentration used for baseflow loading calculations within the SWIL model is lower than the overall median (all regions combined) TN concentration measured throughout the 18 months of sampling for all treatments types except for undeveloped or natural land uses (Figure 52). While the median concentrations observed in the sewer treatment were somewhat similar (24% higher) to the SWIL static value (0.886 mg/L), the measured septic and reclaimed treatments TN median values are 188% to 408% greater than the originally used values. As expected, the only treatment with concentrations below (by 58%) those used in the original SWIL model was the natural one.

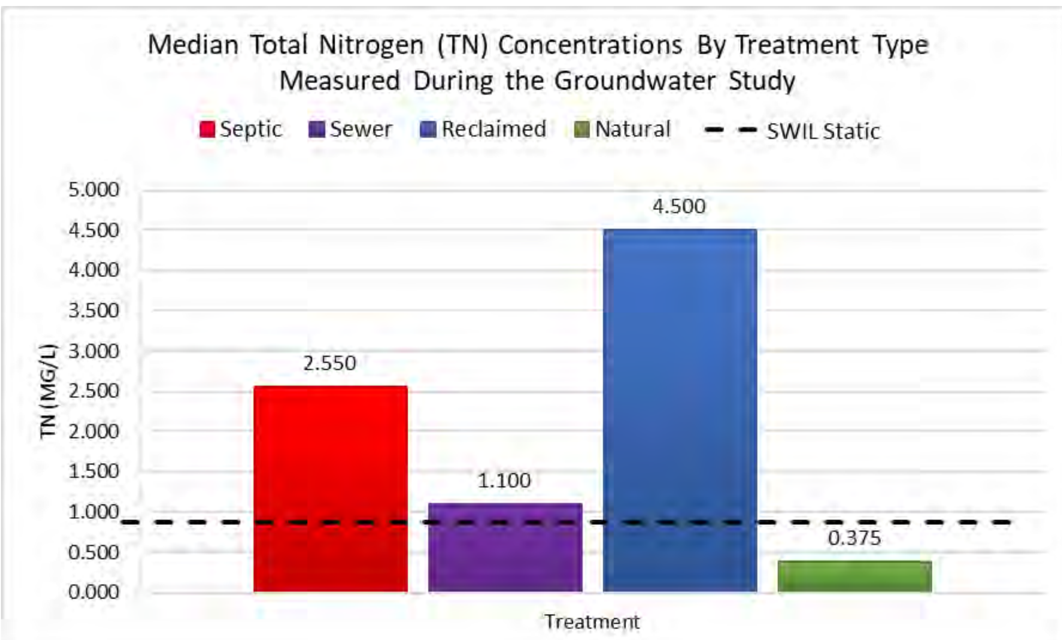


Figure 52. Comparison of the baseline groundwater value used in the SWIL model to the overall median TN concentration from the measured groundwater data of all regions by treatment type.

3.1.3.1.2 DIFFERENCES BETWEEN TREATMENTS BY COMMUNITY

A comparison of treatment types within the various sampling communities shows a majority of the treatments are exhibiting measured median values that exceeded the static TN concentration used by the SWIL model (Figure 53). Regardless of community, all of the natural treatment types fell well below the static SWIL concentration. Additionally, there were two sewer communities that fell below the 0.886 mg/L SWIL static value (Titusville and Suntree). The reclaimed treatment in Titusville is the only treatment with a median concentration slightly higher than the static SWIL value. Medians for the septic and sewer treatment types within the Merritt Island Community were almost identical, and once again higher than the SWIL static value. While the septic and reclaimed treatments were above the static value in the Suntree community, the septic treatment median was higher than the reclaimed, contrasting with the trend observed in other communities. The trends of the Turkey Creek and Melbourne and Satellite Beach are identical to that of the overall median TN concentrations for all regions combined; however, the median TN concentrations at the Turkey Creek community were drastically higher than those of Melbourne and Satellite Beach, with a substantial difference f the septic and reclaimed treatments.

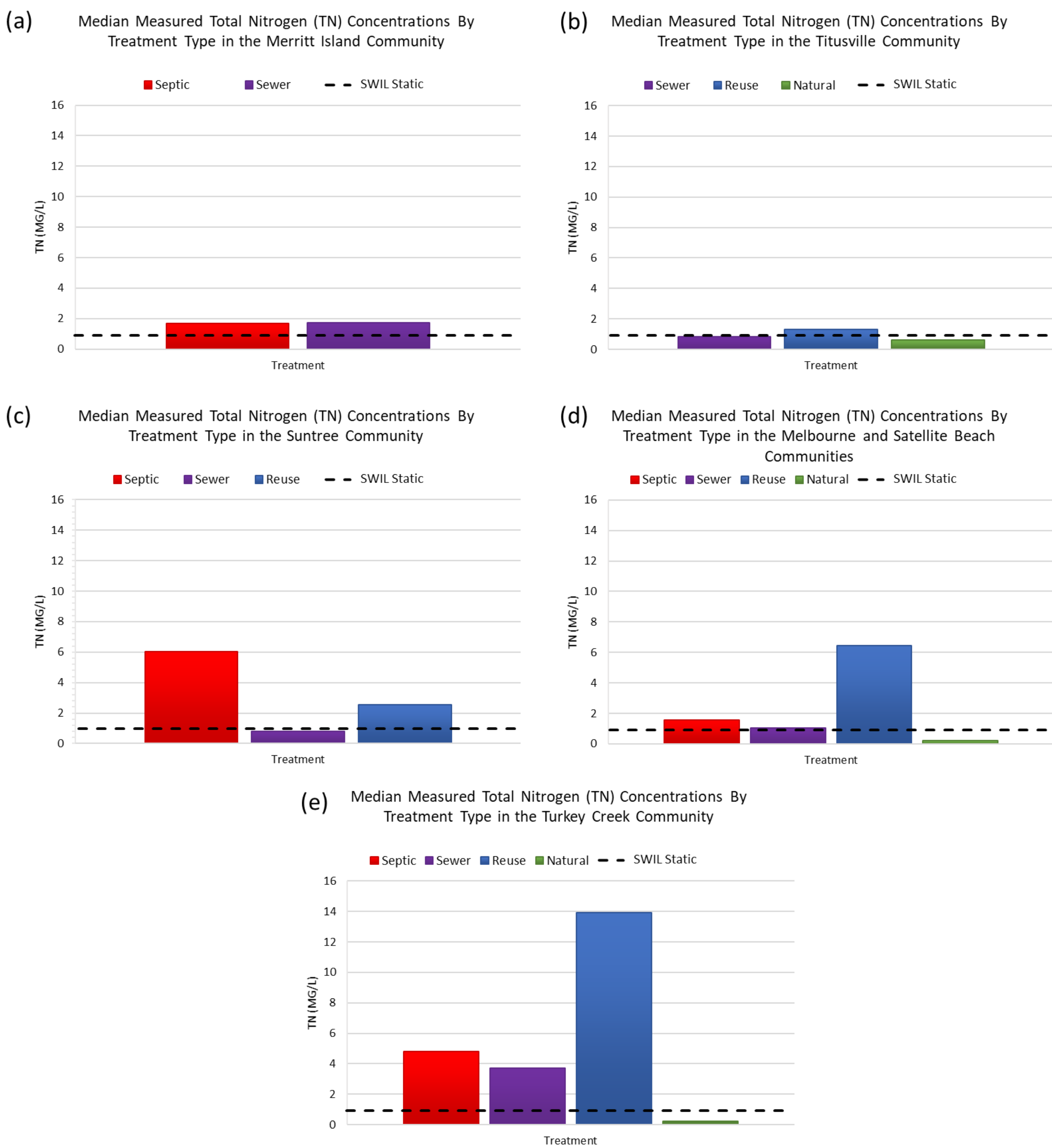


Figure 53. Comparison of the median measured TN concentrations by treatment type against the static SWIL concentration in the (a) Merritt Island, (b) Titusville, (c) Suntree, (d) Melbourne and Satellite Beach, and (e) Turkey Creek Communities.

3.1.3.2 PHOSPHORUS CONCENTRATIONS

3.1.3.2.1 DIFFERENCES BETWEEN TREATMENTS

Patterns in median measured concentrations of TP were not consistent with those observed with TN (Figure 54). While a drastic difference between the septic treatment (0.60 mg/L) and the original model TP value (0.112 mg/L) persisted (difference of 0.48 mg/L or 132%), the remaining treatment types demonstrated values slightly below the median TP concentrations. The sewer and natural treatments have median TP values very similar to the original static value, with respective differences of 0.012 mg/L (10%) and 0.009 mg/L (8%), and the reclaimed treatment had the largest difference of 0.04 mg/L (or 39%).

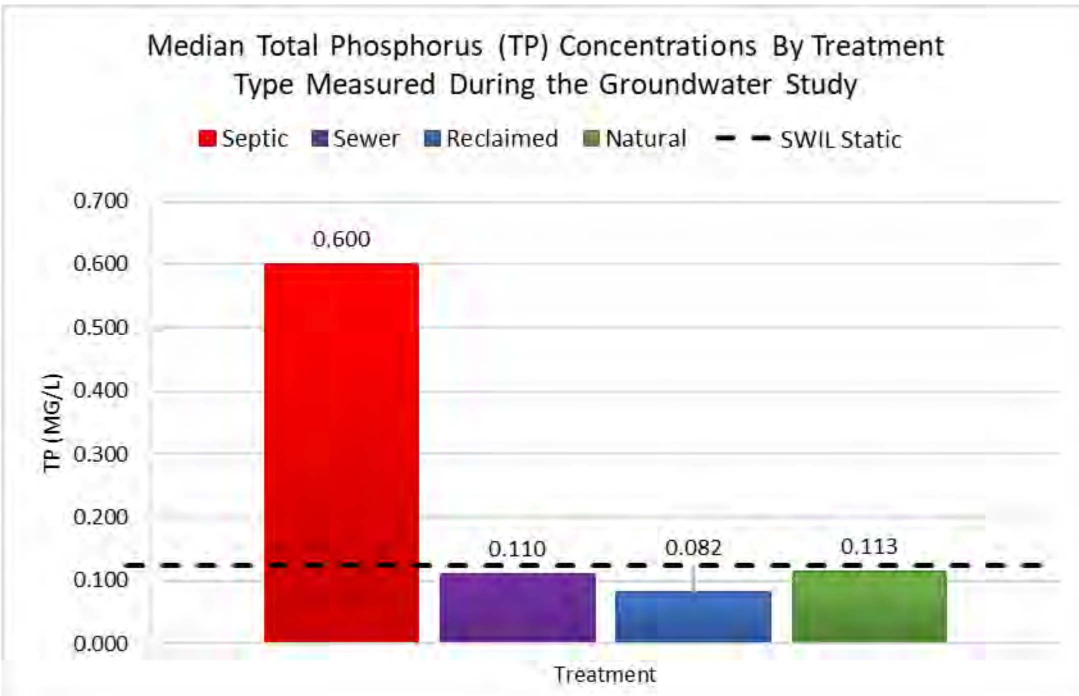


Figure 54. Comparison of the baseline groundwater value used in the SWIL model to the median TP concentration from the measured groundwater data.

3.1.3.2.2 DIFFERENCES BETWEEN TREATMENTS BY COMMUNITY

Comparisons between treatment types demonstrate inconsistent patterns of median TP concentrations between the various sampling communities (Figure 55), with exception of those for all the septic communities. The septic treatment was the only type to consistently exceed the original SWIL concentration, regardless of community, often by several times; concentrations were highest within the Turkey Creek community, while the Suntree, Merritt Island, and Melbourne and Satellite Beach communities were all fairly similar. Medians across all treatments were almost identical in Titusville, with TP concentrations slightly lower in the natural area, and less than the static SWIL concentration. Similarly, the sewer and reclaimed treatments were below the static value in the Suntree community, with the reclaimed treatment having a higher median value than the sewer.

Both treatment types of the Merritt Island Community were above the static value, with larger differences observed in the septic treatment. Melbourne and Satellite Beach is the only area in which the reclaimed treatment has the highest median TP concentrations. Additionally, all but the sewer treatment exceeded the static TP SWIL concentration in this community and were almost identical to that of the overall median TN concentrations. Turkey Creek has the highest measured median TP valued for both the septic and sewer communities, with values several times above the original values used in SWIL for the TP.

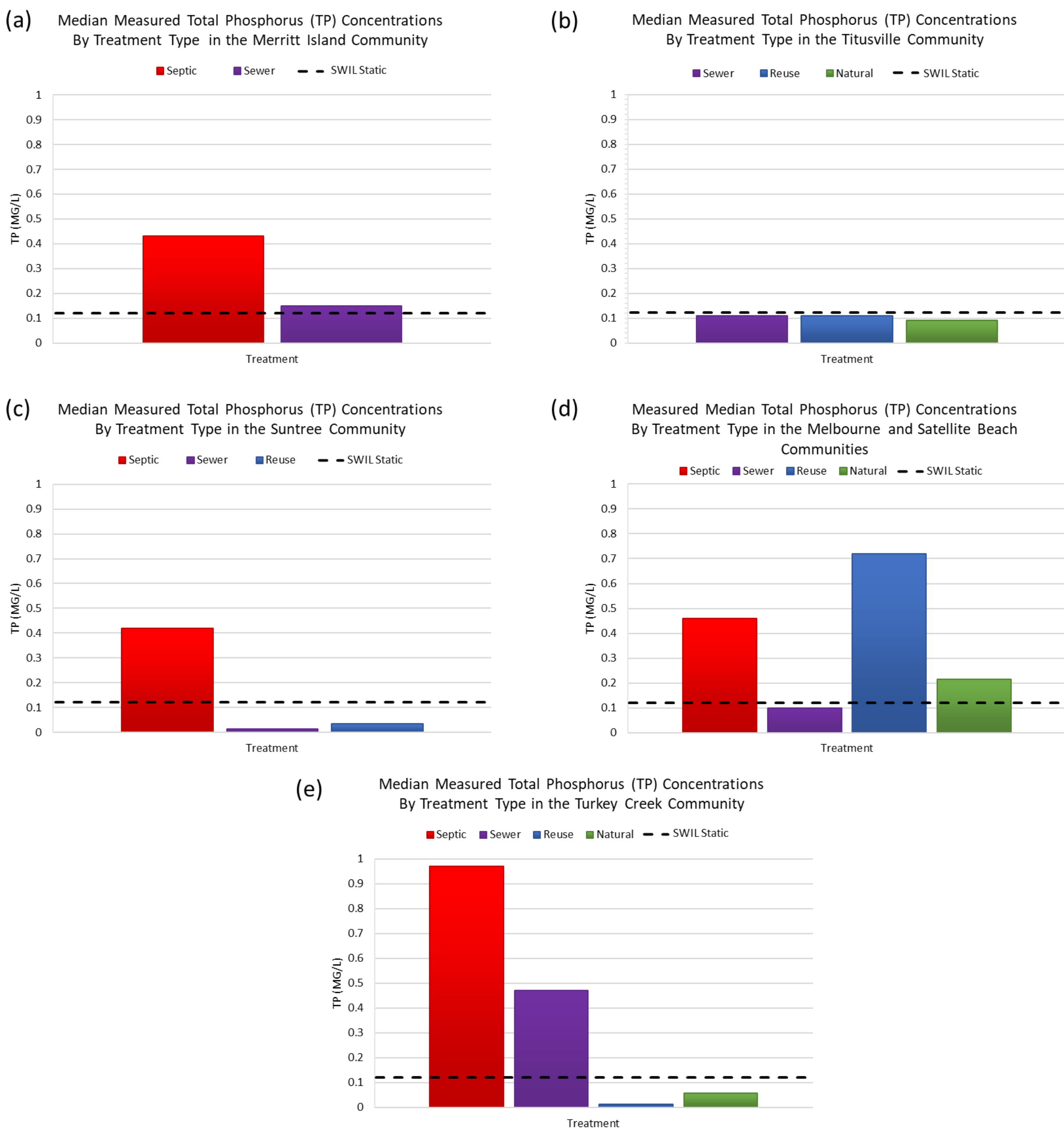


Figure 55. Comparison of the median measured TP concentrations by treatment type against the static SWIL concentration in the (a) Merritt Island, (b) Titusville, (c) Suntree, (d) Melbourne and Satellite Beach, and (e) Turkey Creek Communities.

3.1.3.3 SUBSTITUTION OF SWIL STATIC GROUNDWATER CONCENTRATIONS

While the SWIL model is a great tool to estimate nutrient loads into the IRL, the groundwater concentration values used for the baseflow loading calculations within the model are homogenous and not land use dependent. Even while developing the SWIL baseflow module, it was obvious that some areas within the watershed might have greater nutrient contributions via baseflow than others, based on age of infrastructure, type of sewer (OSTDS or centralized), reclaimed availability and WWTP concentration values, landscape management, soil type, among others. Furthermore, the original groundwater concentration values used in SWIL were also static and did not incorporate seasonal changes as these concentrations were based on a very limited number of sampling events. Luckily, the *in-situ* data collected in this study provides monthly TN and TP concentrations that can be substituted within each monthly output of the grid-level Load Allocation SWIL model.

After comparing the nutrient concentrations measured during the groundwater study to the static concentration used in the baseflow load calculation of the SWIL, differences in overall median values and spatial representations became apparent. As a result, it was decided to perform an experimental analysis to replace the original static uniform TN and TP values by the *in situ* monthly measured medians into the most recent version of the SWIL model. Due to the limited timeframe, only a subsection of IRL basin IR9-11-A was used as representative of the monitored Suntree communities.

The parcel layer was intersected with both Monthly Sum Grid SWIL Layer and Annual Sum Grid SWIL Layer in order to associate the nutrient loads with each parcel. The monthly median measured concentrations of TN and TP were assigned to each parcel by treatment type, then multiplied by the baseflow volume to calculate baseflow loadings of each nutrient within the Monthly Sum Grid SWIL Layer. For future modeling application efforts, it will be important to quantify the potential attenuation that could be ocurring between the measured groundwater concentration data and the Lagoon. It should be noted that there is no natural control area within the Suntree community; thus, the overall median values of natural treatments across all communities were used. Summary statistics were performed to determine the total baseflow TN and TP load for each treatment type.

Comparisons of baseflow (groundwater) nutrient loads estimated by the original SWIL model versus the *in situ* monthly median concentrations were performed for each treatment type within the IR9-11-A watershed basin. Additional analyses were performed to investigate differences in overall nutrient loads resulting from the replacement of groundwater concentrations.

3.1.3.3.1 Comparison of SWIL Static EMCs to Measured Concentrations in Mainland IR9-11-A Basin

Overall, using a single value for groundwater concentration in the SWIL model underestimates the nutrient loadings from baseflow. When comparing the SWIL outputs for baseflow nutrient loadings, there was an overall increase of 84% (22,016 lbs./yr) for TN and an increase of 13% (458 lbs./yr) for TP (Figure 56). It is important to note, that even this more refined method to assign groundwater loading based on treatment type, region, and temporal variability can further be improved with additional data, including a better understanding of soil type, denitrification coefficients, and the impact of rainfall events on the nutrient concentrations.

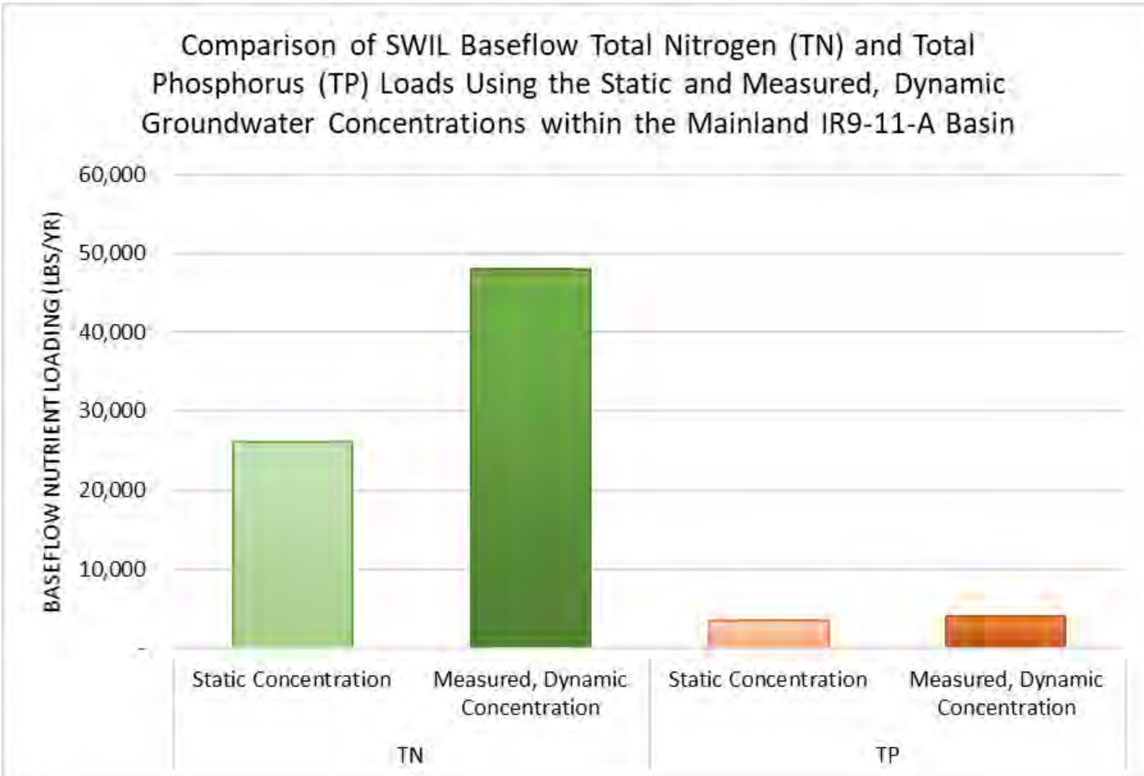


Figure 56. Comparison of SWIL Baseflow Nutrient Loads using the static groundwater concentration incorporated into the model and the measured monthly concentrations within the Mainland IR9-11-A basin.

3.1.3.3.2 COMPARISON OF TN BASEFLOW LOADS BY TREATMENT

The most noticeable shift in contribution was within the septic treatment type (Figure 57), which went from contributing 3,885 lbs./yr to 26,038 lbs./yr to the North Indian River Lagoon, becoming the most important contributor of TN from the entire modeled watershed. This is informative, not only because of the considerable increase in quantitative output, but also because the septic treatment only accounts for 13% of the total modeled area, indicating that the majority of TN loading for this region comes from one of the smallest areas located directly along the IRL shoreline.

There was also a noticeable increase in the estimated contribution of reclaimed treatment to the total loading (an additional 2,054 lbs./yr of TN). Although this increase is not as drastic as that observed for the septic treatment, it does lead to an increase of 206% from the original SWIL model predicted baseflow TN. A decrease of 61% (5,172 lbs./yr) was predicted for the natural areas, which was expected, as there are reduced sources of nitrogen loading in these areas (most in organic form). Differences between the original and refined model outputs for the sewer areas are negligible (~2%).

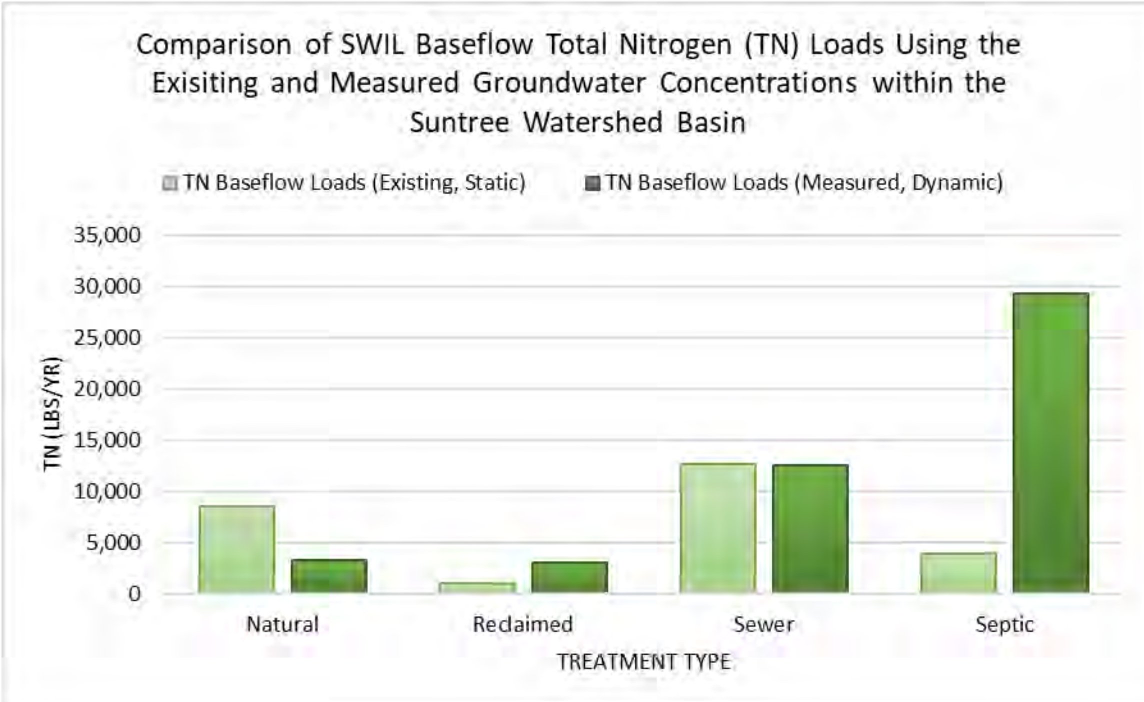


Figure 57. Estimated TN loads contributed by baseflow according to two SWIL model versions within the Suntree watershed basin.

3.1.3.3.3 Comparison TP Baseflow Loads by Treatment

As with TN, the most noticeable shift in contribution was within the septic treatment type (Figure 58), which went from being the third-highest contributor of TP loads (with the original SWIL model) to the highest contributor after refinement with *in situ* data (536 lbs./yr to 2,287 lbs./yr, or 326% increase). There was a substantial decrease in predicted TP loads for the sewer treatment of almost 1,055 lbs./yr (60% decrease). Once again, decreases in estimated TP loads were predicted for the natural areas and reclaimed treatment, but these were relatively small (143 and 97 lbs./yr, respectively).

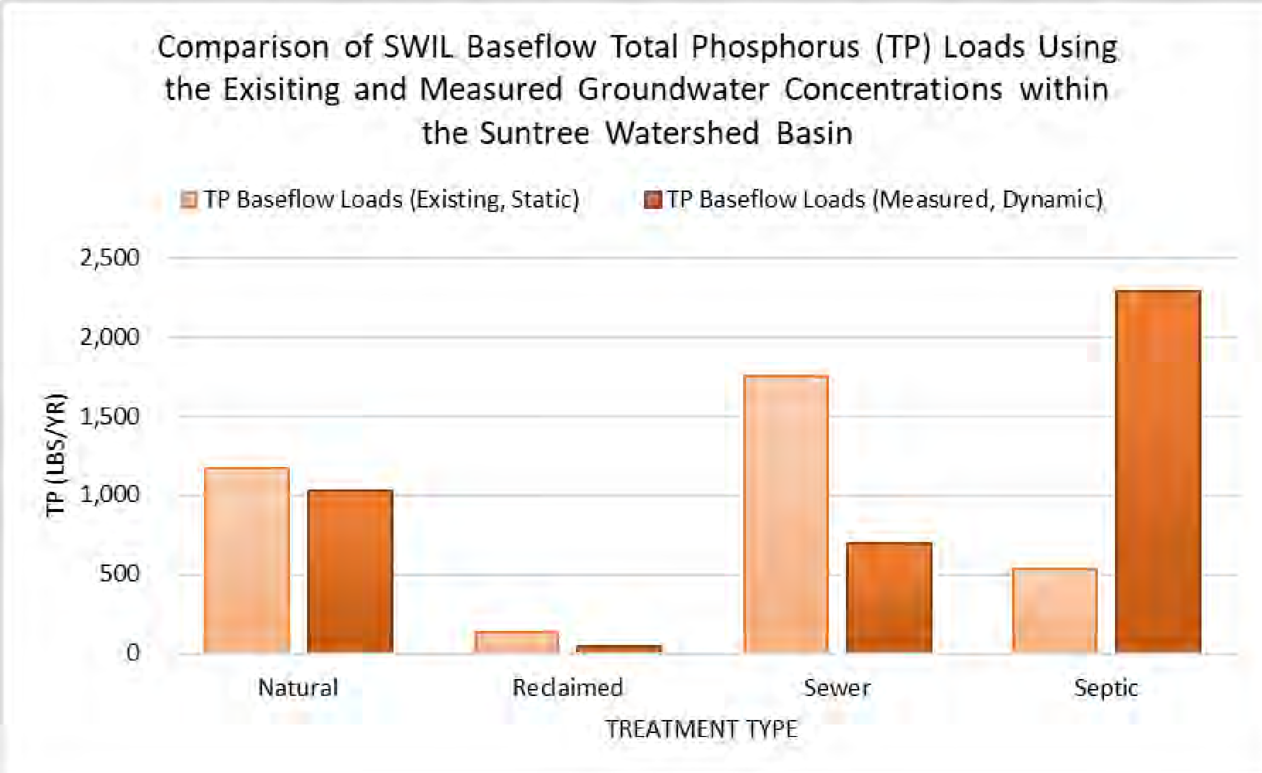


Figure 58. Estimated TP loads contributed by baseflow according to two SWIL model versions within the Suntree watershed basin.

4 CONCLUSIONS AND RECOMMENDATIONS

The extensive modeling efforts performed for this Groundwater Modeling Memorandum allowed several conclusions to be drawn. Some of these might be important for future data acquisition, analysis, and interpretation of the ArcNLET model for management of priority projects to restore the Lagoon. The conclusions are related to the three components of this memorandum report: the ArcNLET calibration effort, the uncertainty modelling, and the refinement of the Spatial Watershed Iterative Loading (SWIL) Model.

ArcNLET:

- ArcNLET severely underestimates nitrogen loading potentials contributed by groundwater sources if not adequately calibrated with measured concentration data.
- Model calibration improved the accuracy of groundwater flow, direction, and plume intensity
- Even with calibration, ArcNLET appears to on the lower side of many modeling efforts, which typically range between 4-19/g/day/septic, likely indicating the use of a high denitrification coefficient, which should be further confirmed
- To better dissect the factors that could be leading to underestimation, the following data should be collected:
    - Input nitrate and ammonia concentration data from septic tanks based on water usage information
    - Transect based groundwater quality with seepage information to follow nitrate and ammonia transport to the receiving waterbody (i.e. Lagoon)
    - Soil hydraulic conductivity values for representative soil types that make up the Lagoon's watershed
    - Long-term groundwater quality data for nitrogen constituents
    - Calibration of the denitrification coefficients used in the model based on laboratory efforts using field collected groundwater samples

- Uncertainty ArcNLET Monte Carlo Simulations:
    - Hydraulic conductivity is a key driver of nitrate transport from septic tanks into receiving water bodies according to the ArcNLET model
    - There is a significant positive linear relationship between hydraulic conductivity and nitrate loads: higher hydraulic values will typically result in higher nitrate load predictions
    - Soil porosity is inversely correlated to soil hydraulic conductivity: soils with high hydraulic conductivity usually have lower soil porosity
    - Soil porosity also an important driving factor, albeit less significant, for nitrate loading, best described by an inverse linear relationship: lower soil porosity typically results in higher predicted nitrate loadings
    - From the limited modeling effort, the relationship between hydraulic conductivity/soil porosity and nitrate loads varies spatially at several scales: regionally (beaches versus mainland) and locally (within communities)
    - Within the beach community, soil parameters (hydraulic conductivity and porosity) have a greater explanatory power in the output nitrate loadings for plumes generated closer to the Lagoon; the same relationship is not as visible for the mainland community

  - Smoothing factor presents a very different relationship to nitrate predictions than the soil parameters, often bidirectional and poorly described by any linear relationship
  - Smoothing factor appears to have most significant impact in nitrate load prediction in the lower end of the range (0-30) with often steeper slopes and a different directionality than at higher ranges
  - Calibration with field collected groundwater water level data is critical to ensure optimum smoothing factor is used, since small changes in the lower end range have yield dramatic different nitrate load outputs
- SWIL Refinement:
  - The original uniform TN concentration used for baseflow loading calculations within the SWIL model (0.886 mg/L) is lower than the overall median TN concentration measured throughout the 18 months of sampling across all developed treatment types, with the largest discrepancies in the septic (2.55 mg/L) and reclaimed (4.50 mg/L) treatments.
  - The uniform TP concentration that was used for baseflow loading calculations in the SWIL model (0.112 mg/L) were closer to those found in this study, with the exception of the septic communities, which had substantially higher TP concentrations (0.6 mg/L).
  - Using a single value for groundwater concentration in the SWIL baseflow component of the model for a subset of the watershed area (mainland area of the IR9-11-A basin) underestimates the potential nutrient loadings contributed by baseflow for both TN (by 84% or an additional 22,016 lbs./yr) and TP (by 13% or another 458 lbs./yr); this number assumes the measured concentration data are homogeneous and reach the Lagoon with minimal attenuation; attenuation data based on seepage information would greatly improve SWIL baseflow refinement.

5 REFERENCES

Amoozegar, A., & Warrick, A.W. (1986). Hydraulic Conductivity of Saturated Soils: Field Methods. Chapter 29, Methods of Soil Analysis, Prt I, 2nd ed. pp. 735-770, A.S.A. Madison, WI.

Fla. Admin. Code R. 64E-6. (2013)

Keene, R. (2015). Martin County Septic System Elimination: Final Report. Stuart, FL: CAPTEC Engineering, Inc.

Listopad, C. (2015). Spatial Watershed Iterative Loading Model Methodology Report. Indialantic, FL: Applied Ecology, Inc. for Brevard County Natural Resources.

Mu, Q., Zhao, M., & Running, S.W. (2011). Improvements to a MODIS Global Terrestrial Evapotranspiration Algorithm. Remote Sensing of Environment, 115, 1781-1800. doi:10.1016/j.rse.2011.02.019

Papadopoulos, C. E., and Yeung, H. (2001). Uncertainty estimation and Monte Carlo simulation method. Flow Measurement and Instrumentation, 12 (4), 291-298. doi: 10.1016/S0955-5986(01)00015-2.

Rios, J.F., Ye, M., Wand, L. & Lee, P. (2011). ArcNLET: An ArcGIS-Based Nitrate Load Estimation Toolkit User's Manual. Tallahassee, FL: Florida Department of Environmental Protection. Retrieved from https://people.sc.fsu.edu/~mye/ArcNLET/users_manual.pdf

Rios, J.F., Ye, M., Wang, L., Lee, P.Z., Davis, H., & Hicks, R.W. (2013). ArcNLET: A GIS-based software to simulate groundwater nitrate load from septic systems to surface water bodies. Computers and Geosciences, 52, 108-116. doi: 10.1016/j.cageo.2012.10.003.

Sayemuzzaman, M. and Ye, M. (2014). Estimation of Nitrogen Load from Septic Systems to Surface Waterbodies in East Palatka, Putnam County, Fl. Technical report submitted to Florida Department of Environmental Protection, Florida State University, Tallahassee, FL., available online at https://people.sc.fsu.edu/~mye/FDEP/EastPalatka.pdf, accessed as of 3/30/2020.

Taylor, C., Yahner, J., & Jones, D. (1997). An Evaluation of On-site Technology in Indiana: A Report to the Indiana State Department of Health. Lafayette, Indiana: Purdue University Department of Agronomy and Agricultural and Biochemical Engineering for the Indiana State Department of Health.

Wang, L., Ye, M., Rios, J.F., & Lee, P.Z. (2012). Sensitivity Analysis and Uncertainty Assessment for ArcNLET-Estimated Nitrate Load from Septic Systems to Surface Water Bodies. Tallahassee, FL: Florida Department of Environmental Protection. Retrieved from https://atmos.eoas.fsu.edu/~mye/ArcNLET/ArcNLETSensitivityUncertainty.pdf

Ye, M., Zhu, H., and Sayemuzzaman, M. (2014). Estimation of Groundwater Seepage and Nitrogen Load from Septic Systems to Lakes Marshall, Roberts, Weir, and Denham. Tallahassee, FL: Department of Scientific Computing, Florida State University.

Ye, M., Sun, H., & Hallas, K. (2017). Numerical estimation of nitrogen load from septic systems to surface water bodies in St. Lucie River and Estuary Basin, Florida. Environmental Earth Sciences, 76, 32. doi: 10.1007/s12665-016-6358-y.

Ye, M, Sun H (2013). Estimation of Nitrogen Load from Removed Septic Systems to Surface Water Bodies in the City of Port St. Lucie, the City of Stuart, and Martin County, Technical report submitted to Florida Department of Environmental Protection, Florida State University, Tallahassee, FL., available online at https://people.sc.fsu.edu/~mye/FDEP/PortStLucieModeling.pdf, accessed as of 8/12/2014.

Zhu, Y., Ye, M., Roeder, E., Hicks, R., Shi, L., & Yang, J. (2016). Estimating ammonia and nitrate load from septic systems to surface water bodies within ArcGIS environments. Journal of Hydrology, 532, 177-192. doi: 10.1016/j.jhydrol.2015.11.017.

Appendix C: Soil Raw Data

Table C-1: Soil data organized by region.

Well ID	Treatment	Region	Carbonate (%)	Organics (%)	Fines (%)
MW RE 158 10-15	Reuse	Melbourne Beach	17.9	3	3.74
MW RE 158 11	Reuse	Melbourne Beach	11.3	3.3	6.2
MW RE 158 5-10	Reuse	Melbourne Beach	16.5	4.6	2.17
MW RE 1750	Reuse	Melbourne Beach	16.5	4.6	5.14
MW RE 182	Reuse	Melbourne Beach	9.6	5.4	1.22
MW RE 182 0.5-1	Reuse	Melbourne Beach	8.2	4.6	0.75
MW RE 182 4.5	Reuse	Melbourne Beach	10.2	3.5	2.21
MW RE 182 4-2	Reuse	Melbourne Beach	10.6	2	1.94
MW RE 182 5-10	Reuse	Melbourne Beach	12.3	1.5	2.65
MW RE-158 15-20	Reuse	Melbourne Beach	11.9	3.2	10.12
MW RE-239	Reuse	Melbourne Beach	17.7	2.4	3.15
PP63	Septic	Melbourne Beach	6.25	3.78	1.30
PP64	Septic	Melbourne Beach	5.30	4.06	4.23
PP65	Septic	Melbourne Beach	6.37	4.83	4.82
PP66	Septic	Melbourne Beach	7.19	4.63	2.20
PP67	Septic	Melbourne Beach	15.00	5.59	82.30
PP68	Septic	Melbourne Beach	9.14	8.01	6.06
PP69	Septic	Melbourne Beach	5.64	4.69	7.89
PP70	Septic	Melbourne Beach	NO DATA	NO DATA	2.60
PP71	Septic	Melbourne Beach	3.99	2.56	4.50
PP72	Septic	Melbourne Beach	11.67	8.91	0.92
PP73	Septic	Melbourne Beach	6.35	4.43	3.89
PP74	Septic	Melbourne Beach	15.06	4.56	11.36
PP75	Septic	Melbourne Beach	4.63	4.31	2.19
PP76	Septic	Melbourne Beach	7.41	6.62	7.85
PP77	Septic	Melbourne Beach	15.68	10.03	6.51
PP78	Septic	Melbourne Beach	14.78	7.62	5.29
PP79	Septic	Melbourne Beach	13.46	8.80	1.73
PP80	Septic	Melbourne Beach	12.90	6.54	0.74
PP81	Septic	Melbourne Beach	4.77	4.44	3.02
PP82	Septic	Melbourne Beach	15.33	8.19	9.04

Well ID	Treatment	Region	Carbonate (%)	Organics (%)	Fines (%)
PP83	Septic	Melbourne Beach	10.46	4.02	9.26
PP84	Septic	Melbourne Beach	8.25	4.87	5.87
PP85	Septic	Melbourne Beach	7.84	4.55	5.47
PP86	Septic	Melbourne Beach	5.79	3.66	1.56
PP87	Septic	Melbourne Beach	18.84	3.27	1.86
PP88	Septic	Melbourne Beach	11.98	9.71	0.95
PP89	Septic	Melbourne Beach	8.86	7.04	2.25
PP90	Septic	Melbourne Beach	41.38	10.45	4.22
PP91	Septic	Melbourne Beach	6.23	5.80	1.81
PP92	Septic	Melbourne Beach	6.20	5.23	9.71
PP93	Septic	Melbourne Beach	5.80	4.33	0.76
PP94	Septic	Melbourne Beach	NO DATA	NO DATA	5.03
PP95	Septic	Melbourne Beach	3.90	2.25	2.02
PP96	Septic	Melbourne Beach	4.49	4.28	1.47
PP97	Septic	Melbourne Beach	5.05	4.69	1.99
PP98	Septic	Melbourne Beach	25.51	9.66	1.34
SP 250	Septic	Melbourne Beach	15.1	3.9	1.56
SP 270	Septic	Melbourne Beach	13.9	3	2.34
SP 275	Septic	Melbourne Beach	14.9	2.3	4.85
MW SE 1710	Sewer	Merritt Island	14.8	4.4	7.99
MW SE 1735	Sewer	Merritt Island	15.6	6.8	4.7
MW SP 1688	Septic	Merritt Island	12.8	3.6	7.2
MW SP 1739	Septic	Merritt Island	12.6	4.2	4.06
PP22	Septic	Merritt Island	39.69	8.38	5.35
PP23	Septic	Merritt Island	34.05	7.29	8.50
PP24	Septic	Merritt Island	26.99	8.91	7.53
PP25	Septic	Merritt Island	47.61	9.00	7.67
PP26	Septic	Merritt Island	42.67	10.38	9.94
PP27	Septic	Merritt Island	40.46	7.26	8.27
PP28	Septic	Merritt Island	53.49	8.47	12.81
PP29	Septic	Merritt Island	22.43	6.96	6.23
PP30	Septic	Merritt Island	22.72	6.71	7.14
PP31	Septic	Merritt Island	9.73	5.82	10.16
PP32	Septic	Merritt Island	42.57	6.43	7.05
PP33	Septic	Merritt Island	27.71	3.50	5.83
PP34	Septic	Merritt Island	43.89	6.77	4.58
PP35	Septic	Merritt Island	42.94	15.07	1.61
PP36	Septic	Merritt Island	47.10	13.35	7.27
MW SE 460	Sewer	Satellite Beach	14.7	5.6	1.66
MW SE 513	Sewer	Satellite Beach	18.3	4.3	15.95

Well ID	Treatment	Region	Carbonate (%)	Organics (%)	Fines (%)
MW SE 523	Sewer	Satellite Beach	10.5	3.8	5.31
MW SE C1	Sewer	Suntree	2.5	1.7	6.96
MW SE C2	Sewer	Suntree	20.1	3.9	7.79
MW SE C3	Sewer	Suntree	24.1	7.6	10.63
MW SP 6155	Septic	Suntree	16.8	2.5	5.66
MW SP 6215	Septic	Suntree	17.6	2.6	6.84
MW SP 6398	Septic	Suntree	4.7	2.8	32.09
PP37	Septic	Suntree	12.97	8.35	6.48
PP38	Septic	Suntree	22.75	6.20	2.62
PP39	Septic	Suntree	26.02	12.60	6.01
PP40	Septic	Suntree	NO DATA	8.31	2.39
PP41	Septic	Suntree	21.22	6.44	6.33
PP42	Septic	Suntree	14.03	3.58	7.32
PP43	Septic	Suntree	18.50	7.96	5.61
PP44	Septic	Suntree	39.79	6.27	8.82
PP45	Septic	Suntree	47.80	6.95	9.43
PP46	Septic	Suntree	54.19	7.22	12.90
PP47	Septic	Suntree	28.82	8.36	9.48
PP48	Septic	Suntree	4.95	3.70	4.07
PP49	Septic	Suntree	8.82	6.88	5.57
PP50	Septic	Suntree	19.53	5.76	4.21
PP51	Septic	Suntree	7.33	5.34	5.17
PP52	Septic	Suntree	16.16	9.77	9.07
PP53	Septic	Suntree	36.94	15.13	3.37
PP54	Septic	Suntree	3.36	2.89	7.16
PP55	Septic	Suntree	12.68	5.45	5.30
PP56	Septic	Suntree	22.98	6.59	12.69
PP57	Septic	Suntree	3.58	3.20	2.18
PP58	Septic	Suntree	11.76	10.85	7.54
PP59	Septic	Suntree	5.45	4.82	3.32
PP60	Septic	Suntree	22.68	16.37	8.57
PP61	Septic	Suntree	17.99	8.26	8.84
PP62	Septic	Suntree	11.89	10.03	5.64
RE FL 2	Reuse	Suntree	15.2	1.9	11.36
RE FL 3	Reuse	Suntree	18.2	2.5	7.94
RE FL1	Reuse	Suntree	24.1	7.6	11.7
MW RE 1319	Reuse	Titusville	18.6	1.8	3.92
MW RE 2091	Reuse	Titusville	16.5	2.9	1.48
MW RE 549	Reuse	Titusville	7.3	3.5	1.85
MW SE 645	Sewer	Titusville	21.7	4.5	13.77

Well ID	Treatment	Region	Carbonate (%)	Organics (%)	Fines (%)
MWEF1	Natural	Titusville	39.7	3.7	2.61
MWEF2	Natural	Titusville	29.4	2.6	7.18
MWSE 680	Sewer	Titusville	3.9	4.9	2.92
SE 540	Sewer	Titusville	NO DATA	NO DATA	2.02
MW RE2456	Reuse	Turkey Creek	0.28	0.36	1.32
MW REC	Reuse	Turkey Creek	0.43	0.5	1.56
MW REC 3	Reuse	Turkey Creek	24.2	1.8	5.95
MW SE 849	Sewer	Turkey Creek	1.35	2.51	0.85
MW SE841	Sewer	Turkey Creek	1.57	1.04	1.64
MW SP1099	Septic	Turkey Creek	0.28	0.46	1.71
MW SP1127	Septic	Turkey Creek	0.22	0.83	1.68
MW TC1	Natural	Turkey Creek	0.4	0.9	1.66
MW TC2	Natural	Turkey Creek	2.9	2.35	0.78
PP1	Septic	Turkey Creek	6.73	2.27	7.60
PP10	Septic	Turkey Creek	16.43	7.77	5.73
PP11	Septic	Turkey Creek	9.54	8.20	7.49
PP12	Septic	Turkey Creek	9.54	8.20	4.96
PP13	Septic	Turkey Creek	5.68	4.94	12.30
PP14	Septic	Turkey Creek	6.28	5.96	5.57
PP15	Septic	Turkey Creek	5.81	5.49	1.60
PP16	Septic	Turkey Creek	7.36	6.73	6.35
PP17	Septic	Turkey Creek	6.97	6.60	7.19
PP18	Septic	Turkey Creek	8.87	8.37	6.24
PP19	Septic	Turkey Creek	5.93	5.34	10.25
PP2	Septic	Turkey Creek	12.89	8.50	4.05
PP20	Septic	Turkey Creek	6.73	6.15	6.18
PP21	Septic	Turkey Creek	15.51	14.69	8.69
PP3	Septic	Turkey Creek	10.44	9.01	5.30
PP4	Septic	Turkey Creek	5.15	4.58	3.77
PP5	Septic	Turkey Creek	8.66	3.51	13.10
PP6	Septic	Turkey Creek	8.63	7.95	9.92
PP7	Septic	Turkey Creek	11.29	10.68	3.97
PP8	Septic	Turkey Creek	4.50	3.69	5.00
PP9	Septic	Turkey Creek	5.51	4.97	6.52
REC 3	Reuse	Turkey Creek	6.8	1.8	24.7

Appendix D: Countywide Analyte Descriptive Analysis Tables

Total Nitrogen

Table D-1: TN (mg/L) statistics for each treatment type across Brevard County. Highest mean and median values are in bold.

Treatment Type	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	0.379	0.350	0.185	0.565	*0.086	1.400
Reclaimed	**6.046**	**4.200**	1.400	9.000	0.130	21.700
Septic	5.600	2.600	1.300	6.675	0.330	37.600
Sewer	1.918	1.200	0.705	3.200	0.260	9.400

*Measured value below the Minimum Detection Level (MDL)

Nitrate/Nitrite

Table D-2: NO_X (mg/L) statistics for each treatment type across Brevard County. Highest mean and median values are in bold.

Treatment Type	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	0.045	0.028	0.025	0.043	*0.025	0.480
Reclaimed	**5.108**	**2.800**	0.037	8.500	*0.025	21.100
Septic	3.479	0.200	0.025	2.800	*0.025	37.600
Sewer	0.544	0.042	0.025	0.525	*0.025	8.600

*Measured value below the Minimum Detection Level (MDL)

Total Kjeldahl Nitrogen

Table D-3: TKN (mg/L) statistics for each treatment type across Brevard County. Highest mean and median values are in bold.

Treatment Type	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	0.345	0.305	0.160	0.525	*0.086	0.930
Reclaimed	0.970	0.620	0.086	1.400	*0.086	4.900
Septic	**2.111**	**1.000**	0.570	2.400	*0.086	9.300
Sewer	1.388	0.720	0.470	2.350	0.140	5.000

*Measured value below the Minimum Detection Level (MDL)

Ammonia

Table D-4: NH_3 (mg/L) statistics for each treatment type across Brevard County. Highest mean and median values are in bold.

Treatment Type	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	0.093	0.065	0.035	0.110	*0.035	0.460
Reclaimed	0.255	0.035	0.035	0.220	*0.035	2.900
Septic	**1.792**	**0.560**	0.052	2.100	*0.035	9.700
Sewer	0.833	0.160	0.035	0.925	*0.035	4.500
*Measured value below the Minimum Detection Level (MDL)						

Total Phosphorus

Table D-5: TP (mg/L) statistics for each treatment type across Brevard County. Highest mean and median values are in bold.

Treatment Type	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	58	0.081	0.073	0.031	0.130	*0.0028	0.200
Reclaimed	114	0.206	0.052	0.014	0.158	*0.0028	1.300
Septic	114	**0.701**	**0.480**	0.250	1.000	0.0290	3.000
Sewer	144	0.159	0.089	0.027	0.210	0.0068	0.680
*Measured value below the Minimum Detection Level (MDL)							

Orthophosphate

Table D-6. PO_4^{3-}(mg/L) statistics for each treatment type across Brevard County. Highest mean and median values are in bold.

Treatment Type	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	0.130	0.120	0.064	0.188	*0.0038	0.490
Reclaimed	0.229	0.086	0.025	0.220	*0.0038	1.300
Septic	**0.747**	**0.570**	0.305	1.100	0.0064	3.400
Sewer	0.181	0.110	0.036	0.260	*0.0038	0.570
*Measured value below the Minimum Detection Level (MDL)						

Fecal Coliform

Table D-7: Fecal Coliform (CFUs/100mL) statistics for each treatment type across Brevard County. Highest mean value is in bold, and highest geometric mean is italicized.

Treatment Type	Mean	Geometric Mean	25th Percentile	75th Percentile	Minimum	Maximum
Natural/Control	2.69	1.25	1.00	1.00	*1.00	73.00
Reclaimed	12.16	*2.04*	1.00	3.00	*1.00	500.00
Septic	**13.47**	1.86	1.00	2.00	*1.00	500.00
Sewer	12.08	1.50	1.00	1.00	*1.00	500.00
*Measured value below the Minimum Detection Level (MDL)						

Appendix E: Analyte Descriptive Analysis Tables for Comparison of Treatment Types within Regions

Turkey Creek

Table E-1: Turkey Creek statistics per analyte for the period May 2018 – November 2019. As PO_4^{3-} and TP were not sampled for until May 2018, this subset was used for analysis of groundwater nutrients. Highest mean and median values are in bold.

Analyte	Treatment Type	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Natural/Control	38	0.1018	0.0350	0.0350	0.1625	0.0350*	0.4600
	Reclaimed	57	0.0396	0.0350	0.0350	0.0350	0.0350*	0.2400
	Septic	57	2.1210	0.9300	0.0350	4.6500	0.0350*	8.3000
	Sewer	57	**3.0770**	**3.4000**	2.1500	3.9000	1.3000	4.5000
NO_x	Natural/Control	38	0.0507	0.0250	0.0250	0.0432	0.0250*	0.4700
	Reclaimed	57	**10.6590**	**14.1000**	3.1000	15.9500	0.0720	21.1000
	Septic	57	4.6600	0.0400	0.0300	4.300	0.0300	37.600
	Sewer	57	0.0272	0.0250	0.0250	0.0250	0.0250*	0.1000
TKN	Natural/Control	38	0.2615	0.2150	0.0860	0.4000	0.0860*	0.7900
	Reclaimed	57	0.1105	0.0860	0.0860	0.0860	0.0860*	0.4400
	Septic	57	2.5410	1.4000	0.7400	4.8000	0.0860*	8.5000
	Sewer	57	**3.7123**	**3.7000**	3.4000	4.1000	2.8000	5.0000
TN	Natural/Control	38	0.2937	0.2350	0.0860	0.4625	0.0860*	0.8600
	Reclaimed	57	**10.6960**	**14.1000**	3.1000	15.9500	0.1300	21.1000
	Septic	57	7.1900	4.8000	1.5500	7.3500	0.7900	37.6000
	Sewer	57	3.6804	3.7000	3.4000	4.1000	0.5800	5.0000
PO_4^{3-}	Natural/Control	38	0.0695	0.0348	0.0048	0.1425	0.0038*	0.1900
	Reclaimed	57	0.0372	0.0140	0.0096	0.0440	0.0038*	0.7100
	Septic	57	**0.9375**	**0.9700**	0.4950	1.2000	0.1300	2.9000
	Sewer	57	0.4091	0.4900	0.2050	0.5400	0.1400	0.6200
TP	Natural/Control	22	0.0791	0.0565	0.0055	0.1700	0.0028*	0.2000
	Reclaimed	33	0.0849	0.0120	0.0075	0.0995	0.0028*	0.6800
	Septic	33	**0.9373**	**0.9700**	0.5800	1.2000	0.1700	2.0000
	Sewer	33	0.4003	0.4700	0.1800	0.5150	0.1500	0.5700
Fecal	Natural/Control	38	2.8900	1.0000	1.0000	1.0000	1.0000*	73.0000
	Reclaimed	57	**3.6000**	1.0000	1.0000	1.0000	1.0000*	60.0000
	Septic	57	1.8070	1.0000	1.0000	1.0000	1.0000*	16.0000
	Sewer	57	1.4740	1.0000	1.0000	1.0000	1.0000*	17.0000

*Measured value below the Minimum Detection Level (MDL)

Beaches

Table E-2: The Beaches statistics per analyte for the period May 2018 – November 2019. As PO_4^{3-} and TP were not sampled for until May 2018, this subset was used for analysis of groundwater nutrients. Highest mean and median values are in bold.

Analyte	Treatment Type	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Natural/Control	36	0.0568	0.0490	0.0350	0.0733	0.0350*	0.1300
	Reclaimed	54	0.0356	0.0350	0.0350	0.0350	0.0350*	0.0530
	Septic	54	**0.7350**	0.0740	0.0360	0.2530	0.0350*	7.5000
	Sewer	54	0.1193	**0.1000**	0.0585	0.1425	0.0350*	0.4100
NO_x	Natural/Control	36	0.215	0.200	0.160	0.275	0.0250*	0.370
	Reclaimed	54	**6.2200**	**6.2500**	3.6500	8.4250	1.7000	10.6000
	Septic	54	2.7390	0.8550	0.3380	3.0750	0.0250*	18.3000
	Sewer	54	1.0080	0.4050	0.0490	1.2000	0.0250*	8.6000
TKN	Natural/Control	36	0.2147	0.2000	0.1600	0.2850	0.0860*	0.3700
	Reclaimed	54	0.2666	0.0860	0.0860	0.4925	0.0860*	1.3000
	Septic	54	**1.1160**	0.6000	0.4800	1.000	0.0860*	6.5000
	Sewer	54	0.6294	**0.6100**	0.4700	0.7425	0.1400	1.4000
TN	Natural/Control	36	0.2492	0.2300	0.1750	0.3375	0.0860*	0.4400
	Reclaimed	54	**6.4410**	**6.4500**	4.2500	8.4250	2.2000	10.6000
	Septic	54	3.8550	1.5500	0.9730	4.0500	0.3300	19.6000
	Sewer	54	1.6320	1.0500	0.5480	1.9250	0.4100	9.2000
PO_4^{3-}	Natural/Control	36	0.1158	0.1200	0.0640	0.1575	0.0270	0.2000
	Reclaimed	54	0.6889	**0.7500**	0.3875	0.9600	0.0810	1.3000
	Septic	54	**0.7033**	0.4100	0.2975	1.1250	0.1900	2.2000
	Sewer	54	0.0811	0.0825	0.0288	0.1100	0.0038*	0.2100
TP	Natural/Control	18	0.2039	0.2150	0.1650	0.2450	0.0900	0.2800
	Reclaimed	27	0.6990	**0.7200**	0.4200	0.9800	0.0920	1.3000
	Septic	27	**0.7330**	0.4600	0.3200	1.1000	0.2300	2.0000
	Sewer	27	0.0970	0.0980	0.0370	0.1300	0.0240	0.2300
Fecal	Natural/Control	36	3.6700	1.0000	1.0000	1.0000	1.0000*	34.0000
	Reclaimed	54	4.0700	1.0000	1.0000	1.0000	1.0000*	111.0000
	Septic	54	**6.9800**	1.0000	1.0000	7.0000	1.0000*	60.0000
	Sewer	54	29.9000	1.0000	1.0000	3.0000	1.0000*	500.0000

*Measured value below the Minimum Detection Level (MDL)

Merritt Island

Table E-3: Merritt Island statistics per analyte for the period May 2018 – November 2019. As PO_4^{3-} and TP were not sampled for until May 2018, this subset was used for analysis of groundwater nutrients. Highest mean and median values are in bold.

Analyte	Treatment Type	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Septic	54	**1.677**	**1.050**	0.723	1.950	0.0350*	7.700
	Sewer	54	0.551	0.088	0.035	0.350	0.0350*	2.900
NO_x	Septic	54	**1.700**	0.025	0.025	0.188	0.0250*	36.700
	Sewer	54	0.746	**0.120**	0.032	1.375	0.0250*	7.700
TKN	Septic	54	**2.055**	**1.600**	0.915	2.400	0.0860*	7.600
	Sewer	54	1.163	0.685	0.375	1.675	0.1600	3.700
TN	Septic	54	**3.746**	1.700	0.950	5.475	0.4400	36.700
	Sewer	54	1.925	**1.750**	0.933	2.375	0.3500	9.400
PO_4^{3-}	Septic	54	**0.643**	**0.365**	0.180	0.978	0.0450	3.000
	Sewer	54	0.160	0.093	0.030	0.180	0.0038*	0.680
TP	Septic	54	**0.6990**	**0.4300**	0.1700	1.0000	0.1000	3.4000
	Sewer	54	0.2014	0.1500	0.0320	0.3400	0.0200	0.5700
Fecal	Septic	27	**33.7000**	1.0000	1.0000	3.8000	1.0000*	500.0000
	Sewer	27	1.8520	1.0000	1.0000	1.0000	1.0000*	34.0000

*Measured value below the Minimum Detection Level (MDL)

Suntree

Table E-4: Suntree statistics per analyte for the period May 2018 – November 2019. As PO_4^{3-} and TP were not sampled for until May 2018, this subset was used for analysis of groundwater nutrients. Highest mean and median values are in bold.

Analyte	Treatment Type	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Reclaimed	54	0.8230	**0.5750**	0.1780	0.9100	0.0350*	2.9000
	Septic	54	**2.6170**	0.4550	0.0350	6.6750	0.0350*	9.7000
	Sewer	54	0.1697	0.1700	0.0350	0.2500	0.0350*	0.6100
NO_x	Reclaimed	54	0.5930	0.0510	0.0250	0.1700	0.0250*	8.4000
	Septic	54	**4.7500**	**0.6200**	0.0600	4.9000	0.0300	32.4000
	Sewer	54	0.3820	0.0260	0.0250	0.1930	0.0250*	6.1000
TKN	Reclaimed	54	2.2130	**2.1000**	1.2500	3.0000	0.4200	4.9000
	Septic	54	**2.7780**	0.9300	0.3720	6.3500	0.0860*	9.3000
	Sewer	54	0.7835	0.6750	0.4875	0.9025	0.2400	3.2000
TN	Reclaimed	54	2.8050	2.5500	1.3500	3.6000	0.4200	11.5000
	Septic	54	**7.5200**	**6.0500**	2.0800	8.6300	0.7600	32.4000
	Sewer	54	1.1540	0.7850	0.5400	1.1000	0.2600	8.2000
PO_4^{3-}	Reclaimed	54	0.0588	0.0270	0.0140	0.0850	0.0047	0.2500
	Septic	54	**0.5049**	**0.2650**	0.1250	0.8375	0.0064	1.8000
	Sewer	54	0.0209	0.0145	0.0067	0.0283	0.0038*	0.2100
TP	Reclaimed	30	0.0814	0.0360	0.0280	0.0965	0.0110	0.3000
	Septic	27	**0.5762**	**0.4200**	0.1600	0.9300	0.0290	1.7000
	Sewer	30	0.0302	0.0145	0.0100	0.0360	0.0068	0.2600
Fecal	Reclaimed	54	**29.0000**	**3.0000**	1.0000	14.3000	1.0000*	500.0000
	Septic	54	12.0400	1.0000	1.0000	1.0000	1.0000*	500.0000
	Sewer	54	26.4000	1.0000	1.0000	2.3000	1.0000*	500.0000
*Measured value below the Minimum Detection Level (MDL)								

Titusville

Table E-5: Titusville statistics per analyte for the period May 2018 – November 2019. As PO_4^{3-} and TP were not sampled for until May 2018, this subset was used for analysis of groundwater nutrients. Highest mean and median values are in bold.

Analyte	Treatment Type	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Natural/Control	36	0.1214	0.1000	0.0892	0.1300	0.0350*	0.4400
	Reclaimed	54	**0.1338**	0.0770	0.0350	0.2200	0.0350*	0.6200
	Sewer	54	0.1223	**0.1200**	0.0350	0.1800	0.0350*	0.3000
NO_x	Natural/Control	36	0.0444	0.0250	0.0250	0.0375	0.0250*	0.4800
	Reclaimed	54	**2.6540**	0.0290	0.0250	3.5750	0.0250*	20.3000
	Sewer	54	0.5840	**0.1060**	0.0330	0.8920	0.0250*	4.4000
TKN	Natural/Control	36	0.5622	0.5600	0.4525	0.6600	0.2900	0.9300
	Reclaimed	54	**1.3370**	**1.1500**	0.8580	1.5000	0.5500	3.9000
	Sewer	54	0.5219	0.4800	0.3575	0.7200	0.1700	1.0000
TN	Natural/Control	36	0.5972	0.6100	0.4700	0.6800	0.2900	1.4000
	Reclaimed	54	**3.9850**	**1.3000**	0.8750	6.4750	0.5500	21.7000
	Sewer	54	1.1020	0.8150	0.5380	1.2000	0.4100	5.3000
PO_4^{3-}	Natural/Control	36	0.0573	0.0555	0.0350	0.0738	0.0038	0.1300
	Reclaimed	54	0.0591	0.0605	0.0238	0.0853	0.0051	0.1900
	Sewer	54	**0.1097**	**0.0780**	0.0357	0.1900	0.0130	0.2500
TP	Natural/Control	18	0.1181	0.0910	0.0700	0.1250	0.0400	0.4900
	Reclaimed	24	0.1017	**0.1100**	0.0350	0.1300	0.0220	0.3500
	Sewer	27	**0.1441**	**0.1100**	0.0530	0.2200	0.0360	0.3400
Fecal	Natural/Control	36	1.5000	1.0000	1.0000	1.0000	1.0000*	9.0000
	Reclaimed	54	**12.4800**	1.0000	1.0000	3.2500	1.0000*	219.0000
	Sewer	54	1.3520	1.0000	1.0000	1.0000	1.0000*	8.0000

*Measured value below the Minimum Detection Level (MDL)

Appendix F: Analyte Descriptive Analysis Tables for Comparison of Regional Difference across Treatment Types

Natural

PCA

Table F-1: Loadings of 6 water quality variables on the first four PCs for the Natural groundwater samples.

Analyte	PC1	PC2	PC3	PC4
NH_3	0.77	-0.18	-0.43	0.09
NO_X	0.39	0.57	0.66	-0.22
TKN	0.94	-0.21	-0.07	-0.13
TN	0.96	-0.02	0.11	-0.17
PO_4^{3-}	0.35	0.68	-0.15	0.62
3 Day Rainfall Sum	0.16	-0.63	0.57	0.49
Variability (%)	44.9	20.9	16.5	12.1
Cumulative %	44.9	65.8	82.3	94.4

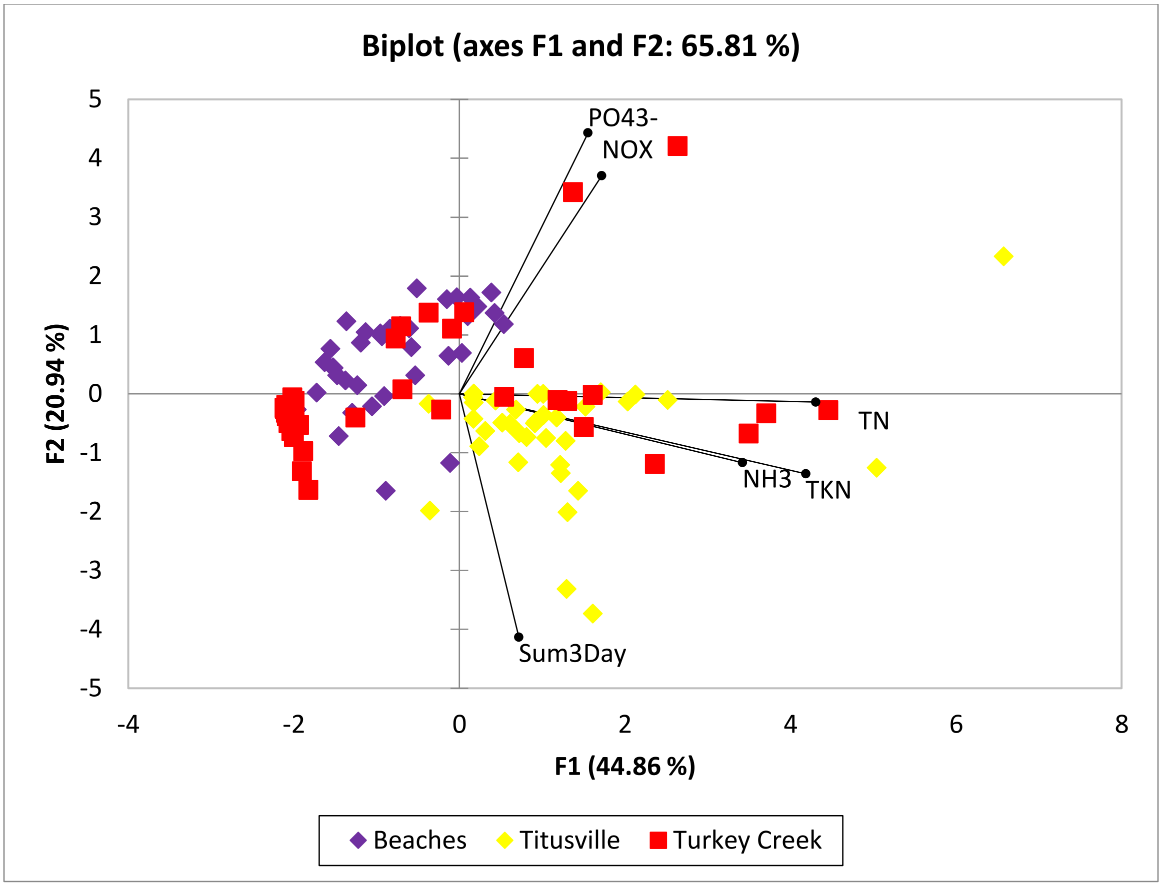


Figure F-1: Coordinates of the Natural treatment PCs based on the region. The color of the dots denotes its classification as shown in the figure legend.

Non-Parametric Analysis

Table F-2. Statistics per analyte for regions containing natural treatments. The highest mean and median values are bolded.

Analyte	Region	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Beaches	36	0.0568	0.0490	0.0350	0.0733	0.0350*	0.1300
	Titusville	36	**0.1214**	**0.1000**	0.0892	0.1300	0.0350*	0.4400
	Turkey Creek	38	0.1018	0.0350	0.0350	0.1625	0.0350*	0.4600
NO_X	Beaches	36	0.0398	**0.0365**	0.0250	0.0475	0.0250*	0.0750
	Titusville	36	0.0444	0.0250	0.0250	0.0375	0.0250*	0.4800
	Turkey Creek	38	**0.0507**	0.0250	0.0250	0.0432	0.0250*	0.4700
TKN	Beaches	36	0.2147	0.2000	0.1600	0.2850	0.0860*	0.3700
	Titusville	36	**0.5622**	**0.5600**	0.4525	0.6600	0.2900	0.9300
	Turkey Creek	38	0.2615	0.2150	0.0860	0.4000	0.0860*	0.7900
TN	Beaches	36	0.2492	0.2300	0.1750	0.3375	0.0860*	0.4400
	Titusville	36	**0.5972**	**0.6100**	0.4700	0.6800	0.2900	1.4000
	Turkey Creek	38	0.2937	0.2350	0.0860	0.4625	0.0860*	0.8600
PO_4^{3-}	Beaches	36	**0.1158**	**0.1200**	0.0640	0.1575	0.0270	0.2000
	Titusville	36	0.0573	0.0555	0.0350	0.0738	0.0038*	0.1300
	Turkey Creek	38	0.0695	0.0348	0.0048	0.1425	0.0038*	0.1900
TP	Beaches	36	**0.2039**	**0.2150**	0.1650	0.2450	0.0900	0.2800
	Titusville	36	0.1181	0.0910	0.0700	0.1250	0.0400	0.4900
	Turkey Creek	38	0.0791	0.0565	0.0055	0.1700	0.0028*	0.2000
Fecal	Beaches	36	**3.6700**	1.0000	1.0000	1.0000	1.0000*	34.0000
	Titusville	36	1.5000	1.0000	1.0000	1.0000	1.0000*	9.0000
	Turkey Creek	38	2.8900	1.0000	1.0000	1.0000	1.0000*	73.0000

*Measured value below the Minimum Detection Level (MDL)

Table F-3. Statistical significance testing for each analyte of all regions containing natural areas. Analytes qualifying for non-parametric testing display median values, while those qualifying for parametric testing display mean values and are an italicized when applicable.

Analyte	Beaches	Titusville	Turkey Creek
***NH_3 (mg/L)**	0.049^{a}	$\mathbf{0.100^{b}}$	0.035^{a}
****NO_X (mg/L)**	$\mathbf{0.037^{a}}$	$0.025^{a,b}$	0.025^{b}
***TKN (mg/L)**	0.200^{a}	$\mathbf{0.560^{b}}$	0.215^{a}
***TN (mg/L)**	0.230^{a}	$\mathbf{0.610^{b}}$	0.235^{a}
***PO_4^{3-} (mg/L)**	$\mathbf{0.120^{a}}$	0.056^{b}	0.035^{b}
***TP (mg/L)**	$\mathbf{0.215^{a}}$	0.091^{b}	0.057^{b}

*Significantly different median at *p*<0.001 using Kruskal-Wallis.
**Significantly different median with p>0.001 and p<0.05 using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts.
Different letters indicate medians with significant differences at p<0.05 within rows. If significant differences were found, the highest value is in bold.

Sewer

PCA

Table F-4: Loadings of six water quality variables on the first four PCs for the Sewer treatment groundwater samples.

Analyte	PC1	PC2	PC3	PC4
NH_3	0.94	-0.18	0.03	-0.24
NO_X	-0.09	0.92	-0.37	0.07
TKN	0.98	-0.09	0.05	-0.14
TN	0.77	0.59	-0.23	-0.08
PO_4^{3-}	0.85	-0.12	0.08	0.50
3 Day Rainfall Sum	-0.01	0.69	0.72	-0.03
Variability (%)	52.6	28.6	12.1	5.7
Cumulative %	52.6	81.2	93.3	99.0

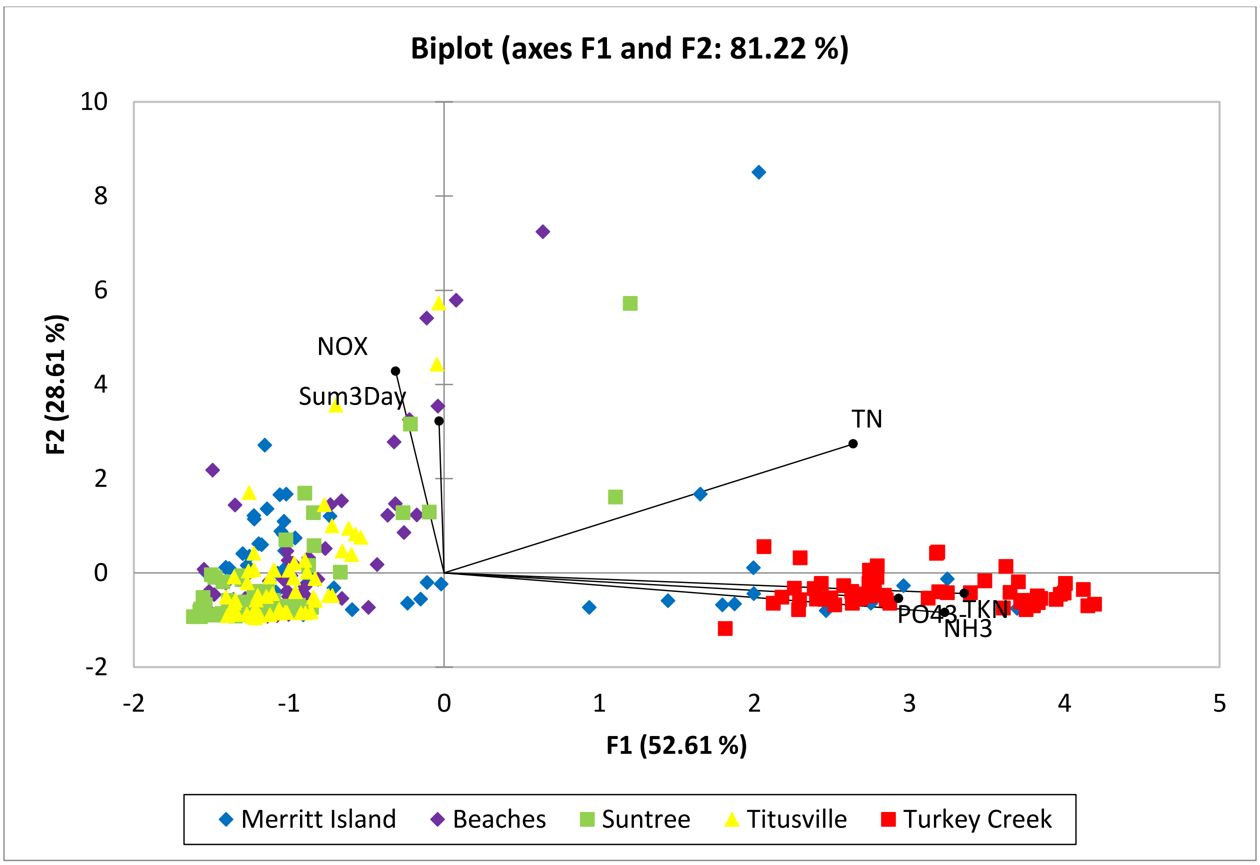


Figure F-2: Coordinates of the Natural treatment PCs based on the region. The color of the dots denotes its classification as shown in the figure legend.

Non-Parametric Analysis

Table F-5: Statistics per analyte for regions containing sewer treatments. The highest mean and median values are bolded.

Analyte	Region	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Merritt Island	54	0.5510	0.0880	0.0350	0.3800	0.0350*	2.9000
	Beaches	54	0.1193	0.1000	0.0585	0.1425	0.0350*	0.4100
	Suntree	54	0.1697	0.1700	0.0350	0.2500	0.0350*	0.6100
	Titusville	54	0.1223	0.1200	0.0350	0.1800	0.0350*	0.3000
	Turkey Creek	57	**3.0770**	**3.4000**	2.1500	3.9000	1.3000	4.5000
NO_x	Merritt Island	54	0.7460	0.1200	0.0300	1.4250	0.0250*	7.7000
	Beaches	54	**1.0080**	**0.4050**	0.0490	1.2000	0.0250*	8.6000
	Suntree	54	0.3820	0.0260	0.0250	0.1930	0.0250*	6.1000
	Titusville	54	0.5840	0.1060	0.0330	0.8920	0.0250*	4.4000
	Turkey Creek	57	0.0272	0.0250*	0.0250	0.0250	0.0250*	0.1000
TKN	Merritt Island	54	1.1630	0.6850	0.3670	1.7000	0.1600	3.7000
	Beaches	54	0.6294	0.6100	0.4700	0.7425	0.1400	1.4000
	Suntree	54	0.7835	0.6750	0.4875	0.9025	0.2400	3.2000
	Titusville	54	0.5219	0.4800	0.3575	0.7200	0.1700	1.0000
	Turkey Creek	57	**3.7123**	**3.7000**	3.4000	4.1000	2.8000	5.0000
TN	Merritt Island	54	1.9250	1.7500	0.8970	2.4000	0.3500	9.4000
	Beaches	54	1.6320	1.0500	0.5480	1.9250	0.4100	9.2000
	Suntree	54	1.1540	0.7850	0.5400	1.1000	0.2600	8.2000
	Titusville	54	1.1020	0.8150	0.5380	1.2000	0.4100	5.3000
	Turkey Creek	57	**3.6804**	**3.7000**	3.4000	4.1000	0.5800	5.0000
PO_4^{3-}	Merritt Island	54	0.1595	0.0925	0.0300	0.2075	0.0038*	0.6800
	Beaches	54	0.0811	0.0825	0.0288	0.1100	0.0038*	0.2100
	Suntree	54	0.0209	0.0145	0.0067	0.0283	0.0038*	0.2100
	Titusville	54	0.1097	0.0780	0.0357	0.1900	0.0130	0.2500
	Turkey Creek	57	**0.4091**	**0.4900**	0.2050	0.5400	0.1400	0.6200
TP	Merritt Island	27	0.2014	0.1500	0.0320	0.3400	0.0200	0.5700
	Beaches	27	0.0970	0.0980	0.0370	0.1300	0.0240	0.2300
	Suntree	30	0.0302	0.0145	0.0100	0.0360	0.0068	0.2600
	Titusville	27	0.1441	0.1100	0.0530	0.2200	0.0360	0.3400
	Turkey Creek	33	**0.4003**	**0.4700**	0.1800	0.5150	0.1500	0.5700
Fecal	Merritt Island	54	1.8520	1.0000*	1.0000	1.0000	1.0000*	34.0000
	Beaches	54	**29.9000**	1.0000*	1.0000	3.0000	1.0000*	500.0000
	Suntree	54	26.4000	1.0000*	1.0000	2.3000	1.0000*	500.0000
	Titusville	54	1.3520	1.0000*	1.0000	1.0000	1.0000*	8.0000
	Turkey Creek	57	1.4740	1.0000*	1.0000	1.0000	1.0000*	17.0000

*Measured value below the Minimum Detection Level (MDL)

Table F-6: Statistical significance testing for each analyte. Analytes qualifying for non-parametric testing (Kruskal-Wallis) display median values.

Analyte	Merritt Island	Beaches	Suntree	Titusville	Turkey Creek
***NH_3 (mg/L)**	0.088[a]	0.100[a]	0.170[a]	0.120[a]	**3.400[b]**
***NO_X (mg/L)**	0.120[a]	0.405[a]	0.027[b]	0.106[a]	0.025[c]
***TKN (mg/L)**	0.685[a]	0.610[a]	0.675[a]	0.480[a]	**3.700[b]**
***TN (mg/L)**	1.750[a]	1.050[a,b]	0.785[c]	0.815[c]	3.700[d]
***PO_4^{3-} (mg/L)**	0.093	0.083	0.015	0.078	0.490
***TP (mg/L)**	0.150	0.098	0.015	0.110	0.470

*Significantly different median at $p<0.001$ using Kruskal-Wallis.

**Significantly different median with $p>0.001$ and $p<0.05$ using Kruskal-Wallis. Pairwise comparisons (SDCF) are indicated by the use of subscripts. Different letters indicate medians with significant differences at $p<0.05$ within rows. If significant differences were found, the highest value is in bold.

Septic

Table F-7: Statistics per analyte for regions containing septic treatments. The highest mean and median values are bolded.

Analyte	Region	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Beaches	54	0.735	0.074	0.038	0.245	0.035	7.500
	Merritt Island	54	1.677	**1.050**	0.723	1.950	0.035	7.700
	Suntree	54	**2.617**	0.455	0.038	6.575	0.035	9.700
	Turkey Creek	57	2.121	0.930	0.035	4.500	0.035	8.300
NO_X	Beaches	54	2.739	**0.855**	0.345	2.725	0.025	18.300
	Merritt Island	54	1.700	0.025	0.025	0.188	0.025	36.700
	Suntree	54	**4.749**	0.620	0.063	4.900	0.025	32.400
	Turkey Creek	57	4.662	0.037	0.025	3.800	0.025	37.600
TKN	Beaches	54	1.116	0.600	0.483	0.988	0.086	6.500
	Merritt Island	54	2.055	**1.600**	0.915	2.400	0.086	7.600
	Suntree	54	**2.778**	0.930	0.393	6.275	0.086	9.300
	Turkey Creek	57	2.541	1.400	0.760	4.800	0.086	8.500
TN	Beaches	54	3.855	1.550	1.025	3.625	0.330	19.600
	Merritt Island	54	3.746	1.700	0.950	5.475	0.440	36.700
	Suntree	54	**7.522**	**6.050**	2.125	8.350	0.760	32.400
	Turkey Creek	57	7.187	4.800	1.600	7.000	0.790	37.600
PO_4^{3-}	Beaches	54	0.703	0.410	0.300	1.100	0.190	2.200
	Merritt Island	54	0.643	0.365	0.180	0.978	0.045	3.000
	Suntree	54	0.505	0.265	0.133	0.810	0.006	1.800
	Turkey Creek	57	**0.938**	**0.970**	0.500	1.200	0.130	2.900
TP	Beaches	27	0.733	0.460	0.320	1.100	0.230	2.000
	Merritt Island	27	0.699	0.430	0.170	0.920	0.100	3.400
	Suntree	27	0.576	0.420	0.170	0.925	0.029	1.700
	Turkey Creek	27	**0.937**	**0.970**	0.590	1.200	0.170	2.000
Fecal	Beaches	54	6.9800	1.0000*	1.0000	7.0000	1.0000*	60.0000
	Merritt Island	54	**33.7000**	1.0000*	1.0000	3.8000	1.0000*	500.0000
	Suntree	54	6.9800	1.0000*	1.0000	7.0000	1.0000*	60.0000
	Turkey Creek	54	12.0400	1.0000*	1.0000	1.0000	1.0000*	500.0000

*Measured value below the Minimum Detection Level (MDL)

Reclaimed

Table F-8. Statistics per analyte for regions containing reclaimed treatments. The highest mean and median values are bolded.

Analyte	Region	N	Mean	Median	25th Percentile	75th Percentile	Minimum	Maximum
NH_3	Beaches	54	0.0356	0.0350	0.0350	0.0350	0.0350*	0.0530
	Suntree	54	**0.8230**	**0.5750**	0.1780	0.9100	0.0350*	2.9000
	Titusville	54	0.1338	0.0770	0.0350	0.2200	0.0350*	0.6200
	Turkey Creek	57	0.0396	0.0350	0.0350	0.0350	0.0350*	0.2400
NO_x	Beaches	54	6.2200	6.2500	3.6500	8.4250	1.7000	10.6000
	Suntree	54	0.5930	0.0510	0.0250	0.1700	0.0250*	8.4000
	Titusville	54	2.6540	0.0290	0.0250	3.5750	0.0250*	20.3000
	Turkey Creek	57	**10.6590**	**14.1000**	3.1000	15.9500	0.0720	21.1000
TKN	Beaches	54	0.2666	0.0860	0.0860	0.4925	0.0860*	1.3000
	Suntree	54	**2.2130**	**2.1000**	1.2500	3.0000	0.4200	4.9000
	Titusville	54	1.3370	1.1500	0.8580	1.5000	0.5500	3.9000
	Turkey Creek	57	0.1105	0.0860*	0.0860	0.0860	0.0860*	0.4400
TN	Beaches	54	6.4410	6.4500	4.2500	8.4250	2.2000	10.6000
	Suntree	54	2.8050	2.5500	1.3500	3.6000	0.4200	11.5000
	Titusville	54	3.9850	1.3000	0.8750	6.4750	0.5500	21.7000
	Turkey Creek	57	**10.6960**	**14.1000**	3.1000	15.9500	0.1300	21.1000
PO_4^{3-}	Beaches	54	**0.6889**	**0.7500**	0.3875	0.9600	0.0810	1.3000
	Suntree	54	0.0588	0.0270	0.0140	0.0850	0.0047	0.2500
	Titusville	54	0.0591	0.0605	0.0238	0.0853	0.0051	0.1900
	Turkey Creek	57	0.0372	0.0140	0.0096	0.0440	0.0038*	0.7100
TP	Beaches	27	**0.6990**	**0.7200**	0.4200	0.9800	0.0920	1.3000
	Suntree	30	0.0814	0.0360	0.0280	0.0965	0.0110	0.3000
	Titusville	24	0.1017	0.1100	0.0350	0.1300	0.0220	0.3500
	Turkey Creek	33	0.0849	0.0120	0.0075	0.0995	0.0028*	0.6800
Fecal	Beaches	54	4.0700	1.0000*	1.0000	1.0000	1.0000*	111.0000
	Suntree	54	**29.0000**	**3.0000**	1.0000	14.3000	1.0000*	500.0000
	Titusville	54	12.4800	1.0000*	1.0000	3.2500	1.0000*	219.0000
	Turkey Creek	57	3.6000	1.0000*	1.0000	1.0000	1.0000*	60.0000

*Measured value below the Minimum Detection Level (MDL)